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Eco-efficient construction and building materials

Life cycle assessment (LCA), eco-labelling and case studies

Edited by
F. Pacheco-Torgal, L. F. Cabeza,
J. Labrincha and A. de Magalhães

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1

Introduction to the environmental impact of construction and building materials

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Abstract: Earth's natural resources are finite and face increasing human pressure. Over the last few decades, concern has been growing about resource efficiency and the environmental impact of material consumption. The construction industry is responsible for the consumption of a relevant part of all produced materials, however, only recently has this industry started to worry about its environmental impacts. This chapter highlights relevant landmarks on sustainable development, materials efficiency and on the assessment of the environmental impact of construction products. An overview on the European Construction Products Regulation (CPR) enforced since the 1 July 2013 is given followed by an outline of the book.

Key words: sustainable development, materials efficiency, environmental impact, LCA, eco-labels, product self-declarations.

1.1 Introduction

Four decades ago, several investigators used a computer model based on the fixed-stock paradigm to study the interactions between population, food production, industrial production, pollution and the consumption of non-renewable resources. They predicted that during the 21st century, the Earth's capacity would be exhausted, resulting in the collapse of human civilisation as we know it (Meadows *et al.*, 1972). Two decades later, an update of this study was published showing that some limits had already been crossed (Meadows *et al.*, 1992). Whilst the particular assumptions and predictions of such studies have been questioned, there is general agreement that may of the Earth's key resources are finite and must be conserved. As a consequence, the concept of 'sustainable development' gained international recognition through the landmark Brundtland Report 'Our common future' (Brundtland, 1987).

Some authors (Clayton, 2001; Choi and Pattent, 2001) have argued that 'sustainable development' is an oxymoron: we cannot have development/growth for the entire world population and at the same time, expect this development to be compatible with protection of the environment. The fact that there have been serious environmental disasters in Europe, such as Stava (1985), Aznalcollar (1998), Baia Borsa (2000) and Kolontar (2010), despite the region's relatively high environmental standards, illustrates this problem. The challenge is also highlighted by continued growth in materials use. Europe has the world's highest net imports of resources per person, and its open economy relies heavily on imported raw materials and energy. In 2007 the total amount of material directly used in the EU economy was more than 8 billion tonnes (COM, 2011a).

It has been estimated that global materials use increased eight-fold in the last century and that current usage is almost 60 billion tons (Gt) of materials per year (Krausmann *et al.*, 2009). Despite this huge historic growth, some authors predict that materials demand will double by 2050 (Allwood *et al.*, 2011). It is important to note that 40% of all materials are used by the construction industry (Kulatunga *et al.*, 2006). The construction industry is expected to continue to grow rapidly. As an example, it is estimated that India will invest US$ 1 trillion in infrastructure between 2012 and 2017 (Chakraborty *et al.*, 2011). In the USA, where around 27% of all highway bridges are in need of repair or replacement, the needs for infrastructure rehabilitation alone are estimated to be over US$ 1.6 trillion during the next five years, (Davalos, 2012). Wang *et al.* (2010) have estimated that construction activities in China consume approximately 40% of its total natural resources and around 40% of its energy and that the country will need 40 billion square metres of combined residential and commercial floor space over the next 20 years – equivalent to adding one New York City every two years (Pacheco-Torgal and Labrincha, 2013a; Pacheco-Torgal and Jalali, 2011).

The World Business Council for Sustainable Development estimates that by 2050, a four- to ten-fold increase in resource efficiency will be needed (COM, 2011a). Over the last few decades, concern has been growing about resource efficiency and the environmental impact of material consumption. As a result, the term 'green materials' became very popular in the construction sector. However, it was not until 2012 that the first life cycle assessment (LCA) investigations into standard structural concrete made using Portland cement started to become available (Van den Heede and De Belie, 2012; Habert *et al.*, 2012). This is despite the fact that it is the most used construction material with output currently about 10 km^3/year. In comparison, the amount of fired clay, timber, and steel used in construction represents about 2, 1.3 and 0.1 km^3, respectively (Flatt *et al.*, 2012). There is still much to investigate concerning the LCA of this material, for example incorporating recent nano and biotech approaches (Jayapalan *et al.*, 2013; Pacheco-Torgal and Labrincha, 2013b).

1.2 Environmental impact assessment

The methodology used to assess the environmental impacts of a given material is known as 'life cycle assessment' (LCA) and 'includes the complete life cycle of the product, process or activity, i.e., the extraction and processing of raw materials, manufacturing, transportation and distribution, use, maintenance, recycling, reuse and final disposal' (SETAC, 1993). The application of LCA has been regulated internationally since 1996 under the International Standards Organisation (ISO) which classifies the existing environmental labels into three typologies – types I(eco-labels, ISO 14024), type II (product self-declarations, ISO 14021), and type III (EPDs, ISO 14025). It should be noted that in 2012, the DG Environment published the draft of a harmonised methodology for calculating of the environmental footprint of products (Del Borghi, 2013). Since the first eco-label, the German Blue Angel, was created in 1978, several others have appeared. However, some authors (Rajagopalan *et al*., 2012) argue that 'the labelling of green materials is confusing and that consumers are suspicious about the environmental claims of manufacturers.'

Hauschild *et al*. (2013) state that the LCA standard ISO 14040-44 is rather generalised and non-specific in its requirements and offers little help to the LCA practitioner in making choices. The weighting process related to decision making as to which environmental impacts are most significant for the process or product in question remains a controversial and inexact science (Johnsen and Løkke, 2013). Other issues also remain as open questions in the LCA methodology (Feifel *et al*., 2010).

Other important subjects must also be taken into account in considering the future environmental impact of construction and building materials. For example, it remains to be seen if the benefits of recycling should be credited to the primary producer or to the user of recycled materials (Huang *et al*., 2013; Chen *et al*., 2010). This is a crucial issue in the context of the Revised Waste Framework Directive (WFD) 2008/98/EC (EU, 2008) which established that by 2020, the minimum recycling percentage of 'non-hazardous' construction and demolition wastes should be at least 70% by weight (Pacheco-Torgal *et al*., 2013). Improving the reuse of raw materials through greater 'industrial symbiosis' across the EU could save €1.4bn a year and generate €1.6bn in sales (COM, 2011a).

The simplifications of LCAs are unlikely to cope well with the increased importance of the environmental impacts of construction and building materials in the context of low energy buildings. (Kellenberger and Althaus, 2009; Blengini and Di Carlo, 2010). It should be noted that higher energy efficiency in new and existing buildings is the key for transformation of the EU's energy system (COM, 2011b). According to the European Energy Performance of Buildings Directive (EU, 2010), all new constructions will have to be nearly zero-energy by 31 December 2020.

Current labelling schemes are only concerned with short term volatile organic compound emissions from building materials. Research on long-term emissions is therefore needed to reduce the level of uncertainty (Skaar and Jørgensen, 2012). It should be remembered that more than 100,000 new chemical compounds have been developed since 1939 and insufficient information exists for health assessments of 95% of the chemicals used in construction products (Pacheco-Torgal *et al.*, 2012).

It is also significant that product replacement may take place over a very short period of time as occupant behaviour is influenced by societal trends. Therefore estimation methods capable of capturing consumer behaviour are a necessary step towards modelling over a lifetime (Aktas and Bilec, 2011). In future, it will be necessary to apply dynamic methods to the LCI or LCIA (Collinge *et al.*, 2013).

1.3 The European Construction Products Regulation (CPR)

The European Union has been in the lead on seven important initiatives to address smart, sustainable and inclusive growth up to 2020 and beyond. One of these: 'A resource-efficient Europe – Flagship initiative under the Europe 2020 Strategy' (European Commission, 2010), highlights the importance of increasing resource efficiency as the key to major economic opportunities, improving productivity, driving down costs and boosting competitiveness.

The EU has recently passed regulations that will make the environmental assessment of construction and building materials mandatory. On 9 March 2011 the European Union approved Regulation 305/2011 (EU, 2011), the CPR, which replaced Directive 89/106/EEC, already amended by Directive 1993/68/EEC, known as the Construction Products Directive (CPD). The new CPR was published in the *Official Journal of the European Union* (*OJEU*) on 4 April 2011. In accordance with Article 68, the CPR entered into force on 24 April, the 20th day following its publication in the *OJEU*. This includes Articles 1 and 2, 29 to 35, 39 to 55, 64, 67 and 68, and Annex IV. However, Articles 3 to 28, 36 to 38, 56 to 63, 65 and 66, as well as Annexes I, II, III and V, will apply from 1 July 2013. Therefore the CPR will be fully enforced without the requirement for any national legislation by 1 July 2013.

A regulation is defined as follows: 'Shall have general application. It shall be binding in its entirety and directly applicable in all Member States'. The CPD statues are 'binding, as to the result to be achieved, upon each Member State to which it is addressed, but shall leave to the national authorities the choice of form and methods'. This means that the UK, Ireland and Sweden will lose their 'opt-out' clause employed under the CPD period.

When the basic requirements of the CPR and CPD are compared, it may be seen that the CPR has a new requirement (No. 7 Sustainable use of natural resources), and that No. 3 (Hygiene, health and the environment) and No. 4 (Safety and accessibility in use) have been refined. This means that commercialisation of construction materials in Europe beyond 2013 will be subject to mandatory environmental assessment. Updated literature will be needed to help the construction industry as this sector generates almost 10% of the European GDP and provides 20 million jobs, mainly in micro and small enterprises, thus playing an important role in the European economy (COM, 2012).

1.4 Outline of the book

This book covers several aspects of the environmental impacts of construction and building materials, including cases studies offering an overview of materials in the wider context of final in-use.

The first part encompasses an overview of relevant issues for LCA, eco-labelling and procurement (Chapters 2–9).

Chapter 2 concerns resource depletion. The scarcity of minerals is assessed in a static way through the reserves to production ratio as well as by a dynamics viewpoint through Hubbert peak models. This chapter includes a new methodology for abiotic resource depletion.

Chapter 3 covers LCA software tools.

Chapter 4 addresses the possibilities and limitations of LCA. It describes the methodological options that will potentially influence the results from the LCA.

Chapter 5 is concerned with LCA and eco-labels.

Chapter 6 describes the EU Eco-label and reviews its history, goals and statistics. Consideration is given to the criteria for obtaining a label. Special attention is given to the Eco-label 'product groups' which are of interest to the construction and building materials sector: floor coverings (hard, wood, textile), paints and varnishes.

Chapter 7 analyses the eco-label type III: Environmental Product Declaration (EPD) which is regulated by the ISO 14025. The methodological aspects of the EPD development process are described. Important EPD Programmes from around the world are provided as well as details on the product category rules (PCR) for construction and building materials. Three case studies of EPDs for the construction industry are presented (concrete, thermal insulation materials and wood boards).

Chapter 8 covers the shortcomings of eco-labelling.

Chapter 9 addresses green public procurement (GPP) which involves the incorporation of environmental requirements during the procurement of services and products by public authorities. It analyses the expansion of GPP into sustainable public procurement (SPP), where social and sustainable development considerations are also integrated, and discusses the implementation of GPP/SPP in the construction sector.

The environmental impact of construction and building materials are the subject of Part II (Chapters 10–16).

Chapter 10 analyses the environmental impact of cementitious materials. A detailed description of Portland cement production process is offered, including its main environmental impacts (CO_2, PM10, SO_x and NO_x emissions). Future improvements (alternative fuels, energy efficiency, CCS technology, cement consumption efficiency) are described and descriptions of SCMs and their environmental impact are presented. The use of alternative binders is also considered, including alkali activated alumino-silicates, calcium sulfoaluminate cements, celitement and cements from magnesium silicates. The reduction of the environmental impact of cementitious materials through improvement of their mechanical strength is also addressed.

Chapter 11 offers an environmental assessment of structural concrete and presents LCA results for concretes with three different types of aggregate: natural gravel, natural crushed and recycled concrete aggregate. The influence of the transport phase and CO_2 uptake during the life cycle of concrete structure is discussed.

Chapter 12 looks at thermal insulation materials and analysis carried out by using SimaPro and the Eco-indicator 99 software characterisation method. This chapter also includes an economic analysis of insulation materials.

Chapter 13 is concerned with the LCA of buildings including PCMs. It evaluates the environmental impact of using PCMs in building envelopes, using the impact assessment method Eco-Indicator 99 which is extracted from the database Eco-Invent 2009. Two different construction systems are assessed: conventional brick and alveolar brick based.

Chapter 14 reviews the LCA of wooded-based building products and considers their manufacturing processes. It discusses design and building with wood and the material and energy flows associated with wood-based construction.

Chapter 15 is concerned with adhesives.

Chapter 16 closes Part II with a closer look at roadway pavements. Results on the LCA over a service period of 30 years for four different pavement types (two concrete pavements and two asphalt pavements) are presented. Environmental implications and green alternatives are discussed.

Part III (Chapters 17–23) deals with environmental impact of particular types of structure.

Chapter 17 compares the environmental strengths and weaknesses of wood and concrete in construction. The use of LCA for wood and concrete buildings design is included. This chapter also includes a case study which compares the LCA of two buildings with the different structural materials (wood and concrete). Open source software, open LCA (GreeDeltaTD) was used for the inventory phase and Eco-indicator 99 was used in the assessment phase.

Chapter 18 reviews prefabricated technologies. Comparisons between prefabricated and non-prefabricated methods are made in relation to their economic, environmental and social impacts. This chapter also includes an environmental assessment of the main technologies used in prefabricating 161 school buildings in Catalonia, Spain, between 2002 and 2009 using a multi-criteria decision-making method.

Chapter 19 examines green wall systems and compares the environmental impact of a bare brick façade with four greening systems:

1. A conventional façade covered with a climber planted at the base of the façade (greened directly).
2. A conventional façade covered with a climber planted at the base of the façade using a stainless steel framework to create a cavity between foliage and façade (greened indirectly).
3. A conventional façade covered with a living wall system (LWS) based on planter boxes filled with potting soil.
4. A conventional façade covered with a living wall system based on felt layers and a conventional façade covered with a living wall system based on mineral wool.

Chapter 20 covers cladding systems. It gives a brief description of the use of the concept of eco-efficiency in the assessment of green cladding systems and presents a systematic assessment of an actual case study.

Chapter 21 compares LCA studies on the impacts of different framing materials with mixed results.

Chapter 22 deals with the LCA of ultra-high performance concrete (UHPC), using the software SimaPro and the Swiss ecoinvent database for life cycle inventory data. The chapter includes different applications of UHPC which are compared to conventional building materials: (a) high rise building columns; (b) five different hot rolled I-beams; (c) two traffic bridge design models and three footbridges.

Chapter 23 closes Part III and covers the LCA of fibre reinforced polymer (FRP) composites. Three LCA case studies from both the composites and civil engineering industries are presented. The first concerns three generic product types: (a) a double curvature monolithic panel, (b) a flat sandwich panel with core, and (c) a complex moulded component. The second is related to a pedestrian bridge in the Netherlands. In the third, FRP decks are examined by comparing the life cycle environmental performance, in carbon terms, with the conventional concrete option.

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2

Mineral resource depletion assessment

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Abstract: This chapter analyses the state of depletion of the Earth's mineral resources. It first examines the production trend of the main mineral commodities produced in the world between 1900 and 2008. The scarcity of minerals is first assessed statically through the well-known reserves to production ratio. This approach is then complemented by a dynamic assessment of mineral resource depletion, using the Hubbert peak model applied to a selection of building materials. Finally, a methodology is presented that includes a material scarcity factor in common accounting practices used in assessing construction projects.

Key words: minerals, production trends, Hubbert peak, reserves, scarcity, accounting.

2.1 Introduction

The Earth's continental crust is the source of the goods essential for industrial civilisation with fuels, metals and non-metallic minerals being the fundamental basis for the technological and economic development of any country. Yet, as Dunham (1974) stated, although the whole continental crust is composed by rocks as solid solutions of minerals, these are not *all* in practice recoverable. Really, only when a combination of natural processes has worked together to produce an enrichment is an ore to be found. Such complex processes operate very slowly when compared to the lifespan of humans as a species. Hence, the non-renewable nature of mineral resources should be clear, at least from a human perspective.

Since at least the latter half of the nineteenth century, great technological innovations have led to the consumption and further dispersion of huge amounts of mineral resources previously concentrated in natural deposits. This activity helped to sustain decades of continuous industrial growth, pushing up the economies of steadily more industrialised countries. Yet it also, more recently, raised concerns regarding resource scarcity with, perhaps, the possibility of total depletion of energy resources provoking the highest levels of anxiety, especially due to the sharp rise in fuel prices, as will be seen later in this chapter. However, non-fuel resources are also being exhausted very rapidly. Indeed, as shown by Morse and Glover (2000), over the span of the last century and only in the US, the demand for metals has grown from a little over 160 million tonnes to about 3.3 billion tonnes.

The general attitude that governed in the past was 'the Earth is nothing more than resources to be used'. Adam Smith's invisible hand (Smith, 1904) has been a guiding principle for those who believe that neo-liberalism (free open markets and trade) ultimately lead to the natural order of things. Nevertheless, in the early 1970s the first Arab oil embargo, the peaking of oil production, together with the studies of the Club of Rome (Meadows *et al.*, 1972), challenged this view and began sounding alarm bells regarding resource scarcity as the limit to economic growth (Menzie *et al.*, 2005).

The aim of this chapter is thus to analyse the extent of depletion of the mineral capital, focusing on construction and building materials. It examines first the production trends of the main mineral commodities produced worldwide between 1900 and 2008. The scarcity of minerals is assessed first under static expectations through the well-known reserves to production ratio and through the degree of depletion of mineral deposits. This approach is complemented with the dynamic assessment of minerals. For this reason, the Hubbert peak model is applied to a selection of building materials. Finally, a methodology is presented to include the scarcity factor in common accounting practices.

2.2 Definition and classification of mineral resources

The whole continental crust is made up of minerals. Yet, only a small part of it is in practice recoverable and forms part of what are commonly known as mineral resources. According to Yoder (1995), all concentrated mineral resources of fuel and non-fuel origin represent only 0.001% of the total mass of the Earth's upper continental crust. The *Encyclopaedia Brittanica* defines a mineral deposit as an aggregate of a mineral in an unusually high concentration. According to this definition, any mineral aggregate with a higher concentration than that of the average crust constitutes a mineral deposit or ore. Other definitions add the qualification that concentrations should be high enough for profitable extraction. But what is meant by profitable? What are the concentration limits that define a deposit as profitable? Indeed, these questions are difficult to answer as many different factors come into play when a given concentration of minerals is considered as a resource.

Probably, the most widely used classification of mineral resources is that proposed by the US Geological Survey (USGS, 1980) and depicted in Table 2.1. Accordingly, a *resource* is a concentration of naturally occurring solid, liquid or gaseous material in or on the Earth's crust in such a form and amount that economic extraction of a commodity from the concentration is currently or potentially feasible. An *identified resource* is one whose location, grade, quality and quantity are known or estimated from specific

Table 2.1 Major elements of mineral resource classification excluding reserve base and inferred reserve base according to the USGS (1980)

Cumulative production	IDENTIFIED RESOURCES			UNDISCOVERED RESOURCES	
	Demonstrated		Inferred	Probability range	
	Measured	Indicated		Hypothetical	Speculative
ECONOMIC	Reserves		Inferred reserves		
MARGINALLY ECONOMIC	Marginal reserves		Inferred marginal reserves		
SUBECONOMIC	Demonstrated subeconomic resources		Inferred subeconomic resources		
Other occurences	Includes non-conventional and low-grade material				

geologic evidence. To reflect varying degrees of geologic certainty, these economic divisions can be subdivided into *measured*, *indicated*, and *inferred*. The sum of measured and indicated deposits is called *demonstrated*. The *reserve base* is defined as that part of an identified resource that meets specified minimum physical and chemical criteria related to current mining and production practices, including those for grade, quality, thickness and depth. And *reserves* are that part of the reserve base which could be economically extracted or produced at the time of determination.

Resources are further classified into economic, marginally economic and subeconomic. *Economic* implies that profitable extraction or production under defined investment assumptions has been established, analytically demonstrated, or assumed with reasonable certainty. *Marginal reserves* are that part of the reserve base which, at the time of determination, borders on being economically producible. Its essential characteristic is economic uncertainty. Included are resources that would be producible, given postulated changes in economic or technological factors. Finally, *subeconomic* refers to the part of identified resources that does not meet the economic criteria of reserves and marginal reserves.

The US Geological Survey and the British Geological Survey are two of the main institutions providing global yearly commodity statistics. The information dates back in some cases to the beginning of the twentieth century or even before. In the case of the USGS, the statistics available include production data as well as amount of world reserves and resources. However, there are important information gaps in the mineral yearbooks since much of the compiled data is incomplete and must be estimated. For this reason, the production and resources figures of past years need to be regularly updated as the information becomes available. As pointed out by the USGS itself, 'The classification of mineral and energy resources is necessarily arbitrary, because definitional criteria do not always coincide with natural boundaries.' Moreover, many resources that are considered uneconomic at present become profitable when commodity prices increase or production costs decrease due to technological improvements or economies of scale. Certainly, it is not easy to determine the quantity of mineral resources and reserves available, particularly since this information has potential strategic value and is thus made confidential by companies and governments.

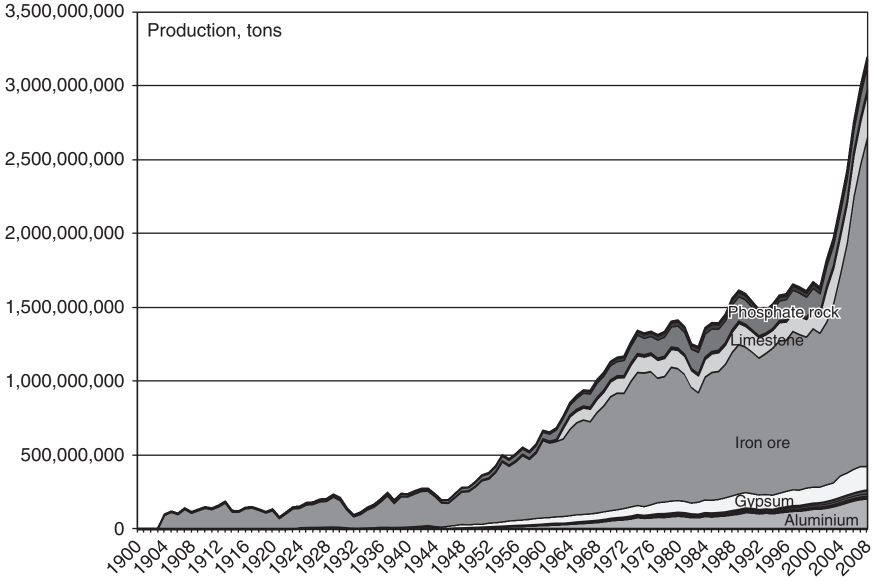
2.3 Trends in mineral use and depletion

From the mineral commodity summaries published in the USGS Mineral Yearbook, a general picture of the production of non-fuel minerals can be obtained. Figures 2.1 and 2.2 show historical cumulative production data of various important minerals from 1900 to 2008.

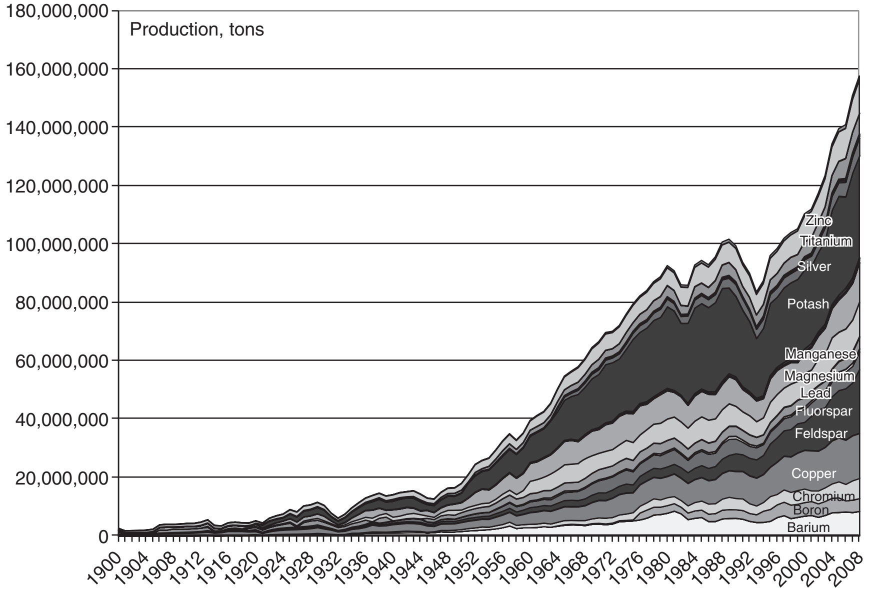
It should be noted that due to lack of information, silica (or sand and gravel – probably the most common building material) and clay minerals (traditionally used as structural building materials) are not included in the list. Such sands have many uses including in glassmaking and for foundry or abrasive applications. Silica and clay minerals are abundant and hence production and resources are difficult to assess.

According to the data shown in Figs 2.1 and 2.2, production of most materials has grown exponentially. The cumulative production passed from around 6 Mtons at the beginning of the twentieth century to 3.2 Gtons in 2008. This trend was taken into account by the International Energy Agency (IEA, 2008) which forecast that demand for materials will at least double current levels by 2050.

Figures 2.1 and 2.2 show that the most extracted commodities are closely related to the building sector. The most extracted mineral so far is iron ore, the basic raw material used for the iron and steel industry, accounting for 67% of the total mineral extraction. Most of this mineral is used in the production of steel. Its abundance in the crust, low production costs and suitable properties such as hardness, flexibility and durability made this material the building block of industrialisation. Specifically, steel is a fundamental construction metal: it forms the structural component in buildings and is applied in the machinery used in the building industry. Aluminium is the other metal that has been extracted on a massive scale historically, especially since the latter half of the twentieth century. It accounts for 6% of the global mineral production in the analysed period and although more expensive (see Fig. 2.5), it can be used as an alternative to steel in many



2.1 Cumulative production of important non-fuel minerals in the period from 1900 to 2008.



2.2 Cumulative production of important non-fuel minerals in the period from 1900 to 2008, excluding phosphate rock, limestone, iron ore, gypsum and aluminium.

applications due to its lower density and corrosion resistance. Copper (0.6%), a key metal in the electric and electronic industry and the alloying metals manganese (0.6%), zinc (0.48%) and magnesium (0.45%) are the other four metals with significant extraction rates.

The building industry also plays a key role in the core consumption of non-metals. Limestone, from which lime is produced, is used amongst many other applications as a raw material in Portland cement and as an aggregate in concrete. It accounts for 7% of the total production and it will undoubtedly continue to be one of the most important raw materials. Gypsum follows limestone in the ranking (6%). This material is produced for a variety of applications but the construction of plasterboards as a finish for walls and ceilings is probably its main use.

In addition to the building industry, agriculture is the other sector that intensively consumes non-metal minerals. Phosphate rock and potash represent respectively 8% and 1.5% of the overall mineral consumption throughout the twentieth century. Both substances are key ingredients for plant and crop nutrients.

But, are there enough resources in the crust to sustain the production trends shown above? A common way to assess the scarcity of minerals is through the parameter resources to production ratio (R/P). This indicator reveals the years until depletion if production and the amount of reserves remain as in the reference year used. This is hence a static view of the state of mineral resources.

It should be stated that it is very unusual for production rates of minerals to remain stable over the years. Indeed the production trends of many minerals, as shown in Figs 2.1 and 2.2, tend towards exponential increases. Furthermore, the relationship between reserve demand and price is well known. Prices rise incentivises exploration and certain deposits that have been classified as uneconomic become profitable again. Additionally, technological improvements may also decrease extraction costs leading to reclassification of reserves. The aforementioned facts are usually claimed by supporters of the opportunity cost paradigm and hence detractors of the fixed-stock paradigm (that assumes that resources are finite and sooner or later shortages will arise). According to the opportunity cost paradigm, future trends in resource and material availability are uncertain. Consequently, the R/P factor can decrease but also increase with time (Söderholm and Tilton, 2012). However, Ericsson (2009) showed that although metal prices rose from 2000 to 2008, despite an associated increase in exploration spending, the rate of discoveries of new deposits declined. This is because most of the more accessible and high-grade mines had already been found and currently exploration takes place in deeper and more remote regions.

Another indicator of the state of mineral resources is the degree of depletion of the known reserves. This factor takes into account the cumulative production throughout history (numerator) and the current reserves plus the cumulative production (denominator). Table 2.2 shows the state of the Earth's mineral reserves and resources according to data published by Kelly and Matos (2011). It should be noted that a '0' in the 1900 production column means either that there was no production or that no records currently exist.

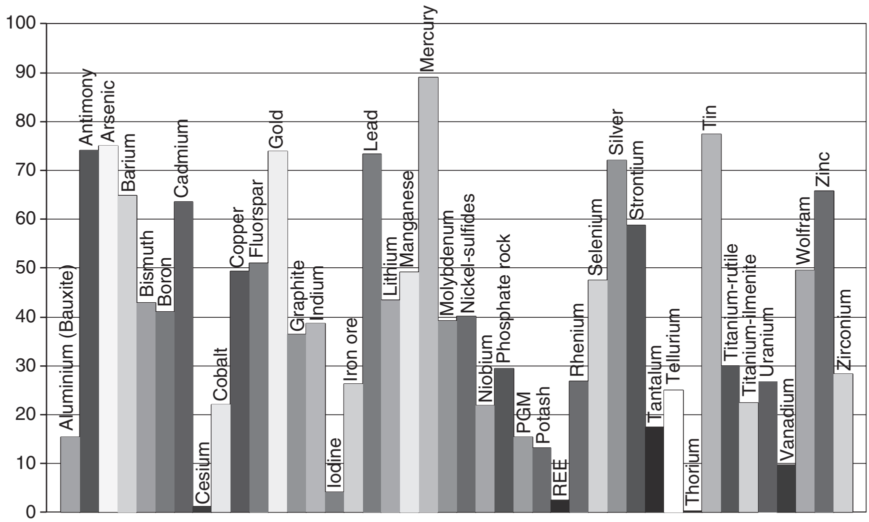
According to Table 2.2 and Fig. 2.3, the most depleted commodities are, in decreasing order: mercury, with 89% of the reserves extracted and an R/P

Table 2.2 State of the main mineral commodities produced worldwide. Elaborated from data in Kelly and Matos (2011)

Resource	Production (year 1900)	Production (year 2008)	Reserves	World resources	R/P	Reserves depletion degree
	(tons)	(tons)	(tons)	(tons)	(years)	(%)
Aluminium (Bauxite)	8.80E+04	2.05E+08	2.70E+10	7.50E+10	132	15.4
Antimony	7.71E+03	1.97E+05	2.10E+06	N.A.	11	74.1
Arsenic	6.17E+03	5.27E+04	1.22E+06	1.10E+07	23	75.0
Barite	0	8.05E+06	1.70E+08	2.00E+09	21	64.8
Beryllium	2.01E+03	1.98E+02	N.A.	>8E+04	N.A.	N.A.
Bismuth	0	7.70E+03	3.20E+05	N.A.	42	42.9
Boron (B_2O_3)	4.69E+04	4.35E+06	1.70E+08	N.A.	39	41.0
Bromine	0	243.000 (Year 2006)	Large	Unlimited (dead sea contains 1 billion tons of bromine)	Large	N.A.
Cadmium	1.40E+01	1.96E+04	5.90E+05	6.00E+06	30	63.5
Cesium	N.A.	N.A.	7.00E+04	N.A.	N.A.	1.2
Chromium	1.65E+04	6.98E+06	2.38E+08	8.16E+09	34	42.4
Cobalt	0	7.59E+04	6.60E+06	1.50E+07	87	22.1
Copper	4.95E+05	1.54E+07	5.40E+08	>3.00E+09	35	49.4
Feldspar	0	2.19E+07	Large	Large	N.A.	N.A.
Fluorspar	0	6.04E+06	2.30E+08	5.00E+08	38	51.0
Gallium	0	1.11E+02	N.A.	1.00E+06	N.A.	N.A.
Germanium	0	1.40E+02	N.A.	N.A.	N.A.	N.A.
Gold	3.86E+02	2.26E+03	4.70E+04	N.A.	21	73.9
Graphite	8.16E+04	1.12E+06	7.10E+07	>8,00E+08	63	36.5
Gypsum	0	1.59E+08	Large	Large	N.A.	N.A.
Hafnium (as HfO_2)	0	2.60E+05	5.60E+08	1.00E+06	2153	3.4
Helium	0	2.95E+04	N.A.	8.79E+06	N.A.	N.A.
Indium	0	5.73E+02	1.10E+04	N.A.	19	38.7
Iodine	0	2.65E+04	1.50E+07	3.40E+07	566	4.2
Iridium	N.A.	N.A.	N.A.	N.A.	N.A.	N.A.
Iron ore	9.55E+07 (Year 1904)	2.22E+09	1.60E+11	8.00E+11	72	26.3
Lead	7.49E+05	3.84E+06	7.90E+07	>1.5E+09	21	73.3

Table 2.2 Continued

Resource	Production (year 1900)	Production (year 2008)	Reserves	World resources	R/P	Reserves depletion degree
	(tons)	(tons)	(tons)	(tons)	(years)	(%)
Limestone	N.A.	2.96E+08	Large	Large	N.A.	N.A.
Lithium	0	3.82E+05	9.90E+06	2.55E+07	26	43.5
Magnesium	0	1.17E+07	N.A.	Large to unlimited	N.A.	N.A.
Manganese	5.92E+05	1.33E+07	5.40E+08	Large	41	49.2
Mercury	1.48E+03	1.48E+03	4.60E+04	6.00E+05	31	89.1
Molybdenum	1.84E+05	1.84E+05	8.60E+06	1.30E+07	47	39.3
Nickel	9.29E+03	1.57E+06	7.10E+07	1.30E+08	45	40.2
Niobium	0	6.29E+04	2.90E+06	N.A.	46	21.8
Osmium	N.A.	N.A.	N.A.	N.A.	N.A.	N.A.
Palladium	2.03E+02	2.04E+02	N.A.	N.A.	N.A.	N.A.
Phosphate rock	3.15E+06	1.61E+08	1.60E+10	N.A.	99	29.5
Platinum group metals	6.62E+00	4.65E+02	7.10E+04	>1,00E+05	153	15.3
Platinum	2.20E+02	2.21E+02	N.A.	N.A.	N.A.	N.A.
Potash	0	3.48E+07	8.50E+09	2.50E+11	244	13.2
Rare earths	1.04E+03	1.34E+05	9.90E+07	Undiscovered resources are thought to be very large relative to expected demand	739	2.4
Rhenium	0	5.65E+01	2.50E+03	1.10E+04	44	26.8
Ruthenium	N.A.	N.A.	N.A.	N.A.	N.A.	N.A.
Selenium	0	1.51E+03	8.80E+04	N.A.	58	47.4
Silicon	2.01E+03	6.16E+06	N.A.	N.A.	N.A.	N.A.
Silver	5.40E+03	2.13E+04	4.00E+05	Large	19	72.1
Strontium	0	4.96E+05	6.80E+06	>1,00E+09	14	58.8
Tantalum	0	1.17E+03	1.10E+05	N.A.	94	N.A.
Tellurium	0	132 (Year 2006)	2.20E+04	N.A.	N.A.	N.A.
Thallium	0	1.00E+01	3.80E+02	6.47E+05	38	51.2
Thorium	N.A.	N.A.	1.30E+07	2.50E+06	N.A.	0.1
Tin	0	2.99E+05	5.60E+06	N.A.	19	77.3
Titanium – Rutile	0	6.21E+05	4.50E+07	1.29E+08	72	29.9
Titanium – Ilmenite	0	6.79E+06	6.80E+08	2.00E+09	100	22.5
Uranium	0	4.39E+04	5.47E+06	1.00E+07	125	26.7
Vanadium	0	5.61E+04	1.30E+07	>6,30E+07	232	9.7
Wolfram	0	5.59E+04	2.80E+06	N.A.	50	49.6
Yttrium (as Y_2O_3)	8.90E+03	8.90E+03	5.40E+05	N.A.	61	N.A.
Zinc	4.79E+05	1.16E+07	2.00E+08	1.90E+09	17	65.8
Zirconium (as ZrO_2)	0	1.28E+06	8.34E+07	N.A.	65	28.4



2.3 Degree of depletion as a percentage of the main non-fuel mineral commodity reserves. Updated from Valero and Valero (2010a).

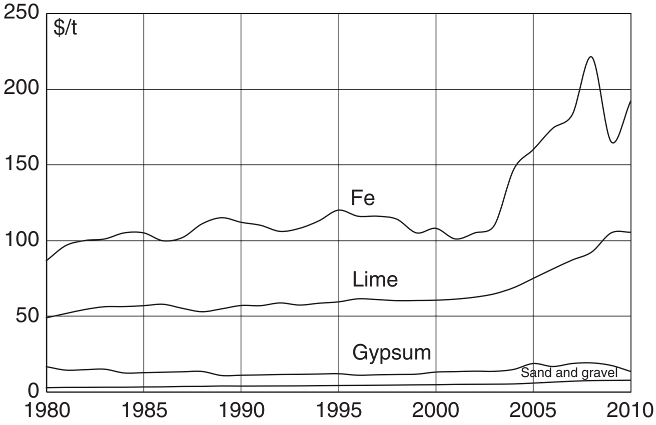
of 31 years, tin (77%, 19 years), arsenic (75%, 23 years), antimony (74%, 11 years), gold (74%, 21 years), lead (73%, 21 years) and silver (72%, 19 years). Conversely, the minerals of caesium, thorium, rare earth elements (REE), iodine, vanadium, potash, platinum group metals (PGM), tantalum, aluminium and cobalt are the least depleted commodities, having extracted less than 20% of their respective reserves. The greatest R/P ratios of the analysed commodities are those of the building materials limestone and gypsum and those of magnesium and bromine. The reserves of the latter substances are so large that no quantification efforts have been devoted to them. Rare earths, with 739 years, followed by iodine with 566, potash with 244, vanadium with 232, platinum group metals with 153 and aluminium with 132 years are the least depleted commodities.

The degree of depletion of minerals depends on two factors: the abundance of the considered mineral reserve and the production rates. Minerals such us iodine, aluminium, iron ore or potash have been extracted to a great extent in the past, but remain in abundance. It also happens that the least depleted minerals usually coincide with those substances for which significant use has been found until recently. However, this situation can change drastically with the development of emerging technologies. For instance, the development of renewable energies or telecommunication technologies is radically increasing the use of REE, PGM, cobalt and tantalum.

Focusing on building minerals, one can see that non-metals (or industrial minerals) such as silica, limestone or gypsum are very abundant in the crust

Table 2.3 Grave to cradle and cradle to entry gate costs of selected minerals (Valero and Valero, 2013)

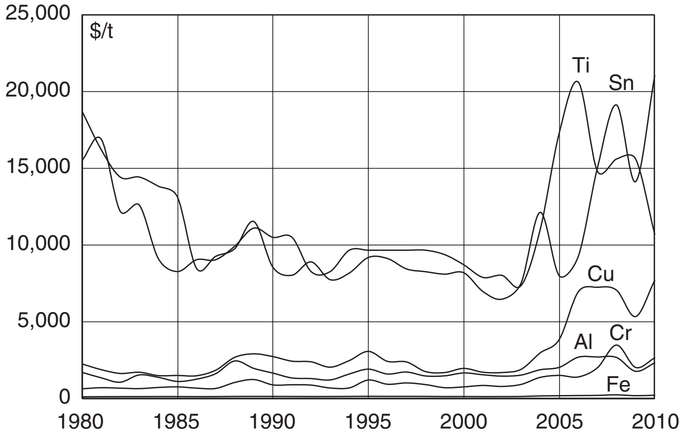
	Exergy replacement costs	Mining and benef.	Smelting and refining
Al-Bauxite	627	10.5	23.9
Chromium	5	0.1	36.3
Copper	110	28.8	21.4
Gypsum	15	0.2	–
Iron ore	18	0.7	13.4
Lime	3	0.4	5.8
Silicon	0.7	0.7	76.0
Tin	426	15.2	11.4
Ti-ilmenite	5	7.2	128.1
Ti-rutile	9	13.8	243.8
Zinc	25	1.5	40.4



2.4 Commodity prices trend of selected mineral commodities in the period between 1980 and 2010.

and no scarcity problems are foreseen either now or in the medium to long term. Moreover, extraction and processing costs are low compared to metals (Table 2.3). This is reflected in the prices of these minerals as shown in Fig. 2.4, where the prices trend of limestone, gypsum and sand and gravel are compared to those of iron (the metal with the lowest market price).

Conversely, metals are not so abundant and extraction and processing costs are significantly higher than those of non-metals (usually by one to two orders of magnitude as shown in Table 2.3). This is reflected in their higher prices compared to industrial minerals (see Figs 2.4 and 2.5). Some important metals used in the building industry are already showing signs of scarcity.



2.5 Commodity prices trend of selected mineral commodities in the period between 1980 and 2010.

This is the case for tin with around 80% of its reserves already being depleted combined with an R/P ratio of 19 years. It should be stated, however, that according to the USGS, world resources are sufficient to sustain recent annual production rates well into the future. Copper is also a much depleted metal. About half of its reserves have been exhausted and its R/P ratio is only 35 years. Although a price rise does not necessarily indicate scarcity because other factors come into play, it is interesting to see in Fig. 2.5 that with some exceptions, commodity prices have generally spiraled upwards in the last three decades. When demand for minerals surpasses supply, prices sooner or later climb until reaching equilibrium. If supply is ultimately limited due to geological scarcity, prices will increase out of necessity if demand grows. So for those most depleted minerals such as tin, copper or fossil fuels (as will be seen in the next section), a rise in commodity price is foreseeable in the near future.

2.4 Dynamic analysis of mineral resource use and depletion: the Hubbert peak model

In the previous section the state of the main mineral resources under static expectations has been shown. As stated previously, this way of assessing scarcity has shortcomings since production and reserve figures are in reality dynamic.

A dynamic analysis of the state of mineral resources entails the inclusion of the time factor. This can be done using the Hubbert peak analysis, named after the geoscientist M. King Hubbert who discovered in the 1950s that the production curves of various fossils fuels over time were all bell-shaped (Hubbert, 1956). The observed trends were based on the fact that no finite resource can sustain, for longer than a brief period, such an exponential rate of growth of production; therefore, although production rates tend to increase exponentially at the offset, physical limits prevent them continuing to do so. The area under the bell-shaped curve thus represents the known reserves or resources of the analysed material. The inflection point where the curve becomes concave downward is commonly referred to as the Hubbert peak. And the year where the peak is reached provides valuable information, since it indicates the point where supply can no longer meet demand. Hubbert's model successfully predicted the peak of oil extraction in the lower 48 states of the US and the subsequent decline of production.

The Association for the Study of Peak Oil and Gas (ASPO) is actively engaged in disseminating studies on this matter. Some prominent ones are those of Campbell and Laherrère (1998), Deffeyes and MacGregor (1980), Bentley (2002) and Aleklett *et al.* (2010). According to these studies, the corresponding peak production of oil could take place within the first decade of the twenty-first century or not much later. As Campbell and Laherrère (1998) argue, from an economic perspective, when the world runs completely out of fuels is not directly relevant: what matters is when production begins to taper off. Beyond that point, prices will rise unless demand declines commensurately.

Although the Hubbert peak theory has traditionally been applied to fossil fuels, there are some, such as Arndt and Roper (1976) or Bardi (2005), who have applied it to non-fuel mineral resources.

Traditionally, the Hubbert peak model plots tonnage of material vs. time. Another way to represent it is through a thermodynamic property called *exergy* (denoted B) over time. Technically, exergy is the maximum useful work that can be extracted from a system as it is brought into equilibrium with its surroundings, i.e. to the so-called 'dead state'. It can also be expressed as the minimum useful work required to bring the system into its initial state from those conditions in the dead state. Thus exergy measures the level of departure of the system from the surrounding environment. In fact, exergy is a measure of the quality of things. The farther a system is removed from a reference environment, the more exergy it contains. To take an example, a natural waterfall in the mountains has a lot of exergy since it is at an elevated height with waters of maximum purity compared to the dead state which would be (for the sake of argument) saltwater flowing on a level with 35 ppt of salinity. Another illustration is that of a mine which also has a high exergy content as it is a highly concentrated mineral system compared to the average mineralogical composition of the crust. Therefore the exergy of a mineral deposit measures the minimum amount of energy required to form the mine in composition and concentration from the dispersed state of the environment, i.e. from the upper continental crust.

Apart from this, exergy has another advantage in that all important features that make a mineral valuable (composition, concentration and tonnage) are unified in a magnitude measured in energy units, such as tons of oil equivalent (toe). Furthermore, exergy allows for systems to be aggregated and disaggregated deliberately and this is definitely not possible if tonnage is used as a form of assessment, whereby one would effectively be adding pears to oranges such as in the case of the collective assessment of gold and iron production. In short, if exergy is used, the orders of magnitude are comparable. Prominent studies that have used exergy for the assessment of natural resources are those of Szargut *et al.* (2002), Finnveden and Ostland (1997), Wall and Gong (2001) and Ayres *et al.* (2003).

However, as stressed by Valero and Valero (2010a), exergy only provides the minimum theoretical values that could eventually be reached if man-made technology were completely efficient. Yet man's technology is still very far removed from this point. So in order to overcome this deficiency, exergy replacement costs (denoted with B^*) rather than exergy (B) alone are used. The former are defined as the energy required to restore the mine in composition and concentration from the crust with currently available technologies.

As further stated in Valero and Valero (2010a), the bell-shaped curve is better suited to minerals if it is fitted to exergy or exergy replacement costs over time instead of mass production of the metal commodity over time. Oil quality keeps near constant with extraction, whereas other non-fuel minerals do not (a mineral's concentration decreases as a mine is increasingly exploited). As such, exergy is a much better unit of measure than mass, since it accounts not only for quantity, but also for ore grades and mineral composition. Furthermore, if the Hubbert model is applied to the exergy replacement costs, the technological factor of extracting and refining the mineral is also taken into account.

Hubbert's bell-shaped curve as a time function can be described through the generic Gaussian curve:

$$f(t) = \frac{R}{b_0\sqrt{2\pi}} \cdot \exp^{-\frac{1}{2}\left[\frac{t-t_0}{b_0}\right]^2} \qquad [2.1]$$

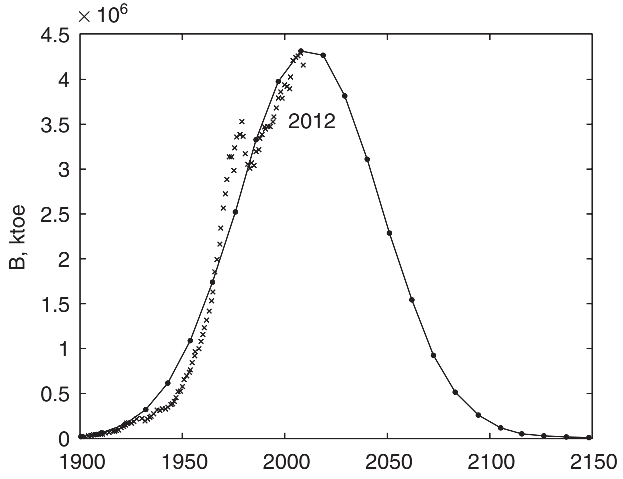
where parameters b_0 and t_0 are the unknowns and R is the reserves or resources of the analysed commodity.

The Hubbert peak model is now applied to the main metals used in the building industry, namely iron, aluminium, zinc, tin, copper, titanium and chromium. The peak of oil is also provided, as this commodity is the raw material necessary for the production of plastics. Industrial minerals are not listed since their reserves are so large that no quantification for R exists. As demonstrated in Eq. [2.1], either reserves or world resources can be used for estimating mineral peaks. However, considering world resources instead of reserves, combined with the variation of production over time, would provide a more realistic picture of the state of the commodities. With both figures, a range of probable peaks can be provided; the lower one corresponds to data based on available reserves, while the upper one is based on world resources. That said, world resource figures are rather speculative, since they are very difficult to assess. Table 2.4 shows the results obtained showing the theoretical peak considering available reserves and world resources as published in USGS (2010). The regression factors of each fit are shown (R^2) in addition to the observed empirical peak if it has already occurred. Figures 2.6 through 2.10 show a selection of the studied curves.

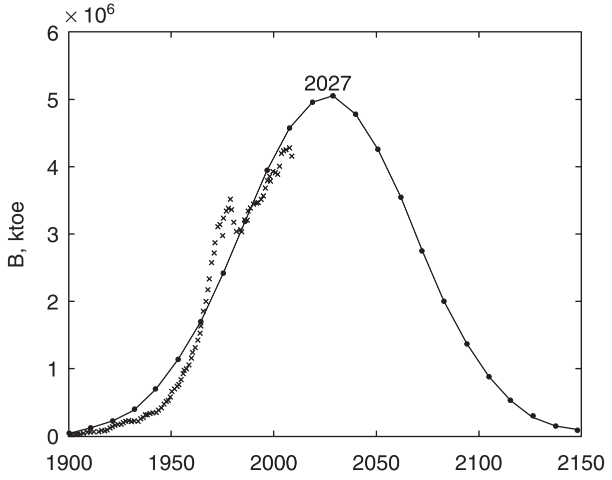
The regression factors of the Hubbert models carried out are over 0.89 except for tin. According to Table 2.4, the theoretical peak based on reserves could have been reached for tin (1979), zinc (1999), oil (2012) and copper (2012). And indeed the theoretical (2012) and observed peak (2011) for oil almost coincide. Yet no empirical peaks have been reached for tin and zinc. This might be because even if reserves are low, world resources are thought to be large enough for mining companies to believe with certainty that reserves could continue to increase. This may also be the reason for the low regression factor of tin. For the remaining studied minerals, chromium and titanium extracted from rutile are expected to reach their peaks in 2015 and 2028, respectively. The remaining commodities show peaking years after 2040. Considering world resources, the peak is displaced on average around 60 years, with a range between 15 years for oil and 134 years for chromium. The peak of iron ore is displaced 75 years, copper 58 and aluminium only 38.

Table 2.4 The peak of production of some important building materials worldwide

	Theoretical data				Empirical data
	Reserves peak	R^2	W.R. peak	R^2	Observed peak
Aluminium	2050	0.98	2088	0.98	–
Chromium	2015	0.96	2149	0.97	–
Copper	2012	0.95	2068	0.98	–
Iron	2040	0.91	2115	0.92	–
Oil	2012	0.97	2027	0.97	2011
Ti-ilmenite	2040	0.96	2082	0.96	–
Ti-rutile	2028	0.89	2069	0.86	–
Tin	1979	0.53	–	–	–
Zinc	1999	0.92	2062	0.98	–



(a) Proven reserves

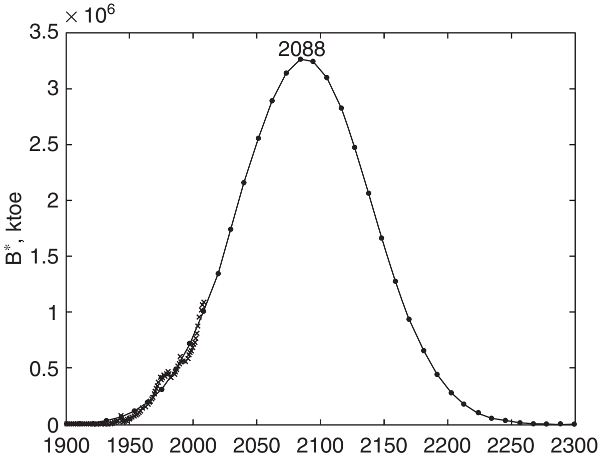


(b) Additional recoverable resources

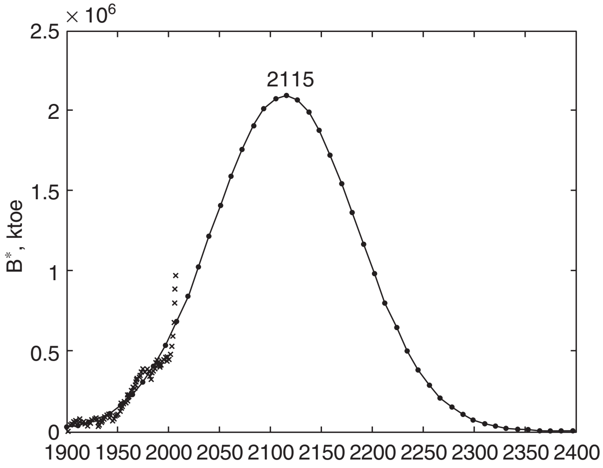
2.6 The Hubbert peak applied to world oil production. Data based on proven reserves (a) and additional recoverable resources as registered in Masters *et al.* (1994) (b).

Hence, even if reserves increase considerably until reaching the upper bound of world resources, the peak of production of the analysed commodities would be postponed on average by just two generations.

It should be stated that the Hubbert peak is a theoretical model that may predict future supply/demand dynamics but with certain limitations. If the integral under the Hubbert curve is equal to reserves or world resources, the curve obtained is a geologically-driven one. Conversely, if production



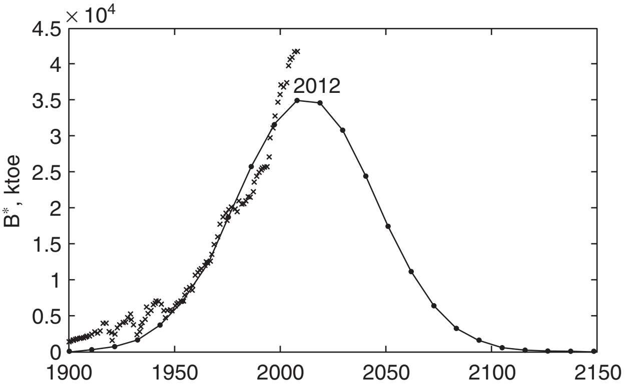
(a) Aluminium



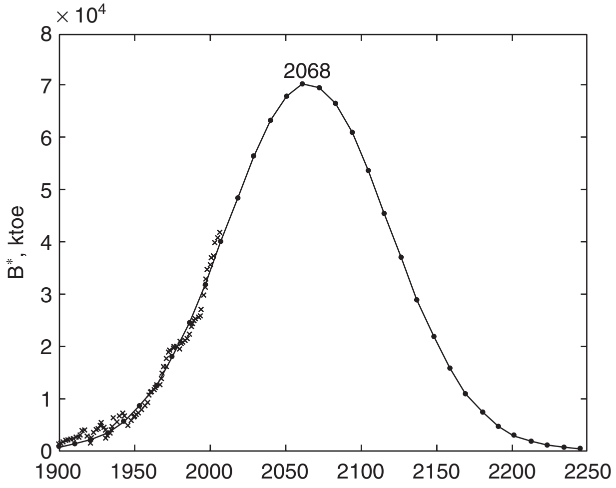
(b) Iron

2.7 The Hubbert peak applied to aluminium (a) and iron (b) world resources. Data obtained from USGS (2010).

curves are fitted with a bell-shaped curve without any restriction, the curve obtained would be economically-driven. Such curves are provided, for instance, in the study of Bardi and Pagani (2008). The Hubbert peak model as used in this chapter (with the constraint that the integral of the curve is equal to reserves or world resources) fits satisfactorily when the main criterion of future supply/demand dynamics is geological availability. In general, regression factors increase with the peaking year, as the empirical points fall into the foot of the curves. When the peak is in sight, different factors other than pure geological scarcity usually come into play. Höök



(a) Reserves



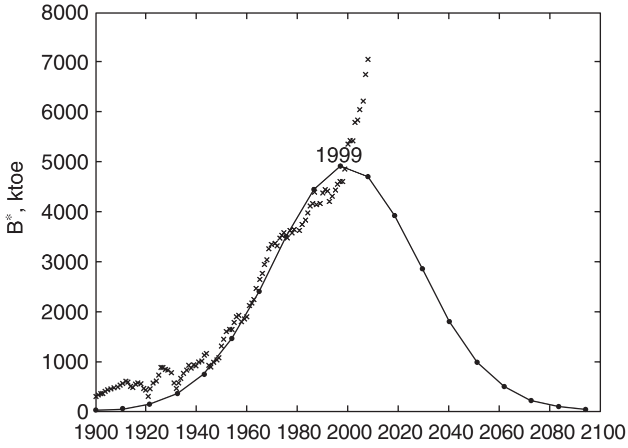
(b) World resources

2.8 The Hubbert peak applied to copper reserves (a) and world resources (b). Data obtained from USGS (2010).

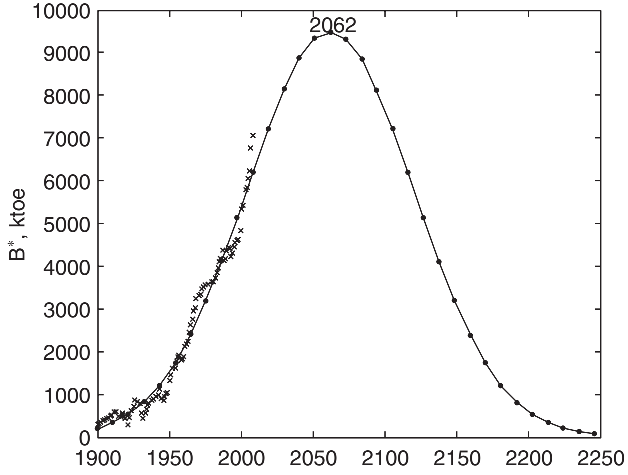
et al. (2010) claim that peak (oil) is a theory backed by phenomenological evidence, including geology, reservoir physics, fluid mechanics, statistical physics, economics and actual observations.

2.5 From grave to cradle: A new approach to assess and account for mineral depletion

Throughout this chapter, it has been seen that production of raw materials has increased exponentially over this and the last century. Minerals have constituted the building block of industrialisation and have been



(a) Reserves

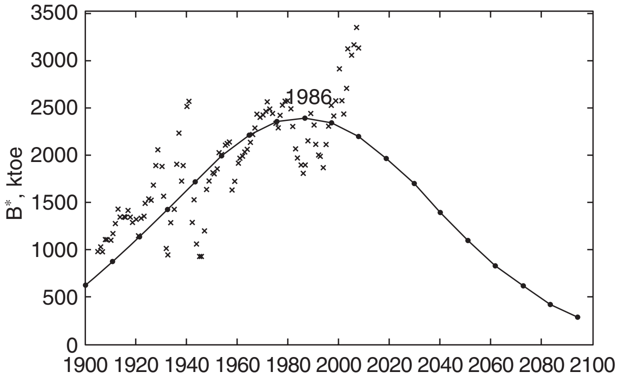


(b) World resources

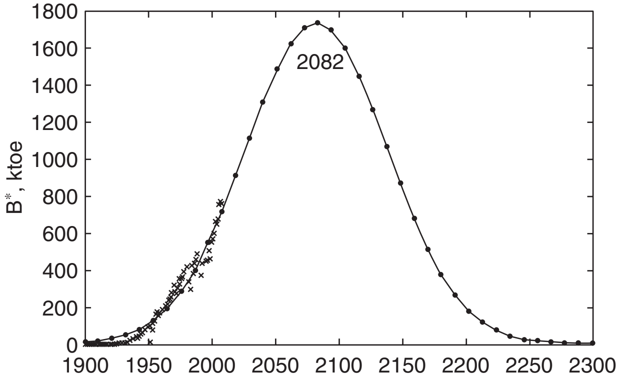
2.9 The Hubbert peak applied to zinc reserves (a) and world resources (b). Data obtained from USGS (2010).

fundamental in the development of many nations. However, despite their importance, minerals have been treated as a limitless free good provided by Nature leaving some critically scarce. An adequate management of mineral resources is urgently required with scarcity assessment tools needing to be developed and implemented in conventional accounting systems.

Consequently, the life cycle assessment (LCA) methodology could become a good starting point. LCA methodology assesses the environmental impacts associated with a product, process or service throughout its life



(a) Tin



(b) Titanium-ilmenite

2.10 The Hubbert peak applied to tin (a) and ilmenite (b) world resources. Data obtained from USGS (2010).

by inventorying material resources, energy inputs and environmental issues, through a cradle to grave approach. The US Environmental Protection Agency (EPA) (2003) divides the product life cycle into the following stages:

- cradle to entry gate (mining and processing)
- entry gate to exit gate (product manufacture) and
- exit gate to grave (product use, recycling and disposal)

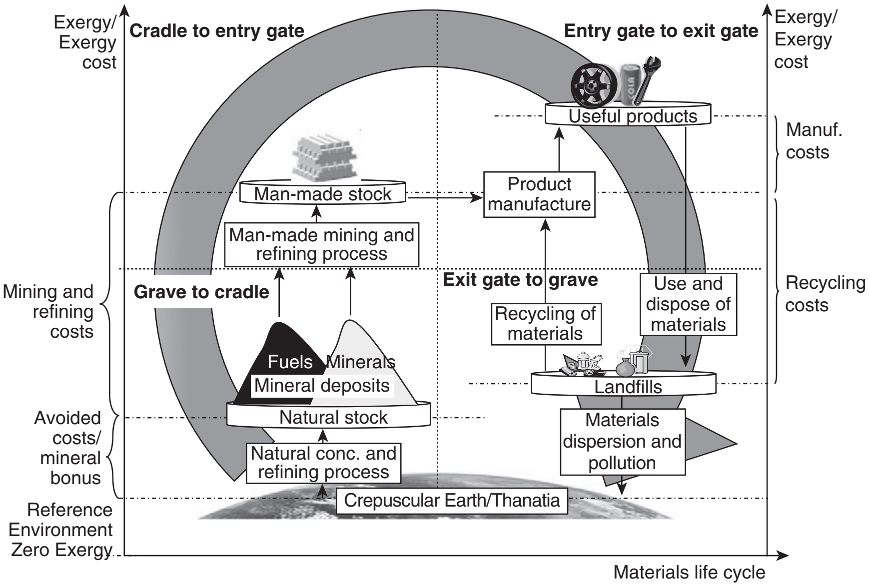
Valero and Valero (2010c) stated that the energy, water and materials required for mining and refining minerals is well reflected in the cradle to entry gate stage. While the relative importance of this stage compared to the other ones increases as the ore grade of the deposits decreases, because more effort is needed for extracting the same amount of mineral, this approach only partially takes into account the problem of mineral scarcity, leading to the following question: How can we assess the fact that extracting a highly concentrated resource today will force future generations to extract a less concentrated resource leading to an increased impact on environment and economy?

A way to tackle this problem is to include a new stage in the LCA methodology: the grave to cradle. The idea behind this is to assess the exergy replacement costs (exergy required with available technologies) to replace minerals dispersed throughout the crust after the useful life of products comes to an end, to the initial conditions of composition and concentration at which they were found in the mines.

Figure 2.11 shows in a schematic way the four stages in the 'grave to grave' approach. During millions of years, the Earth has formed through natural concentration and refining of processes the current mineral stock. Man extracts raw materials from those concentrated mineral deposits (in the cradle to entry gate stage), saving huge amounts of energy. The natural stock is then converted into man-made stock which is further manufactured producing different commercial products in the entry gate to exit gate stage. When the life of those products comes to an end, they are in the best case recycled or disposed of in landfills, i.e. the 'grave'.

Conventional accounting methodologies in the best case scenario only take into account the three costs appearing in the cradle to grave approach (1 – mining, mineral processing, smelting and refining; 2 – manufacturing and 3 – eventual recycling). These costs are real and belong to the so-called 'over the rainbow' component in the grave to grave LCA methodology (Valero and Valero, 2010c). However, an additional cost should arguably be included if the scarcity factor of minerals is taken into account: the exergy replacement costs, which represent the 'effort' that Nature spent in concentrating minerals in mines. It is in fact an avoided and imaginary cost and can be understood as the debt man acquires with future generations for exhausting high-grade mines. This approach belongs to the 'down the rainbow' component of the grave to grave LCA methodology.

In order to assess those imaginary avoided costs, the conditions of the 'grave' should be fixed. As Hubbert (1956) stated, 'soon all the oil is going to be burned and all the metals mined and scattered' and it is this state which is the starting point for the exergy replacement costs calculation. In Valero and Valero (2010c) and Valero *et al*. (2011), a model of a degraded planet where all resources have been burned and all minerals dispersed is developed. This planet is Thanatia (named after 'death' in Greek). Thanatia is composed by an atmosphere with a carbon dioxide content of 683 ppm and a mean surface temperature of 17°C. Its hydrosphere is assumed to have the current chemical composition of seawater. And finally Thanatia's continental crust is modelled with a composition and concentration of the 294 most abundant minerals found on today's Earth. In fact, it is a first



2.11 Graphical representation of the 'grave to grave' concept (Valero and Valero, 2013).

model of the average mineralogical composition of the Earth's crust. So it is Thanatia which is the starting point, or reference, for the assessment of abiotic resource depletion. The exergy costs measured from Thanatia give a measure of the quality of the resource and constitutes a useful tool for classifying resources according to their depletion states. The detailed methodology is described in Valero and Valero (2010b, 2013).

Table 2.3 shows the exergy replacement costs compared to mining and smelting costs for a selection of building materials. As can be seen in Table 2.3, the exergy replacement costs, i.e. the imaginary cost reflecting the effort to reproduce the mineral capital from the degraded state of Thanatia have the same order of magnitude as real processing costs.

A factor that influences the magnitude of exergy replacement costs is the current state of technology. The calculation of grave to cradle costs assumes that the same technology is used in the real (over the rainbow component) as well as in the imaginary part (down the rainbow component). This fact is highlighted with aluminium. Even if the average grades of bauxite ore in the crust and in the mines (1.38E-03 and 7.03E-01, respectively) are similar to those of chromite and main chromium ore (1.98E-04 and 6.37E-01, respectively), the elevated irreversibility of the production process of aluminium leads to greater exergy replacement costs.

Grave to cradle costs also increase with the difference between the ore grade of the original mine and that of Thanatia. Therefore, a mineral that is very scarce in the crust such as copper (where the average grade of its main ore grade chalcopyrite in the crust (in Thanatia) is 6.64E-05 and in the deposits 1.67E-02) has high exergy replacement costs. Conversely, silicon in the form of quartz is very abundant in the crust and the difference between the average concentration in Thanatia and in the deposits is low (2.29E-01 vs. 6.50E-01). Hence, as the mineral deposits become depleted and their ore grades tend to those in Thanatia, exergy replacement costs tend to decrease. This is why exergy replacement costs can be seen as a measure of 'thermodynamic rarity' and why the methodology encourages preserving those mines with high grades for future generations.

2.6 Conclusions

The Earth has been treated as a free department store in materials, as there was, and still is, a huge information gap about how much is extracted and how much is left. It has been seen in this chapter that for most important minerals, geological surveys provide information about production, tonnage and, in the optimal scenario, ore grades. Adjectives like measured, hypothetical, proved, indicated, inferred, etc., usually accompany information about resources and reserves. However, the borders that limit what is actually recoverable and what is not are very uncertain, as they strongly depend on market prices, technological developments, the discovery of new deposits or even the arbitrary decisions of mining companies. Hence, the real available quantity of a certain mineral is very difficult to ascertain, although it most likely lies between its reserve (minimum) and resources value (maximum).

Production trends of minerals in the period between 1900 and 2008 generally exhibit exponential growth behaviour. It is therefore important to know whether the amount of recoverable resources is enough to sustain this increasing demand. Industrial minerals such as sand and gravel, lime, gypsum or clay are so abundant that no scarcity problems are in sight at least in the medium to long term. However, quantities of other building materials, in particular metals, could become critical if consumption continues to grow exponentially. Iron ore and aluminium, meanwhile, two essential metals in the building industry, have been extracted on a massive scale in the past but remain abundant. Their R/P ratios and degree of depletion of their reserves suggests no reason to worry about future scarcity. Conversely, the reserves of tin, chromium, zinc or copper show R/P ratios of less than 50 years. This means that if production continues to climb and no significant amounts of reserves are found, commodity prices will presumably spiral upwards.

But as seen in the chapter, the R/P factor is just a static view of resources with the dynamics of the production trends of minerals better shown through Hubbert peak models. The application of the latter to those reserves (i.e., the minimum available) of the main building metals and oil suggests that their production peaks might be reached before the middle of the twenty-first century. If an analysis undertaken considers world resources instead of reserves (i.e., the upper available limit), the peak is displaced on average around 60 years. This proves that the limiting factor in resource availability is the exponential growth of demand. Thus, supply will hardly satisfy this demand, even if more mineral deposits are found, reserves are reclassified into resources with technological development and price rise, substitutes for critical materials are found, etc. Moreover, the environmental and social externalities associated with the extraction of increasingly scarce minerals could become even more critical than the eventual commodity shortages. Therefore, as pointed out by Allwood *et al*. (2011), material efficiency should be urgently increased through longer-lasting products, product upgrades, modularity and remanufacturing, component re-use or the designing of products with less material. In this endeavour, a good management of resources will play a key role.

At the end of the chapter, a new methodology for abiotic resource depletion has been presented. The so-called ‘grave to cradle’ approach should complement conventional LCA analysis by including an additional stage: the effort that one would need to invest to restore minerals from the world’s most depleted state (whereby commodities have been thrown away and dispersed throughout the crust) into the initial conditions of composition and concentration at which the raw materials (commodities in their unprocessed form) were found in mines. This methodology is a decision-making tool. It provides an estimation of the value of the mineral capital as if one were to be charged with the job of reproducing or replacing it. Only by doing this, can one realise how difficult it is. This gives a sense to the ethics of conservation, because arguably one should not destroy that which one does not know how, or is not prepared, to construct.

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3

Life cycle assessment (LCA) of sustainable building materials: an overview

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Abstract: The construction industry is one of the largest exploiters of both renewable and non-renewable natural resources. It was inevitable that it would find itself at the centre of concerns regarding environmental impact. The process and operation of building construction consumes a great deal of materials throughout its service life cycle. The selection and use of sustainable building materials play an important role in the design and construction of green building. This chapter sets out to present an overview of sustainable building materials and their impacts on the environment. It also discusses the life cycle assessment as a methodological principle and framework, and its limitations for the analysis of sustainable building materials.

Key words: green building materials, LCA, sustainable construction, sustainable development.

3.1 Introduction

Environmental economics and sustainable development have become central concerns to people from all disciplines and in all countries (Cole, 1999). Many environmental discussions centre on the concept of ecologically sustainable development (ESD) since the major oil crisis of the 1970s (Balderstone, 2004). ESD, from a building development viewpoint, is the efficient use of resources to meet the requirements and needs of present and future generations whilst minimizing the adverse effects on the natural environment.

Buildings have a direct impact on the environment, ranging from the use of raw materials during construction, maintenance and renovation to the emission of harmful substances throughout the building's life cycle (Balaras *et al.*, 2005). The construction industry and the environment are inextricably linked. It was inevitable that the industry has found itself at the centre of concerns regarding environmental impact. Environmental problems associated with resource consumption have been extended from the local scale of indoor air pollution to a global scale of contribution to climate change and ozone depletion.

Construction is not an environmentally friendly process and it has major impacts on the depletion of natural resources and on the emissions of greenhouse gas (GHG) as a result of fossil fuel combustion. In the US, the construction sector is the third largest industry sector in terms of contributions to GHG emissions (Li *et al*., 2010). Globally, it depletes 40% of natural materials, consumes 40% of the total primary energy, 15% of the world's fresh water resources, generates 25% of all wastes and emits 40–50% of GHG (Ramesh *et al*., 2010; Mokhlesian and Holmén, 2012).

To minimize the industry's environmental impacts, the use of sustainable building materials has become the main focus of research and development in achieving the goal of sustainable construction. This is one way in which the construction industry can make a responsible contribution towards protecting the environment (Du Plessis, 2007). Achieving the goal of sustainable construction is not about restricting the total amount of construction, but to pay more attention to how the design and selection of sustainable building materials can complement the environment to improve living quality, user health and comfort. The selection of sustainable building materials should not only focus on the performance specifications but also select materials that have the lowest GHG emissions (Ip and Miller, 2012).

Ecological building design is characterized by the use of natural materials rather than man-made materials that require energy in the process, and has an emphasis on healthy, non-toxic specification to minimize pollution (Halliday, 2008). Ideally materials should also be implemented into passive design and environment control such as thermal mass. Berge (2009) states that an acceptable goal of achieving a sustainable future is the drastic reduction and shift in the use of raw materials. This is particularly important when considering scarce and non-renewable resources. Equally important is the reduction of wastage and losses during the manufacture of materials, the construction process and also throughout the service life of the completed building. The recycling of materials during demolition of a building must also become regulated. The recycling process should be carefully planned and managed to ensure that these materials can be taken care of, maintaining them to their original quality rather than disposing of them.

This chapter aims at reviewing the impacts of building materials on the environment and the importance of sustainable building materials in enhancing the goal of sustainable construction for the design and construction of green building. This chapter presents the methodological principle and framework of the life cycle assessment (LCA) for the analysis of sustainable building materials. The chapter reviews LCA tools, studies and concludes with limitations of the LCA method.

3.2 The environmental impact of building materials

3.2.1 Sustainable building materials

The building and construction industry consumes great quantities of raw materials and energy. According to Bribian *et al*. (2011), approximately 24%

of global raw materials were consumed by the industry. Traditional building materials, including steel, concrete, aluminium and glass, are high energy content materials. Buildings impact on the environment during their whole life cycle and the choice of materials used will impact on their overall performance. They are used in various stages, from initial construction through to the operation stage when the buildings are maintained and refurbished to preserve their normal functioning until the end of their service life. Therefore, it is an area where designers can have significant inputs if they are properly informed in the area of incorporating sustainable building materials into the design of the building.

According to Franzoni (2011), the selection of building materials plays an important role in achieving the goal of sustainable development in the construction industry. Choosing materials with high contents of embodied energy entails an initial high level of energy consumption in the production process, which is associated with high levels of GHG emissions (Bribian *et al.*, 2011). With the implementation of energy efficient design, the initial embodied energy of building materials becomes a more important consideration (Thormark, 2006). Thormark (2006) further states that more attention should be given not only to the operating energy of a building but also to the material choice. However, according to Saghafi and Teshnizi (2011), the selection of sustainable building materials is the most difficult and challenging task. It is beyond the time and skill of the project team to incorporate environmental aspects of building materials and construction technologies. The process will involve analysing and developing environmental profiles of building materials and their interaction with the environment. This is an area that is largely unknown due to the large number of variables and variations.

There are a number of research studies addressing the problems with materials selection. However, most of these studies have failed to properly and adequately establish the definition of 'sustainable building materials'. Even today there is no universally accepted definition of 'sustainable building materials' (Franzoni, 2011; Saghafi and Teshnizi, 2011). This makes it very difficult to establish principles and guidelines that enable sustainable building materials to align with the principles and goals of sustainable development in construction.

Sustainable building materials are often regarded as materials that are natural and offer specific benefits to the users in terms of low maintenance, energy efficiency, the improvement of occupant health and comfort, the increase of productivity whilst being less harmful to the environment. However, according to Franzoni (2011), natural materials are not necessarily green materials such as asbestos, radon and turpentine. They are natural materials but they are harmful to the built and natural environment in various forms and ways. Therefore, sustainable building materials are materials that are environmentally friendly or environmentally responsible materials (Spiegel and Meadows, 1999; Franzoni, 2011). Accordingly, sustainable building materials are mainly materials that are from renewable sources rather than from non-renewable sources. They must also be sustainable during their whole life cycle and require the use of less energy in the manufacturing process. During the life cycle, these materials must not release pollutants or other emissions that impact on human health and comfort.

3.2.2 The life cycle impact of building materials

Building materials impact on both the built and natural environments throughout their life cycle. According to Franzoni (2011), material selection can take place both at an early stage to establish strategies and undergo market research for sustainable building materials as well as at the design development stage where these materials are incorporated into the design. Esin (2007) states that sustainable building materials should be considered at an early stage and must be evaluated from a life cycle perspective as the decision on material choices will impact the overall performance of the building. The life cycle of building materials is often referred to as a 'cradle-to-gate' analysis. The material life cycle relates closely to the pre-use phase of a building and it includes the raw materials extraction, manufacturing process, delivery to the construction site, installation on site, as well as further materials required during the operation stage for maintenance, refurbishment and renovation (Franzoni, 2011).

The raw materials extraction stage relates to the mining and harvesting of raw materials from the natural environment. The extraction process often generates dust, noise, disruption and is a nuisance locally. The process has direct impact on the environment in various ways. It disturbs the immediate natural habitat, flora and fauna, landscape character and causes pollution to the ground and surface water (Halliday, 2008). On the mining site, the work environment has serious impacts on the health and well-being of workers and the people within the vicinity. The process is also usually high in primary energy consumption due to the high level use of machinery and this energy is usually considered a part of the initial embodied energy of the extracted materials.

Extracted raw materials will require a manufacturing process to transform them into usable materials for construction purposes. The process has attracted serious concerns from people from a social and environmental viewpoint. It requires significant inputs of energy and other resources in the manufacturing process and usually outputs with solid wastes and other pollutants that may be harmful to the built and natural environment. These outputs can both be toxic and have little or no further usage. This area attracts much attention on how these processes can be improved to be more environmentally and socially friendly. Packaging of building materials for distribution to the building site is also detrimental to the environment as there are few packaging material options that are biodegradable or can be safely burnt (Halliday, 2008). It represents a significant waste of resources and has serious impacts on landfill.

The installation of materials on site is a process that involves cutting and fitting materials for a building. This process generates waste and construction waste has become a serious environmental problem. The figures generated for construction-related waste amount to approximately 30% in the US, 35% in Canada and 50% in the UK (Ekanayake and Ofori, 2004; Esin and Cosgun, 2007). The waste is often generated due to inefficient management on site or in the workshop and the production of off cuts. Most of this waste can be recycled and reduced if properly planned and managed. The site installation stage also requires energy which will also be considered in the embodied energy calculation.

Most of the construction-related waste is unnecessary according to Sterner (2002), who says that this waste has a high potential for recovery and reuse. However, due to the economic nature of the building industry, every stage of the construction period is kept to a minimal. The depletion of natural resources by the building industry is a topic of serious discussion as much of the recyclable material from building sites ends up in landfill sites.

Transportation is often involved in various stages of the life cycle of a material. Transportation is required between the site of extraction, processing/production and construction. The corresponding accumulation in transport miles associated with the consequent embodied energy and emissions are an area of concern. Smaller and regionally based plants and distribution yards will generally improve the overall environmental burden of a building (Berge, 2009). According to Esin (2007), each stage of the manufacturing process and the final assembly on site requires transportation, and the energy consumption can take up to approximately 2.2% of the life cycle primary energy consumption. Locally sourced and manufactured materials have the potential to make a significant difference to the overall environmental impacts associated with each stage.

During the operation stage of a building, materials are also involved in maintenance, refurbishment and renovation up until the end of the building's service life. The materials used during the operation stage are closely related to the materials selection. Some materials may be high in capital cost and initial embodied energy but have low maintenance requirements. However, other materials may be the other way round. The decision will be subjected to the user and client preferences. The correct choice of materials may not only impact on the maintenance requirements but may also have a harmful effect on occupants and the environment throughout the building life span, such as the release of volatile organic compounds (VOCs) and other substances due to chemicals used during the manufacturing process.

End of service life can be discussed from two aspects. It may refer to the end of service life of a particular material where further maintenance may not restore its original function or it may be uneconomical to maintain. The other aspect of end of service life may refer to the entire building even though some of the materials may still be operating and functioning. The eventual replacement and destruction of materials will end up in landfill. The treatments, coatings or preservation that was used may transform natural materials into toxic wastes, dismissed into the air and water. Therefore, materials that are recyclable and biodegradable should be selected and used in order to reduce landfill requirements and the release of pollutants.

It is important that building materials are selected not just to serve their intended functions when they are newly installed but also for an acceptable length of time. Some of the building materials may last for the entire life of a building but others may only be functional for a few years before renewal is required. The service life of a material may relate to particular conditions that reflect on the complexity of their chemical and mechanical properties. Natural materials are generally lower in embodied energy intensity and toxicity levels than man-made materials. Products will become sustainable if low embodied energy natural materials are involved (John *et al.*, 2005).

3.2.3 Sustainable building material selection criteria

Sustainable building and development involves considering the whole life cycle of buildings that are designed to minimize all adverse impacts on the built and natural environment through sustainable building design and material selection. The selected building materials impact on the environment throughout the life cycle of a building. This life cycle can be divided into five stages: feasibility, design, construction, operation and demolition. Materials are used during the construction stage of a building through to the operation stage which involves building maintenance. The replacing, renewing and renovating of building materials and components remain until the end of service life of the building. Fundamentally, materials play an important role in enhancing the overall performance of a building and in achieving the goal for sustainable construction in the industry.

Buildings are responsible for a substantial amount of material and energy consumption. However, the environmental properties of materials have not traditionally been a design or construction priority (Guggemos and Horvath, 2005), where the cost, performance characteristics and aesthetics are the main items that determine material selections in buildings (Bayne and Taylor, 2006). In a study for developing an optimization model for sustainable materials selection, Florez and Castro-Lacouture (2013) state that the choice of materials can take place during the feasibility stage when different technological solutions and environmental matters can be incorporated early in the decision-making process.

In most projects, material selection takes place at the design development stage when specification and working plans are produced. Indeed, selecting appropriate materials for a building is an integral part of good design and the key to good design lies in realizing the importance of occupant health and comfort, and harmonizing it with the inherent properties of materials. Consequently, evaluating the properties of sustainable building materials and their impacts on the environment becomes central to the design and construction of green buildings. As stated by Sturges (1999) designers have gradually turned more attention to material selection and regarded it as an important part of the design. However, it may be at a stage that is either too busy or too late to consider sustainability options of building materials, whilst more opportunities may exist if sustainability of building materials can be considered early in a building life cycle.

Methodologies and tools have been developed to optimize material selection in situations where the best combination of physical and mechanical properties is required and this helps designers to make objective decisions about material selection (Halliday, 2008). Sturges (1999) suggests using the same principles to optimize material selection by taking account of environmental considerations and focusing on achieving the set sustainability goals to minimize impacts on the environment. This methodological framework usually takes a life cycle approach in selecting sustainable building materials. These tools are limited in use due to the complex nature of sustainability in construction and the environmental characteristics of most materials, which are closely related to the sourcing and handling of resources, the way in which they are used, and the care that goes into their detailing and maintenance. The environmental profile of sustainable materials is largely unavailable or incomplete, and this is particularly serious in developing countries.

In addition to considering environmental matters in material selection in construction, Abeysundara *et al.* (2009) suggest that the LCA approach for sustainable material selection should also take consideration of economic and social factors. They developed an evaluation matrix to assist decision makers in selecting sustainable materials while balancing environmental, economic and social factors in the assessment. Anastaselos *et al.* (2009) support this view in assessing thermal insulation solutions to include environmental, economic and social aspects.

Florez and Castro-Lacouture (2013) further suggest that factors to be considered in sustainable material selection should not be limited to economic, environmental and social aspects, but be inclusive of other subjective factors such as metaphysical and cultural aspects. They have developed an optimization model that integrates both objective and subjective factors in the evaluation process for sustainable material selection. With the widening of the scope of sustainable material selection, the evaluation process has now moved towards a full integration of all aspects emerging during the lifetime of a building and its elements (Anastaselos *et al.*, 2009).

Sustainable material selection is clearly a multi-criteria subject. However, since there is no universally accepted definition of a sustainable building material, there is no clear philosophy that precisely articulates the criteria of materials for sustainability. Esin (2007) and Franzoni (2011) state that sustainable building materials are materials related to resource and energy efficiency in the manufacturing process and that these materials should pollute less and have no negative impact on human health. It becomes apparent that sustainable building materials are related to the following criteria:

- resource efficiency;
- energy efficiency (including initial and recurrent embodied energy, and GHG emissions); and
- pollution prevention (including indoor air quality).

Resource efficiency is all about utilizing the finite resources in the most effective and efficient manner whilst not restricting economic growth on society. Research indicates that certain resources are becoming extremely rare. Therefore the use of remaining stocks should be handled cautiously, especially where they are known to support threatened habitats, or where these resources are known to be used should take precedence (Halliday, 2008). Most of these rare materials can be substituted by other less rare or renewable materials if they are carefully planned and designed at an early stage (Berge, 2009).

Sustainable building materials should also be designed and manufactured to be long lasting as well as requiring low maintenance throughout their lifetime. This is achieved either by designing for durability or by rehabilitating existing building materials for an extended life. There is good information available on detailing to enhance durability, but materials and components need to have worth for reuse, and this may place an emphasis on the specifications for good quality materials in the first instance.

Esin (2007) suggests that resources used for the production of building materials should have been sourced locally in order to save energy and the associated emissions from the transportation of these materials to the project site. Sustainable building materials should be reusable or recyclable in that they can be easily dismantled at the end of their useful life. Gao *et al*. (2001) state that the production of building materials which contains recycled content is important in terms of conserving natural resources and saving embodied energy. However, where recycling contributes to additional and avoidable pollution should be avoided.

Energy efficiency is another important consideration for sustainable building materials. The production and use of energy has become a growing source of environmental concern and research has demonstrated that the production of energy is closely related to the degradation of the environment (Hammond, 2000; Tiwari, 2001). The wide use of fossil fuels, to some degree, has polluted the atmosphere. However, improving the energy efficiency of the building alone may not result in the maximum potential reduction in energy consumption, because there is a substantial portion of energy trapped in the upstream and downstream production of building materials.

The energy use in building materials is often related to the embodied energy embedded in the recovery of raw materials and the manufacturing of building materials together with on-site construction energy and transportation. The embodied energy intensity of building materials varies from region to region and from production plant to production plant, depending on energy sources, technology use and the manufacturing process. The embodied energy of building materials includes initial and recurrent embodied energy. Initial embodied energy relates to the building materials used for construction, whilst recurring embodied energy is required during the operation stage. This refers to the embodied energy used to produce materials for replacement, repair and maintenance over the building's effective life. It is measured during the building's economic life after occupancy.

Both initial and recurrent embodied energy play an important role in the energy efficiency of building materials on a life cycle approach. According to Huberman and Pearlmutter (2008), embodied energy represents between 10 and 60% of the total energy used during the lifetime of the building. Treloar *et al*. (2001) state that embodied energy is significant because it occurs immediately and the total energy consumed in the production of building materials can equate to, over the life cycle of a building, the temporary requirements for operation energy. Fully identifying the nature and content of the embodied energy intensity will allow designers and building material manufacturers to improve production processes in order to minimize energy consumption.

Nowadays people spend more than 90% of their time indoors and therefore indoor conditions have important implications for user health, well-being and performance (Frontczak and Wargocki, 2011). It is well-documented from research that building materials play an important role in determining the indoor air quality. The emission of VOCs such as formaldehyde from building materials are regarded as serious problems that affect human health and comfort and productivity (Li and Niu, 2005; Lee and Kim, 2012). Therefore sustainable building materials are materials that do not have negative implications for the built and natural environment. Materials that do contain embodied pollution can impact throughout their material lifetime on employees during the manufacturing process, to building occupants through off-gassing or leaching in use and eventual pollution through recycling or disposal. Materials that emit low or no carcinogens, reproductive toxicants, or irritants should be used.

Sustainable building materials are about choosing materials manufactured from resource-efficient processes such as choosing materials of a low embodied energy content, using locally provided and renewable energy sources, as well as selecting materials that contribute fewer amounts of GHG emissions to the atmosphere. Commonly used building materials, such as steel, concrete and aluminium, consume energy and release CO_2 during the production process. Therefore research into new material production, manufacturing methods, recycling of building materials and using low embodied energy materials has become extremely important. In addition, techniques and technologies are improved so that natural resources are minimized and conserved in buildings.

Resource minimization and conservation are considered important aspects in the construction industry. In considering the importance of sustainable building materials in green building design and construction, Berge (2009) suggests switching to the exploitation of smaller deposits of raw materials. This is because small-scale exploitation often contributes less damaging activities to the environment. Halliday (2008) suggests using the ‘5 Rs’ approach to deal with issues related to resource conservation; they refer to refuse, reduce, reuse, recycle and repair.

Refuse refers to setting guidelines on what are and are not acceptable materials in buildings. In 2003, the European Commission released the integrated product policy (IPP) to identify products within the construction sector that have the greatest lifetime environmental impact potential. The IPP has led to the development of environmental product declarations (EPD) to communicate the environmental performance of materials from a life cycle perspective. The EPD is being recognized by the USGBC LEED program and is being encouraged by the European Union.

Reuse of building materials deals with a serious resource issue. Reduce relates to the reduction in the use of resources, space or elements. This need not undermine a good design solution, such as by reducing the amount of mechanical services. It involves the adaptation of existing buildings instead of demolition and reuse of salvaged materials to minimize raw material consumption. The reuse of materials involves the consideration of the material and joining techniques so as to enable the reuse and replacement of components, either in parts or as a whole. Thormark (2000) studied the environmental impacts of reused building materials and the component of a single-family dwelling and concludes that the environmental impacts were about 55% of the impacts that would have been caused if all materials had been sourced as new.

When the reuse of a component is not possible, it may still be possible to recycle it in whole or in parts. An example includes concrete which is crushed into aggregates for road base. Saghafi and Teshnizi (2011) state that recycling is an effective strategy for reducing environmental impacts in the construction industry as it will help to close the material loop. They developed a method to assess energy savings as a factor for assessing the potential benefits for recycling building materials. This was to assist designers and contractors who could compare and select sustainable materials. Finally, repair is a strategy that aims to reduce a wasteful lifestyle by overhauling and refurbishing to extend the useful life.

3.3 Life cycle assessment (LCA) and sustainable building materials

3.3.1 LCA

As previously discussed, sustainable building materials play an important role in the environmental performance of a building and impact a building at various stages during the building's life cycle. Therefore the analysis and choosing of sustainable building materials has also been based on a life cycle assessment (LCA).

LCA has a long history and has been used in addressing problems ranging from excessive consumption of global resources, both in terms of construction and building operation, to the pollution of the surrounding environment (Ahn *et al.*, 2010). Sustainability is an important consideration in construction and the concept of sustainability in this area is about creating and maintaining a healthy built environment. At the same time, sustainability is also about focusing on minimizing resources and energy consumption, thereby reducing the damage done to the environment.

There are a great number of tools for environmental assessment of the built environment, ranging from construction material selection, energy labelling and indoor air quality to a whole building assessment, and then to an urban-scale built environment assessment (Forsberg and von Malmberg, 2004). Reijnders and van Roekel (1999) and Forsberg and von Malmberg (2004) have generally classified current built environment assessment tools into two groups: qualitatively based building rating systems, and quantitative tools using a physical life cycle approach with quantitative input and output data on flows of materials and energy. In both groups there exists a diverse variety of concepts all over the world.

The most appropriate and accepted method used to produce a holistic assessment of the environmental impacts associated with a building and building materials is the LCA (Cole, 1998; Junnila *et al.*, 2006; Horne *et al.*, 2009). Cole (1998) states that the LCA approach is the only legitimate basis on which to compare alternative materials, components and services in buildings. Page (2006) suggests that the LCA method is a more appropriate means to measure the environmental burden of buildings over other tools as it has the ability to implement a trade-off analysis to achieve a reduction in overall environmental impacts rather than merely a simple shift of impacts.

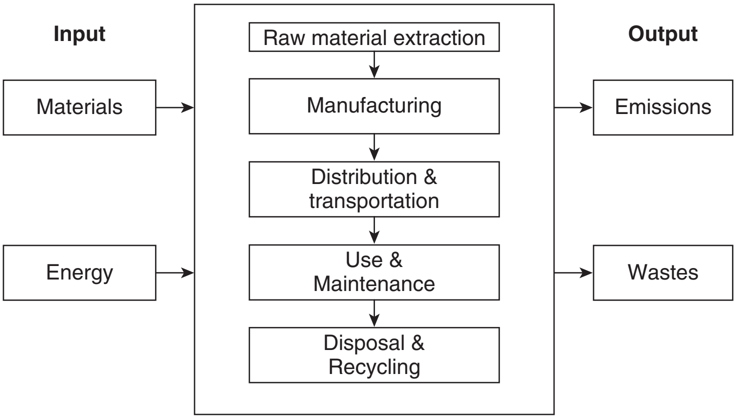
LCA has been widely used in Europe and the United States, initially for product comparison, but its current application has been extended to include government policy, strategic planning and product design (Kohler and Moffatt, 2003; Scheuer *et al.*, 2003). LCA permits an evaluation of how impacts are generated and distributed across various processes throughout the life cycle where data on environmental qualities were previously lacking (Puettmann and Wilson, 2005). Ortiz *et al.* (2009) state that LCA is an important tool for assessing impacts and has been used in the building sector since 1990. The methodology was eventually adopted for the development of building assessment tools around the world.

3.3.2 Principles and framework

LCA is a methodology used to analyse complex processes which focus on dealing with the input and output flows of materials, energy and pollutants to and from the environment from a life cycle perspective (Perez-Garcia *et al.*, 2005; Puettmann and Wilson, 2005; Wei *et al.*, 2008; Saghafi and Teshnizi, 2011). Figure 3.1 demonstrates the process of an LCA study.

LCA is best defined as a systematic approach to evaluate the environmental burdens associated with a product, process or activity by identifying and quantifying energy and materials used and wastes released to the environment over the whole life cycle from 'cradle to grave', i.e. from extraction of raw materials to the ultimate disposal of waste from a product, process or activity (Klopffer, 2006; Xing *et al.*, 2008). Therefore the principle objectives of LCA are:

- to quantify and evaluate the environmental performance of a product or a process and to help decision makers choose among alternatives; and
- to provide a basis for assessing potential improvements in the environmental performance of the system so as to modify or design a system in



3.1 Input and output flows of materials, energy and pollutants from a life cycle perspective.

order to decrease its overall environmental impacts. This can be done in an overall sense or targeted to improve specific stages during the life cycle.

The idea for the LCA was conceived in Europe and in the USA in the late 1960s and early 1970s. It was not until the late 1980s and early 1990s that LCA received wider attention in response to increased environmental awareness and concern for energy usage (Azapagic, 1999). There was a need for a more sophisticated approach to analyse complex environmental issues. LCA originated from net energy analysis studies to predict future supplies of raw materials and energy resources over a life cycle approach. During the early studies, energy consumption and efficiency were the main focus and energy-related waste emissions were not considered (Azapagic, 1999). Since the early 1970s, wastes and emissions generated by the production processes were taken into account (Fay and Treloar, 1998; Treloar *et al.*, 2001).

The LCA methodology was originally developed by the Society of Environmental Toxicology and Chemistry (SETAC, 1993) to improve the science, practice and application of reducing resource consumption and environmental burdens associated with a product, packaging, process or activity (Weir and Muneer, 1998; Johnstone, 2001). In addition, it enables the identification and quantification of energy and materials used and waste released to the environment over the whole life cycle (Klopffer, 2006).

In the early 1990s, concerns over inappropriate claims of LCA results by product manufacturers resulted in action taken by SETAC to develop a standard methodology for conducting LCA studies. In 1997 the

International Standards Organization (ISO) published the first ISO 14040 series to standardize the guidelines and principles on the LCA methodology (Perez-Garcia *et al.*, 2005). In 2006 the ISO 14040 series was revised. The ISO 14040 governs the principles and frameworks, whilst ISO 14044 provides the requirements and guidelines for conducting LCA studies (ISO, 2006a, 2006b). The ISO approach contained in these Standards is fundamental to the standardization and therefore the generalizability of findings pertaining to all LCA studies and life cycle inventory (LCI) studies.

The Standard states that the overarching aims of LCA include:

- recognizing opportunities to improve environmental performance of products at various stages of their life cycle;
- assisting in decision making in the industry for strategic planning, priority setting, product design or redesign;
- selecting relevant indicators of environmental performance and developing measurement techniques; and
- marketing, e.g. an environmental claim, eco-labelling scheme or environmental product declaration.

In procedural terms and in accordance with the Standard, LCA starts with a definition of the functional unit and then a quantitative inventory of all inputs and outputs is performed. It is followed by a classification and impact assessment and, finally, an evaluation of the environmental impact of the system being studied (Bribian *et al.*, 2009). The process of conducting an LCA is well documented and received in the industry as a tool to provide a picture of the interaction of an activity within the environment and to facilitate environmental improvements (Azapagic, 1999).

According to ISO 14040, LCA has four stages (Weir and Muneer, 1998; Perez-Garcia *et al.*, 2005; Ortiz *et al.*, 2009). The first stage involves developing a goal definition and scope. This stage includes defining study objectives, products and their alternatives, system boundary choice, environmental parameters and a data collection strategy. The second stage is an inventory analysis and this stage includes data collection and treatment, quantifying materials, energy inputs and waste emissions, and preparing inventory tables where a system's material and energy balance is calculated. The third stage is the life cycle impact assessment (LCIA), which involves classifying the inventory table into impact categories, aggregation within the category, normalization, weighting different categories where the system's potential environmental impacts are evaluated, and modelling category indicators. The LCIA gives a quantitative assessment of the environmental impacts based on either endpoint (problem-oriented) or midpoint (damage-oriented) approaches (Baumann and Tillman, 2004). The final stage is an improvement assessment and this stage may involve undertaking a sensitivity analysis to test the results, setting a prioritization and feasibility assessment to reduce the environmental burden, and formulating recommendations for improvement.

Drawing from the commentary in the Standard, LCA typically does not address the economic or social aspects of a product and the nature of choices and assumptions may be subjective; models used for the inventory analysis or to assess environmental impacts are limited by their assumptions. Furthermore, accuracy may be limited by the accessibility or availability of relevant data, or by data quality. Generally, the information developed in an LCA study should be used as part of a more comprehensive decision process or used to understand the broad or general trade-offs (ISO, 2006a).

3.3.3 LCA studies

LCA methodologies have made more complicated and sophisticated the assessment processes for eco-efficient design, and enhanced decision making and reduction of environmental impacts. Within the industry, impacts occur from the extraction and manufacturing of building materials, to the manufacturing processes, operation and eventual disposal at the end of life cycle. LCA is widely acknowledged as a framework to systemically analyse and develop strategies to improve the understanding of the environmental impacts of this sector.

LCA has been used in many studies in the building sector as an environmental tool for comparative assessments of whole buildings. Cole (1999) has used the LCA approach to assess the initial embodied energy consumption and GHG emissions of alternative wood, steel and concrete structural systems for buildings, with concrete being the highest contributor on both issues. Gustavsson and Sathre (2006) conducted similar research to develop an LCA-based methodology to study factors affecting the energy and CO_2 balances of concrete and wood frame buildings. Results suggest that the use of wood and wood by-products is an effective means for reducing fossil fuel use and net CO_2 emissions into the atmosphere. Thormark (2006) studied how material choice may affect embodied energy levels and the recycling potential in an energy efficient housing project. The conclusion was that having a prolonged lifetime and/or making the choice of materials with less embodied energy levels could considerably reduce the impact.

LCA has also been used extensively to analyse and compare building materials and systems. Based on the LCA framework, Saghafi and Teshnizi (2011) were able to establish the potential recycling energy as a factor for assessing the recycling value of materials by taking into account material selection, construction and deconstruction technologies, and the frequency of recycling. Wei *et al.* (2008) and Kellenberger and Althaus (2009) used the LCA to examine the flow of material, energy and pollutants for different building components. Wu *et al.* (2005) developed an LCA-based method to assess the environmental impacts of building materials. It was based on environmental profiles, in which environmental impacts are categorized and the green tax was used to study the inter-correlations across different categories.

Haapio and Viitaniemi (2008) used the LCA to analyse how different structural solutions and building materials affected the results of the environmental assessment of a building over the building's life cycle. The research highlighted the importance of the choice of the building's service life. Ip and Miller (2012) conducted an LCA study on the construction of hemp-lime walls and established their life cycle impact on climate change in the UK. The results showed that this hemp-lime wall not only compensated for the GHG emitted during the growing and manufacturing processes, but also enabled the storage of CO_{2e}.

Ardente *et al.* (2008) conducted an LCA study of a kenaf-fibre insulation board and results indicated that this insulation system has a significant reduction in environmental impacts compared to traditional insulating materials using synthetic materials. Anastaselos *et al.* (2009) developed an assessment tool to assist designers with the option to select and evaluate building materials to implement thermal insulation solutions and measured their environmental performance, energy efficiency and cost. The outcomes of these studies have a profound impact in utilizing a more environmentally friendly insulation material that can help to reduce emissions to the environment and operation energy consumption.

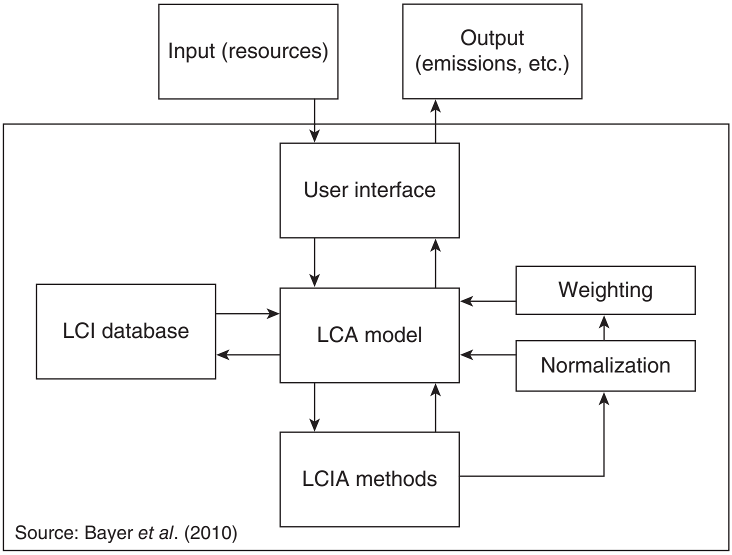
3.3.4 LCA tools

LCA is a complex and expensive methodology. The LCA framework provides an avenue to obtain quantitative data and indicators to facilitate comparisons to improve material performance for building design and construction. Owing to the subjectivity and complexity, and the varying needs of users, there are LCA-based tools developed to simplify the process of conducting an environmental assessment of building materials. Bayer *et al.* (2010) state that these tools can be defined as environmental modelling software that develops and presents a life cycle inventory. The impact assessment results from a laborious analytical process. These tools usually follow closely to the ISO standards and other accepted LCA guidelines. These tools take inputs in the form of material take-offs, convert them into mass and attach this mass value to the LCI data available from an LCI database.

Anastaselos *et al.* (2009), Ortiz *et al.* (2009) and Bayer *et al.* (2010) define LCA tools into three levels. Level 1 refers to the tools that are in the form of evaluating environmental performance at a material level. They aim at identifying environmental characteristics of building materials. They can be used as comparison tools and are largely used for materials selection. These tools are found as software packages in various forms of development and use. Commonly-known software tools include GaBi from Germany, SimaPro from the Netherlands, TEAM from France, EDIP from Denmark and LCAiT from Sweden, and they provide a framework for evaluating and comparing building products.

Level 2 is related to whole building design decision-making tools, on a life cycle approach. They are software packages used to assess environmental impacts from a whole building approach that take input in terms of building geometry and building assemblies. The result is aggregated for the entire building and presented in the form of environmental impacts. This is due to the different life cycle stages or the contribution of the building towards a particular impact. They are generally capable of comparing several design options for a building program and are generally helpful during the initial design (Bayer *et al.*, 2010). Some examples of tools include LISA from Australia, Ecoquantum from the Netherlands, Envest from the UK and ATHENA from Canada. Figure 3.2 presents the configuration of LCA tools at Levels 1 and 2.

Level 3 is often regarded as the environmental building assessment framework that assesses the performance of a building on a set of predetermined criteria. The environmental building assessment framework has been widely used to promote green building design and construction and popular frameworks like Leadership in Energy and Environmental Design



Source: Bayer *et al*. (2010)

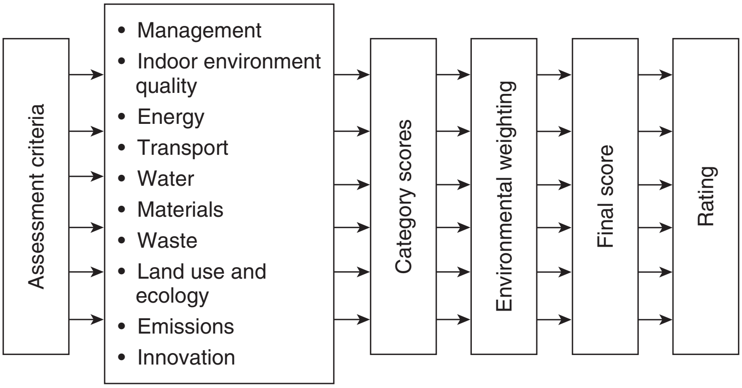
3.2 Levels 1 and 2 LCA software tools.

(LEED) in the US, the Building Research Establishment's Environmental Assessment Method (BREEAM) in the UK, GreenStar in Australia and the internationally developed Green Building Challenge (GBC) (Ding, 2012). This environmental building assessment framework is a rating system aimed at certifying buildings to provide an indication of their environmental performance. The system is based on credits and points. A building is evaluated and points are awarded if the requisites are met. These points are given towards the certification.

Sustainable building material selection plays a role in the environmental building assessment framework but this framework may not be adequate with regard to enhancing the use of sustainable materials (Franzoni, 2011). Encouraging the use of sustainable building materials seems to be one of the biggest challenges facing designers in the environmental building assessment framework. Saghafi and Teshnizi (2011) state that material selection criteria are the most notable failures of building assessment systems in maintaining a clear objective. Additionally, points allocated for sustainable materials selection do not encourage designers and assessors to pay more attention to sustainable building materials as only approximately 9%, 13% and 16% of the total points were allocated respectively in BREEAM, LEED and GreenStar. Figure 3.3 below presents the framework for environmental building assessment tools.

3.3.5 Limitations of LCA

LCA is a systematic and consistent approach for analysing the environmental impacts of building materials. It is becoming a more popular tool used to improve performance from a life cycle perspective. LCA is a complex



3.3 Framework for Level 3 environmental building assessment tools.

process as it quantifies and compares inflows of materials and energy and outflows of emissions associated with building materials at different spatial scales and in different contexts (Finnveden *et al.*, 2009). There are limitations for the use of the LCA in the environmental assessment of building materials (IEA, 2004; Keeler and Burke, 2009). The evaluation process is time-consuming and complex. Despite the great variety of LCA tools, Cole (2010) states that due to data limitations and the large variety of building techniques, these LCA tools are inadequate in modelling and analysing the environmental impacts of all phases in a building.

One of the limitations of LCA is that the calculations are excessively complicated and expensive, and this limitation has restricted its extensive use in the building sector (Bribian *et al.*, 2009; Malmqvist *et al.*, 2011). Proposals were developed to simplify the methodology in a way to improve its current uptake in the construction industry (Liu *et al.*, 2003; Bribian *et al.*, 2009; Kellenberger and Althaus, 2009; Malmqvist *et al.*, 2011).

LCA involves a life cycle approach in the assessment and therefore the choice of the service life of building materials is critical. However, it is difficult to determine the choice of the life expectancy of a building material. The life expectancy is affected by many variables such as user patterns, the maintenance cycle, climatic conditions and detailing and workmanship during design and construction. Haapio and Viitaniemi (2008) state that the impact of the service life of building materials affects the environmental impacts of a building. However, this is an area that has been neglected; how the service life influences the results of most LCA studies has not often been included.

Although LCA is an ideal tool for analysing the environmental impact of building materials, the method relies heavily on the availability and completeness of LCI data (Finnveden *et al.*, 2009; Bayer *et al.*, 2010). The data of the manufacturing process is also a major issue of data quality as it varies greatly from one region to another and from one production plant to another (Esin, 2007; Saghafi and Teshnizi, 2011). Additionally, the current LCI databases represent conditions in industrialized countries which may not be equally applicable to developing countries if they want to conduct LCA studies. There is a large gap in the LCI data in these countries. Therefore the use of these tools and the database in developing countries may not lead to correct decisions without having substantial modifications made to them.

The LCA method is currently focusing on material and energy flows and the associated impacts, but not all types of impacts are equally well covered in a typical LCA (Finnveden *et al.*, 2009). The methodology does not sufficiently assess many sustainability issues such as social and economic concerns, and site-specific impacts such as biodiversity, land use or freshwater sources. In addition, an economic analysis has not been adequately addressed in any of the LCA studies (Abeysundara *et al*., 2009; Anastaselos *et al*., 2009). Ortiz *et al*. (2009) further state that LCA studies have been undertaken for analysing environmental impacts of building materials but the results of these studies were not fully comparable due to different study scopes and system boundaries (Saghafi and Teshnizi, 2011).

Saghafi and Teshnizi (2011) state that the LCA methodology does not appropriately tackle the closed-loop behaviour of materials, as it neither addresses whether a product or building can be disassembled and recycled, nor the recyclability of these materials. The framework does not assess correctly when materials are recycled, because recycling in a system occurs where the waste from one production function may become the raw material in another. It is because the occurrence of building material recycling only takes place in the distant future.

LCA is a systematic and consistent approach for analysing the environmental impacts of a building and is becoming a popular methodology to improve performance. The LCA is a useful tool in assessing building-related environmental impacts and it will be more effective if it is complemented with qualitative assessment tools such as existing environmental building assessment tools to deal with its limitations.

3.4 Conclusions

Ecologically sustainable development is a major concern that embodies both environmental protection and management. The concept of ecologically sustainable development is fundamental in developing the goal of sustainable construction. However, the concept itself is somewhat vague and broad. Generally speaking, sustainable construction concerns attitudes and judgement to help ensure long-term environmental, social and economic growth in the built environment. In buildings, it involves the efficient allocation of resources, minimum energy consumption, low embodied energy intensity in building materials, reuse and recycling, and other mechanisms to achieve effective and efficient short- and long-term use of natural resources. The improvement in the environmental performance of buildings will indeed encourage greater environmental responsibility and place greater value on the welfare of future generations. Within the context of sustainable construction, the selection and use of sustainable building materials play an important role.

Construction is one of the largest end users of environmental resources and one of the largest polluters of man-made and natural environments. By far the selection of sustainable building materials is the most difficult and challenging task facing the industry. Developing the environmental profile of materials and their interaction with the environment require detailed analysis of the environmental impacts on a 'cradle-to-grave' basis of these materials. Life cycle assessment is a comprehensive and systematic method for evaluating the environmental impacts of building materials. This method quantifies and compares inflows of materials and energy and outflows of emissions of materials on a life cycle perspective for possibilities of improvement and providing guidelines for materials selection. The ultimate goal of LCA is to provide users with comprehensive results and enable the evaluation of specific improvement alternatives, based on economic, social and environmental criteria. LCA case studies on commonly used building materials have been carried out in developed countries but there are insufficient studies in developing countries using local data. Therefore, more LCA studies of building materials are to be developed and used in order that environmental and energy considerations can be implemented worldwide.

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4

Life cycle assessment (LCA) of the building sector: strengths and weaknesses

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Abstract: This chapter discusses possibilities and limitations of life cycle assessment (LCA) in the building sector. Through standardisation, LCA has gained global support as a most important tool for furthering more environmentally friendly choices in the sector. However, there are several limitations of LCA at a macro level such as choice of unit of analysis and at a micro level related to methodological and practical matters. Overall and inherent possibilities and limitations in the steps of LCA analyses are scrutinised. The flexibility of the method gives LCA the power to inform decision makers and the public about the environmental performance of buildings and building products.

Key words: LCA, building material, building product.

4.1 Introduction

What is an environmentally friendly material for building and construction? The question is increasingly being posed by customers, researchers, building owners, politicians, entrepreneurs and others. The answer is that it all depends: on the specific site where the material is to be employed; on the energy sources used in the production of the material; on the lifetime of the material; on the other materials and systems with which the material will interact; and a range of other issues including the assessment method used to answer the question. Life cycle assessment (LCA) is one such method which aims to scrutinise the environmental friendliness of products and services. LCA is now employed in ever more applications from intra-company decisions to large-scale political decisions. However, LCA cannot provide a single answer that can be used in all circumstances but must be adapted to different situations. A decision situation where an engineer is to choose between two well-known materials for an outer wall is very different from a situation where the environmental properties of a material with unknown use in the future need to be assessed. This chapter will describe the possible uses of LCA in the environmental assessment of materials. We will, however, also include a discussion on the limitations that should be taken into consideration when applying LCA in the building sector.

The first section describes the general possibilities and limitations of LCA. Thereafter, possibilities and limitations inherent in the steps of LCA work processes are examined. Finally, a concluding section sums up the information and gives general advice as to how to conduct a critical reading of an LCA study. The chapter is directed towards readers with a superficial understanding of LCA who want to know more about where the important assumptions and methodological choices are located. However, this should also remind readers more familiar with LCA, or even performing LCAs, about benefits and pitfalls. Readers with no knowledge of LCA may gain from reading some basic texts first, for instance the ISO standards 14040-44 (ISO, 2006) or *The Hitch Hiker's Guide to LCA* (Baumann and Tillman, 2004).

4.2 The overall strengths and limitations of life cycle assessment (LCA)

LCA is an environmental assessment method intended to capture the environmental performance of products, services or product systems. It provides a flexible framework for a large number of different analyses, ranging from details in production processes to systemic political decisions. The framework has been developed to be able to capture the inflows and outflows from activities in all life cycle stages associated with a product or product system and all environmental impacts related to these inflows and outflows. One of the greatest advantages of LCA is the ability to avoid so-called 'problem shifts', both with respect to environmental impacts and to life cycle stages. The possibility to compare different analyses and to use LCA in business is ensured by its inclusion in organisations with global outreach. Three issues are scrutinised in the following, namely the possibilities and limitations of:

- running different analyses;
- avoiding problem shifts; and
- widespread use and standardisation.

4.2.1 Different analytical approaches

The applicability of LCA is discussed in relation to different themes and various uses. Regarding themes, area waste management (Ekvall *et al.*, 2007; Blengini *et al.*, 2012), building materials and buildings (Guggemos and Horvath, 2005; Dimoudi and Tompa, 2008; Huberman and Pearlmutter, 2008; Upton *et al.*, 2008; Ortiz *et al.*, 2009) and electricity production (Rossi *et al.*, 2012) have been examined. These discussions partly overlap with other themes, such as policymaking (Tillman, 2000; Bribián *et al.*, 2009;

Pacheco-Torgal *et al.*, 2013), product improvement (Haapio and Viitaniemi, 2008b) and product labelling (Rajagopalan *et al.*, 2012). Different uses require different approaches to modelling systems. In the *General Guide for Life Cycle Assessment – Detailed Guidance* (EU JRC, 2010), hereafter referred to as the ILCD guide, different decision contexts are introduced to explain why it may be sensible to employ different modelling approaches and data sources in an LCA. The guide discriminates between three different contexts, termed: (A) micro-level decision support; (B) meso/macro-level decision support; and (C) accounting. These three contexts are used in the guide to structure the presentation of LCA methodology, showing that they will have consequences for choosing what system(s) to include, and how to include them, in an analysis.

Two main modelling principles exist for LCA: *attributional* and *consequential*. According to the ILCD guide '*the attributional life cycle model depicts its actual or forecasted specific or average supply chain plus its use and end-of-life value chain. The existing or forecasted system is embedded into a static technosphere*' (EU JRC, 2010, p.71). The consequential life cycle model '*depicts the generic supply chain as it is theoretically expected in consequence of the analysed decision. The system interacts with the markets and those changes are depicted that an additional demand for the analysed system is expected to have in a dynamic technosphere that is reacting to this additional demand*' (EU JRC, 2010, p. 71). These definitions may seem overly theoretical but the main difference lies in the former striving to capture the actual data in a product system, while the latter investigates a hypothetical system where data should reflect changes in a product system. Weidema *et al.* (2009) state that '*the term consequential describes a modelling approach that seeks to describe the consequences of a decision*' (p. 6). This should not be confused with the decision context as laid out by the ILCD guide. The keyword in the citation is not 'decision', but 'consequences'. Both the attributional and the consequential approach can support decisions even though the former does not focus on changes incurred by the decision *per se*. A visible difference between the two approaches is the nature of the data used as input to analyses. In the attributional approach, average data should be employed to describe the relevant process, while in the consequential approach, marginal data is more appropriate.

Both these modelling approaches are relevant and necessary and it is futile to argue which one is better in general. The question is rather which one is the best in a given situation. The most important aspect in answering this question is whether or not the analysis addresses the dynamics of change and refers back to the decision contexts described by the ILCD guide.

If one wishes to consider the performance of a material, the study is usually based on an attributional approach, which is the predominant approach in the building sector. But if the goal is to examine the wider system consequences of introducing a new material on a large scale, a consequential approach must be adopted. The framework should allow for reproducibility but also wider scenario building. It should be applicable for strict comparisons of two products fulfilling the *same function* as well as generating an understanding in a *dynamic perspective*. The latter refers to environmental performance in a world where basic systems are changed, for instance a change in transport and energy systems from fossil to renewable sources. The European Commission has initiated strategies to harmonise and explain life cycle assessment methodology, which have led to documents on best practice in LCA (EU JRC, 2010).

The ambiguous, sometimes contradictory, results based upon LCA assessments reflect partly the complexity of the building sector as such, but also, in spite of the efforts to standardise LCA assessment approaches, inherent ambiguities in the LCA methodology as it is presently used. The LCA methodology opens up windows of opportunities to make various choices which will eventually influence the results of the actual studies being performed.

4.2.2 Avoiding problem shifts

Those performing LCAs in the building sector are all potential actors along the value chain, for instance material producers, entrepreneurs, owners, politicians, etc. LCA is used to support different decisions by these various actors in the building sector. It is not obvious to all what the consequences of decisions made at one place in the value chain will be. When reducing one environmental load, the measures may lead to increased emissions elsewhere in the value chain.

This is precisely where the main advantages of LCA lie, in identifying such issues that are not visible at first sight. These issues can be classified into two main categories for product systems:

1. capturing all life cycle stages where impact occurs in order to identify the important stages and associated processes, and
2. capturing several environmental impacts simultaneously to identify the important environmental issues.

A general conclusion and assertion that has been verified is that energy use for operation contributes to 70–90% of the total during the lifespan (Rønning *et al*., 2001, 2007; Vold *et al*., 2006; Fernandez, 2008; Barrett and Wiedmann, 2007; Sartori and Hestnes, 2007; Dimoudi and Tompa, 2008; Bribián *et al*., 2009; Ortiz *et al*., 2009). As stated, LCA provides the opportunity to include all life cycle stages and at the same time capture the consequences that the introduction of one or more measures at one stage has on others. Due to increasingly stringent political energy requirements and other changes, energy use for the operation is likely to decrease over time as we are heading towards low-energy buildings as a consequence of different measures. Sartori and Hestnes (2007) have performed a literature review where one main conclusion was that low energy buildings are more energy efficient than conventional buildings, although energy use for production of (upstream) materials increases.

On the basis of these findings, energy consumed during production, transportation and construction of a building can be relatively more important in an LCA. On the other hand, we often see that it is not the case that all life cycle stages are included regarding (low energy) building. In addition, the perception and understanding of the term 'user stage' in LCA can be ambiguous as it often only refers to energy for heating and cooling for operation of the building. Thus, as we are heading towards low-energy buildings, the environmental impacts and energy consumption associated with the maintenance, replacement and development phases are of greater importance than the operation of the building. High replacement rates of materials with high embodied energy will have a greater impact on life cycle performance. Thus, including all life cycle stages in LCA will have a vital impact on the ability to capture the effects of introducing measures.

The second category for avoiding problem shift, the capturing of several environmental impacts simultaneously, intends to minimise the risk that introducing a measure to reduce one impact increases another. This seems promising but *several* does not mean *all* environmental impacts, and it is still potential impacts and not effects on people and nature which are calculated. Impact on land use, biodiversity, indoor climate and exposure to toxic chemicals are some examples of those impacts known to occur during the life cycle of a building or building material, but are often troublesome to include in the LCA.

In relation to the low-energy building debate, we observe that environmental assessments of buildings and materials largely take place in the context of capturing embodied energy and greenhouse gases (Statsbygg, 2011). This is an example of choosing not to exploit the strength of LCA.

Capturing a large amount of activities, components and environmental impacts also has pitfalls. It makes analyses complex and difficult to penetrate and merely checking which processes are included can be a cumbersome task, especially since the documentation of the system being studied in many instances is lacking. Of course, it also causes problems for LCA practitioners because of the amounts of data required and the understanding needed of a range of different environmental issues. This is further discussed in the sub-sections below.

4.2.3 Widespread use and standardisation

The LCA method has been subject to standardisation both by ISO and CEN. The process of standardising LCA approaches began in the early 1990s and today the two standards SO14040 and ISO 14044 are generally adopted. In addition, a standard for communication of results from LCAs has been developed, known as Type III declarations or environmental declarations (EPD), ISO 14025. This standard addresses all products while ISO 21930 focuses on EPD for building products. A closely related standard to the ISO 14000 series has been developed: the ISO 15686 series Buildings and constructed assets – Service life planning. These standards are intended to complement the ISO 14000 series by describing how environmental standards may be implemented in building projects.

LCA is referred to as a backdrop and instrument for policy making by the EU Commission and UNEP, respectively. In relation to EU energy and building policy, several new CEN standards for sustainable buildings based upon LCA have been developed. This reflects a growing recognition of the wider systemic consequences of isolated processes undertaken by individual actors in the building sector. For more than two decades LCA has been developed and used as the predominant method for assessing the environmental performance of different building products and constructions during their lifetime.

In order to avoid flaws in assessments due to the interests of actors, the ISO standards include procedural measures to be applied when different products are compared. Ekvall *et al*. (2007) point out that the international standardisation process helps to reduce what can appear to be the arbitrariness of the methodology, but important methodological choices still remain free to be made in the individual study. It all depends on how goal and scope are defined, as discussed below.

From this description of general possibilities and limitations of LCA, we move to the more specific possibilities and limitations inherent in the method.

4.3 Strengths and weaknesses within LCA methodology

Current literature reveals that even if two LCAs apparently study the same kind of building, the results may differ significantly. How can this be explained? Performing LCA analyses of building materials and buildings requires methodological considerations on several dimensions, leading to a variety of explanations of differences in results. In the following sub-sections we will highlight some of the methodological explanations by organising the discussion according to the four phases of LCA as laid out by ISO 14040, namely:

1. Goal and scope definition;
2. Inventory analysis;
3. Impact assessment; and
4. Interpretation (ISO, 2006).

4.3.1 Goal and scope definition

As explained under the general possibilities and limitations of LCA, the goal of the LCA study determines several of the subsequent methodological choices. This may sound alarming – as if LCAs can be tuned to provide whatever results one may choose. However, it is more connected to a realisation that, for example, a study of an existing material in order to improve a production process requires a different approach from a study to guide policy making for buildings in the future.

When the goal is defined and the appropriate analytical method chosen, the next choice concerns the functional unit. In contrast to most assessment methods coupled to the environment, LCA has a product-oriented focus. Other assessment methods such as risk assessment or environmental impact assessment are mostly connected to point sources, i.e. individual production sites. Such methods can therefore reveal nothing about the viability of the production site to fulfil the needs and wants of users of products. LCA offers the possibility to make the assessment more relevant to end consumers.

The first century Roman architect Vitruvius (1960) once said that buildings must be solid, useful and beautiful. In our modern times the need for changing the functionality of buildings has become an increasingly important issue. An office building may be a school building tomorrow or the room solution may not meet the new tenant's requirements. Thus, Vitruvius's statement can be supplemented with 'buildings must be adaptable over time'. In this context Rønning *et al.* (2007) define sustainable buildings as those that *function* optimally for their purpose over time, while using the optimal amount of resources. The sustainable building should:

- function optimally – the users' needs should be met effectively
- be suitable for its use
- be flexible to adapt to changing needs and user requirements over time
- have optimal resource use, i.e. low material and energy consumption, low carbon emissions, etc.

A meaningful and valid description of these qualitative and quantitative aspects when analysing buildings or building materials and products is not possible without an understanding of the function of the product or system with respect to the needs or requirements of the end users. What is 'meaningful' depends on the scope. The functional unit (FU) is a quantified description of the performance of the product systems, for use as a reference unit in this context. Thus, one asks questions that make it possible to quantify both the qualitative and quantitative aspects of the function in relation to end use; e.g. 'what', 'how much', 'how well' and 'for how long'. In this context, the ILCD Handbook gives the following example of a functional unit (EU JRC, 2010): 'Lighting 10 square metres with 3,000 lux for 50,000 hours with daylight spectrum at 5,600 K'.

Given an indoor paint as an example where the scope is to substitute a chemical which will not influence the property of the paint, then one litre could be a suitable FU. But if the FU is to reflect the end use and the functionality when comparing two paints, 'the amount of paint necessary to cover 1 square metre of wall surface with a given quality and always "up to date" colour during 15 years' may be a more appropriate choice. In this case, the functionality (what), quality (how well) and time aspects (for how long) are reflected and quantified by the FU.

Some products are applied in a context where other conditions are guidance factors, rather than the product's capability to fulfil the required functionality. An example could be surface protection of an aeroplane. Due to safety requirements, repainting of the plane is necessary long before the actual need for maintenance connected to the lifetime of the paint. This kind of information is vital both for customers, as they can demand products which fulfil their needs and requirements, and for producers for better understanding their customers' needs as input to product development and marketing.

How can this be applied to building products or materials, or to a whole building during its lifetime? The new CEN standard EN15978:2011 introduces a functional equivalent intended to give a representation of the required technical characteristics and functionality of the building. According to this standard, the functional equivalent of a building or an assembled system shall include at least the following aspects:

- building type (e.g., office, factory, etc.)
- required service life
- relevant technical and functional requirements (e.g., the specific requirements of regulations, client, etc.)
- pattern of use.

Rossi *et al.* (2012) confirm this approach and highlight that each building is a unique product and its characteristics have to be included when defining the functional unit. It is of course a challenge to quantify all aspects related to the function of a material or building. One has to predict the future; what are future environmental impacts as a consequence of building design, user behaviour and changes in use pattern, re-design and other aspects related to adaptability, etc. Rønning and Lyng (2011) discuss how different types of buildings will need different approaches when defining the time aspect and scenarios in relation to the functional unit. The time aspect in this context refers both to the service life itself, which will vary considerably between, for example, an opera building, a warehouse and an office building, and to the service life period, which reflects the need for changes or maintenance during the required service life.

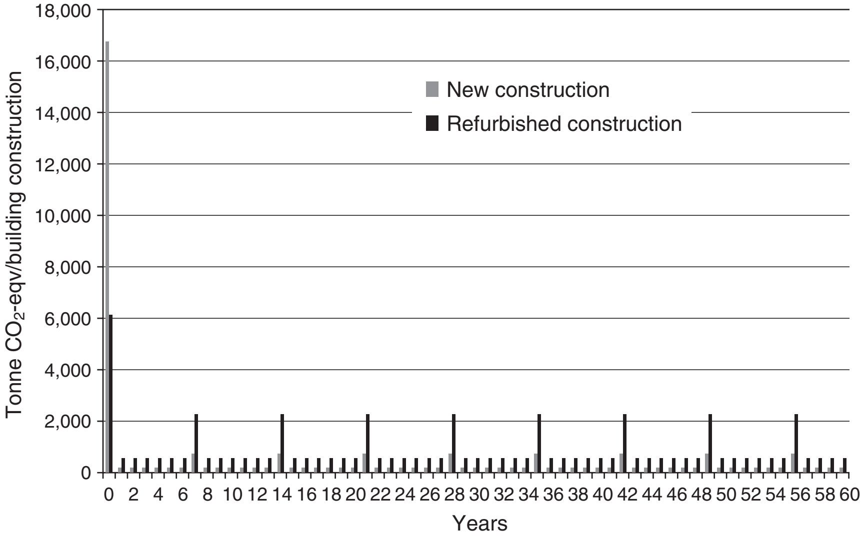
The required service life for which the LCA is undertaken is of great importance for the results as service life is usually relatively long (Erlandsson and Borg, 2003; Haapio and Viitaniemi, 2008a; Klunder, 2002). The choice of service life period (SLP) has a vital influence on the LCA results. The range in SLP is found to be between 20 and 100 years in the literature (Sartori and Hestnes, 2007). The definition of service life is a part of boundary setting in LCA and a premise for defining which life cycle stages and which activities should be included when developing a scenario for the future.

In fact decisions made in the feasibility or planning phase of the building process will affect factors such as the need for heating, maintenance, repair, etc., during the building's lifetime. Such activities will again increase the emissions and resource use from, for example, the production of new building materials and products; this is known as the embodied energy.

There are now strict guidelines for how to develop scenarios. This is done case by case. An example of how scenarios for the life cycle may be developed in an LCA is given in Fig. 4.1 (Rønning and Vold, 2008). In this study, LCA was used as one input to the decision on whether one should renovate an existing office building or demolish it and construct a new building. The state of the building and how it is and will be used were the main focus in constructing the scenario.

Since the average rental period for office buildings in Norway is 7 years, these buildings are subject to extensive rebuilding due to new tenants' needs and requirements. How extensive the rebuilding processes will be depends on the degree of adaptability of the building. In this case the existing building had a very low degree of adaptability and therefore the maintenance and changes required by new tenants needed relatively greater efforts. Thus, the contribution to global warming increased since the amount of materials needed for rebuilding of the poorly adaptable building will be greater than in the case of one with better adaptability.

Total GHG emissions related to the new construction are about 60% lower than in the case of refurbishing the old building, but if only the construction stage is considered, the conclusion will be the opposite as seen in the figure. This case illustrates that the emissions related to the maintenance



4.1 Global warming potential measured as CO_2 equivalents during the 60 years life time for the two constructions (Rønning and Vold, 2008).

and development of a building make a vital contribution to the total emissions during lifetime. If the life cycle stages of maintenance, replacement and development are excluded from the analysis, the total environmental performance throughout the life span of the building will thus be underestimated.

As discussed in Section 4.2 some LCAs define operation as limited to use of energy for heating and cooling (Blengini and Di Carlo, 2010). Others also include lighting and the use of technical equipment and appliances (Scheuer *et al*., 2003). In the first case one may question whether lighting and use of appliances are in fact included since it is difficult to specify the total energy used for different purposes. Other operation-related activities such as cleaning and inspection of technical equipment form part of the daily operation of the building, but are seldom included in LCAs. At the same time design and choice of materials and equipment will affect the other life cycle stages, e.g. the need for and type of cleaning (Rønning and Vold, 2008).

To summarise, the goal and scope definition phase lays the premises for the LCA by the methodological choices involved. As LCA is an iterative process, the goal and scope will probably change in the course of carrying out the study. This again affects the 'how', 'what' and 'why', and the study will thus potentially be refined. A side effect of the iterative process is that knowledge building related to the system studied can be achieved.

4.3.2 Inventory analysis

The goal and scope definition and the definition of functional unit and system boundaries form a framework for the inventory analysis. In this phase one enables the actors to examine and enhance their knowledge about, for example their own production as data regarding energy use, input of raw materials, loss in production, the functionality of their product with respect to the user's needs, etc. Thus, this phase encourages dialogue among different actors. Also, by defining and collecting data from suppliers, it emphasises the understanding of what defines and causes the product's embodied energy and how accumulated emissions increase. Of course if all the work is performed by an external LCA practitioner, communication and dialogue with clients will be vital for the dissemination of the new knowledge acquired.

To decide which data to collect regarding upstream and downstream processes is an extensive task. First of all, data selection and collection can be approached methodologically from two different perspectives: top-down, based on input–output life cycle assessment (IO-LCA) analysis or bottom-up, based on process life cycle assessment (P-LCA).

Input–output (IO) models represent the monetary transactions between industry sectors in mathematical form. IO models indicate what goods or services (or output of an industry) are consumed by other industries (or used as input). Hendrickson *et al*. (2006) exemplify this as:

> consider the industry sector that produces automobiles. Inputs to the automobile manufacturing industry sector include the outputs from the industry sectors that produce sheet metal, plate glass windshields, tires, carpeting, as well as computers (for designing the cars), electricity (to operate the facilities), etc. In turn, the sheet metal, plate glass windshield tire, etc. industry sectors require inputs for their operations that are outputs of other sectors, and so on. Each of these requirements for goods or services between industry sectors is identified in an IO model.

Thus, one captures infrastructure and other indirect emissions which would otherwise be excluded.

The factors that make the IO-LCA method an efficient and robust tool also limit its use for life cycle assessment. The results of an IO-LCA analysis represent the impacts from a change in demand for an industry sector. The given industry sector consists of several industry types. Thus, the data is not describing a specific industry and if the industry concerned is not representative of the sector or the sector is not homogeneous, this approach to modelling leads to uncertainty rather than providing accurate or holistic information. Only a limited number of environmental effects are included. The information is based on publicly available data for monetary flows and emissions measured and reported from different industries. The IO data are not modelled annually and a limited number of countries have statistics available in this form. One consequence is that an IO model for one country is used as a data source to describe other not necessarily comparable countries.

The second and most commonly used approach is the P-LCA. This approach calculates emissions and energy use based on the inflows and outflows from activities and process units in all life cycle stages associated with a product or product system and all environmental impacts related to these inflows and outflows. In other words, it is based on a traditional mass and energy balance approach. In the building sector P-LCAs have been the most common approach. Also in P-LCA, data selection and collection represent challenges for several reasons. Firstly, the construction sector does not have a tradition of evaluating their projects on a mass basis, only in economic terms. There are thus no key figures or experience-based calculations to lean on in this respect. Secondly, in a feasibility phase it will generally not be known which materials will be chosen. Last but not least, environmental data are not available for all building materials and components.

Based upon experiences both from P-LCA and IO-LCA, a third approach, 'hybrid LCA', has been developed. This approach avoids the 'narrow' system boundaries in P-CLA and thereby the omission of contributory factors outside these boundaries. Hybrid LCA combining P-LCA and IO-LCA models has demonstrated its value as a complementary tool to traditional inventory methods in LCA to overcome the lack of data and to include embodied emissions (Guggemos and Horvath, 2005; Sharrard *et al.*, 2008). Here process information collected in physical life cycle inventories is linked with monetary flows in economic models.

The selection of data is of vital importance for the LCA results. In the early days of LCA it took months of effort to access and collect both specific and generic data. Today, various databases and tools for calculating emissions are available to facilitate the performance of an LCA, and because producers are now often familiar with LCA, there is a greater willingness to share data. However, missing data has been and still is a challenge in LCA. If two products are to be compared and some of the data is lacking for one of the products, the conclusions may be the opposite of what is actually the case.

The development of databases has been a great success for developing and distributing data. One often distinguishes between specific data, for example from an individual producer, and generic data which can be an average of all producers in a region or from a database (e.g., Ecoinvent). Some of these data are common to many building products and this availability makes the LCAs based on these data more comparable. On the other hand, databases must be updated and not a source for preserving outdated data and knowledge.

Even when updated, several data sets in the same database may address the given activity or material. This is the case for electricity, and as it is widely used in relation to buildings and building materials, the choice of electricity model will have a vital influence on the results of the LCA. Various mixes of energy sources for producing electricity are available in databases. These sets of data give huge variations in GHG emissions from the use of electricity (Raadal *et al.*, 2011). Then, as presented in Section 4.3.4, simulation of the effects of different choices of data representation is one way of overcoming this limitation.

The process of 'defining scenarios' is and always has been to some extent a part of the design process, both for buildings and even when improving or developing new building materials. The LCA approach provides the opportunity to capture these issues through both qualitative and quantitative knowledge, which is then systematised and utilised in defining scenarios that quantify the activities. It is of course challenging to quantify and collect data for all materials, energy use, etc., in a way that correctly and realistically predicts scenarios for the given case. As an example, technical service life for building products is vital when developing new building materials or products. Studies of service life in practice show it often differs significantly from the service life predicted by producers (Bjørberg, 2009; Haagenrud, 2006). This is both a possibility and a limitation depending on how the actors approach it. On the one hand, if defining incorrect scenarios means that one does not take into account when and how maintenance or replacement might occur, this may weaken the robustness of the results. The lack of data may be overcome by simulations of different assumptions. On the other hand, investigation of service life in practice will increase knowledge of why changes in the life cycle occur and may be vital input to, for example, product development.

In the method of life cycle costing (LCC) these aspects are often handled and defined as scenarios for operation, maintenance and development, and expressed in monetary terms. Combining LCC and LCA enables the identification of the different building materials, energy use, maintenance, etc., for defining the scenarios given in an LCC model. But, in order to quantify the inputs, a slightly different approach from that of LCC must be implemented. When a developer performs an LCC assessment, the costs from historical projects often form the basis. Likewise, the existing LCC web tools are based on experience and historical data in order to give the output as monetary values.

The comprehensiveness of LCA is undoubtedly one of its main advantages. However, it also creates the need for a huge amount of data as all elements are ultimately interconnected. The data needed are of various origins and at different levels. The selection of data is of vital importance for the results. As more studies are performed using more data sources, the availability for future projects will increase.

4.3.3 Impact assessment

The impact assessment part of an LCA serves to aggregate the inventory data to aid interpretation by assessing the potential impacts of a life cycle. The main advantage of aggregating emissions into impact categories is to reduce the large number of emissions into a much smaller set of categories. The first step is classification where the question is 'Which emissions contribute to which impact categories?' The second is characterisation: 'How much do the different emissions contribute to the given potential impact expressed by the impact categories?' Potential impact is presented as, for example, 'acidification potential' and not 'how many fish will die' as a consequence of the given number of SO_x emissions. One does not take into account the buffer capacity of the river or lake. The third step consists of weighting, where the question is: 'Which potential impacts are most important?'

Although the actual assessment of environmental impacts must take place after the inventory is finished, the choice of impact categories must be made in the goal and scope stage. As discussed in previous sections, LCA captures several environmental impacts simultaneously. It gives the possibility to map the influence one measure has on several impacts. For example, reducing energy for operation by increasing the insulation properties of a material may lead to a reduction of greenhouse gases or global warming potential in the user phase, but increase the use of chemicals in the production phase.

Climate change and climate mitigation are on the political agenda and are highly focused in both international and national environmental policies. This again is reflected in business priorities, also in the building sector. Thus, LCA often focuses on a limited number of impact categories, e.g. global warming potential (GHG emissions) and embodied energy. The risk of a unilateral focus on one environmental challenge is that others are ignored. Which life cycle stage contributes most to a given impact category will differ from one category to another. In the context of the low energy debate which will involve a stronger focus on, for example, building material selection, the use of non-renewable resources may become a more critical factor in LCA than today. Again, depending on scope, ignoring impact categories may lead to a suboptimal understanding of the most important environmental aspect and where in the life cycle it occurs.

Human activities have impacts on the environment which may not easily be quantified in an LCA. Examples of such impacts are noise, encroachment during construction periods, effects on the ecosystems and indoor climate. Given the significant consumption of resources in the construction sector, impact categories related to the depletion of non-renewable resources, such as land use, are also particularly relevant for building-related LCA studies (Rossi *et al.*, 2012). However, databases and calculation tools used for inventory analysis or to assess environmental impacts may not be available for all potential impacts or applications. Moreover, even if they are available, there are several models for calculating one given impact. This is problematic as it requires specialised knowledge to select and use the methods correctly.

When all impacts are calculated, what is the most important one? There is no definite answer to this. It all depends on which impact is given more weight than others and by whom. This is to a large extent a matter of interests and values. The comprehensiveness of LCA and its ability to capture several impact categories is one of its main advantages. Furthermore, as more studies are performed with improved data sources and models, availability for future projects will improve.

4.3.4 Interpretation

The benefits of an interpretation step in the LCA are primarily to help the reader to understand and identify the most significant issues. In relation to the previous three LCA phases, examples of general questions to be asked are:

- *Goal and scope definition*: Is analysis in accordance with the defined goal and is the goal in accordance with the needs of intended users in order to achieve appropriate decision making, both at political and technical level? Do system boundaries include all relevant life cycle stages and do they cover the relevant geographic area? Are realistic scenarios defined? Is the functional unit defined satisfactorily?
- *Inventory items*: What are the most important items and where in the life cycle do they appear?
- *Impact categories*: Are all relevant impacts included and do they give the same conclusions?

The answers are complex and entangled since LCA modelling allows for considerable variations in terms of calculation methods and the results obtained may thus also vary. The intended application of the study, how the system boundaries are determined (which phases to include/exclude), the assumptions made, cut-off criteria, the available data and the quality of the data, and finally the intended audience, will all affect the results. They may also be influenced by the values and perspectives of the LCA practitioner and whoever commissions the LCA. These limitations are also not unique to LCA. As stated by Ekvall *et al.* (2007), several methods for environmental systems analysis have been developed to support different types of decisions, and most of these involve similar problems.

This makes it important that these aspects are documented and assessed in the interpretation step. In the interpretation the robustness of the choices can be tested by simulation of, for example, other data, system boundaries, calculation methods, characterisation factors, etc. This not only strengthens the results of the LCA, it may also give vital input to choose the best solution for optimising the system. Even the flexibility in methodological choices may weaken the credibility of LCA at first sight; one should not underestimate the importance of the process of performing the LCA itself. It is here one has to understand the product or system and actors in a broader context. When methods for future studies are integrated in the LCA, the methodology not only assesses scenarios, but Ekvall *et al.* (2007) state that it also assists in *developing* the scenarios that are to be assessed. This is in accordance with the experiences of Rønning and Vold (2008) where LCA was used to communicate choices in a feasibility phase.

LCAs are mostly used for documenting the consequences of already established choices and decisions or completed construction projects, and are to a lesser extent used as a planning tool for simulation of consequences of different choices in various phases of the construction process or throughout the lifetime of a building.

The environmental performance of a building or an individual building material depends on many factors, such as how it is designed, what it is made of, where it is located, how it fits into the surroundings and how it is used. These factors must be considered in the interpretation phase in a comprehensive way in order to determine if the environmental profile of a building or an individual building material is 'satisfactory' with respect to the scope defined. However, what is considered satisfactory is generally not defined, and will at all times reflect political targets, environmental guidelines, the ambitions of the developers, etc. At the same time, these factors will often partially explain the results; for example, the design will affect the use of energy for heating or the amount of materials needed.

Several studies conclude that when performing LCAs of buildings for planning purposes, especially regional planning, one should include transport activities related to the use of the building. Transport may contribute as much as 50% of the total energy use (Norman *et al.*, 2006; Selvig and Cervenka, 2008). Cole (1999) found that worker transportation could account for 10–80% of total construction energy. However, in current LCAs the transport of users is typically overlooked (Stephan *et al.*, 2012), while the transport of construction workers is not insignificant and is excluded in most studies (Blom *et al.*, 2010). On the other hand, LCA standards do not require transport of users (or workers) to be included in LCAs as a consequence of localisation.

The interpretation phase in LCA is at first sight aimed at the individual building or material which is being examined. Haapio and Viitaniemi (2008a) have performed a literature review on different calculation tools for environmental evaluations based on LCAs of entire buildings. This study shows that LCA results are dependent on the tool used, and that a comparison between results from different tools is impossible and not recommended by the authors.

However, there are a number of LCA studies that compare building materials, such as wood versus concrete or steel (Upton *et al.*, 2008; Guggemos and Horvath, 2005). The functional units may be equal, but the system boundaries are often not comparable as some of the life cycle stages are excluded. The argumentation for the exclusion is not always clear. However, some argue that 'energy for heating is equal for both buildings'. This is clearly a weakness when comparing building systems based on different building materials, especially when comparing results for two different LCAs. On the other hand, if the transparency is satisfactory one may draw general conclusions and knowledge from such studies.

Rajagopalan *et al.* (2012) state that LCAs are too case-specific with respect to functional unit, system boundaries, specific scenarios for a specific type of building, etc. Thus, an LCA cannot be replicated and general conclusions are not transferable to other building projects. On the other hand, the flexibility of LCA enables a large number of different analyses. Often the results are intended to be communicated to a wide audience. Thus, the results and outcome of the LCA have to be presented with a certain transparency and clear interpretation to ensure that the audience understands that figures and results may vary depending on the intended use of the LCA.

4.4 Conclusions

The use of LCA as an assessment tool has become common as the holistic model for assessing energy and environmental performance of buildings or building materials. Presently, it has become the main approach for answering the question of which material is most environmentally friendly for constructions and buildings. The fact that LCAs are used in applications from intra-company decisions to large-scale political decisions indicates that LCA has gained broad acceptance by actors in the building sector. During the last 20 years the method has been improved and refined, and has been subject to a number of strategies in order to create global standardisation of the method.

The comprehensiveness of LCA is undoubtedly one of its main advantages. The comprehensive approach offered by the LCA approach makes it possible, at least at a theoretical level, to confront a number of issues relating to how to assess complex problems by including value chains with vast amounts of data from different sources. At the same time, the approach addresses a number of potential and actual environmental impacts. Likewise, it allows a number of actors to become involved in efforts to create environmentally friendly solutions based upon scientifically grounded knowledge and methods. Results from LCA analyses point out the most important environmental problems, and where in the life cycle these problems should be identified. Informing about LCA approaches is thus an arena for communication and learning about environmental issues, offering input to different decision makers about when and how in the life span of a product environmental problems and possible improvements might occur.

As indicated by several authors, despite (or maybe because of) the holistic approach LCA offers, inherent ambiguities in the methodology represent limitations as it is presently used. This is becoming obvious in the current work on standardising LCA methodology. The limitations can be identified at different levels of analysis: at a macro level, the ambitions of global standardisation are confronted with the complexities and ambiguities of the building sector. Here, there is a need for 'translation processes', by opening up the standardisation formats to local and contextual characteristics. At the micro level, there are several pragmatic obstacles to using the full potential of LCA: there is not enough time to capture all relevant data, not sufficiently sophisticated methods to capture all relevant environmental impacts, no consensus (or too many methodological choices) to determine which environmental impacts are more important than others, not enough harmonisation to compare results even for the same material across studies.

From a number of examples presented in the current literature, it may be seen that even when studying the same kind of building or building materials, LCA results may differ significantly. As illustrated in the discussion above, there are a number of methodological options that will potentially influence the results from the LCA. In our presentation we structured these options according to the stages in the LCA method. As we see it, several conditions must be fulfilled in order to reduce the variations in the results from the LCA method. Firstly, a clarification of goal(s) will be necessary as a starting point. Secondly, the definition of scope, including the choice of functional unit, is the defining reference for the analysis to be performed. Here, the system boundaries must be defined in an unambiguous way. Thirdly, data selection and collection must be transparent, valid and reliable. The fourth stage in LCA is the impact assessment. In spite of the complexities of aggregating emissions into impact categories, there is currently a large extent of agreement for several important categories. Finally, in the interpretation stage the results of the LCA must be related to the goals and scope of the analysis.

In summary, we argue that in order to ensure more use of life cycle considerations, focus should be on the challenges along two axes: on the one hand, to strengthen the credibility of the underlying data and calculation methods of LCAs, and on the other hand to facilitate the use of results in actual construction processes, companies' product development and overall priorities at the national and local levels. Clarification of which environmental information decision makers need in the various phases of the construction process is vital in this context. However, we would claim that LCA provides a wealth of possibilities. The flexibility of the method coupled with already existing models and data give LCA the power to inform decision makers and the public about the environmental performance of products and services to a considerable extent.

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5

Using life cycle assessment (LCA) methodology to develop eco-labels for construction and building materials

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Abstract: The increase in both the number and complexity of products of all kinds is a main cause of the current environmental problems. Manufacturers and distributors use different types of eco-labels to voluntarily provide environmental information about their goods. By providing accurate and relevant information, eco-labels aim at helping consumers to identify those products and services with a better environmental performance in comparison with other equivalent products or services. A life cycle approach should be applied when defining the rules for awarding eco-labels, no matter the type, to avoid impact shifts between different impact categories, life cycle stages or geographical areas.

Key words: life cycle thinking, life cycle assessment, eco-labels, environmental product declarations (EPD).

5.1 Introduction: life cycle thinking and eco-labels

Products are responsible for a range of impacts on the environment (global warming potential, ozone layer depletion, acidification, toxicity, etc.) across their complete life cycle. The increase in both the number and complexity of products of all kinds is a main cause of current environmental problems. For this reason, the scope of environmental policies has been extended in recent years, going from end-of-pipe and cleaner production initiatives to an integrated life cycle perspective (European Commission, 2008). This perspective considers all life cycle stages of products from raw material extraction, through manufacturing and distribution, to use and end-of-life treatment and final disposal. Life cycle thinking seeks to identify lower environmental impacts by products and services across all life cycle stages in order to avoid burden shifting, i.e. 'minimising at one stage of the life cycle or in a geographic region, or in a particular impact category, while helping to avoid increases elsewhere' (European Commission, 2010).

By providing accurate and relevant environmental information, eco-labels aim at helping consumers to identify those products and services with a better environmental performance in comparison with other functionally equivalent products or services. Manufacturers and distributors use different types of eco-labels to voluntarily provide information about the environmental performance of their goods. A life cycle approach should be applied when defining the rules for awarding eco-labels, no matter the type, in order to avoid impact shifts between different impact categories, life cycle stages or geographical areas. By looking at the whole product's life cycle, environmental impacts are addressed in an integrated way at the point where they are likely to be most effective in reducing the environmental loads and saving costs for business and society (European Commission, 2003). However, not all the different types of eco-labels embrace the life cycle thinking approach in a comprehensive and systematic way. In addition, not all the environmental logotypes or claims that can be found associated with products or services are truly 'eco-labels'. The latter concept should be applied only to those labels and declarations which provide relevant, verifiable and reliable environmental information. On the other hand, an environmental label or declaration may take the form of a statement, symbol or graphic and may be available on a product or package, in advertising, technical brochures, etc. (ISO 14020, 2000).

5.2 Life cycle assessment (LCA)

The life cycle assessment (LCA) methodology is defined by ISO 14040-44 standards and explained in detailed guidelines such as the *ILCD Handbook* (IES, 2010). According to the ISO 14040 standard definition, LCA consists of a 'compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle' (ISO 14040, 2006). Therefore, LCA is a tool for the practical implementation of the life cycle thinking approach, independently or in combination with other related methodologies such as life cycle costing (LCC) and social life cycle assessment (SLCA).

The origins of LCA can be traced back to studies conducted in the 1960s and 1970s aimed at optimizing energy and raw material consumption (Raugei and Gazulla, 2009). Since then, its methodology has been intensively developed and an increasing number of companies use it in internal decision-making processes (Pennington *et al.*, 2007; Finnveden *et al.*, 2009). LCA results may be useful inputs to a variety of decision-making processes, including product development and improvement, strategic planning, public policy making and environmental labels and declarations (ISO 14040, 2006).

Four steps form the LCA methodology according to ISO 14040. It has to be noted that LCA is an iterative methodology, where each step is dependent on the results and methods used in the other previous or subsequent steps.

1. ***Goal and scope definition***: the objectives and scope of the analysis should be clearly stated, including the definition of the functional unit, the system boundaries, the necessary data, the basic assumptions or the intended verification procedures. This first step is of great importance, as the subsequent stages depend on it.
2. ***Life cycle inventory (LCI)***: in this step all the input and output flows (i.e., materials, energy and emissions) of the various processes included in the system boundaries are accounted for. To this end, different sources of data can be used, such as data obtained through direct measurements, data obtained from previous studies available in up-to-date and verified international LCI databases or data inferred by means of educated guesses and estimates, based on previous analyses and the analyst's experience (Raugei and Gazulla, 2009). As it is not always feasible to obtain data of the desired quality, the methodology utilizes sensitivity analysis in order to verify whether the doubtful data may have a relevant impact on the reliability of the results.
3. ***Life cycle impact assessment (LCIA)***: in this step, all the inputs and outputs of the LCI are classified into different impact categories, according to the type of impact that they may have on the environment. Examples of impact categories are: abiotic resource depletion, global warming, acidification, stratospheric ozone layer depletion, eutrophication, etc. This classification results in several long lists of different inputs and outputs which may appear many times in different groups, as a single input/output can potentially contribute to more than one environmental impact category. Once classified, for each impact category, the different inputs and outputs are characterized, i.e. converted to reference compounds (e.g., CO_2 for the global warming category) which, if emitted, would have a quantitatively comparable effect on the environment to one unit of the actual system input/output being considered (Raugei and Gazulla, 2009). As a result of the characterization, a unique impact indicator is calculated for each impact category. A third, optional, phase of the LCIA is normalization aimed at providing an indication of the comparative relevance of the environmental impact caused by the system under assessment in the different impact categories. To this end, the characterized results are divided by the total yearly impact of a reference region (normally, a region, a nation or even the entire world). Because extrapolation rather than empirical estimates are used in the normalization, this stage is affected by high degrees of uncertainty (Raugei and Gazulla, 2009). Finally, normalized indicators can be multiplied by weight factors and then summed to produce one overall environmental impact indicator. It has to be noted that the choice of weighting factors is based on value-choices and is not scientifically based and, therefore, should not be used in LCA studies intended to be

used in comparative assertions for disclosure to the public (ISO 14044, 2006).

4. ***Interpretation***: the fourth and final step in LCA should consist of the identification of significant environmental issues, an evaluation (completeness, sensitivity and consistency checks) and the drawing of conclusions, limitations and recommendations.

The LCA methodology has been developed and matured during the last 20 or 50 years, but there are still several areas where further development would be useful, such as methods for assessing impacts on ecosystem services, weighting methods and quality assurance of LCA databases (Finnveden *et al.*, 2009).

5.3 Types of eco-labels and their relation to LCA

Different types of labels with environmental information are present in the market, each with different characteristics and specific objectives. Eco-labelling programmes have been developed in almost every industrialized country and also in some developing countries (Cobut *et al.*, 2013). This chapter is focused on voluntary eco-labels related to products and services which, according to the International Standardization Organization (ISO), are classified into three types: Type I (certified eco-labels), Type II (product self-declarations) and Type III (environmental product declarations). As acknowledged by ISO, all these different types of eco-labels pursue a common objective, i.e. to foster the demand and production of products with a lower environmental impact through the communication of verifiable and reliable information, stimulating different market players towards continuous environmental improvement (ISO 14025, 2006). As a quantitative environmental assessment methodology, LCA can be used in the definition of the award criteria of different types of eco-labels, as outlined below.

5.3.1 Type I (third-party)

According to ISO 14024, Type I eco-labels indicate the environmental preference of the product or service within its product category, based on multiple criteria over its entire life cycle and defined by a third party. Several Type I eco-label schemes, such as EU Eco-label, Nordic Swan (Nordic countries) or Blue Angel (Germany) have been developed worldwide in the last decades, a majority of them administered by national or regional public organizations. These schemes are intended to promote those products representing the best environmental performance achieved in the addressed markets and consequently the awarding criteria consider both environmental and market relevant information. Revised criteria are periodically published in order to ensure that the eco-labelled products have their environmental impact among the lowest in their corresponding categories.

For each product category, Type I eco-label schemes identify the most environmentally relevant areas and establish the specific criteria and thresholds which will differentiate the best products from the rest. For instance, the Nordic Ecolabelling scheme states that the most relevant environmental aspects associated with computers are mainly due to power consumption, the amount of waste produced and the use of hazardous substances such as flame retardants and lead (Nordic Ecolabelling, 2012). Consequently, the requirements that a computer must fulfil to be awarded the Nordic Ecolabel focus on aspects such as power consumption, design for upgradeability and disassembling, contents of heavy metals and plastics and their additives, recycling of discarded products and performance on noise level, ergonomics and electric and magnetic fields. For each of these aspects, the scheme establishes specific criteria and thresholds, the compliance of which must be proven by manufacturers willing to have their products awarded the eco-label.

Despite the ISO 14024 standard establishing that the life cycle of a product has to be considered, it does not stipulate to what extent LCA methodology has to be followed and, therefore, its application varies from one Type I scheme to another (Scheer and Rubik, 2005). For instance, the EU Eco-label criteria are determined on a scientific basis considering the whole life cycle of products, taking into account the most significant environmental impacts and the net environmental balance between the benefits and burdens at the various life stages of the products (Regulation EC 66/2010). According to the procedure for the development and revision of EU Eco-label criteria, the relevance of the environmental impacts associated with the product group is based on new or existing LCA studies. Therefore, when developing criteria for new product categories, one of the tasks to be undertaken is the gathering of existing LCA studies related to the product category under assessment and if such studies are not available, LCA studies must be developed anew. The results of such studies as well as other sources of scientific information are combined with market information in order to ensure that the eco-labelling criteria correspond indicatively to the best 10–20% of the products available on the Community market in terms of environmental performance.

Other Type I eco-labelling schemes do not establish such requirements in the development of criteria, so proper LCA studies (i.e., according to ISO 14040-44 standards) may or may not be used for identifying the relevant areas or setting the threshold values. On the other hand, applicants willing to have their products awarded the eco-label do not need to develop LCA studies to prove the environmental benefits associated with their products, but can prepare other types of documentation and perform tests proving the fulfilment of the awarding criteria.

5.3.2 Type II (self-declarations)

Type II eco-labels are developed directly by manufacturers or distributors in order to provide environmental information about their products or services. Self-declarations may consist of a written statement or a symbol referring to an environmental aspect of a product, a component or the product's packaging. In this case, there is no certification by a third party, as it is a self-declaration developed internally by companies. Nevertheless, the information provided should be verifiable, accurate and relevant. ISO 14021 (1999) provides guidance on how to develop Type II eco-labels, for example it states that vague or non-specific claims or which broadly imply that a product is environmentally beneficial shall not be used; therefore, claims such as 'green', 'environmentally friendly' or 'non-polluting' are not adequate. The standard also provides guidance on how to use specific claims such as 'compostable', 'recyclable', 'reduced resource use', 'reduced water consumption' or 'waste reduction', or on how to use symbols to make environmental claims.

Prior to making the claim, the manufacturer shall implement evaluation measures to achieve reliable and reproducible results necessary to verify the claim. Once the claim is disclosed, the manufacturer may voluntarily release to the public the information needed to verify the claim or, if not, disclose it upon request (ISO 14021, 1999). In that context, LCA may be used to provide scientifically sound information for the development and verification of the claim. However, the fact that Type II eco-labels are not third-party verified lowers their level of reliability in comparison to the other eco-label types.

According to Scheer and Rubik (2005), Type II eco-labels could be regarded as 'do-it-yourself' labelling, i.e. a business marketing approach to inform consumers of the environmental qualities of their products. Abundant examples of environmental self-declarations can be found in the packaging of different products; however, not all of them are ISO 14021 complying. In fact, numerous examples of product declarations that have no meaning, are misleading or deceitful can be found in the market, confusing consumers and undermining the effectiveness of true claims and labels (UNOPS, 2009). In that sense, guides aimed at promoting clear, accurate and relevant environmental claims in marketing and advertising in order to help consumers to make informed choices have been developed (DEFRA,

2010) as well as guides to detect potential 'greenwashing' practices in professional purchasing processes (UNOPS, 2009).

5.3.3 Type III (environmental product declarations)

ISO 14025 standard defines environmental product declarations (EPD) or Type III eco-labels as an 'environmental declaration providing quantified environmental data using predetermined parameters and, where relevant, additional environmental information' (ISO 14025, 2006). The Type III eco-label format is totally different from the previously explained eco-labels, as EPDs consist of a written report providing both qualitative and quantitative information about the product. The quantitative information is based on the LCA methodology and can be used either by producers in order to improve their products, or by consumers to help them make informed purchase decisions. Independent verification of the EPDs ensures that these declarations contain relevant and verifiable LCA information based on the ISO 14040 series of standards.

Because EPDs can be developed for any product without implying that it has a better environmental performance in comparison to an average product, it is the comparison between products that triggers the desired continuous improvement. Then, in order to ensure that this comparison is done correctly and to avoid that EPDs contain misinformation, it is necessary that the same calculation rules are applied during the EPDs' development process. In that sense, predetermined parameters contained in an EPD (inventory results, impact category indicators and others) are based on the ISO 14040 series on LCA, as well as on the product category rules (PCR) applicable to the specific product. PCR documents set the procedure and requirements for the development of LCA and EPD for specific product categories. In the elaboration of PCR documents, appropriate LCA studies should be collected or produced and interested parties (e.g., product manufacturers, purchasers, users, associations, public agencies, etc.) involved.

As in the case of Type I eco-labels, several national and international programmes have been developed around the world for the development of Type III eco-labels, a majority of them within the construction sector. The operators of such programmes (from professional associations to certification bodies) shall establish transparent procedures for the independent verification of the Type III environmental declarations (ISO 14025, 2006) to ensure that EPDs have been correctly developed.

In general, these schemes have been developed according to ISO 14025 but, as foreseen in this standard, all of them have created their own programme rules and PCR documents. As a result, the EPDs of these programmes differ in their content, calculation rules and format and, therefore, are not directly comparable or combinable. It is expected that the recently approved EN 15804 (2012), which provides core product category rules for all construction products, will contribute to a major harmonization of EPDs within the construction sector. One of the key aspects defined by the EN 15804 standard is the impact assessment methodology, including the specific impact categories and environmental parameters to be declared.

Carbon footprint

Due to the urgent need to accelerate efforts to reduce anthropogenic greenhouse gas (GHG) emissions, companies are assessing the carbon footprint of their products and services to better understand their contribution to climate change. The main standards developed for the calculation of the carbon footprint of products and services (WBCSD, 2011; BSI, 2011) are based on the LCA methodology. However, the limitation of the assessment scope to just one impact category (i.e., global warming) does not prevent the undesired burden shifting pursued by the life cycle thinking approach and, therefore, carbon footprint can unfairly promote products that do not necessarily have a better overall environmental performance (European Commission, 2010).

Like other single-attribute eco-labels, carbon footprint adopts a life cycle orientation. However, it does not have the holistic approach of Type I or Type III eco-labels, which aim at evaluating all aspects of environmental impact over the whole life cycle of products or services (Cobut *et al.*, 2013).

5.3.4 Conclusions

As has been explained, life cycle assessment (LCA) may be used during the development of the criteria that need to be fulfilled in order to obtain a Type I eco-label, whereas it is not needed to demonstrate that a certain product or service meets the required eco-label criteria. On the other hand, Type II eco-labels are not necessarily based on LCA results, as manufacturers or distributors can decide which specific information they want to communicate. However, in such cases, to undertake an LCA study may help companies to identify the main environmental hotspots of their products as well as assess the potential advantages that a certain product entails in comparison to similar products. Finally, Type III eco-labels (or EPDs) require that an LCA study is undertaken following specific pre-defined calculation rules (called product category rules). Therefore, only in the case of Type III eco-labels is LCA a prerequisite for award of the eco-label as a means to inform about the relevant environmental aspects of the product throughout its life cycle.

Type III eco-labels allow the comparison of the environmental performance of products awarded, as quantitative and verified LCA results are included in the declaration. By contrast, two different products awarded the same Type I eco-label may not be compared and, therefore, apart from knowing that they both represent better options in the market, it cannot be established which of them has the lower environmental impact. As Cobut *et al.* (2013) state, the remaining limitation of Type I eco-labels is the lack of information enabling the differentiation between products bearing the same eco-label as well as the link between environmental criteria and the

Table 5.1 Relation of LCA methodology to different types of eco-labels

	Type I	Type II	Type III
Description and purpose	Indicates the environmental preference of the product or service within its product category, based on multiple criteria over its entire life cycle and defined by a third party	Self-declarations referring to an environmental aspect of a product, a component or the product's packaging	Environmental declaration providing quantified environmental data using predetermined parameters and, where relevant, additional environmental information
ISO standard	ISO 14024	ISO 14021	ISO 14025
Life cycle assessment methodology:			
• is used when defining the awarding criteria	Not in all cases	No	Yes
• is used to demonstrate that the product/service fulfils the awarding criteria	No	Not in all cases	Yes
• is used to provide quantitative environmental information allowing the comparison of different eco-labelled products/services	No	No	Yes

real environmental impacts of the product. In some cases, that lack of quantitative information may make decision making within the context of procurement processes more difficult. On the other hand, the more simplified and aggregated information provided by Type I and Type II eco-labels is mainly addressed to final consumers, while Type III eco-labels are developed for business-to-business communication, as the technical and detailed level of EPDs better suits the time and competence available for purchasing decisions of professional consumers (Baumann and Tillman, 2004).

The research of Rajagopalan *et al.* (2012) shows that currently, when the LCA methodology is applied, eco-labelled products do not always have lower impacts than their non-green counterparts. Therefore, eco-labelling award criteria based on a quantitative life cycle approach (i.e., LCA) should be applied in order to ensure that the eco-labelled products are the best option in terms of environmental performance. This comprehensive application is carried out in the case of Type III eco-labels, but not in all cases in Type I schemes. On the other hand, in order to reduce uncertainty in eco-labelling schemes, the availability and quality of LCA data needs to be improved (Rajagopalan *et al.*, 2012).

5.4 Environmental certification programmes for buildings

Labelling schemes for sustainable or green buildings (such as BREAM, LEED, HQE or DGNB) have been developed over the past two decades (Fullana *et al.*, 2008), and are seen as a major driver for innovation and implementation of sustainable thinking in the construction sector (Wittstock *et al.,* 2012). In general, life cycle information is taken into account when defining the rules for awarding green buildings certifications; however, the LCA methodology is used only in some cases (such as DGNB, for instance) for assessing the whole building. In addition, some of these schemes require environmental quantitative information on the used products which can be provided using EPDs. However, as Cobut *et al.* (2013) observed in their research, life cycle oriented eco-labels are seldom included in environmental certification programmes for buildings while single-attribute eco-labels are the first required environmental criteria for certain materials (such as FSC in the case of wood products).

5.5 Future trends

Relevant and reliable environmental information can help consumers to make informed purchasing decisions if this information is also simple to understand. An excess of information can overwhelm consumers and, in that sense, aggregation allows the communication of environmental information to be simplified. However, as explained above, aggregation of environmental impacts has little scientific meaning and, in addition, is less transparent. One of the findings of a recent study commissioned by the European Commission (2013) in that field is that absolute values by themselves are not sufficient and scales should be used when communicating multi-criteria environmental information. In that context, proposals for combining the characteristics of Type I and Type III eco-labels are being developed. Type I eco-labels are more simple to interpret and can be printed on the products' packaging. On the other hand, Type III eco-labels allow a direct comparison between similar products if the same LCA calculation rules have been applied.

Since 2007, within its national commitment to the environment (le Grenelle Environnement), the French government has launched a series of activities to reduce climate change and energy consumption in France. One of these activities is to ensure that consumers have access to reliable, objective and comprehensive information on the overall environmental performance of products (including the emissions of GHG and the consumption of natural resources). To achieve this, experimentation has been conducted in 2012 with the collaboration of more than 150 companies. As a result, several proposals for labels and declarations for communicating environmental information combining the advantages of Type I, Type II and Type III eco-labels have been developed. Within the HAproWINE LIFE project, D'Souza *et al.* (2012) suggest that wine producers first carry out an LCA study and EPD of their products and then, by comparison to average reference values of the different environmental impact categories corresponding to the product category, companies award an Type I eco-label for their wines if they satisfy an excellence criteria (i.e., if they have lower environmental impact than the average). Such a scheme, which may be applied to other product sectors, implies that the Type I eco-label criteria should be based on LCA results of individual products. To this end, average environmental impacts of product categories should be known.

In order to encourage consumers to make informed purchase decisions that stimulate manufacturers to improve their products towards sustainability, information about the environmental, economic and social life cycle performance of goods should be disclosed to the public. The combination of LCA, LCC and SLCA methodologies is a first step in that direction; however, the level of maturity and data availability for their combined application is still low.

Finally, smartphone technology may be very useful in business-to-consumer communication, enhancing access to detailed information when making the purchasing decision at the point of sale. It is expected that obtaining real-time purchase input will become more common in the future (European Commission, 2013). Rating initiatives such as GoodGuide have developed smartphone applications to help consumers quickly evaluate and compare products on their health, environmental and social performance, and make purchasing decisions that reflect their preferences and values.

5.6 Sources of further information and advice

5.6.1 Institutions

- UNEP-SETAC Life Cycle Initiative is an international partnership between the United Nations Environment Programme (UNEP) and the Society for Environmental Toxicology and Chemistry (SETAC) in order to bring science-based life cycle approaches into practice worldwide.
- The Global Ecolabelling Network (GEN) is a non-profit association of Type I ecolabelling organizations from around the world.
- The Global Environmental Declarations Network (GEDnet) is an international non-profit association of Type III environmental declaration organizations and practitioners.
- The International Life Cycle Academy (ILCA) aims to promote the use and good practice of life-cycle based sustainability assessment worldwide through education of students and practitioners to the highest scientific and ethical standards.

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6

The EU Ecolabel scheme and its application to construction and building materials

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Abstract: The aim of this chapter is to describe the EU Ecolabel, an environmental label developed by the European Commission to promote environmental excellence in products and services. The chapter first introduces the working EU Ecolabel: its history, criteria setting procedure and GPP (green public procurement). The link between the EU Ecolabel and national ecolabelling schemes is discussed, followed by a focus on construction and building materials covered by EU Ecolabel criteria. The products are listed, showing the main problems and an explanation of possible future trends is given.

Key words: sustainability, environmental labelling, EU Ecolabel criteria, GPP, construction and building materials.

6.1 Introduction

The EU Ecolabel is an evaluation/communication instrument developed by the European Commission during the last 20 years to support ‘business to consumers’ environmental initiatives as set out in the latest revised Regulation (EC) No. 66/2010 of the European Parliament and of the Council of 25 November 2009. The EU Ecolabel is a ‘Type I’ environmental label, according to the ISO 14020 classification (ISO 2000) and is intended to be a voluntary market tool for promoting environmental excellence in products and services in a rigorous and standardised way. It is part of the sustainable consumption and production policy of the Community. Its aim is to reduce the negative impact of consumption and production on the environment, health, climate and natural resources, stimulating producers to verify the means by which products can achieve environmental excellence for environmentally conscious consumers (EU Regulation No. 66/2010). The scientific reliability of the EU Ecolabel is based on the use of the life cycle assessment methodology (LCA) for establishing the environmental criteria used in assessing products. The EU Ecolabel is therefore a recommended reference for establishing green public procurement action plans for the public purchasing of environmentally friendly products by EU member states.

The first part of the chapter covers the history of the EU Ecolabel, its aims and related statistics. This is followed by the development and revision criteria for obtaining the label according to the current Commission Regulation. Stakeholders directly involved during working sessions are identified and existing procedures analysed. Green public procurement (GPP) and its connections with the EU Ecolabel criteria are introduced, together with the relation between the EU and national Ecolabels, mainly ISO Type I labels.

Particular attention is given to those Ecolabel product groups which are of interest to the construction and building materials sector. In Section 6.7, attention is focused on those products or services that can be awarded the label, from coverings (hard, wood, textile), paints and varnishes, to tourist accommodation and office buildings (currently in progress). The available EU Commission decisions for these products and services are analysed and useful information, approaches and principal criteria are reported. Finally, possible future trends for the EU Ecolabel and construction materials are reported.

6.2 The EU Ecolabel and the European Commission policy for sustainability

In its strategy for sustainable development, the European Commission has identified measures for responding to the key challenges of unsustainable trends and the required actions. The goal of a more sustainable model for production and consumption remains one of the most significant challenges for the Commission's strategy on future sustainability. The recent launch of the product environmental footprint (PEF) methodology reinforces the vision of a 'A resource-efficient Europe' (European Commission, 2011b) and a renewed policy is expected by early 2013. Pragmatic methods for increasing resource productivity and decoupling economic growth from both resource use and environmental impacts, while taking a life cycle perspective, are likely to be forthcoming (European Commission, 2012).

In practice, the EU aims to break down the link between economic growth and environmental degradation. To this end, several strategies have been proposed and remain under continuous improvement:

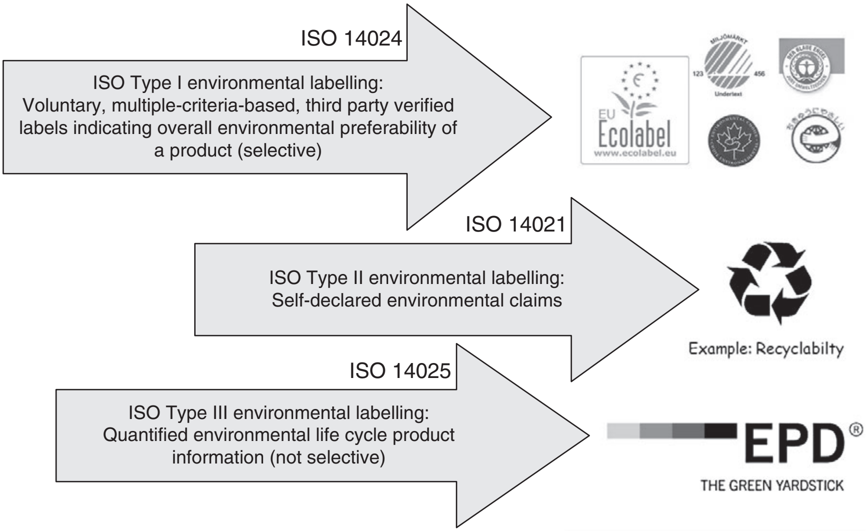
- promotion of environmental innovations at industrial level;
- promotion of information about environmental friendly technologies;
- encouragement of green public procurement;
- developing and promoting the Ecolabelling of products and services.

The EU Ecolabel is therefore to be considered part of a wider sustainable consumption and production policy for promoting environmental excellence which can be trusted by consumers. The slogan used by the Commission on the EU Ecolabel home page is:

> 'The EU Ecolabel helps you identify products and services that have a reduced environmental impact throughout their life cycle, from the extraction of raw material through to production, use and disposal. Recognised throughout Europe, EU Ecolabel is a voluntary label promoting environmental excellence which can be trusted' (www.ecolabel.eu).

The environmental labels (identified as 'Type I' by ISO 14020) and the environmental product declarations (or EPD, identified as 'Type III' by ISO 14025) are tools that enable the identification of products or services which offer reliable information on environmental impact throughout their life cycle, from the extraction of raw material through production, use and disposal. However, the EU Ecolabel differs from the EPD as it guarantees environmental excellence (Fig. 6.1) as defined by the European Commission.

The EU Flower is issued by competent bodies based in each member state of the European Union under European Commission surveillance. An organisation from everywhere in the world may apply for the EU Flower and start the application with any competent body belonging to the member state in which the product is placed on the market. A success story is, for instance, represented by an enterprise from Hong Kong which was awarded



6.1 Environmental claims classification by ISO 14020.

the Ecolabel for soaps, shampoos and hair conditioners by the French competent body (AFNOR).

6.3 History and goals of the EU Ecolabel scheme

In 1992, a common EU standardised Ecolabel was launched under the Council Regulation (EEC) No. 880/92 of 23 March 1992 on a Community Award Scheme. Since then, amendments have been proposed to increase the effectiveness of the scheme and EC Regulation No. 66/2010 of 25 November 2009 represents the third revision.

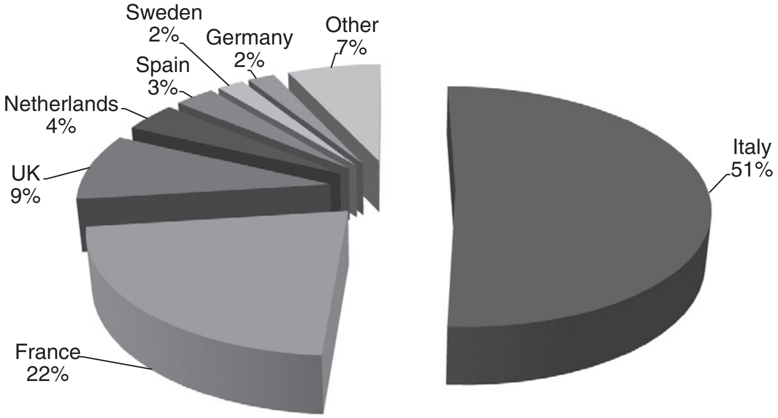
The Regulation establishes a voluntary B2C (business to consumer) scheme intended to:

- promote products and services which have a reduced impact on the environment throughout their life cycle, from the extraction of raw material through production, use and disposal;
- pay particular attention to toxicology;
- consider social and ethical aspects (where relevant);
- obtain widespread agreement on critical points for a single product group and the means of managing them;
- contribute to improving the integration of environmental considerations into markets and provide consumers with better information on the environmental impact of products.

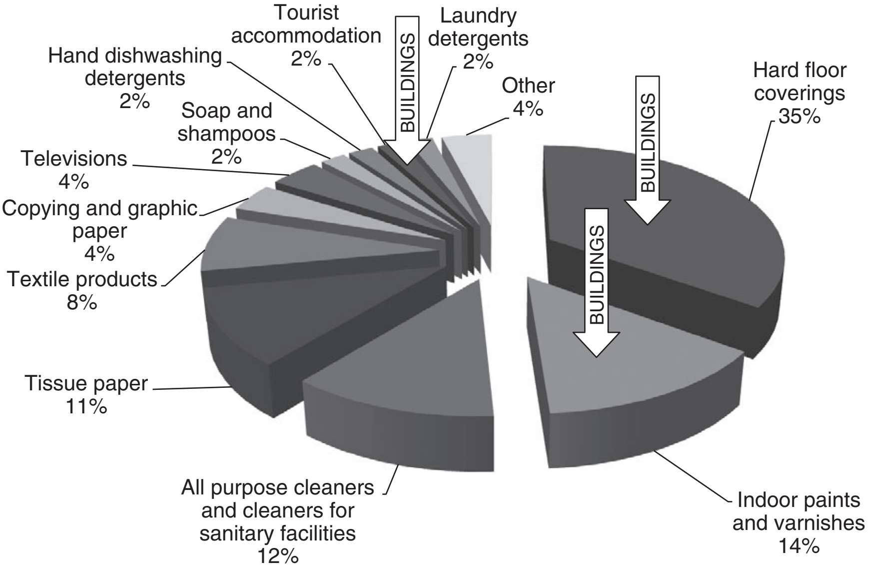
To obtain the EU Ecolabel, a product group must fulfil the following conditions:

- it shall represent a significant volume of sales and trade in the internal EU market;
- it shall involve, at one or more stages of the product's life, a significant environmental impact on a global or regional scale and/or of a general nature;
- it shall present significant potential for effecting environmental improvements through consumer choice as well as an incentive to manufacturers or service providers to seek a competitive advantage by offering products which qualify for the Ecolabel;
- a significant part of its volume shall be sold for final consumption or use.

It should be noted that the criteria for a particular product group are designed to apply to the best 10–20% of products available on the market, based on their environmental performance. These criteria therefore need to be updated and revised on a regular basis to ensure they remain relevant and sufficiently stringent to apply to the above mentioned group of products.



6.2 EU Ecolabelled products issued in Europe as of January 2012.



6.3 EU Ecolabelled products, January 2012 broken up into different product groups. European Commission, Facts and Figures. Available at: http://ec.europa.eu/environment/ecolabel/facts-and-figures.html (accessed 21 November 2012).

During the last five years, the number of licences issued has increased fivefold (from 249 in 2005 to 1,375 in 2011[1]). Italy is the most active European country, having more than 50% of EU Ecolabelled products (Fig. 6.2). Among the EU Ecolabelled products, approximately 50% belongs to the construction sector (Fig. 6.3).

[1] European Commission, Facts and Figures. Available at: http://ec.europa.eu/environment/ecolabel/facts-and-figures.html (accessed 21 November 2012).

6.4 EU Ecolabel establishment procedures and criteria

The assigning of an ecological quality award to a product or a service begins with the links between stakeholders such as Commission plans and GPP goals and the needs of business and manufacturing associations. The initiating activities are usually concerned with the need to involve all potentially interested parties in the dissemination of knowledge so as to build up a team of experts: the so-called 'Ad Hoc Working Group' (AHWG). The AHWG is therefore created to develop a transparent and wide discussion with:

- relevant stakeholders;
- concerned manufacturers;
- consumers;
- environmental associations at the European level.

Its goal is to support the criteria definition by providing technical advice to the whole of the European Union Ecolabelling Board (EUEB). The EUEB consists of the representatives of the competent bodies of all the member states (see below) and other interested parties. In collaboration with the Commission, the EUEB is responsible for developing, revising, publishing and promoting Ecolabel criteria for product groups.

Other organisations which are involved in the EU Ecolabel procedures are:

- *Competent body forum:* in accordance with article 13 of the EU Ecolabel Regulation, the Commission set up a working group composed of representatives of competent bodies to facilitate the exchange of experience and information, particularly in the areas of awarding the EU Ecolabel and market surveillance.
- *Regulatory Committee:* this consists of representatives from the member states and the European Commission. The Commission cannot adopt criteria before voting takes place by qualified majority in the EU Ecolabel Regulatory Committee.

A scheme of all activities arising from the EC's initial decision to develop EU Ecolabel criteria for a new product group to the final publication of ecological criteria in the *Official Journal*, may be summarised as follows (the detailed procedure is also available in Annex I of the Regulation):

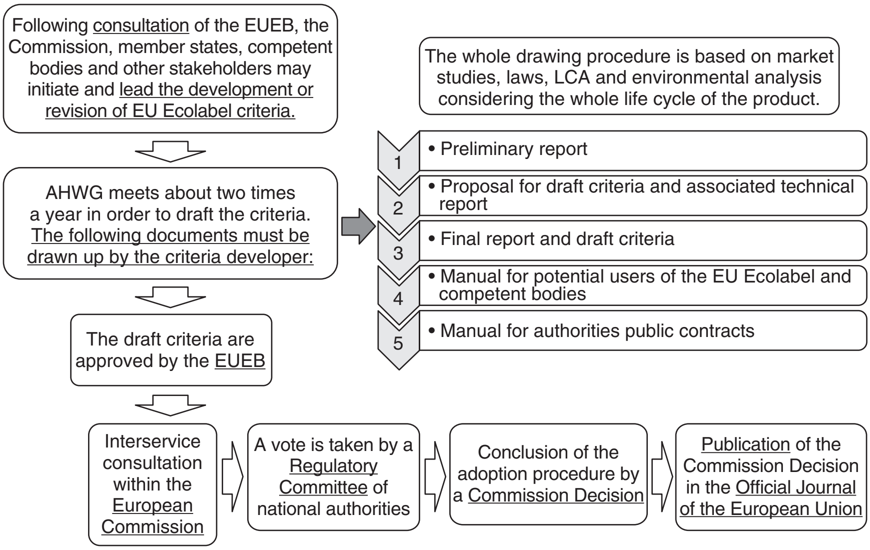
- The Commission co-ordinates the activities in strict co-operation with a leading party. In principle, any interested party may lead the development or revision of EU Ecolabel criteria, providing that common procedural rules are followed and the process is co-ordinated by the Commission. While the leading party is usually a national CB, this work has recently been assigned to the Product Bureau of the Joint Research

Centre (JRC). This is the reference centre of the European Commission's science service which provides customer-driven scientific and technical support for the conception, development, implementation and monitoring of EU policies. The EUEB acts as the main reference and decision-making body, together with the Commission which adopts the final criteria. The last call for tenders supporting work in revising Ecolabel and GPP criteria for six selected product groups was recently launched by the Directorate General Joint Research Centre (DG JRC) through the Institute for Prospective Technological Studies (IPTS).

- The leading party, including JRC, usually nominates technical support with solid LCA experience and expertise and begins with the dissemination of the project among all the stakeholders. The preparatory work includes a feasibility study, environmental and market studies, improvement analysis and revision of the existing life cycle analysis, or implementation of new analysis as necessary. The EU Ecolabel criteria are then established based on these scientific data.
- All CBs in Europe participate in the AHWG meetings and assist in the dissemination of the project in their own countries by providing technical support for the development of the ecological criteria.

Figure 6.4 shows the principal steps of the procedure for EU Ecolabel criteria.

Because the purpose of the EU Ecolabel is the promotion of products with reduced environmental impacts throughout their life cycle, the main



Following consultation of the EUEB, the Commission, member states, competent bodies and other stakeholders may initiate and lead the development or revision of EU Ecolabel criteria.

The whole drawing procedure is based on market studies, laws, LCA and environmental analysis considering the whole life cycle of the product.

AHWG meets about two times a year in order to draft the criteria. The following documents must be drawn up by the criteria developer:

1. Preliminary report
2. Proposal for draft criteria and associated technical report
3. Final report and draft criteria
4. Manual for potential users of the EU Ecolabel and competent bodies
5. Manual for authorities public contracts

The draft criteria are approved by the EUEB

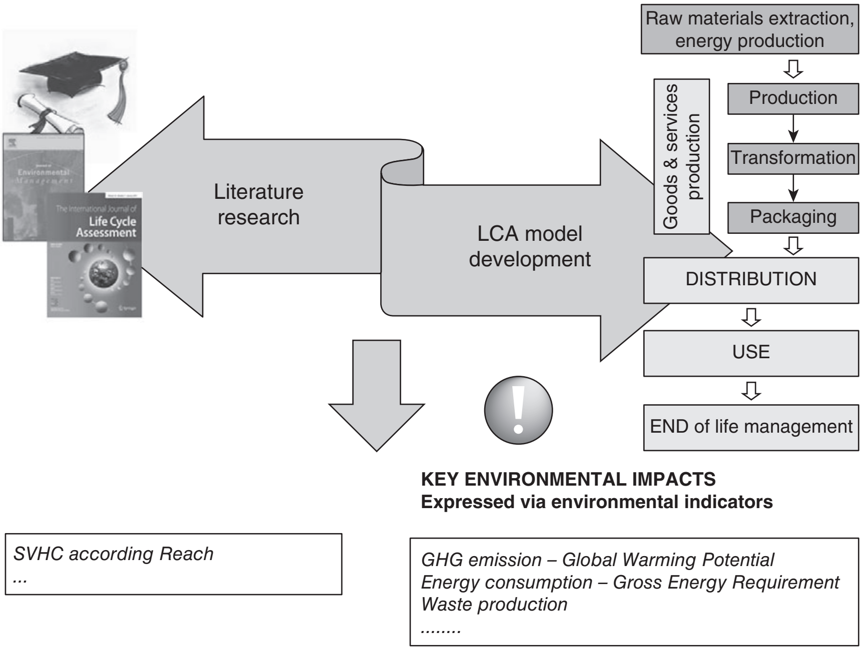
Interservice consultation within the European Commission

A vote is taken by a Regulatory Committee of national authorities

Conclusion of the adoption procedure by a Commission Decision

Publication of the Commission Decision in the Official Journal of the European Union

6.4 EU Ecolabel criteria-making procedure.



6.5 Process to define the key environmental impacts by means of the LCA approach.

contributors to that environmental impact must be identified according to those criteria. The analysis usually starts with bibliographical research and consultation with those stakeholders having specific competencies in the field, including direct data collection along the supply chain and the main processes involved in the product system. The LCA analysis is then used to quantify the product system environmental burden (Fig. 6.5).

Within the EU Ecolabel Regulation, life cycle assessment considerations play a key role in:

- highlighting 'hot spots' over the entire production chain of the products/services concerned, and
- identifying the most appropriate set of ecological criteria for the above mentioned preliminary conditions.

There follows the specific task of covering issues related to human health as well as other social/non-environmental impacts of products not covered by the LCA analysis. Through literature review, stakeholder dialogue and the skills of criteria development leaders, the non-LCA environmental impact indicators are analysed for assessment and discussion of the relevant impacts. Where these are significant, they are taken into account during the criteria development.

6.5 EU Ecolabel and green public procurement (GPP)

The Commission has identified the potential of green public procurement (GPP) as an important instrument for promoting environmentally friendly products and services and encouraging the eco-innovation which contributes to sustainable development. Green public procurement is a process by which public authorities procure goods, services and works which will have a reduced environmental impact throughout their life cycle when compared to others with the same function. It is important to ensure that criteria used by member states are similar to avoid distortion of the single market and a reduction in community competition.

GPP is a voluntary instrument and member states and public authorities may determine the extent to which they implement it. Its use may present problems as some purchasers are unfamiliar with environmental issues and may have difficulties in integrating environmental criteria into the tendering process. It may also prove difficult to verify the fulfilment of environmental requirements provided by tenderers. To overcome this problem, the EU Ecolabel, other 'Type 1' or 'ISO 14024' labels (e.g., the Nordic Swan, The Blue Angel, NF Environment, Milieukeur) and 'single-issue' labels (such as energy efficiency labels) may be used as appropriate instruments to provide the source for identifying environmental criteria and describing them. The criteria can then be directly inserted into tendering documents according to specific guidelines and may be divided into two types, core and comprehensive. The core criteria address the key environmental aspects and are designed to require minimum additional verification or cost increases. The aim of the comprehensive criteria is the purchase of the best available environmental products (European Commission, 2011a). In this context, the added value of the EU Ecolabel is clear:

- The environmental (cut-off) criteria have been developed based on solid scientific evidence and in co-operation with all relevant stakeholders; the Ecolabel gives information about specific limits and 'assessment and verification' of criteria. All analyses and assessments are regulated by European and international approved standard such as EN, ISO, CEN or other certificated systems or regulations.
- EU Ecolabel criteria can be used as GPP criteria and can therefore be inserted directly into green tendering documents.
- Products that meet the environmental criteria are easy to detect as those with EU Ecolabel automatically fulfil the requirements.

At present, there are 19 available GPP criteria, ranging from copying paper to mobile phones. The European Commission has also developed GPP criteria for some product groups related to building materials such as windows, thermal insulation and wall panels.

6.6 EU Ecolabel and national ecolabelling schemes

Both the previous and newer versions of the EU Ecolabel Regulations foresee collaboration between the EU Flower and national ecolabelling schemes for EN ISO 14024 Type I labels. Article 11 of the EU Ecolabel Regulation No. 66/2010 states that 'EU Ecolabel criteria shall also take into account existing criteria developed in officially recognised ecolabelling schemes in the Member States'. This consideration has also been formulated in Annex I of the Regulation where the different procedures for development and revision of EU Ecolabel criteria are stated, as in Procedure B 'Shortened procedure where criteria have been developed by other EN ISO 14024 Type I ecolabelling schemes'.

In practice, it is possible to develop EU Ecolabel criteria taking existing criteria as the basis for a given product group in a national ecolabelling scheme (EN ISO 14024 Type I). This enables a simpler procedure as only one AHWG meeting will be necessary if requested by a member state. The main challenge is to ensure that the criteria are formulated within a national ecolabelling scheme which will be workable and applicable in all member states. The criteria of national ecolabelling schemes are based on specific national standards, industrial characteristics and consumer attitudes which may differ from country to country.

In addition to this specific procedure for developing the criteria, the methodology generally requires reference to existing initiatives, providing examples of workable criteria already in place. For this reason, the existing ecolabelling ISO Type I criteria are considered as possible benchmarks in developing EU Ecolabel criteria.

6.7 EU Ecolabel for eco-efficient construction and building materials

As it is a typical B2C tool, the construction and building materials sector can only be partially affected by the EU Ecolabel. It is likely that a clear interaction with GPP will exist only when business operators are interested in considering the possibility of applying for it. It is therefore important to make information available about the products used in construction and the general construction criteria and management which will minimise environmental impact in terms of resources consumed, emissions and waste produced during the operational life of the structure.

The traceable product and service groups belonging to this sector are:

- coverings (wooden, hard, textiles);
- indoor and outdoor paints and varnishes;
- tourist accommodation;

- office buildings (in progress);
- road construction (only GPP criteria).

An overview of each of the above listed groups is given to provide the following essential elements:

1. Types of products/services considered.
2. The meaning of current criteria and the area of expected improvements.

Details of the definition of product and criteria assessment and verification are reported in the EU Ecolabel Regulations published in the *Official Journal of European Union*. References about these documents can be found below.

There are three other groups which are mostly related to domestic systems:

- heat pumps;
- sanitary tapware;
- toilets and urinals (waiting for publication-voted on 20 June 2013).

These three groups are not analysed in this chapter (see Section 6.9 for links).

6.7.1 Coverings

This group provides criteria for three different product categories (Table 6.1):

- wooden floor coverings (Commission Decision 2010/18/EC);
- hard coverings (Commission Decision 2009/607/EC);
- textile floor coverings (Commission Decision 2009/967/EC).

Criteria have generally been developed according to single specific product characteristics. Even where the typical destination of these products is similar, each is specific because it will be made from different raw materials using different production processes.

A general view of the criteria for 'Coverings' which compares all product categories (wooden, hard floor and textile coverings) is presented in Table 6.2. A commentary on these common criteria is proposed as follows:

- ***Raw materials***: A short explanation of raw material criteria for 'Coverings' is reported in Table 6.3.
- ***Production process* (only for wooden floor coverings and hard coverings)**: A short explanation of production process criteria for 'Coverings' is reported in Table 6.4.
- ***Use phase***: In order to control the release of dangerous substances during the life cycle of coverings, the release of some substances,

Table 6.1 Products considered in the group ‘Coverings’

Wooden floor coverings
- Wood and timber coverings
- Laminate floorings
- Cork coverings
- Bamboo floorings

Hard coverings
- **Natural products** (including marble, granite and other natural stones)
- **Processed products,** divided into:
  - § **Hardened products**
    - o agglomerated stones
    - o concrete paving units
    - o terrazzo tiles
  - § **Fired products**
    - o ceramic tiles
    - o clay tiles

Textile floor coverings
Family of carpets, defined as ‘floor covering’ commonly installed with tacks or staples, or by adhesives. Include
- Woven fabric
- Knitted fabric
- Needle-tufted fabric

Table 6.2 Criteria for ‘Coverings’ category

Wooden floor coverings	Hard coverings	Textile floor coverings
Raw materials	Raw material (extraction)	Raw materials
–	–	Production of all materials
–	Raw material selection	–
–	Finishing operations	–
Use of dangerous substances	–	–
Production process	Production process	–
–	Waste management	–
Use phase	Use phase	Use phase
Packaging	Packaging	–
Fitness for use	Fitness for use	Fitness for use
Consumer information	Consumer information	Consumer information
Information appearing on the Ecolabel	Information appearing on the Ecolabel	Information appearing on the Ecolabel

Table 6.3 Raw materials criteria for 'Coverings' category

Wooden floor coverings
In a sustainability context it is important to evaluate the management of forests, remembering that wood is a not permanent renewable resource.[a] Ecolabel raw material criteria report that wood must originate from forests that are managed in a way to implement the principles and measures according to certified sustainable forest management.
Hard coverings
The raw materials shall comply with some requirements related to extraction activity project and environmental recovery such as authorisation for the extraction activity, environmental recovery plan and/or environmental impact assessment report; a map indicating the location of the quarry, the declaration of conformity to Council Directive 92/43/EEC (1) (habitats) and Council Directive 79/409/EEC. Moreover, only for natural products, EU regulations assess raw material extraction evaluating six indicators (using a matrix) relating to water recycling, land use, natural resource waste, noise, and air/water quality.
Textile floor coverings
Beyond giving advice on risk phrases in raw materials, the regulation imposes restrictions about chemical substances used for fibre treatments (for wool treatments, polyamide fibres, polyester, polypropylene, foam rubber, vulcanised foams) and concentration of formaldehyde.

[a] 'Not permanent renewable resources' means that there is a flux (input flux) comparable with human life, which creates continuously new material. A condition where the input flux is smaller than the extracting flow implies that, in a variable period, the resource may run out.

Table 6.4 Production process criteria for 'Coverings' category

Wooden floor coverings
Limits for process energy consumption, according to product family (wood floor and bamboo coverings, laminate floor coverings, cork coverings) are reported using a score method (algorithms set by the Regulation). Moreover, the Regulation imposes that information about waste management, in terms of type and quantity of waste recovered, disposed or reused shall be reported in a specific report.
Hard coverings
Limits, in terms of specific consumption (MJ/kg) for process energy, according to product family (for agglomerated stones and terrazzo tiles) are reported. For ceramic and clay tiles, a limit for energy used in the firing process is established. Moreover, the Regulation imposes limits for freshwater specific consumption, air emissions and water emission as well as specification for cement use.

Table 6.5 Use phase criteria for 'Coverings' category

Wooden floor coverings
- Volatile organic compounds emissions (TVOC, TSVOC, Total VOC without LCI)
- formaldehyde from panels made of cork, bamboo or wood fibres

Hard coverings
- Lead (Pb) only for glazed tiles
- Cadmium (Cd) only for glazed tiles

Textile floor coverings
- Volatile organic compounds emissions (TVOC, TSVOC, Total VOC without LCI)

Table 6.6 Packaging criteria for 'Coverings' category

Wooden floor coverings
Packaging must be made out of easily recyclable material or materials taken from renewable resources intended to be reusable.

Hard coverings
Paperboard used for the packaging of the final product should be reusable or made out of 70% of recycled materials.

depending on the floor material, is limited by this criterion. Substances released and limited by this criterion are shown in Table 6.5.

- ***Packaging* (only for wooden floor coverings and hard coverings)**: Regulation set criteria for packaging materials are shown in Table 6.6.
- ***Fitness for use***: The product shall be fit for use and this evidence may include data from appropriate ISO, CEN or equivalent test methods. The kind of use shall be clearly specified.
- ***Consumer information***: This part of the Commission Decision shows the information to be reported on the packaging and/or in documentation accompanying the product, such as recommendations for its use and maintenance, indication of the route of recycling or disposal, information on the EU Ecolabel.
- ***Information appearing on the EU Ecolabel***: The last part of the Commission Decision shows which information and data about product hot spots shall be reported in the Box 2 of the Ecolabel. Statements that shall be reported are listed in Table 6.7.

The aim and specific purposes of other criteria will be reported in the following paragraphs.

Wooden floor coverings specific criteria

Wooden floor coverings criteria, not reported for other covering materials, are as follows:

Table 6.7 Information appearing on the Ecolabel for 'Coverings' category

Wooden floor coverings	Hard coverings	Textile floor coverings
Sustainable managed forests and reduced impact on habitats,	Natural products:	
Hazardous substance restricted,	Reduced impact of extraction on habitats and natural resources,	Hazardous and toxic substance restricted,
Production process energy saving,	Limited emission from finishing operations,	Production process energy saving,
	Improved consumer information and waste management.	Limited pollutant emissions to water,
Lower risk to health in the living environment.	Processed products:	Lower risk to health in the living environment.
	Reduced energy consumption of production processes,	
	Reduced emissions to air and water,	
	Improved consumer information and waste management.	

- ***Use of dangerous substances***: The limitation of dangerous substances in raw wood and plant treatments is reported (absence of risk phrases in raw materials and the absence of halogenated organic binding agents, azidirin and polyaziridins, based on lead, cadmium, chrome (VI), mercury and their compounds, arsenic, boron and copper, organic tin). There are limitations on substances used in the coating and surface treatments (chemicals, adhesives, formaldehyde emissions, pesticides, biocides).

Hard coverings specific criteria

Hard coverings criteria, not reported for other covering materials, are as follows:

- ***Raw material selection* (for all hard coverings products)**: In addition to restriction on risk phrases in raw materials, the Commission Decision imposes limitations on the use of some substances such as additives (for glazing and tiles only) and the presence of asbestos and polyester resins in the materials.

- ***Finishing operations*** (**for natural products only**): Natural product finishing operations shall respect some limits in terms of air emissions (particulate and styrene), water emissions (suspended solid and cadmium) and waste recycling.
- ***Waste management***: Production plants shall have a system (documented and explained in the application form) for handling the waste and residual products of production. The system shall include information on procedures for separating and using recyclable materials from the waste stream, procedures for recycling materials for other uses, procedures for handling and disposing of hazardous waste.

Textile floor coverings specific criteria

Textile coverings criteria, not reported for other covering materials are as follows:

- ***Production of all materials***: Restrictions on risk phrases for flame retardants and pesticides are given. There are also limitations on dyes and pigments/composition, water emissions and substances that could affect water emission values. Finally, energy consumption (process energy calculated according to a technical appendix reported in the Commission Decision) shall be smaller than an imposed limit (this criterion is reported for hard covering and wooden covering materials in another section (Production process).

6.7.2 Indoor and outdoor paints and varnishes

Criteria are divided into nine points for both indoor and outdoor products and are shown in Table 6.8.

The general aims of EU Ecolabel criteria for these categories, as listed in the Regulations are:

- the efficient use of the product and the minimisation of waste;
- reducing environmental and other risks (such as tropospheric ozone) by reducing solvent emissions;
- reducing the discharges of toxic or polluting substances into watercourses.

There are set levels for these criteria. Even where criteria names are the same for these products, the limits and specification may be different. Criteria names are shown in Table 6.9.

- ***White pigments***: Commission Decisions limit the content of white inorganic pigments. (The limits are not the same for outdoor and indoor paints and varnishes).

Table 6.8 Product considered in the groups 'Indoor and outdoor paints and varnishes'

Outdoor paints and varnishes – Commission Decision 2009/543/EC (valid until June 2014)

According to EU Commission Decision, the product group '*outdoor paints and varnishes' shall comprise 'outdoor decorative and protective paints and varnishes, wood stains and related products for use on buildings and outdoor furniture, floors and fencing (...), for use by do-it-yourself and professional users; and that are primarily developed for outdoor use and marketed as such. This includes, inter alia, floor coatings and floor paints; products which are tinted by distributors at the request of amateur or professional decorators; tinting systems; decorative paints in liquid or paste formulas which may have been pre-conditioned, tinted or prepared by the manufacturer to meet consumers needs, including wood paints, wood and decking stains, masonry coatings and metal finishes (excluding anti-corrosion finishes and primers) as well as primers (and undercoats) of such product systems (...)*'.

Indoor paints and varnishes – Commission Decision 2009/544/EC (valid until June 2014)

According to EU Commission Decision, the product group '*indoor paints and varnishes' shall comprise 'indoor decorative paints and varnishes, wood stains and related products (...), intended for use by do-it-yourself and professional users and primarily developed for indoor use and marketed as such. This includes, inter alia, floor coatings and floor paints; products which are tinted by distributors at the request of amateur or professional decorators; tinting systems; decorative paints in liquid or paste formulas which may have been preconditioned, tinted or prepared by the manufacturer to meet consumer's needs, including primers and undercoats of such product systems (...)*'.

Table 6.9 Criteria for 'paints and varnishes' category group

Outdoor paints and varnishes	Indoor paints and varnishes
White pigments Titanium dioxide Volatile organic compounds (VOC) Volatile aromatic hydrocarbons (VAH) Heavy metals Dangerous substances Fitness for use Consumer information Information appearing on the Ecolabel	As 'Outdoor paints and varnishes'

- ***Titanium dioxide***: The Commission Decisions limit the emissions of SO_x and the production of sulphate and chloride waste in titanium dioxide pigment manufacturing. (These substances are the same in both outdoor and indoor paints and varnishes but the limits are different).
- ***Volatile organic compounds (VOC)***: Volatile organic compounds shall not exceed fixed values according to product classifications given by Directive 2004/42/EC (limited values and limits may change according to outdoor or indoor use).
- ***Volatile aromatic hydrocarbon (VAH)***: Volatile aromatic hydrocarbons shall not be directly added to the product before or during tinting. Ingredients containing VAH can be used only if the VAH content of the final product does not exceed a fixed value (substance and limits are the same for both outdoor and indoor paints and varnishes).
- ***Heavy metals***: Cadmium, lead, chromium VI, mercury, arsenic, barium (excluding barium sulphate), selenium, antimony, cobalt (excluding cobalt salts) shall not be used as an ingredient of the product or tint (substance and limits are the same for both outdoor and indoor paints and varnishes).
- ***Dangerous substances***: Commission Decisions set rules and limits for some specific matter (see Table 6.10). Rules, substance and limits are also in this case the same for both outdoor and indoor paints and varnishes.
- ***Fitness for use***: As with 'Coverings', these product groups shall meet criteria about their usability. The requirements for fitness for use in

Table 6.10 Matter of analysis for dangerous substances

The product
The product shall not be classified as very toxic, toxic, dangerous to the environment, carcinogenic, toxic for reproduction, harmful, corrosive, mutagenic or irritant.
Ingredients
There are restrictions on risk phrases and risk classification for ingredients.
Alkylphenolethoxylates (APEOs)
APEOS shall not be used in the product before or during tinting.
Isothiazolinone compounds
The content of isothiazolinone compounds is limited by Commission Decisions.
PFAS, PFCA, PFOA
Perfluorinated alkyl sulfonates (PFAS), perfluorinated carboxylic acids (PFCA) including perfluorooctanoic acid (PFOA) and related substances listed in the OECD 'Preliminary lists of PFOS, PFAS, PFOA, PFCA', related compounds and chemicals that may degrade to PFCA are not permitted in the product.
Formaldehyde
Halogenated organic solvents
Phthalates

Table 6.11 Fitness for use criteria for 'Indoor and outdoor paints and varnishes' category

Outdoor paints and varnishes	Indoor paints and varnishes
Spreading rate	Spreading rate
–	Wet scrub resistance (only for indoor use)
Resistance to water	Resistance to water
Adhesion	Adhesion
Abrasion	Abrasion
Weathering	–
Water vapour permeability	–
Liquid water permeability	–
Fungal resistance	–
Crack bridging	–
Alkali resistance	–

paints and varnishes for both indoor and outdoor use are shown in Table 6.11.

- ***Consumer information***: Information that shall be reported on the packaging and/or on documentation accompanying the product is given below. The following information shall come with the product:
  - use, substrate, conditions of use of the product, including advice on preparatory work;
  - recommendations for cleaning tools and waste management in order to limit water pollution;
  - recommendations on storage conditions after opening, including safety advice if appropriate;
  - recommendations on preventive protection measures for the painter;
  - for thick decorative coatings, a text informing that these are paints specially designed to give a three-dimensional decorative effect;
  - text advising that unused paint requires specialist handling for safe environmental disposal;
  - advice on the correct primer or base paint for darker coatings.
- ***Information appearing on the EU Ecolabel***: Information that shall be listed in Box 2 of the EU Ecolabel, is reported in Table 6.12.

6.7.3 Tourist accommodation

This category differs from the groups previously analysed but its design rules are relevant for the construction sector and are reported in Table 6.13. Criteria for the category, which are formally and numerically almost the same as those for Campsite and Tourist accommodation services, set limits for three specific stages of the life cycle of the accommodation service:

Table 6.12 Information appearing on the Ecolabel for 'Indoor and outdoor paints and varnishes' category

Outdoor paints and varnishes	Indoor paints and varnishes
Good performance for outdoor use	Good performance for indoor use
Hazardous substances restricted	Restricted hazardous substances
Low solvent content	Low solvent content

Table 6.13 Services considered in the group 'Tourist accommodation'

Campsite service – Commission Decision 2009/564/EC (valid until November 2015)

'The product group "campsite service" shall comprise, as a main service provided for a fee, the provision of pitches equipped for mobile lodging structures within a defined area. It shall also comprise other accommodation facilities suitable for the provision of shelter to lodgers and collective areas for communal service if they are provided within the defined area'

Tourist accommodation service – Commission Decision 2009/578/EC (valid until November 2015)

'The product group "tourist accommodation service" shall comprise the provision, for a fee, of sheltered overnight accommodation in appropriately equipped rooms, including at least a bed, offered as a main service to tourists, travelers and lodgers. The provision of overnight sheltered accommodation may include the provision of food services, fitness and leisure activities and/or green areas'

- purchasing;
- provision of the service;
- waste management.

In particular, criteria, according to Commission Decisions, aim at:

- limiting energy consumption;
- limiting water consumption;
- limiting waste production;
- promoting the use of renewable resources and of substances which are less hazardous to the environment;
- promoting environmental communication and education.

The criteria set environmental limits in five specific areas:

- energy systems;
- water systems;
- waste management;
- general management of the structure;
- structural materials.

In term of construction materials, the Commission Decisions define criteria and limits for:

- *Energy efficiency of buildings:* The tourist accommodation shall comply with the national legislation and local building codes related to energy efficiency and the energy performance of buildings.
- *Window insulation:* All windows shall have an appropriate degree of thermal insulation according to the local regulations and climatic conditions and shall provide an appropriate degree of acoustic insulation.

6.7.4 Office buildings

Although the EU Ecolabel criteria for office buildings are still under development, discussion of their provisional content according to the most recently circulated document is relevant: 'Final draft proposal for the development of ecological criteria for office buildings' is available on the European Commission website (Boyano Larriba *et al.*, 2012).

The draft criteria define an office building as follows:

> An office building is a building which contains administrative, financial, technical and bureaucratic activities as core representative activities. The office area must make up a vast majority of the total buildings gross area dedicated to this purpose providing a service to other companies or to individuals. Therefore, it could have associated other type of spaces, like meeting rooms, training classes, staff facilities, technical rooms, etc. Excluded from this definition are parking areas that are not counted in this total buildings gross area.

Criteria for the 'Office buildings' category cover the following issues:

- energy consumption;
- material selection and hazardous materials;
- indoor air quality and well-being;
- waste management;
- water management;
- corporate criteria (information to end users).

In focusing on construction products, the draft Commission Decision sets criteria which address the environmental impacts of the production stage, but which also consider other indirect measures which may contribute to reduction of the environmental impact throughout the entire life cycle.

Criteria established and reported in the document are:

- ***Use of construction materials complying with certain environmental criteria***: At least 80% of the cost of major construction elements shall be selected using environmental sustainability criteria. Construction products or construction materials with verified environmental information (Type I or Type III ISO labels) shall be selected.

- ***Material recovery potential of the construction components***: At least 80% in weight of waste generated at the construction phase and the end of the service life of the building shall be prepared for re-use, recycling and other material recovery, including backfilling operations.
- ***Recycled, reused and/or recovered content in the construction products and materials***: At least 50% of the cost of construction components will consist of products and materials containing at least 30% of recycled, reused and/or recovered materials.
- ***Hazardous substances and materials in the construction components***: The draft criteria ban product with some risk phrases and limit the concentrations of hazardous substances in construction materials.
- ***Substances listed in accordance with Article 59 of Regulation (EC) No. 1907/2006***: The work-in-progress criteria impose specific concentration limits for some substances identified as being of 'very high concern'.
- ***Responsible sourcing of construction materials***: At least 80% by value and weight of finishing materials and products used within stairs, windows, external and internal doors, skirting and panelling must be responsibly sourced. These criteria shall not be applied for insulation materials, fixings, adhesives and additives and construction products or materials that account for less than 10% by weight of a key area.

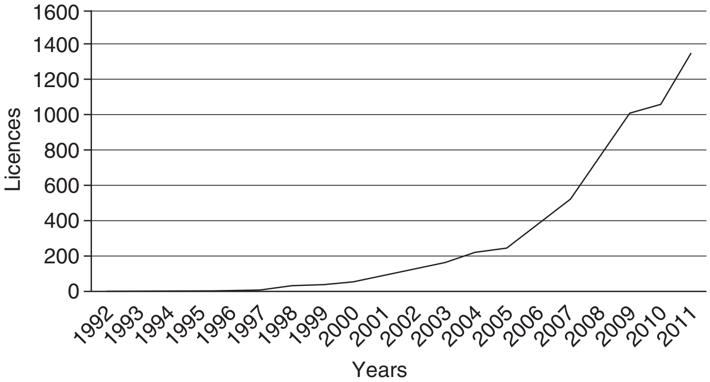
6.8 Future trends

If EU Ecolabelled products are analysed in terms of the number of licences issued per year (Fig. 6.6), a positive growth rate emerges which is expected to continue in the future. This trend is the direct consequence of two factors:

1. Stakeholder and consumer attention to environmental problems is increasing.
2. A wider number of product groups are eligible for the award.

In 2011, the European Commission published a new EU Ecolabel work plan for 2011–2015 which lays out a proposed expansion of the Ecolabelling programme. Its purpose is to set a number of realistic and achievable objectives for the following years. It includes a strategy and a non-exhaustive list of product groups for consideration regarding the priorities for future Ecolabelling. These products include groups which are concerned with construction and building materials:

- hydronic heating systems (already in progress);
- insulation;
- water heating systems.



6.6 Trend of licences issued from 1992 to 2011. European Commission, Facts and Figures. Available at: http://ec.europa.eu/environment/ecolabel/facts-and-figures.html (accessed 21 November 2012).

These are indicated as a 'non-exhaustive list of product groups ranked by environmental priority order for Ecolabelling with priority assessment'. It is likely that windows and wall panels will also be included as the European Commission has developed GPP criteria for these product groups.

In addition to the EU Ecolabel for construction and building materials, the Commission has developed a complementary strategy for building sustainability and has engaged CEN to develop a standard for this assessment. CEN TC 350 'Sustainability of construction works' has been developing a European series of standards for sustainability assessment in terms of:

- environmental performance;
- social performance;
- economic performance.

Both the EU Ecolabel and TC 350 use the LCA as the instrument for environmental assessment and for the provision of reliable indicators on:

- use of natural and secondary resources and waste management (including material recycling and other recovery operations);
- energy use (including use of renewable energy);
- climate change and other environmental impacts on nature.

Although the scope of the two schemes is different, their integration is logical. The EU Ecolabel uses the LCA approach for criteria definition to assess hot spots and for verifying criteria improvements in defining goods and services as environmentally friendly. TC 350 work has been developed on the principle of providing a common EU framework and sound standardisation practices and building assessment calculation. Despite their different goals, both schemes are directed towards the Commission's objective of making zero energy buildings the new European standard and of contributing to a more sustainable society through sustainable building.

The Commission aims to promote the construction sector as a driving force in job creation and at the same time, to prevent trade barriers within the internal market that could be caused by a lack of consistency between the requirements of member states in areas such as public procurement. By the end of 2013, it is expected that a 'Sustainable Buildings Communication' will provide the Commission's vision for bringing together tools such as TC 350, Ecodesign measures, Ecolabel criteria for buildings/construction, GPP or incentives and other private schemes, in order to define a comprehensive building assessment system.

6.9 Sources of further information and advice

Sources of further information are provided here. Additional information can also be found in the documents and websites reported in Section 6.10.

6.9.1 GPP

- European Commission, *GPP criteria.* Available at: http://ec.europa.eu/environment/gpp/gpp_criteria_en.htm (accessed 21 November 2012).
- European Commission, 2011a, Buying *Green!, Green Public Procurement in Europe*. Available at: http://ec.europa.eu/environment/gpp/pdf/buying_green_handbook_en.pdf (accessed 21 November 2012).

6.9.2 Product environmental footprint (PEF)

- European Commission, *Product Environmental Footprint.* Available at: http://ec.europa.eu/environment/eussd/product_footprint.htm (accessed 21 November 2012).
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6.9.3 TC 350

- European Committee for Standardization, *CEN TC 350 Sustainability of construction works*. Available at: http://www.cen.eu/cen/Sectors/

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6.9.5 Life cycle assessment (LCA)

- JRC European Commission – IES, 2010. *ILCD Handbook, General Guide for Life Cycle Assessment – Detailed Guidance.* Available at: http://lct.jrc.ec.europa.eu/pdf-directory/ILCD-Handbook-General-guide-for-LCA-DETAIL-online-12March2010.pdf (accessed 21 November 2012).

6.9.6 GPP and road construction and signs

- European Commission, 2011. *Road Construction and Traffic Signs – Green Public Procurement Product Sheet Other*. Available at: http://ec.europa.eu/environment/gpp/pdf/road_construction_and_traffic_signs_GPP_product_sheet.pdf (accessed 21 November 2012).

6.9.7 Heat pumps criteria

- *Commission Decision of 9 November 2007 establishing the ecological criteria for the award of the Community eco-label to electrically driven, gas driven or gas absorption heat pumps.*

6.9.8 Sanitary tapware criteria

- See: http://ec.europa.eu/environment/ecolabel/products-groups-and-criteria.html

6.9.9 Toilets and urinals (in progress)

- See: http://ec.europa.eu/environment/ecolabel/products-groups-and-criteria.html

6.10 References and further reading

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Commission Decision 2009/578/EC of 9 July 2009 establishing the ecological criteria for the award of the Community Ecolabel for tourist accommodation service.

Commission Decision 2009/607/EC of 9 July 2009 establishing the ecological criteria for the award of the Community Ecolabel to hard coverings.

Commission Decision 2009/967/EC of 30 November 2009 on establishing the ecological criteria for the award of the Community Ecolabel for textile floor coverings.

Commission Decision 2010/18/EC of 26 November 2009 on establishing the ecological criteria for the award of the Community Ecolabel for wooden floor coverings.

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6.11 Appendix: abbreviations

LCA	Life cycle Assessment.
AHWG	'Ad Hoc Working Group', bringing together all stakeholders who participate in the EU Ecolabel criteria development for specific product groups: industry, experts, NGOs, public authorities and other interested parties.
CB	Competent body. Each member state designates a body within government ministries or outside, responsible for carrying out the tasks provided for the EU Ecolabel Regulation and ensures that they are operational. They are in charge of the verification process.
EUEB	European Union Ecolabelling Board, consisting of the representatives of the competent bodies of all the member states.
GPP	Green public procurement, to provide a clear, verifiable, justifiable and ambitious environmental criteria for products and services, based on a life cycle approach and scientific evidence base.
ISO	International Organization for Standardization.
PEF	Product environmental footprint, a multi-criteria measure of the environmental performance of a product or service throughout its life cycle.

7

Environmental product declaration (EPD) labelling of construction and building materials

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Abstract: The aim of this chapter is to present the eco-label Type III: Environmental product declaration (EPD), regulated by the ISO 14025 (2006), and its application to label construction and building materials. The purpose of the EPD is to provide quantified environmental information about the life cycle of a product in order to facilitate environmental comparison between products that perform the same function. There are many different EPD programmes around the world, all of them operating by product category rules (PCR), which set out the specific guidelines and requirements for the development of the LCA study. As a result, environmental indicators for different impact categories through the life cycle of a product are obtained and included in the EPD in addition to other environmental information.

Key words: environmental declaration, ISO 14025, construction and building materials.

7.1 Introduction

The main goal of environmental certification is to promote the demand and supply of those products or services that cause less impact on the environment by communicating accurate and verifiable information. Environmental certification also meets the increasing requirements for information transparency which arise from ever more restrictive legislation on how information has to be communicated to consumers.

Environmental declarations and eco-labels, based on the ISO 14000 series of standards, are the main instruments that can be used to communicate environmental information about a product or service in a reliable, accurate and simplified way. This chapter is focused on Type III environmental declarations, which are regulated by ISO 14025 (2006). This standard sets out the principles and procedures for developing:

- Type III Environmental declaration programmes
- Type III Environmental product declarations

The purpose of environmental declarations is to provide quantified environmental information about the life cycle of a product in order to facilitate environmental comparison between products that perform the same function. These declarations have the following characteristics:

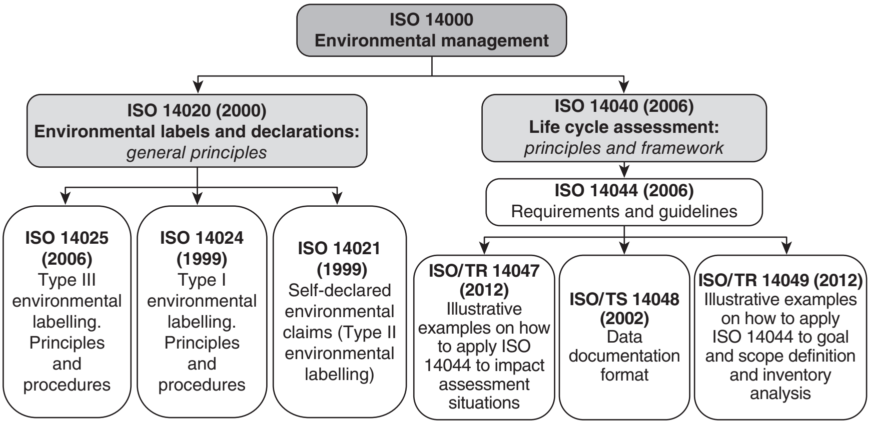
- they provide environmental product information about the entire life cycle of a product or service based on a life cycle assessment (LCA) in accordance with the ISO 14040 (2006) series of standards;
- they are based on independently verified systematic data. This means that a third party should assess the validity and quality of the data with respect to the functional unit and the scope of the underlying LCA study;
- they are presented as a set of indicators related to different impact categories describing the environmental performance of the product;
- they are voluntary processes;
- they are primarily aimed at business-to-business communication, although they can also be used in business-to-consumer communication; and
- they are subject to the management of a programme operator.

As will be discussed below, there are many different environmental declaration programmes around the world. All of them rely on a set of operating rules known as product category rules (PCR), which set out the specific guidelines and requirements for the development of the LCA study; this in turn allows environmental indicators to be obtained for different impact categories. The purpose of an EPD in the construction sector is to provide the basis for assessing buildings and other construction works, and to assist in identifying those construction products which cause less stress to the environment considering the whole building life cycle.

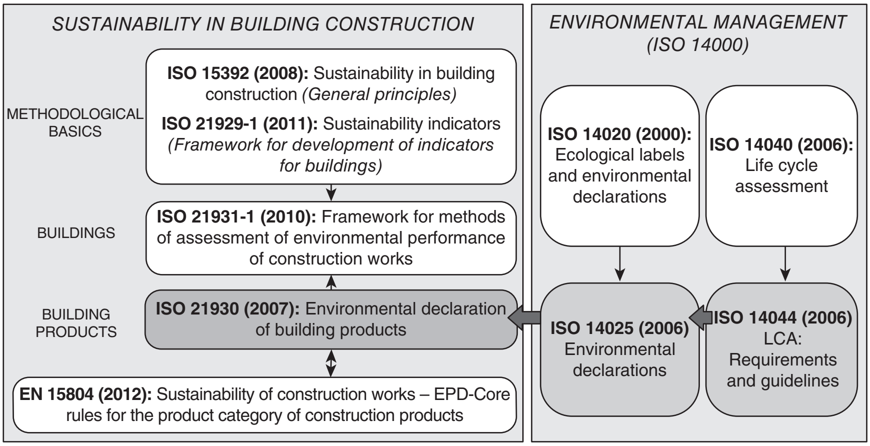
7.2 Regulatory framework

The regulatory framework for Type III environmental declarations is included in the ISO 14000 series of standards, as shown in Fig. 7.1. The ISO 14025 (2006) standard establishes the principles and specifies the procedures for developing Type III environmental declaration programmes and Type III environmental product declarations (EPDs). It specifically establishes the use of the ISO 14040/44 (2006) series of standards to obtain and manage the environmental information needed to develop such declarations.

In the field of construction and building materials, the ISO 21930 (2007) standard complements ISO 14025 (2006), as it provides more specific requirements for the EPD of building products. Moreover, ISO 21930 (2007) provides a framework and the basic requirements for PCR of



7.1 Regulatory framework for environmental certification.



7.2 Regulatory framework for Type III environmental declaration of building products.

building products as defined in ISO 14025 (2006). Unlike the family of standards for evaluating sustainability in buildings, which considers environmental, social and economic aspects of buildings and products, ISO 21930 (2007) focuses on environmental impacts when developing declarations.

The relation between the international standards concerned with EPDs, LCA and sustainability in construction and building materials sector is shown in Fig. 7.2. At the European level, the CEN's Technical Committee for the Sustainability of Construction Works (CEN/TC 350) was set up 'to provide a method for the voluntary delivery of environmental information for construction'. The EN 15804 (2012) is the latest and one of the most significant publications in this field. This standard provides a structure to ensure that EPDs of building materials are presented in a harmonised way across Europe, thus minimising barriers to trade. Beyond that, this standard provides a consistent method to supply environmental information on building products that can then be combined with data from other products to evaluate buildings.

7.3 Objectives and general principles

The main purpose of Type III environmental declarations is to provide quantitative and quality-assured environmental data in order to give information that can be used to make fair comparisons between the environmental performances of products. It must be stressed that Type III EPDs do not provide criteria on the environmental preference of one product over another or minimum requirements to be achieved by the product.

The more detailed objectives of EPD are defined as follows:

- To provide LCA-based information and additional information on the environmental aspects of products over their entire life cycle.
- To provide information about the environmental performance of products and services with objectivity, comparability and credibility.
- To encourage the continuous improvement of environmental performance of products over time.

Therefore, environmental declarations encourage the demand and supply of those products that cause less stress on the environment through the communication of verifiable and accurate information, thereby stimulating the potential market for environmentally enhanced products.

According to ISO 14025 (2006), Type III EPD shall be based on the nine guiding principles listed and defined in Fig. 7.3.

7.4 Environmental product declaration (EPD) methodology

The EPD development process involves the four stages shown in Fig. 7.4. The first step begins with the selection of a Type III environmental declaration Programme and the research of the PCR for the product that is to be declared. If there is no PCR for such product category, it must be created. Next, the EPD draft must be produced based on the application of the LCA methodology. This draft should meet EPD programme rules and the specific PCR for that product category. Finally, a verification process must prove (before EPD publication) that data collection and the application of the LCA methodology are conducted in accordance with the related PCR and meet all ISO requirements.

GUIDELINES/PRINCIPLES OF EPD (ISO 14025 (2006))

Relationship to ISO 14020 (2000) → the principles set out in ISO 14020 (2000) shall be applied when developing an EPD.

Voluntariness → EPD programmes shall be voluntary in nature.

Based on life cycle approach → EPD shall include all environmental aspects from their life cycle. The quantified environmental product information in a Type III environmental declaration shall be based on the results of an LCA in accordance with the ISO 14040 (2006) series of standards.

Information modules → The LCA-based data (inputs and outputs) used to compile the environmental profile in EPD configuration shall be referred to as information modules and shall represent the whole or a portion of the life cycle of the evaluated material.

Openness and consultation → EPD programmes shall implement a formal and open consultation mechanism for the participation of interested parties.

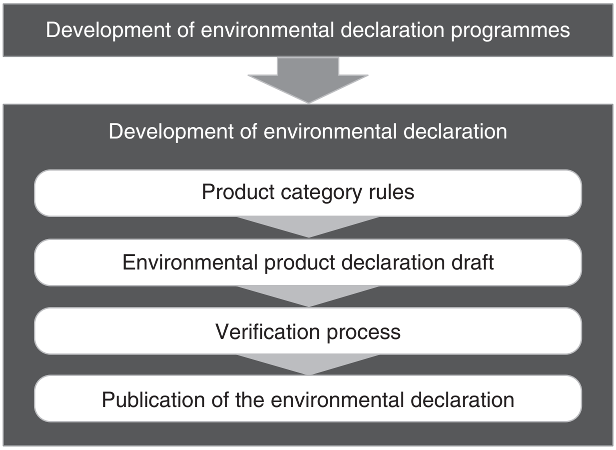
Comparability → EPD results must be comparable in order to allow users to choose products with the best environmental performance. To ensure comparability is achieved, the standard encourages the harmonisation of programmes and the development of mutual recognition agreements.

Verification → In order to ensure the content of EPD is appropriate and verifiable, the programme administrator shall specify the procedures for reviewing the PCR, the EPD programme, the LCA and LCI data and the results from EPD.

Flexibility → EPD shall be flexible, since the contents of EPD can be amended as necessary and as required by the company/organisation after due external review and verification.

Transparency → EPD programmes must be able to demonstrate transparency throughout all stages of their development and operation, thus implying that information shall be made available to interested parties.

7.3 Guiding principles of the Type III EPD following ISO 14025 (2006).



7.4 Stages of the EPD development process.

7.4.1 Development of the EPD programme

The Type III EDPs are voluntary processes which are developed by a programme operator to manage their production. An environmental declaration programme sets the minimum requirements and defines the involvement of interested parties, the independent inspection body and the format of

CONTENT OF OVERALL PROGRAMME INSTRUCTIONS

- Goal and scope of programme
- Programme administration and programme operator committee
- Intended audiences
- Involvement of interested parties
- Rules for selecting and calculating the product categories
- Data and documentation management
- Data confidentiality management
- PCR development, maintenance and verification procedure
- EPD verification procedure
- Periodic review of programme instructions
- Funding sources and programme fees

7.5 Minimum content of instruction programme document.

the declaration in accordance with ISO 14025 (2006). Moreover, it includes procedures for the development of PCR, all in compliance with ISO standards.

The programme operator is a body (or bodies) consisting in the interested party (e.g., company, industrial sector or independent body), which defines specific programme instructions and supervises the EPD process. In particular, programme operators shall ensure the open consultation and the selection of competent independent verifiers, maintain a transparent and readily available library of their published PCR and EPDs, and harmonise these documents between other programmes. Note that the operator may or may not conduct EPD studies.

In order to manage the environmental declaration programmes, programme operators shall prepare a document with overall programme instructions. These instructions, usually called 'programme operator rules', should include at least those detailed in Fig. 7.5.

7.4.2 Product category rules

Product category rules are a set of operative rules and guidelines that are applicable to a product category (group of products that can fulfil equivalent functions) and which describe the requirements and calculation guide for the application of the LCA methodology to obtain environmental indicators, which make up the environmental declaration. The PCR aim to provide the EPD with reliable, consistent and comparable information and to ensure the communication of relevant environmental performance of products and materials to relevant audiences.

PCR DEVELOPMENT PROCESS

- Gather previous PCR and LCAs on the topic
- Get interested parties together (announcing the development of the PCR on the website can lead to contacts with other interested parties)
- Identify the system function and functional unit
- If necessary, develop the LCA in accordance with ISO 14040/44 (2006)
- Write the PCR based on existing LCAs
- Have a panel review the PCR (consultation procedure by interested parties)
- Approval of PCR documents by technical committee (independent experts in LCA from academia and the business sector)

7.6 Stages of PCR development process.

The use of PCR ensures the comparability of LCA-based data from environmental declarations within a product category. To do so, the PCR set the functional unit, the scope, the boundaries, the life cycle inventory data and the impact categories, all of which are needed for the LCA study. PCR configuration shall be performed in accordance with ISO 14040/44 (2006), ISO 14025 (2006), ISO 21930 (2007) and EN 15804 (2012).

In accordance with ISO 14025 (2006), Fig. 7.6 shows the detailed stages that make up the development of PCR. Note that, to develop PCR documents, an open and participatory process must be performed. Interested parties that usually participate in these co-operation meetings are:

- companies and organisations in co-operation with other parties and organisations,
- institutions involving LCA experts in close co-operation with companies or branches and interested organisations, or
- single companies or organisations.

As a result of this process, PCR are obtained for each product category. PCR are valid for a pre-determined period of time as of the date of approval (normally for a period of 3–5 years). The PCR must be reviewed and updated once they have expired. The main content of the PCR is summarised in Fig. 7.7.

7.4.3 EPD draft

Environmental declarations include quantified information about the impact on the quality of the environment produced throughout the life cycle of a product or service. Therefore, such a declaration shall consider all the environmental impacts associated with its entire life cycle from

PCR CONTENT

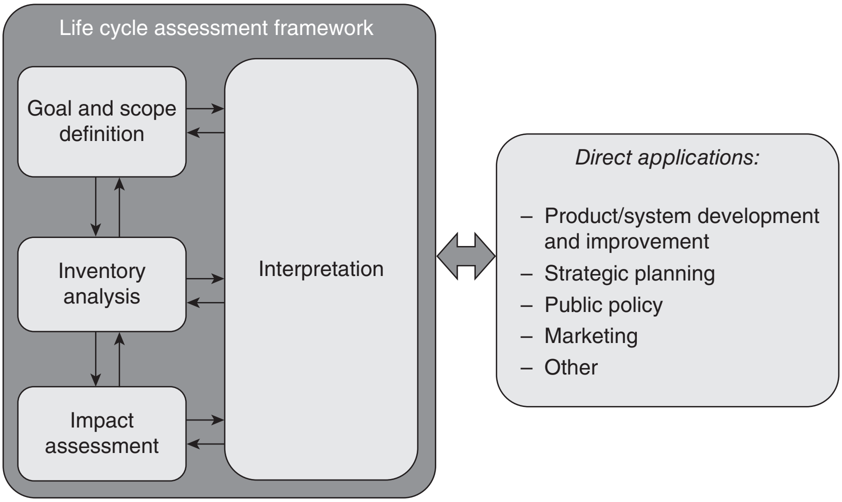
- Product category definition
- Goal and scope definition for the product LCA
    - *Functional unit*
    - *Declared unit*
    - *System boundary*
    - *Description of data*
    - *Units*
    - *Criteria for the inclusion of inputs and outputs*
- Inventory analysis
    - *Data collection*
    - *Calculation procedures*
    - *Allocation of material and energy flows and releases*
- Impact category selection and calculation rules
- Additional environmental information
- Materials and substances for declaration
- Input-data collection instructions
- Format and content of EPD
- Life cycle stages not included in EPD
- Period of validity

7.7 PCR content.

cradle to grave, such as raw material acquisition, energy use and efficiency, content of materials and chemical substances, emissions to air, soil and water, and waste generation. In order to maintain consistency between declarations made by different manufacturers regarding the same product, it is essential to follow the same guidelines established in the PCR corresponding to the product category concerned.

An environmental declaration is focused on obtaining environmental indicators via the application of the LCA methodology, regulated by the ISO 14040-44 (2006) standard, which proposes the stages shown in Fig. 7.8:

I. The definition of the goal and scope of the study.
II. The compilation and quantification of the relevant inputs and outputs of the system (life cycle inventory, LCI).
III. The evaluation of the magnitude and significance of the potential environmental impacts of the system based on the inputs and outputs (life cycle impact assessment, LCIA).
IV. The interpretation of the results, in which the findings are combined according to the goal and scope in order to reach conclusions and recommendations.



7.8 Phases of an LCA according to ISO 14040 (2006).

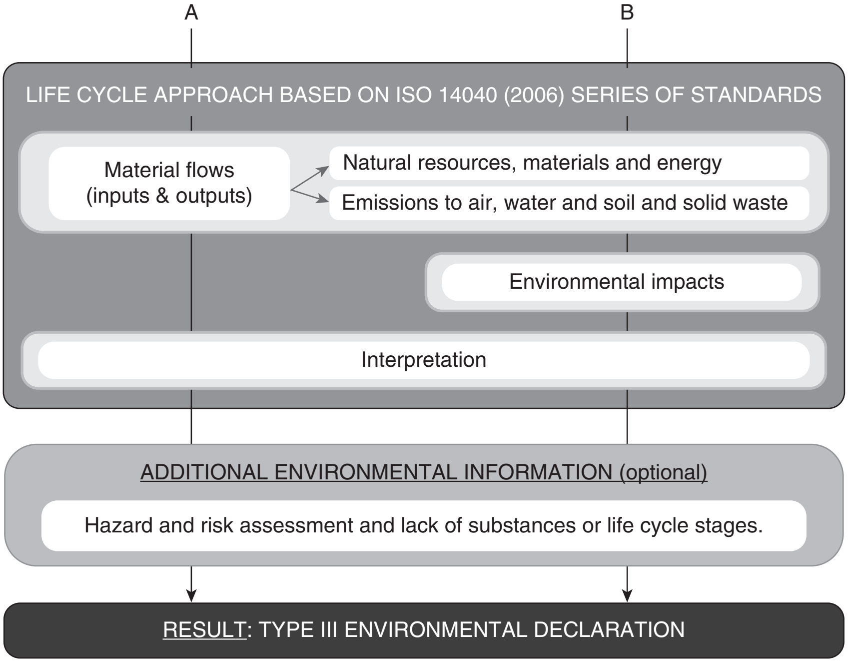
ISO 14025 (2006) proposes the two methodological options shown in Fig. 7.9:

- *Option A*: Life cycle inventory. The EPD results are expressed as an identification of inputs (natural resources, materials and energy) and outputs (emissions to air and water, and solid waste).
- *Option B*: Life cycle impact assessment. The EPD results are expressed by means of environmental indicators for different impact categories (global warming, ozone layer depletion, acidification, eutrophication, etc.), depending on the LCIA method applied.

Results from other analysis tools, which may or may not be derived from the application of the LCA methodology, might be optionally included to provide alternative information that gives a complementary perspective to a Type III environmental declaration. This could include, for example, relevant indicators of sustainable development such as economic or social elements.

7.4.4 Verification process

The purpose of the verification process is to determine whether an environmental declaration is in accordance with related standards (ISO standards, or other internationally recognised standards and regional or national standards), the programme instructions and the PCR for the product category under consideration. The verification procedure should be transparent and conducted by an independent inspection body (internal or external

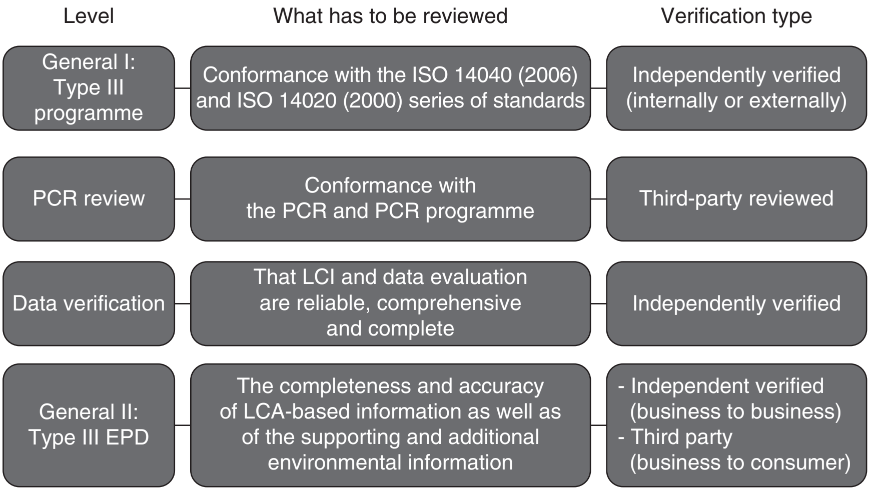


7.9 Methodological options for Type III environmental declarations and programmes.

to the organisation), which was not involved in the development of the EPD.

The independent inspection body shall prepare a verification report, which shall reflect the declaration's compliance with standards and programme requirements, as well as the quality and appropriateness of the information provided in the Type III EPD. Note that EPD data should be valid and scientifically correct. If the programme administrator determines that, according to the verification report, some aspect of the EPD configuration process is inadequate, then the declaration should not be published. Figure 7.10 shows the levels of verification, what is to be verified at each level and the inspection methods that are required in each case.

As Fig. 7.10 states, EPDs are intended for public distribution and should provide a transparent and credible basis for product comparison, based on LCA data. An EPD can be either 'business-to-bussines' (B-to-B) or 'business-to-consumer' (B-to-C), depending on the use (Bergman and Taylor, 2011). Most EPDs are categorized as B-to-B, give LCA information on environmental inputs and outputs up to the end of the manufacturing process and the EPD itself does not need to be third-party verified. However, B-to-C EPDs give LCA information on environmental inputs and outputs on manufacturing and through application in end use and to final



Level	What has to be reviewed	Verification type
General I: Type III programme	Conformance with the ISO 14040 (2006) and ISO 14020 (2000) series of standards	Independently verified (internally or externally)
PCR review	Conformance with the PCR and PCR programme	Third-party reviewed
Data verification	That LCI and data evaluation are reliable, comprehensive and complete	Independently verified
General II: Type III EPD	The completeness and accuracy of LCA-based information as well as of the supporting and additional environmental information	- Independent verified (business to business) - Third party (business to consumer)

7.10 Levels of verification.

disposition after use (such as recycling, valorisation and/or landfill), and the EPD itself must be third-party verified.

7.4.5 Publication of the EPD

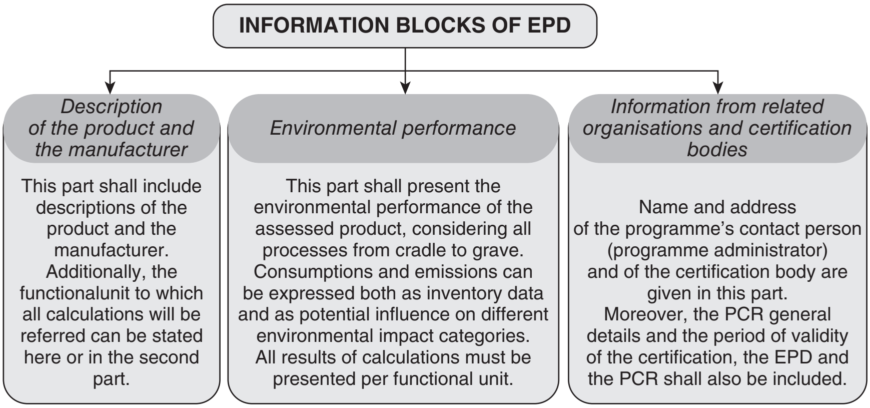
Once the environmental declaration has passed the verification process, it is sent for publication in the corresponding EPD programme. Regardless of the programme, the EPD has to include the three main blocks of information, which are detailed in Fig. 7.11. Additionally, the EPD shall be accompanied by the following information:

- information which demonstrates that the establishment of the functional unit, the pre-set categories of parameters and the product-specific requirements match the scope, principles and procedures set out in the related international standards;
- the methods to verify declared information; and
- information which demonstrates that interested parties participated in the process and that their views were taken into consideration.

In order to include all the information that makes up an EPD, a possible index of EPD content, based on recommendations in international standards, is proposed in Fig. 7.12.

7.5 EPD programmes around the world

Nowadays, there are many Type III EPD programmes around the world. Each programme develops its own PCR, which serve as the basis for the



This part shall include descriptions of the product and the manufacturer. Additionally, the functionalunit to which all calculations will be referred can be stated here or in the second part.

This part shall present the environmental performance of the assessed product, considering all processes from cradle to grave. Consumptions and emissions can be expressed both as inventory data and as potential influence on different environmental impact categories. All results of calculations must be presented per functional unit.

Name and address of the programme's contact person (programme administrator) and of the certification body are given in this part. Moreover, the PCR general details and the period of validity of the certification, the EPD and the PCR shall also be included.

7.11 Information in an EPD.

TYPE III ENVIRONMENTAL DECLARATION CONTENT

1. Producer details, including reference for further information
2. Product description and content declaration
3. EPD publication date and period of validity
4. The name of the EPD progamme and programme administrator/operator
5. References to the PCR, including PCR publication date and period of validity
6. Life cycle-based data (LCI and LCA data):
    Life cycle inventory data (LCI):
        Inputs: raw material, energy, water, etc.
        Outputs: water, air and soil emissions
    Impact categories data (LCIA):
        Global warming
        Ozone layer depletion
        Acidification
        Eutrophication
        Photochemical oxidation
        Abiotic depletion
    Other:
        The omitted life cycle stages
7. Additional environmental information
8. Standard comments on comparability
9. Information about verification process and body:
    Information on the review of the PCR and the agency responsible for carrying it out.
    Name and organisation of the coordinator and contact information.
    Independent verification of the declaration and data, according to ISO 14025 (2006).
    Third party verifier (optional for business-business communication and compulsory for business-consumer communication).
10. References to further information

7.12 Type III EPD content.

preparation of the EPDs for each product category, depending on their market demands. Table 7.1 lists some of the most important EPD programmes from around the world, and includes interesting information such as the programme operator, the number of PCR documents and licences developed since they were launched, and the countries where the programme is recognised. Links to the official websites are also included.

As shown in Table 7.1, the KEITI-EPD programme from Korea along with the IBU-EPD programme from Germany and The International EPD® System from Sweden are the ones with the largest number of products that have already been certified. The International EPD® System is likewise the programme which has created the most PCR since its launch. It is worth mentioning that many EPD programmes are based on this Swedish system, which is the most widely accepted around the world, since it is recognised in many countries from both inside and outside Europe.

Note that most of the EPD programmes include some PCR for the construction and building materials product categories. Furthermore, some programmes, such as DAPc from Spain, IBU-EPD from Germany or BRE from United Kingdom, have been created exclusively for the certification of construction and building products and/or building systems. To know which products and materials are considered in the PCR for the construction sector, the exhaustive analysis presented in the next section was carried out.

7.6 Product category rules (PCR) for construction and building materials

To find out which product categories related to the construction sector have already developed product category rules, a review of the PCR developed by the main international EPD programmes in relation to construction and building materials is discussed below. It is important to point out that each EPD programme arranges and classifies the PCR differently. For example, the IBU-EPD programme sets out the PCR related to construction and building materials in three blocks: 'Basic materials and precursors', 'Building products' and 'Building services engineering'. The International EPD® System, on the other hand, has only four PCR in the 'Constructions, construction products and construction services' category, but 33 building and construction materials are registered in other categories such as 'Metals' or 'Ores, minerals, stones', and so forth.

The PCR currently in force that are related to construction and building materials and which were developed by the international programmes listed in Table 7.1 are presented in Table 7.2. As can be seen from Table 7.2, some product categories have a greater number of developed PCR. In other words, there are some categories which have been regarded by more

Table 7.1 Examples of Type III environmental product declaration programmes in different countries

Name and logo	Year	Country	Number of		System operator/type		Where it is found	Link
			PCRs	EPDs				
DAPc	2008	Spain	2	12	Dtp. de Medio Ambiente e Instituto de la Construcción de la Generalitat de Cataluña	Government	Spain	http://csostenible.net/sistema_dapc/
The International EPD® System	1999	Sweden	212	249	International EPD Consortium	Government	Belgium, Greece, Italy, Netherlands, Sweden, Switzerland, Taiwan and United Kingdom	http://www.environdec.com/
ECO LEAF	2002	Japan	78	180	Japan Environmental Management Association for Industry (JEMAI)	Non-profit	Japan	www.cfp-japan.jp
KEITI-EPD	1997	Korea	30	277	Korean Environmental Labelling Association (KELA)	Certification organisation	Korea	www.keiti.re.kr
EPD-NORGE	1989	Norway	17	112	Norwegian EPD foundation	Government	Norway	www.epd-norge.no

MRPI	2000	The Netherlands	(not available)	88	MRPI-bureau	Certification organisation	The Netherlands	http://www.mrpi.nl/
IBU-EPD	2004	Germany	69	280	IBU – Institute Construction and Environment e.V.	Non-profit	Germany, Sweden and Austria	www.bau-umwelt.de
BRE	2001	United Kingdom	1	42	BRE GLOBAL	Non-profit	Czech Republic, Denmark, France, Netherlands, UK and USA	www.bre.co.uk
MVD	2009	Denmark	7	(not available)	MVD-secretariat at Dansk	Government	Denmark	www.mvd.dk
EDF-EPD	1997	Taiwan	51	(not available)	Environment and Development Foundation (EDF)	Non-profit	Taiwan	http://www.edf.org.tw/

Table 7.2 Building and construction products which have product category rules in force[a]

PCR	EPD programmes							
	ECO LEAF	IBU	EPD©	EPD-Norge	KEITI-EPD	EDF-EPD	DAPc	MVD
Electricity (wires, cables, circuit breakers, disconnectors, etc.)	5		8		1	1		1
Insulation materials (thermal and acoustic)	1	5	1	1			1	
Aggregates (stones, clay, sand and asphalt)	2	2	2	1				
Cement		1	1					
Other textile materials and precursors (synthetic carpets, textile floor tiles, etc.)	2	2				1		
Products related to concrete, mortar and grout		1	1	1				
Masonry and related products (bricks, tiles, etc.)		4	2					
Construction adhesives and coatings, including corrosion protection		3	1					
Paints and varnishes			2					
Doors, windows, shutters, and related products		3	1	1				1
Façades		8						
Fixings (wall plugs made of plastic and metal)		1						
Flat glass, profiled glass and other glass products		1				1		
Wall and ceiling finishes (fibre cement, fibre concrete and mineral panels)		2		1				
Wall and ceiling finishes (glass mesh, coverings and lightweight boards)		3	1					
Wall finishes and floor coverings (ceramic tiles and vinyl tiles)	1	2				1	1	

Membranes and kits (waterproofing, etc.)		3		1				
Metallic products (structural steels, aluminiums and building metals)		4	1	1		3		
Normal/lightweight/autoclaved aerated concrete products		5						
Roof coverings		7	2					1
Room divider systems		1						
Interior facilities (laminates)		1						
Plastic products (plastic boards)	2	1	1					
Gypsum products (plasterboard)		1						
Structural timber products (fibreboard)		1	2					
Wood panels and products		2		1				
Products for artificial lighting		1						
Sanitary facilities (toilets, bidets, taps, etc.)	4	3	2		1			
Ducts, pipes, etc.			2					
Central heating boilers and water heaters			1					
Air conditioners/refrigerators			2		1	1		
Mechanical equipment for buildings			1	1				
Highways, streets and roads			1					
Railways			1					
Chemical products for building			1	1				
	17	**68**	**37**	**10**	**3**	**8**	**2**	**3**

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[a] The BRE programme from United Kingdom and the MRPI programme from The Netherlands have been excluded from the analysis presented here. The reasons for this omission are, on the one hand, that the British programme has only one generic PCR called ‘construction products’ and, on the other, the Dutch programme does not make its PCR available to the general public.

programmes when developing their PCR. These categories are: electric components (wires, cables, etc.), insulation materials, metallic products, roof coverings and sanitary facilities, which include elements ranging from taps to toilets. From Table 7.2 we can also see which programmes have a greater number of PCR related to building and construction materials. These programmes are, in descending order, IBU-EPD from Germany (68 PCR), the International EPD® System from Sweden (37 PCR) and the ECOLEAF from Japan (17 PCR).

7.7 Case studies: EPD for construction and building materials

In the following, the methodology described in the previous stages has been applied to different construction and building materials in order to show the results provided by different EPD. Specifically, the next subsections show the set of operative rules and guidelines included in the PCR and the EPD results (environmental indicators) for construction materials such as concrete, thermal insulation materials and wood boards. Note that in order to preserve the privacy of the manufacturers of the analysed products, the names of the companies have been omitted.

7.7.1 Concrete

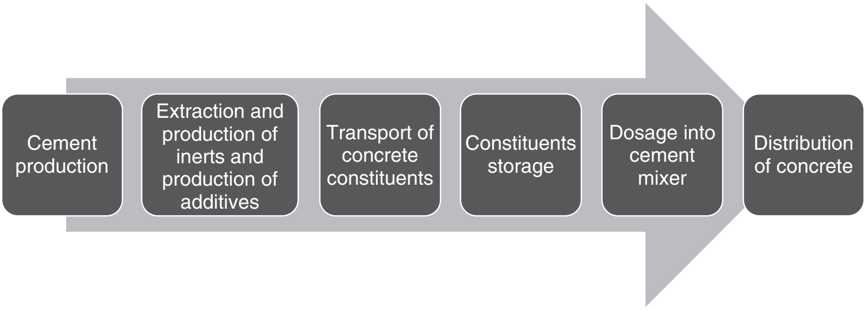
This subsection presents two EPDs for concrete, which are obtained from different EPD programmes. One of them is certified by The International EPD® System and the other by the EPD-Norway programme. The main characteristics of both declarations, as well as the main environmental results included in each EPD, are shown in Table 7.3. The methodological basis for both EPDs is based on the document PCR 2005:07 (2005). This document describes the scope and goal of the LCA and sets the parameters for the assessment of the environmental performance needed for the development of an EPD for the product group 'Concrete'.

According to PCR 2005:07 (2005), the unit processes that were considered to obtain the EPD results, which are shown in Table 7.3, are those illustrated in Fig. 7.13. Note that the use and the end of life of concrete are not included in the scope of the LCA study. That is to say that the cradle-to-gate stage must be included in any EPD, and should remain the same for a given manufacturing location, irrespective of where the product is used. The transport to construction site and construction stage modules may vary depending on where the product is delivered and how it is used in the building or construction works. For this reason, this information based on a specific scenario is optional.

Table 7.3 EPD for concrete: The International EPD® System vs EPD-Norway

	EPD®	EPD-Norway
Description of analysed product in each EPD programme	Concrete mixture formed by mixing cement, coarse and fine aggregates and water, with or without the addition of additives and recycled materials such as fly ash.	Concrete mixture
Validity of the declaration	19/11/2007–19/11/2010	26/01/2010–26/01/2013
Functional unit	The functional unit consists of 1 m^3 of concrete. The environmental performance must be for that quantity of the product.	

	Environmental indicators			
	EPD®		EPD-Norway	
Global warming potential	2.31E02	kg of CO_2 eq.	2.12E02	kg of CO_2 eq.
Ozone layer depletion potential	3.00E-06	kg of CFC_{11}eq.	0.00E00	kg of CFC_{11}eq.
Acidification potential	7.80E-01	kg of SO_2 eq.	3.67E-01	kg of SO_2 eq.
Eutrophication potential	9.00E-02	kg of PO_4^{3}eq.	5.00E-02	kg of PO_4^{2-}eq.
Photochemical oxidation potential	6.00E-02	kg of C_2H_4 eq.	2.60E-02	kg of C_2H_2eq.
Primary energy, non-renewable	1800.71	MJ	705	MJ
Primary energy, renewable	78.27	MJ	663	MJ



7.13 System boundaries of the LCA of concrete.

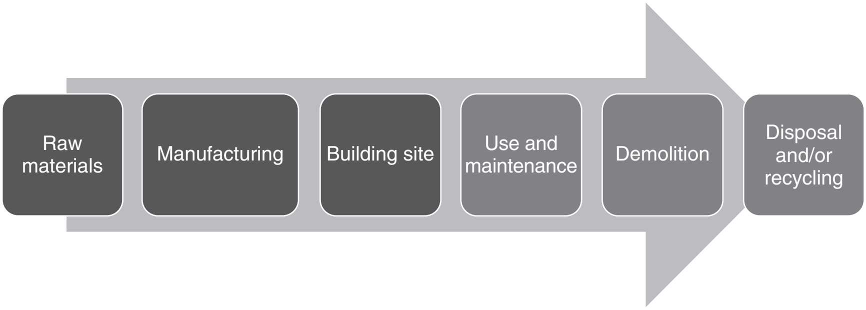
7.7.2 Thermal insulation materials

Below, three EPDs for thermal insulation materials, which are obtained from different EPD programmes, are presented. The first one is certified by DAPc programme, the second one by the EPD-Norway programme and the last one by IBU-EPD programme. Again, the main characteristics of such declarations, as well as the main environmental results included in each EPD, are summarised in Table 7.4. In this case, the methodological basis for each one of the analysed EPDs is based on its own PCR document: the DAPc programme is based on the PCR-001 (2010) for 'thermal insulation products', the EPD-Norway on the NPCR-012 (2007) for 'insulation materials', and the IBU-EPD programme on the PCR (2007) for 'mineral insulating'. The three PCR documents describe the scope, the goal and the main parameters for the LCA study needed for the development of each EPD.

The unit processes which are considered by all programmes when obtaining the EPD results are the raw material extraction and the manufacturing phase. DAPc also considers the installation on the building site, including the transport of the materials to the workplace. The complete life cycle of thermal insulating materials is illustrated in Fig. 7.14 and the environmental indicators included in each EPD in Table 7.4.

7.7.3 Wood boards

The EPD results for three kinds of wood fibreboards and three of wood particle boards which were obtained from different EPD programmes are shown in Tables 7.5 and 7.6, respectively. Two of the declarations are certified by the IBU-EPD programme and the other ones by The International EPD® System. The main characteristics of such declarations, as well as the main environmental results included in each EPD, are summarised in Tables 7.5 and 7.6.



7.14 System boundaries of the LCA of thermal insulation materials.

Table 7.4 EPD for thermal insulation materials: DAPc vs EPD-Norway vs IBU-EPD

	DAPc	EPD-Norway	IBU-EPD
Description of analysed product in each EPD programme	Compact non-hydrophilic panel of glass wool, uncoated, 50 mm thick, 1350 mm long, 600 mm wide and 18 kg/m^3 of density	Glass wool panel produced from recycled glass and used to insulate walls, ceilings and floors. The glass wool is elastic and has a thickness of 37 mm/m^2 with a heat resistance of 1 m^{2}K/W.	Glass wool insulation panel with synthetic crystalline mineral fibres. The structure consists of a silicate melt. The average fibre diameter is 3 to 6 microns. The fibre length can be up to few centimetres.
Validity of the declaration	28/10/2010–28/10/2015	28/08/2007–26/08/2012	04/08/2008–04/08/2011
Functional unit	1 m^2 of thermal insulation material. The environmental performance must be for that quantity of the product.		

Environmental indicators

	DAPc		EPD-Norway		IBU-EPD	
Global warming potential	2.60E+00	kg of CO_2 eq.	5.94E-01	kg of CO_2 eq.	1.77E+00	kg of CO_2 eq.
Ozone layer depletion potential	1.81E-07	kg of CFC_{11}eq.	6.39E-12	kg of R11eq.	8.86E-08	kg of R11eq.
Acidification potential	1.68E-02	kg of SO_2 eq.	1.29E-03	kg of SO_2 eq.	6.70E-03	kg of SO_2 eq.
Eutrophication potential	2.34E-03	kg of PO_4^3eq.	1.66E-04	kg of PO_4^3eq.	1.10E-03	kg of PO_4^3eq.
Abiotic resource depletion potential	1.99E-02	kg of Sb eq.	–		–	
Photochemical oxidation potential	1.52E-03	kg of C_2H_6 eq.	1.50E-04	kg of C_2H_4eq.	3.40E-04	kg of C_2H_4eq.
Primary energy, non-renewable	48	MJ	10.5	MJ	28.76	MJ
Primary energy, renewable	4.98	MJ	–	MJ	1.34	MJ

Table 7.5 EPD for wood fibreboard: IBU-EPD vs The International EPD® System

	IBU-EPD	EPD®$_1$	EPD®$_2$
Description of analysed product in each EPD programme	Resin board, three layers of wooden panels elongated oriented strand, called micro-filaments. The plates are bonded with polyurethane resin. The panels are manufactured in thicknesses of 6-40 mm; the density of the plates is about 614 kg/m^3.	The product is plain Medium Density Fibreboard with an average density of 840 kg/m^3. The panels are manufactured from lignocellulose fibres obtained from carefully selected wood, bonded together with synthetic resins under pressure at high temperatures.	The product is melamine-coated Medium Density Fibreboard with an average density of 840 kg/m^3. The panels are manufactured from lignocellulose fibres obtained from carefully selected wood, bonded together with synthetic resins under pressure at high temperatures.
Validity of the declaration	08/12/2008–08/12/2009	15/12/2010–17/12/2013	15/12/2010–17/12/2013
Functional unit	1 m^3 of wood board	1 m^3 of plain fibreboard	1 m^2 of melamine-coated fibreboard
	The environmental performance must be for that quantity of the product.		

Environmental indicators

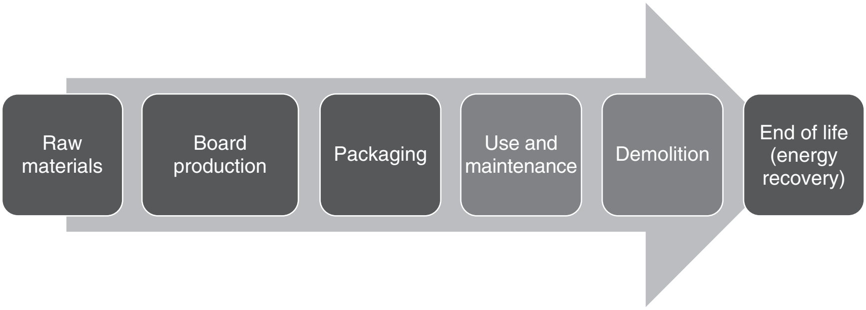
	IBU-EPD		EPD®$_1$		EPD®$_2$	
Global warming potential	–5.38 E02	kg de CO_2 eq.	–8.18E02	kg of CO_2 eq.	–3.48 E00	kg of CO_2 eq.
Ozone layer depletion potential	–7.59E-06	kg de R11 eq.	4.30E-05	kg of R11eq.	1.90E-07	kg of R11eq.
Acidification potential	1.10E+00	kg de SO_2 eq.	4.68E-00	kg of SO_2 eq.	2.07E-02	kg of SO_2 eq.
Eutrophication potential	1.80E-01	kg de PO_4^3 eq.	3.35E-01	kg of PO_4^3eq.	1.50E-03	kg of PO_4^3eq.
Photochemical oxidation potential	9.59E-02	kg de C_2H_4 eq.	6.21E-01	kg of C_2H_4eq.	2.70E-03	kg of C_2H_4eq.
Primary energy, non-renewable	–7,651	MJ	11,044	MJ	51.37	MJ
Primary energy, renewable	12,564	MJ	4,919	MJ	22.63	MJ
Electricity consumption	–		501	kWh	2.25	kWh

Table 7.6 EPD for wood particle board: IBU-EPD vs The International EPD® System

	IBU-EPD	EPD®$_3$	EPD®$_4$
Description of analysed product in each EPD programme	Particle board is a panel-shaped wood-based material, manufactured in a flat-pressing process by means of compression under heat of small wood particles with adhesive. The gross density of uncoated particle board is 655 kg/m^3.	The product is plain particle board with an average density of 700 kg/m^3. The panels are manufactured from lignocellulose fibres obtained from carefully selected wood, bonded together with synthetic resins under pressure at high temperatures.	The product is melamine-coated particle board with an average density of 700 kg/m^3. The panels are manufactured from lignocellulose fibres obtained from carefully selected wood, bonded together with synthetic resins under pressure at high temperatures.
Validity of the declaration	25/03/2012–25/03/2013	17/12/2010–17/12/2013	17/12/2010–17/12/2013
Functional unit	1 m^3 of uncoated particle board	1 m^3 of plain particle board	1 m^2 of melamine-coated particle board
	The environmental performance must be for that quantity of the product.		

Environmental indicators

	IBU-EPD		EPD®$_3$		EPD®$_4$	
Global warming potential	–5.46 E02	kg de CO_2 eq.	–9.10E02	kg of CO_2 eq.	–1.44 E01	kg of CO_2 eq.
Ozone layer depletion potential	–1.45E-05	kg de R11 eq.	2.90E-05	kg of R11eq.	4.90E-07	kg of R11eq.
Acidification potential	1.33E+00	kg de SO_2 eq.	5.44E-00	kg of SO_2 eq.	9.02E-02	kg of SO_2 eq.
Eutrophication potential	2.91E-01	kg de PO_4^3 eq.	8.60E-01	kg of PO_4^3eq.	1.49E-02	kg of PO_4^3eq.
Photochemical oxidation potential	1.48E-01	kg de C_2H_4 eq.	4.30E-01	kg of C_2H_4eq.	7.49E-03	kg of C_2H_4eq.
Primary energy, non-renewable	–8,482	MJ	6,877	MJ	136.64	MJ
Primary energy, renewable	12,977	MJ	1,800	MJ	35.53	MJ
Electricity consumption	–		183	kWh	3.27	kWh



7.15 System boundaries of the LCA of wood boards.

In this case, the methodological basis for each of the analysed EPD is based on its own PCR document: on the one hand, those EPDs from The International EPD® System are based on the PCR 2003:8, version 1.0, for 'Wood particleboard', although PCR for 'Fibreboard and particle board of wood or other ligneous materials' is currently being developed by The International EPD® System and its publication is pending. On the other hand, the IBU-EPD programme is based on the PCR-01 (2009) for 'Wood materials'.

Both PCR documents describe the scope, the goal and the main parameters for the LCA study needed for the development of each EPD. The unit processes which are considered by both programmes when obtaining the LCA results are the production phase including from extraction of the raw material to the packaged product at the factory gate, and the end of life which is assumed to be energy recovery at a biomass plant. The stage related to the use of boards is excluded in the present declarations. The complete life cycle of wood boards is illustrated in Fig. 7.15.

For this product category, the PCR developed by The International EPD® System states that it could be interesting to include in the EPD some additional environmental information regarding:

- the energetic content of the particle board (energetic declaration), and
- the recycling properties of the product (recycling declaration), including information about the dismantling of products and reuse, suitable procedures for recovery of selected parts of the entire products, method for reuse of the product or the proper handling of the product as waste at the end of its life cycle.

7.8 Conclusions

This chapter has focused on one of the environmental certification tools that the construction sector can apply to communicate the environmental performance of its products: environmental declaration, which is regulated by ISO 14025 (2006). To date, several environmental declaration programmes have developed PCR for different categories of materials/products related to the construction sector, which gives an idea of the extent to which it is involved in promoting and communicating the environmental performance of their materials and products.

Environmental declarations can be used in different ways by the stakeholders involved in the construction supply chain:

- Designers: criteria for selecting a material according to the environmental profile of the EPD.
- Manufacturers: communication, in a credible and transparent way, of the environmental performance of the products.
- Purchasing and procurement: verification of the required specifications related to environmental performance.
- End users: criteria to purchase based on the environmental information provided by the EPD.

The intended users of EPD are professionals of different kinds, not necessarily knowledgeable about environmental issues and some efforts have been taken in order to facilitate understanding by developing interpretation keys of certified environmental declarations (Steen *et al*., 2008) and increasing its use both for B-to-B and B-to-C communication.

Finally, it is important to note that public procurement can be an instrument to extend the use of EPD. Registered environmental declaration by any of the EPD programmes described in this chapter proves that certain environmental standards are fulfilled. In public procurement it is possible to ask for goods and services to have this information. However, any procurement criteria need to be in line with the information that can be provided by a certified environmental declaration.

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8

Shortcomings of eco-labelling of construction and building materials

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Abstract: In recent years, there has been a proliferation of eco-labels of differing kinds, resulting in customer fatigue. This chapter discusses the common shortcomings of different eco-labels, especially in the context of construction materials. From the same context, the four most important shortcomings of an eco-label are identified, demonstrating that an eco-label must be comprehensive, clear, credible and customer focused. Further, with the proliferation of eco-labels, clarity is the most important attribute. Eco-labels of important construction materials are discussed.

Key words: shortcomings, eco-labels, construction materials, cement, steel, paints.

8.1 Introduction

Eco-labelling refers to the process of assigning ecological labels containing sustainability information about the environmental friendliness of a product (which may be goods or services). The purpose of eco-labelling is to guide customers of the product (intermediate or ultimate) to make environmentally conscious decisions when considering the product for purchase. The approach was initiated by Germany's Blue Angels Eco-Labelling programme in 1978 (The Blue Angel, 2012) and since then, a deluge of non-governmental organisations (NGOs) that perform eco-labelling have come into existence. A summary of European Union eco-labels is given in Chapter 6. According to the World Resources Institute and Big Room (2012), there are more than 435 eco-labelling organisations in 197 countries in over 25 business sectors. In a survey conducted by Big Room in 2010, it was found that eco-labels were distributed in the ratio 11:45:1:43 in Asia-Pacific, Europe, Latin America, and North America, respectively (WRI and Big Room, 2010). An abundance of such programmes testifies to an inclination from consumers to make environmentally conscious decisions. In fact, it has been shown in surveys (Palin, 2012; Lathrop and Centener, 1998) that consumers from all walks of life treat the issue of sustainability with importance when shopping.

However, the overwhelming number of eco-labels creates a confusing scenario for both manufacturers and consumers who wish to pick an eco-label while setting manufacturing guidelines and purchasing, respectively. Many eco-labels also suffer from a lack of transparency, resulting in ambiguity in the representation of sustainability information, further complicating matters. This state of affairs is additionally exacerbated by proprietary issues of different firms, because of which manufacturing information is not released openly (Gaussin *et al.*, 2011). Eco-label abuse ('green-wash') is common where the environmental friendliness of the product is assessed by the manufacturing firm itself, through which process undeserving products are labelled sustainable (Lathrop and Centener, 1998). Self-declared labels are classified as Type II labels in ISO 14020 guidelines.

Eco-labels are widely prevalent today, appearing in different forms and applicable to different product categories. They are often used to quantify the sustainability properties of construction materials that are predominantly manufactured artificially using natural and synthetic raw materials, which impact the environment negatively (Dutil *et al.*, 2011). Production of construction materials is an extremely energy-intensive task, hence different eco-labelling schemes have been devised by different countries to quantify the sustainability of the production of the raw materials and the building as a whole (Harris, 1999; Crawley and Aho, 1999; Tam, 2007). Furthermore, with the growing population and demand for residential/industrial infrastructure by societies, eco-labelling of construction materials has become an important consideration today.

This chapter discusses the shortcomings of various forms of eco-labelling and will focus on building and construction materials. Section 8.2 outlines the typical shortcomings that constrain the effectiveness of eco-labels. Section 8.3 provides a brief overview of typical building materials used in construction and the practices that will make their production more sustainable. Section 8.4 gives an overview of different types of building eco-labelling programmes used in different parts of the world. This is followed by a critical review and conclusion.

8.2 Typical shortcomings of eco-labels

The wide variation in different categories of products sold today, combined with the conviction that environmentally sustainable business practices create a 'win-win-win' (Elkington, 1994) situation for the company, customer and the environment, has resulted in a proliferation of many eco-labels. However, these labels have come with associated challenges. The authors have recognised the 4Cs of eco-labelling that are necessary for their utility. They are comprehensiveness, clarity, credibility and customer focus. The following sections give an overview.

8.2.1 Comprehensiveness

A well-known measure of sustainability is the ecological footprint. Calculation of this value or metric involves a set protocol which eventually produces a single outcome: the number of hectares of biologically productive land required to create a product. The measure also includes the theoretical amount of land required to reverse the damage caused by greenhouse gases that result from human consumption/development. While intuitive and concise in nature, the use of an ecological footprint as a metric of sustainability has been criticised (Fiala, 2008). The practice of converting greenhouse gases to hectares of land has been questioned, especially because all greenhouse gases are converted to equivalent amounts of carbon dioxide during calculation of the metric. The ecological footprint is deemed arbitrary and comparative in nature at best.

This argument suggests the use of carbon footprint (CF) as the preferred metric for eco-labelling. In fact, CF has been widely adopted to represent product sustainability, and an example may be found in the Casino Group carbon index (Gaussin *et al.*, 2011). However, CF suffers several shortcomings and might be a grossly inadequate measure of the environmental impact (EI) of the construction materials industry. For example, CF as a metric is an insufficient measure of EI of deforestation (Gaussin *et al.*, 2011) and sand mining, both of which are significant activities in the construction/construction material industry. This is because both activities cause negligible carbon dioxide emissions, but cause serious harm to the environment and eco-system. For example, the Indian reptile 'Gharial' is almost extinct due to sand mining (IUCN, 2007). The challenges involved in formulating a measure of EI lie in encompassing all harmful effects of all activities involved in the manufacture of products.

Studies have suggested that the comprehensiveness of sustainability information does not affect consumer demand for eco-labelled products such as apples (Blend and Ravenswaay, 1999). However, it is clear from the shortcomings of ecological and carbon footprints that creation of an accurate eco-label for a product requires a comprehensive assessment of sustainability. While it is ultimately the producer's choice to present all this information on the package, the eco-label must be comprehensive to avoid false impressions that result from incomplete representation of sustainability information.

8.2.2 Clarity

While it is important to perform a comprehensive analysis of sustainability, it is equally crucial to present the data on the eco-label clearly. Unclear information makes it hard to determine the environmental friendliness of a product from reading the eco-label (Saane *et al.*, 2008) and indeed, use of words like 'bio-degradable' on an eco-label without suitable quantification may result in inadvertent 'greenwash', the term used for inaccurate or unsubstantiated environmental claims. For example, a common practice exemplifying ambiguous representation of sustainability information involves use of vague and potentially meaningless terms like 'environmentally friendly' in product eco-labels, giving the consumer a false impression. Note that this is inherently inaccurate in most situations where manufacture involves any processing of raw materials.

The problem can be demonstrated by considering a hypothetical situation in which two manufacturers (A and B) produce the same product in two different ways. The manufacturing routine adopted by A utilises three harmful chemicals, as opposed to just one harmful chemical produced by the routine adopted by B. In this situation, although the product is less damaging, use of a product eco-label with the words 'environmentally friendly' by manufacturer B would be fallacious due to the ill effects of the single harmful chemical. Although B causes less harm to the environment, product B nevertheless fails to be completely 'environmentally friendly'.

The Fair Trade Commission (FTC) in the USA sets clear guidelines for representing the environmental impact of products for eco-labelling. The FTC covers aspects such as ozone friendliness, recyclability, recycled content, refillability, compostability, degradability and source reduction on its website (FTC, 2012). For example, it says that a product bearing the term 'recyclable' is ambiguous and must be qualified, in that it must be clearly stated whether the product or its packaging or both are recyclable. Ambiguity in eco-labels is a serious issue given the different kinds of input materials required and effluents produced during product manufacturing and usage.

An example of ambiguity in eco-labelling, resulting in greenwash, is shown in Fig. 8.1. Such placards are typically found in hotels and let the tenants choose if they wish to have fresh sheets every day. The idea is for the tenants to place the placard on the bed to indicate that a change of sheets is required, the inference being that not doing so would reduce consumption of chemicals (laundry detergent inherently harmful to the environment). The practice gives a sense of extra effort taken by the respective hotel to protect the environment, whereas, in reality, it was found that hotels put minimal effort into reducing energy-wasting activities, and this placard was in fact only a cost-saving endeavour.

8.2.3 Credibility

The aforementioned issues of clarity originating from the lack of an easy to understand metric of environmental impact, combined with a massive

GREAT green

Your Choice Makes a Difference!

Your room will be serviced everyday and of course your bed linens and towels are fresh upon your arrival. During your stay we invite you to help reduce the extraordinary amounts of water and detergent used each day to launder bed linens and towels.

Bed linens will be changed when this card is placed on your bed in the morning.

Towels will be exchanged when placed on the shower floor.

Thank you for joining the effort to save the environment!

8.1 An example of greenwash advertisement in a hotel laundry placard.

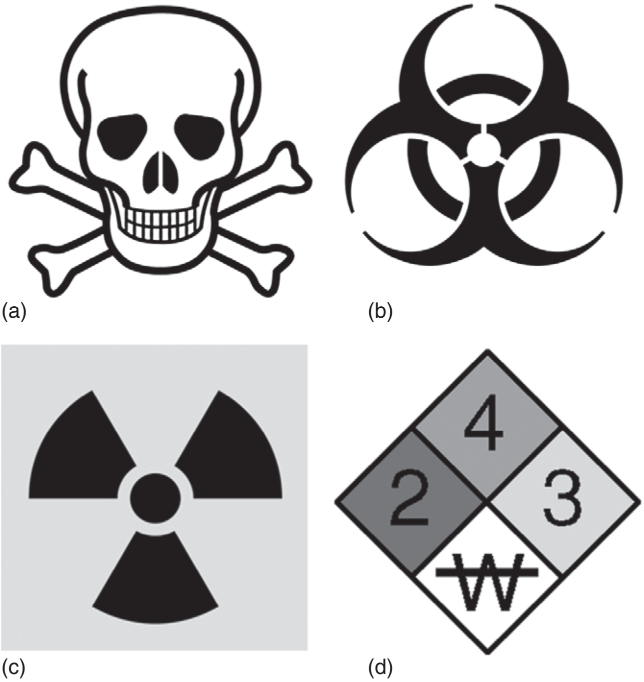
number of available eco-labels, have caused fatigue amongst consumers. This state of affairs is further exacerbated by incidents of greenwash (Karliner, 2001) that can arise from voluntary eco-labelling without third-party verification and proprietary issues ensuring secrecy of product manufacture routes (Gaussin *et al*., 2011). It must be noted that ISO 14020 clearly distinguishes between eco-labels that are self-declared (Type II by ISO 14020) and third-party verified (Types I and III by ISO 14020). The combined result of all these factors is a lack of credibility of Type II eco-labels. To highlight this point, consider the task of summarising nutritional information available on the packaging of processed foods. This task involves representing a sizeable amount of significant information related to ingredients that went into the product into an easily readable format, akin to that in eco-labels. A strict set of guidelines is available for preparation of this 'nutrition label' (FDA, 2012). In the US, the Food and Drug Administration (FDA) also acts as a regulatory body, penalising product manufacturers for false advertisement of product ingredients on their nutrition labels. Furthermore, the easily quantifiable nature of information present in nutrition labels makes it easier for representation in a concise manner. The aforementioned strict guidelines ensuring uniformity and accuracy in representation, combined with ease of representation, have resulted in a mainly trustworthy nutrition label that can be used for comparison purposes at least (Reinagel, 2011). It is noteworthy that in a study conducted by the University of Minnesota (Melnick, 2011), researchers found that more than 70% of participants looked at nutrition labels when purchasing groceries. Evidence suggests that eco-labels take time to gain the acceptance of the consumer masses

(Teisl, 2002). It is reasonable to expect that reputation matters significantly for eco-labels for establishing credibility where the aforementioned incidents of greenwash slow down the process. A simple method that boosts the credibility of any eco-label is third-party verification. Such eco-labels that have been verified by an independent third party are identified by the International Standards Organisation as Type I eco-labels.

8.2.4 Customer focused

To ensure clear representation of sustainability data, technical information must be translated into a form easy to understand by the common consumer who may have little or no technical knowledge of manufacturing techniques. Although clarity of the data is crucial as described, customer focus must be borne in mind to make the effort useful. Common examples of customer-focused representation are shown in Fig. 8.2.

The 'skull and crossbones' (shown in Fig. 8.2) is a common symbol indicating toxicity, and is printed on bottles containing toxic substances. This symbol is widely prevalent in popular culture and represents death. Misuse/



8.2 Hazard labels depicting (a) toxicity, (b) bio-hazard, (c) radiation hazard, and (d) National Fire Protection Agency 704 'fire diamond'.

ingestion of any material bearing this hazard sign may be fatal. It is to be noted here that due to the nature of the hazards related to misuse of materials with packaging bearing this symbol, its presence/absence is enough to signify the dangers involved. In other words, any person using a harmful chemical would be immediately aware of the dangers involved due to the presence of this symbol, without knowing the chemical composition of the product. This symbol elucidates the type of clarity that manufacturers of poisonous materials must bear on their packaging.

The bio-hazard symbol (Fig. 8.2(b)), radiation exposure (Fig. 8.2(c)) and National Fire Protection Agency (FPA) 704 (Fig. 8.2(d)) are also clear in representation, at least within groups of people dealing with such hazards (although FPA 704 has been called subjective in nature). It must, however, be recognised that young children (unfamiliar in its use) could not recognise radiation hazards upon seeing its symbol (Fig. 8.2(c)). This demonstrates that any person who is not aware of technical terminology will not be able to objectively judge a product's environmental friendliness, even if the eco-label bore comprehensive and accurate details.

These arguments show that any eco-label must be designed in accordance with the four criteria (4Cs) listed here, in that a label must be comprehensive, clear, credible and customer focused. The comprehensive nature of an eco-label would in fact also assure its consistency, whereby the EI of the product calculated by different means would be similar. However, to be of practical value, eco-labels must also be compact so as to be suitably positioned on the packaging.

8.3 Building materials

With growing populations throughout the world, there is a constant demand for residential and industrial infrastructure, whereby building materials have come into the focus of sustainability measurement efforts. As compared to other consumer products (e.g., groceries), assessing the sustainability of building materials presents the additional challenge of quantifying environmental impacts associated with their performance. For example, certain building materials are more effective heat insulators and will result in less energy consumption associated with heating in cold environments (Morel *et al.*, 2001). Such materials should therefore perform well in EI assessment during the eco-labelling process, even in situations where their manufacture involves a marginally higher release of pollutants. Furthermore, the impacts associated with material transport play an important role in the determination of sustainability of building products. A building made with locally available materials like rock and sand quarried/mined from the construction site is more sustainable than one made with bricks and cement/concrete that had to be transported to the site. Of course, ill effects of the latter practice are twofold because cement and concrete have to be chemically produced and involve release of effluents into the atmosphere during manufacture. However, the practice of using locally available resources (rock/sand) also affects local ecology. As stated before, sand mining has been associated with the decline in population of the Indian reptile Gharial (IUCN, 2007). Use of locally available rock may also affect the ecology in an unforeseen manner.

These arguments suggest that parameterisation of building material sustainability is more challenging and 'non-linear' than for other manufactured products. It is for this reason that life cycle assessment (LCA) techniques are often used while judging the EI of buildings (Mora, 2007). Furthermore, because of the varying (and sometimes conflicting) ways of representing LCA data in different European countries, a representation guideline (EN 15084) has been prepared that 'streamlines' environmental product declarations (EPD) in Europe. Quantification of EI of buildings using LCA entails a detailed analysis of sustainability associated with its construction, lifetime and reuse/demolition phases. The following section gives an overview of different building materials that are typically used in construction, and opportunities for sustainable practices in their production. However, other building sustainability measures have also been analysed, as it might be inferred from earlier arguments that a simple eco-labelling scheme of building materials may not be comprehensive enough. Because the construction of buildings involves the procurement of land, it is reasonable to expect that the governments of various countries are involved in the process of construction and hence in eco-labelling. Eco-labels of common building materials are segregated into product categories.

8.3.1 Cement

Cement is a common building material widely used as a component in concrete, mortar, stucco and grout. It is made by heating a mixture of limestone and clay in an oven and then grinding it to a very fine powder, thereby mixing calcium sulphate which tunes the setting time. It acts as a binder when used with other materials (cement, sand and water are the constituents of mortar) and holds bricks/stones together after setting, whereby it finds utility as a construction material.

Production of cement is an energy intensive process and adversely affects the environment (WBCSD, 2012). A mixture of limestone (extracted from quarries) and clay is progressively heated in a massive kiln to temperatures reaching 2,000°C. This step involves expenditure of fuel, which has to be continuously fed into the kiln to maintain such a high temperature, and results in the release of carbon dioxide into the air. The kiln produces a substance called clinker, which is ground to a fine powder in the final stage of manufacture, which again releases particles into the atmosphere. In addition to these aspects, naturally occurring harmless chromium (Cr^{3+}) present in cement raw materials is oxidised to a harmful (Cr^{6+}) state that affects humans through direct contact while building (Estokova and Palascakova, 2012). Therefore, an eco-label that represents sustainability of cement must take these aspects into consideration during sustainability calculations.

It was proposed that LCA be used for measurement of environmental impact during production and use of cement (Nisbet and Geem, 1997). This is probably because of the complications associated with isolation and parameterisation of different aspects of cement use and production that are grouped together. For example, it is suggested that the ecology of limestone quarries be subject to restoration after they have been exhausted. However, more recently, the metric commonly used for measuring sustainability of cement (like any other product) has been carbon dioxide emissions, arising predominantly from the heat treatment of cement raw materials in the kiln (Rosenthal, 2007). The Cement Sustainability Initiative (CSI), operating under WBCSD, began a system called 'Getting the Numbers Right' (GNR), which is an information system (WBCSD and CSI, 2006) that (whilst maintaining confidentiality) keeps track of the carbon dioxide emissions of the cement manufacturing firms. This was done with the purpose of helping members of the CSI identify factors that may be institutional in checking emissions. The environmental impact arising from production and use of cement obviously presents a challenge to environmental engineers. It is expected that controlling releases of carbon dioxide to fewer than 610 pounds per tonne of cement produced would be difficult.

8.3.2 Steel

Steel is one of the most commonly utilised building materials, and finds use as a structural element in, for example, truss beams, reinforced cement concrete, windows and plumbing. Apart from functioning as a construction material, steel is also used widely as a raw material for a host of other industrial and household goods. The manufacture of steel by the basic oxygen process involves melting iron ore in a furnace and then blowing oxygen through it, whereby impurities and carbon percentage are reduced to standards permissible in steel. Owing to the rapid increase in the demand for steel and the fact that steel is one of the most heavily utilised industrial raw materials, there exists a special need for sustainable steel manufacturing and eco-labelling. This argument is bolstered by the fact that steel can be recycled completely and used indefinitely. Furthermore, of the two kinds of steel structural elements typically produced (flat type as in sheets and long type as in rods), the long type can easily be made with high amounts of recycled materials.

Sustainable practices in steel manufacture may originate in one of the following stages of production (ECNZ, 2012):

- *Use of recycled content in making steel*. It is well known that use of recycled content generally helps to curb pollution levels during manufacture. This has significant implications in the production of wrought steel because gathering the necessary raw materials mostly involves mining iron ore, which is an ecologically damaging step. Furthermore, processing the iron ore into wrought alloy also involves steps during which energy is consumed (e.g., melting the ore, etc.) and is therefore unsustainable if not performed conscientiously with an environmental perspective (e.g., using sustainable sources of energy). Therefore, recycling of materials is advocated for the sustainable production of steel (long wrought products like rods, etc.).
- *Reusing water*. It is advised that water is reused and only the amount that evaporates be restored.
- *Reporting hazardous substances in steel*. The New Zealand eco-labelling trust (ECNZ, 2012) requires that hazardous materials that are found in steel be reported. Furthermore, it also requires that initiatives be taken that minimise the levels of hazardous substances in steel. The steel that is produced must also not be treated with any hazardous substance like mercury.
- *Controlling effluents released to water*.
- *Controlling emissions to air*. ECNZ needs the gas emitting from steel manufacturing to be treated in a facility whereby the particulate matter in the final emissions falls below 20 mg/m^3, emissions levels of dioxins and polychlorinated bi-phenyls (PCBs) are reported annually, and discharges to air are demonstrably sustainable.
- *Energy management*. The New Zealand eco-labelling trust specifies a maximum of 500 kWh be used to produce a tonne of liquid steel. However, the trust does not specify any origins for the specified amount.
- *Waste management*. ECNZ requires that its licence holders annually report the amount of waste recovered for reuse, recycling and the amount disposed of in landfills, along with the initiatives used to reduce waste generation and improve recovery/recycling. ECNZ also requires that all ferrous waste be diverted from waste stream and recycled.

It is clear from the arguments in this section that there is plenty of room in the steel industry for sustainable practices. In fact, given the high demand for steel in construction and other industries, such practices are bound to make a difference to ecology and the environment. However, it is once again surprising to see that few eco-labels specialise in measuring the sustainability of steel. The given state of affairs concerning eco-labelling of steel and cement probably arises from the fact that the energy consumed during their production comprises only 2% of what will be used by a building during its 60-year lifetime (Eaton and Amato, 1998).

Although eco-labels specific to steel are scarce, it must be pointed out that measures are being taken to produce construction steel more efficiently. In fact, steel today is 40% stronger and more efficiently made compared with ten years ago (AISC, 2013). However, it would only be didactic to point out that the steel industry is one of the very few industries that tends to refrain from sustainable practices and to manufacture as cheaply (and therefore 'dirtily') as possible (Busse, 2004), thus making the necessity for an eco-label for steel even greater.

8.3.3 Paint

While not a conventional building material, it is well known that most construction projects use some kind of paint when the building project nears completion. This activity may be performed with the motive of safeguarding iron structures from rusting or for aesthetic purposes. Recently, highly reflective paint has also been used to minimise the need for artificial lighting, whereby sunlight entering green buildings can provide the necessary illumination (Hewitt, 2003). For the sake of a comprehensive review, the eco-labelling of paint is described in this section.

Paints necessarily contain a binder that cures to form a film on the wall onto which they are applied, and come in various forms. A comprehensive discussion of their manufacturing techniques from the perspective of sustainability is beyond the scope of this chapter. Primary damage to ecology during paint manufacture arises from the manufacture of chemicals that form their ingredients, and release of volatile organic compounds (VOCs) after application. The Thailand Environmental Institute (TEI) has prepared 'core criteria' for the sustainability of paints where they prescribe the use of no formaldehyde, aromatic halogenated compounds and heavy metals in paints and the use of lead in paint containers. They also prescribe maximum levels of VOCs that may be released to the atmosphere after paint has been applied. Additionally, the European Union Eco-label details limits on white pigment content (in the presence/absence of TiO_2), SO_2, heavy metals (banned), and other dangerous substances (The European Union Eco-Label, 2012).

8.4 Eco-labelling of buildings

From the arguments in the previous sections, it can be concluded that eco-labelling of building materials is a challenging task, given that many components used in the manufacture of different materials originate from widely separated sources. Furthermore, as mentioned above, the building material industry incurs a measly 2% energy cost of the actual amount of energy that will be expended during the lifecycle of a building. This suggests that eco-labelling of building materials is a ‘moot point’ (Hewitt, 2003). However, given that the construction materials industry is expanding, there is benefit in practising sustainable manufacturing. The trend, however, has been to label the entire building for sustainability using more impactful eco-labels. Some of them are discussed in this section.

8.4.1 Comprehensive Assessment System for Build Environment Efficiency (CASBEE)

Comprehensive Assessment System for Build Environment Efficiency (CASBEE) is a green building eco-label method that has been used in several cities in Japan. This eco-label is based on calculation of a metric called building environment efficiency (BEE), found by dividing parameters of build environment quality (Q) by build environment load (L). The parameter L, which is closely associated with the damage to the environment during building construction and use, comprises a parameterisation of recycled materials, existing infrastructure and sustainably grown wood during construction. The BEE parameter also comprises factors such as natural resource usage (sunlight, rainwater, etc.) and takes into account efforts to conserve water during building use. Extensive details are available on their website (CASBEE, 2012). A BEE value (>3) produced using the procedure described suggests that a building is sustainable.

An interesting aspect of CASBEE is parameterisation of efforts to restore the original ecological conditions on the site where the building was constructed. This includes restoration of original plantation on site (within building campus, building wall and roof construction). The eco-label also tries to assess the effect of construction on different species in the building site. Similar species are chosen and greenery around the building is designed to match the characteristics of these species whereby these might be relocated.

8.4.2 Leadership in Energy and Environmental Design (LEED)

LEED is an internationally recognised green building programme that certifies buildings according to their ecological footprint upon their environment. LEED is an internationally accepted green building criteria and has been used in the USA, Canada, Brazil and Mexico. LEED awards points to buildings on a 136-point scale, depending on how they perform on different criteria of sustainability. Eventually, all points are totalled and a certification level (LEED gold, LEED platinum, etc.) is awarded depending on cumulative performance. Building categories in LEED are diverse and include homes, healthcare, hospitability, new construction and mid rises among others. Note that, in LEED, applicants have the option of choosing certain criteria on which they wish to be evaluated, while a select few are mandatory. For example, for the new construction building segment, the version 4 criteria of LEED specifications state regulation of indoor water use mandatory, whereas protection and restoration of natural habitats prevalent in building sites is optional.

8.5 Conclusions

It is clear that accurate eco-labelling of construction materials (and buildings constructed from them) is a formidable task, due to the multitude of criteria that must be borne in mind while making quantitative calculations for the purpose of eco-labelling. Obviously, the situation is exacerbated by the recent proliferation of eco-labels and the lack of regulation by independent third parties. Consequently, it is safe to say that the most important attributes of a successful eco-label lie in the 4Cs (comprehensiveness, clarity, credibility and customer focus). While a quantification of the performance of different eco-labels based on the 4Cs is beyond the scope of this chapter, examples have been identified where eco-labels performed quite well. For example, the ECNZ (see Section 8.3.2) for steel addresses clarity and comprehensiveness to a fair degree in its eco-label. It might be argued that, of the 4Cs pointed out, clarity (of representation) of an eco-label is the most important consideration for its successful utilisation. This is especially because of the massive amounts of new products that are designed and manufactured every day, each having its own subtle ecological features. Disparity between the proliferating eco-labels (especially for new products) can be called almost consequential but can be partially negated by clarity in representation. Specifically, this attribute of an eco-label will allow the customer to recognise its origins and make better ecological judgements. As discussed in Section 8.2.2, the origins of this disparity (resulting from lack of clarity) are deeply and firmly rooted within the frameworks of corporate law that secure the confidentiality of their function.

This chapter has reviewed the shortcomings of eco-labelling from the context of construction and building. The 4Cs of eco-labelling that are necessary for utility were identified. It was realised that any eco-label must be comprehensive, clear, credible and focused towards a customer segment to be complete. Subsequently, eco-labelling of important building materials (cement, steel and paint) was discussed. Using such instances, it was acknowledged that eco-labelling of building materials is less ‘linear’ than other manufacturing products, as building materials must last a long time and can leave a tremendous end-of-life signature if not properly designed.

They also affect the energy performance of the building and the ecology of the surrounding environment. Subsequently, eco-labelling of buildings was discussed in brief. This was followed by a critical review, where it was identified that the most important aspect of any eco-label is its clarity.

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9

Green public procurement (GPP) of construction and building materials

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Abstract: Green public procurement (GPP) involves the incorporation of environmental requirements during the procurement of services and products by public authorities. This chapter provides an overview on the growth of GPP in recent years by focusing on its expansion towards sustainable public procurement (SPP), where sustainable development considerations are integrated. Furthermore, the chapter discusses the implementation of GPP/SPP in the construction sector, and issues related to approaches in procurement procedure, GPP/SPP criteria, and the link between criteria and functional objectives of contracts. The chapter highlights that such issues need to be addressed during the progress towards SPP.

Key words: green public procurement, construction sector, sustainable public procurement.

9.1 Introduction

The construction sector is a significant user of natural resources and energy. The Worldwatch Institute's State of the World 2012 highlights that the construction industry consumes more than one-third of global resources (Taipale, 2012). The Organization for Economic Co-operation and Development (OECD) indicates that the construction sector accounts for around 25–40% of final energy consumption in OECD countries. As a result of vast consumption of resources and energy, the sector has been greatly responsible for environmental pollution and problems related to sustainability. With increasing global concerns over sustainability issues, the construction sector is required to be proactive in improving its environmental performance. The sector is thus compelled to address many questions regarding the approaches to handle environmental problems within the overall design process, the suitable materials to be used, eco-efficient practices, and other such issues and aspects that influence its contribution to the improved environment (Ball, 2002). Furthermore, the construction sector is pushed by government authorities to adopt several policy instruments that contribute to the improvement of environmental performance. One such policy instrument is green public procurement. Green public procurement (GPP) is defined by the European Commission (henceforth referred to as the Commission) as 'a process whereby public authorities seek to procure goods, services and works with a reduced environmental impact throughout their life cycle when compared to goods, services and works with the same primary function that would otherwise be procured' (CEC, 2008).

This chapter provides an overview on the growth of GPP in recent years, and deals with its implementation in the construction sector. The Commission has identified construction as one of the priority sectors for GPP. However, a statistical report published by PricewaterhouseCoopers *et al.* (2009) showed that the GPP levels are relatively low in the construction sector for most of the member states. The statistical report also concluded that green procurement in construction can result in negative financial impact, which indicates that cost reductions can be achieved by procuring green. Similar conclusions have also been drawn in another study conducted by Testa *et al.* (2011). They surveyed 78 European firms operating in the construction sector, and the results indicated that there is a positive effect of GPP on private companies' business performance, which led them to conclude that policy makers should strengthen the use and diffusion of policy instruments such as GPP. However, a Swedish study has suggested that the cost-effectiveness argument must be used carefully, primarily due to the heterogeneity in the environmental performances of potential bidders, which leads to differences in the magnitude of the investments needed to comply with environmental requirements (Lundberg and Marklund, 2012). Nevertheless, the OECD has considered GPP to be particularly important in areas where no other policy instruments are feasible, and identified its potential to improve the environmental performance of the construction sector (OECD, 2003). What can be interpreted from these studies is that there is a need to enhance the construction sector's capacities to plan and implement GPP. At the outset, it is important to understand where GPP stands in the array of environment and sustainability-related statutes so that the construction sector is prepared for the anticipated opportunities and challenges.

Therefore, this chapter begins by introducing GPP as a policy instrument, highlighting its evolution towards sustainable public procurement (SPP), wherein social and sustainable development considerations are also integrated in procurement decisions, followed by an overview of the policy contexts in the European Union (EU) and certain OECD countries. The chapter subsequently discusses the implementation of GPP/SPP in the construction sector, and presents issues that require detailed attention. Furthermore, the chapter identifies and substantiates the need to inter-link GPP/SPP with environmental assessment instruments supporting planning and decision-making processes. The chapter concludes by focusing on areas for improvement to facilitate progress towards SPP.

9.2 Green public procurement (GPP) and sustainable public procurement (SPP) as policy instruments

9.2.1 Evolution of GPP towards SPP

As a policy instrument, GPP gives public authorities such as municipalities an opportunity to procure eco-efficient services and products from their contractors and suppliers. The more a public authority is informed and acquires competence and knowledge in developing GPP practices, the greater its willingness to experiment with such new practices and introduce GPP criteria in the tenders (Testa *et al.*, 2012). However, the challenge has been to connect GPP to the overall concept of sustainable development. In 2006, the OECD surveyed the implementation of measures related to its 'recommendation on improving the environmental performance of public procurement'. The OECD study revealed that all but three respondents (Japan, the Netherlands and the United States) had incorporated GPP as a part of broader policy on sustainable development (Johnstone and De Tilly, 2007). Moreover, studies have questioned the focus on sustainability in GPP. For instance, Powell *et al.* (2006) indicated that green procurement in many organisations included one or two pillars of sustainability (largely focusing on the environment), with inadequate consideration of social and economic issues. The consideration of social issues might include, for instance, sourcing products/services to benefit the economic well-being of local communities, and ethical purchasing of materials from developing countries where employees have safe and humane working conditions (Meehan and Bryde, 2011).

The integration of social issues in GPP is a substantive concern, and furthermore, a procedural concern has been raised elsewhere in the Nordic countries. A Nordic study argues that one of the reasons for not achieving the intended ambitions of GPP is its promotion and development as an environmental policy instrument, and thereby its isolation from most other public procurement policy making. Therefore, a strong need for mainstreaming GPP has been identified (Bergman *et al.*, 2012). There are a few other procedural concerns related to procurement contracts that are discussed in the subsequent sections in this chapter. The substantive concern regarding the integration of social issues led many governments to enhance the scope of GPP and progress towards SPP. The United Nations defines sustainable procurement as a procurement wherein 'an organization uses it buying power to signal preferences to the market by its choice of goods and services which meet sustainable development criteria' (United Nations, 2008). Many countries have adopted the national action plan for SPP and set targets. For instance, Austria adopted the naBE-Action plan in the year 2010. The municipalities in the Netherlands aim towards 100% SPP in the year 2015. Poland developed a national action plan for SPP that included activities to be implemented during the years 2010–2012.

In terms of the construction sector, if GPP should contribute to 'truly sustainable buildings', then it needs to address 'the multi-dimensional aspects of sustainability, from the time a building is planned and constructed, throughout its operating lifespan, until it is demolished or renovated' (Taipale 2012). Therefore, the transformation of GPP to SPP holds challenging promises with respect to linking construction procurements to sustainable development. For instance, a study conducted (for the Commission) to assess the national building regulations shows that ethical trading throughout the supply chain is regulated only in Cyprus, Germany and the Netherlands at the national level (Vermande and van der Heijden, 2011). Would the implementation of SPP enable the incorporation of such social requirements during the procurement procedure? This depends on how far the issues surrounding GPP and now SPP will be addressed in the future. For instance, one of the crucial issues relates to the selection of the approach within the procurement procedure for incorporating social considerations. There are three approaches, which include technical specifications, award criteria and contract performance conditions. In the procurement procedure, technical specifications indicate the prerequisites to submit a tender, and the award criteria enable the procurer to compare the relative advantages of different tenders by giving weights to the criteria and scoring each tender on the basis of the level of fulfilment of each criterion. Furthermore, the contract performance conditions are those that are included in the contract to indicate how the contract work is to be performed.

An advocacy agency called ClientEarth argues that a comprehensive incorporation of sustainable development considerations is hindered by the tendency in the Commission Directive 2004/18/EC (on public procurement) to stipulate what concerns can and cannot be stated in technical specifications and/or award criteria. In many cases, social issues have been relegated to contract performance conditions, wherein the contracting authority is not able to assess compliance with these conditions as part of its selection of eligible tenders. Such restrictions on the type of concerns to be stated in the technical specifications/award criteria have been suggested with the intention to maintain the relevance of the incorporated considerations to the functional objectives or use of the procured services, supplies or works (ClientEarth, 2012). In the light of such intriguing issues, the evolution of GPP towards SPP needs to be challenged as to whether this progress 'is as good as things can or should get' (Sutton and Preece, 1998).

9.2.2 Innovation through GPP

GPP has also been attributed to stimulate innovation in environmental technologies, products and services (CEC, 2008). Innovation is not only about the implementation of a new product; it could also be the introduction of a significantly improved process, marketing method or organisational method in business practices (Bröchner, 2010). According to the OECD, handling innovation systems requires the potential to manage information creatively in response to the market and societal needs. The OECD approach also insists that governments should address systemic failures that block the functioning of innovation systems and obstruct the flow of information. These systemic disruptions emerge from institutional rigidities that are based on asymmetric information and communication gaps and lack of networking (OECD, 1999). Such gaps and lack of integration are evident in the construction sector. For instance, Vermande and van der Heijden (2011) point out that the construction sector is highly fragmented with a wide range of trades and professions.

Therefore, if GPP is to stimulate innovation in the construction sector, it should also address the complexities associated with this fragmentation. The potential for eco-innovation through GPP was explored by the Nordic Council of Ministers (2010). Their recommendations for stimulating innovation in the construction sector through GPP are focused on identifying appropriate models for dialogue (between procurers and tenderers) in tender processes for construction work; considering a wider use of framework agreements with selected eco-innovative suppliers, where the procurer is willing to test and buy products without traditional guarantees (share risks); extensively using specific programmes that promote innovation and eco-innovation, such as the EU-funded lead market initiative; and activating instruments such as tax reductions, direct financial support and regulation.

9.3 Policy context in the EU

Several policy documents in the EU have highlighted the importance of integrating environmental considerations into public procurement ever since 2001. However, this section identifies recently introduced ‘policies’ that support the implementation of GPP in the EU. In this section, the term ‘policies’ includes Commission Communications, Directives, Green Papers, European Parliament resolutions, among others. The EU introduced a policy called the lead market initiative (LMI) in 2007. The LMI sought to address six emerging markets including renewable energies, bio-based products and sustainable construction. The aim of the LMI is to streamline legal and regulatory environments, and bring new products and services into the market by enabling market access and developing measures to facilitate the aggregation of demand. The LMI deployed a diversified set of policy instruments, which *inter alia* included public procurement, and developed action plans for each of the six markets (CEC, 2007). Furthermore, under public procurement, encouragement of GPP has been identified as an action plan for certain markets (EU, 2011).

One of the most important policy documents concerning EU GPP in recent years is the Commission's Communication entitled 'Public procurement for a better environment', published in 2008. The objective of this Communication is to 'provide guidance on how to reduce the environmental impact caused by public sector consumption and to use GPP to stimulate innovation in environmental technologies, products and services' (CEC, 2008). Although the Communication objective is highly motivating from an environmental perspective, there has been a need to link green procurement decisions with policies concerning biodiversity.

BRE (2011) has identified and described the relevance of the Birds Directive/Commission Directive 2009/147/EC on the conservation of wild birds (adopted in 2009) to GPP. What particularly relates to GPP in this directive is the focus on the protection of habitat. The extraction of raw materials can lead to habitat destruction if activities such as mining and quarrying are not undertaken in a controlled manner. In addition, the Birds Directive emphasises the establishment of a coherent network of special protection areas (SPAs) comprising all the suitable habitats for bird species. Furthermore, all these SPAs form an integral part of the NATURA 2000 network, which is an EU-wide network of nature protection areas that aims to protect Europe's threatened species and habitats (EC, 2003). With regard to biodiversity concerns in GPP, it is also relevant to consider the EU biodiversity strategy to 2020 and the April 2012 resolution adopted by the European Parliament on the EU biodiversity strategy. The EU biodiversity strategy aims to halt the loss of biodiversity in the EU by 2020. The resolution urges the Commission to implement measures to reduce the negative impact of EU consumption patterns on biodiversity, and enhance the contribution of EU trade policy to conserving biodiversity (EC, 2012). Besides the 2020 biodiversity target, GPP has links also with the sustainable growth targets set for 2020. These targets are addressed in the Europe 2020 strategy.

In June 2010, the European Council adopted the Europe 2020 Strategy. The aim of this strategy is to recover from the economic crisis by focusing on three priorities: developing an economy based on knowledge and innovation; promoting a more resource efficient sustainable growth; and stimulating a high-employment economy delivering social cohesion. The Commission has put forth 'seven flagship initiatives to catalyse progress under each priority theme' (EC, 2010a). According to ClientEarth (2011a), three of the seven flagship initiatives have identified public procurement as an important instrument to contribute to achieving sustainable development in the EU. These three initiatives include 'resource efficient Europe', 'innovative union' and 'integrated industrial policy'.

The primary role of the flagship initiative 'resource efficient Europe' is to help decouple economic growth from resource use and its environmental impact, support the shift towards a low carbon economy, increase the use of renewable energy sources, and promote energy efficiency. This initiative highlights GPP as one of the policies to increase resource efficiency (EC, 2011a). 'Innovative union' sets out to improve conditions and access to finance for research and innovation, with the aim to ensure that innovative ideas can be transformed into products and services that create growth and jobs. Innovative union identifies GPP as an area where it will be possible to address factors that influence innovation via procurement. Some of these factors involve incentives that favour low-risk solutions, lack of knowledge regarding successful procurement of new technologies and others (EC, 2010b). The flagship initiative on 'integrated industrial policy' aims to stimulate economic recovery by ensuring a thriving industrial base in the EU and, furthermore, seeks to enable the transition to a low carbon and resource efficient economy. The integrated industrial policy endorses the wider use of GPP in order to develop the EU market for environmental goods and services (EC, 2010c).

Within the Europe 2020 strategy, the role of public procurement (especially GPP) is thus considered to be significant to support the shift towards a resource efficient and low carbon economy.

In the context of these ambitious objectives of the Europe 2020 Strategy, the Commission developed a Green Paper in 2011 to discuss the scope for the modernisation of EU public procurement policy. The Green Paper discusses some of the challenges concerning GPP. For instance, the diverging approaches by member states on GPP might be an impediment for certain suppliers, who have to adapt to different frameworks in different countries, thereby hindering the development and sale of eco-efficient products and services (EC, 2011b).

9.4 Policy context in selected countries

9.4.1 Introduction to the countries

A recent OECD study reports that although 24 member states among the 34 OECD countries have introduced environmental requirements in the technical specifications and 18 countries include them in the award criteria, less than half of the OECD countries have not established a standard definition for green procurement. Even though green policies are at the forefront, only six countries have defined green procurement in law:

Denmark, France, Italy, Japan, Luxembourg and Slovenia. Most of the other countries that have defined green procurement have done so in an environmental policy or strategy document. Furthermore, codes of practice have been adopted in only ten OECD countries: Austria, Denmark, France, Korea, the Netherlands, New Zealand, Poland, Slovenia, Spain and Sweden (OECD, 2011). This section provides an overview and discussion on the policy contexts in five of the ten countries that have adopted the code of practice for GPP. They include Poland (Central Europe), the Netherlands (North West Europe), Sweden (Northern Europe), New Zealand and Korea. These five countries have been selected so as to represent different parts of Europe and outside Europe. Moreover, this selection has been made with an intention to present different approaches involved in managing and developing the policy instrument, and also to highlight some of the issues involved in these approaches. However, this is not a comparative analysis of the GPP/SPP policy contexts in these countries.

9.4.2 Policy context in Poland

During the implementation period of Poland's national action plan on GPP for 2007–2009, there was an increase in the level of implementation of GPP. In 2006, the public procurement office conducted an analysis of the contract documents (sample size: 400), and estimated that 4% of the contracts had included environmental criteria. Further, in 2009, another analysis of the contract documents (sample size: 600) was conducted to review the execution of the action plan. This analysis revealed that 10.5% of the contracts had incorporated environmental criteria. Green procurement was evident in, among others, some of the services related to improving thermal efficiency in buildings and modernising water/sewage systems. Despite the increase in GPP at the national level, its status in comparison with the leading member states remains low. This proved the need to continue activities undertaken within the action plan for GPP for 2007–2009. Therefore, a new action plan was developed, with a view to incorporate social considerations as well, and thus it is called the national action plan on SPP for 2010–2012. The goal of this plan is to 'promote solutions for contract award procedures, which can positively influence the eco-innovations and pro-social behavior'. The general objectives are focused on increasing the implementation level of GPP (to 20%) and SPP (to 10%) at the national level, and creating demand for products that meet high environmental standards and innovative environmental technologies. The detailed objectives aim at an increase in the number of public contract awards that incorporate social considerations; awareness creation regarding GPP and SPP; increase in the number of units applying a verified environmental management system (EMS) and national products certified by the Polish e*koznak* and/or EU *ecolabel* (PPO, 2010).

The national action plan has set targets for achievement, indicators, schedule, and also assigned agencies that will be responsible for the actions.

In addition, the country's national environmental policy for 2009–2012 has identified GPP as a direction for action in order to activate the market to protect the environment. Certain approaches to activate the market towards environment protection are evident in the objectives of the national action plan on SPP. One such objective concerns the popularisation of EMS. EMS is a management tool that facilitates a company or an organisation to identify the environmental impacts resulting from its activities and to improve its environmental performance (NCSI, 2009). However, in terms of EMS in the construction sector, a Spanish study shows that the EMS application at construction sites seems to be considered as a formality and a tactic to access the tender of contracting organisations, rather than as a genuine commitment towards improving the environmental performance of companies in the construction sector (Rodríguez *et al.*, 2011). Furthermore, Lam *et al.* (2011) have suggested that the mere promotion of EMS in the construction industry may not ensure the incorporation of sufficient environmental considerations.

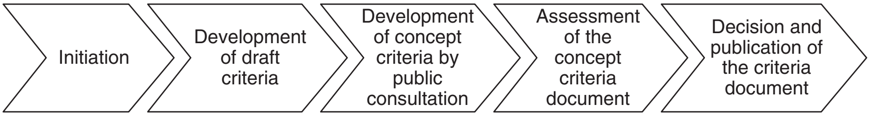
It should be noted that the objectives in the national action plan have been stated separately for GPP and SPP, which, however, raises certain questions. For instance, does the segregation of SPP and GPP in the national action plan indicate the lack of recognition of the 'interdependence of ecological, social and economic systems'? (Hutchins and Sutherland, 2008). Moreover, the SPP action plan objectives intend to create demand for certain certified eco-efficient products. However, Vermande and van der Heijden (2011) have stated that Poland's national building regulations do not include any specific voluntary or mandatory requirements concerning the sustainability of construction products. In terms of regulations, Vermande and van der Heijden (2011) refer to all those central/state/regional regulations (laws, ordinances, decrees, standards, codes, approved documents, guidance) imposing mandatory or advisory requirements or provisions on the planning, design, execution, maintenance and use of construction works. Furthermore, they have pointed out that even though there are existing assessment standards for eco-efficient construction products, such products cannot be used by the developers as a way to confirm compliance. Such inconsistencies might prove to be a challenge for the implementation of SPP in the construction sector, and also, may not be specific to Poland.

9.4.3 Policy context in the Netherlands

The Dutch national government had set an ambitious goal to achieve a target of 100% sustainable procurement by the year 2010. The municipalities and provinces both aim for 100% by 2015 and had their interim targets to achieve 75% and 50%, respectively, by 2010 (NL Agency, 2010a). The standard procedure for the development of EU GPP criteria is led by the Commission through a formalised process involving data collection, consultation, adoption and publication. However, the criteria development process can also be led by member states and stakeholders after obtaining Commission approval. In the Netherlands, a public agency called the NL Agency, which facilitates government authorities in various ways to achieve the SPP objectives, is primarily involved in the SPP criteria development process. This process includes five phases (see Fig. 9.1).

In the initiation phase (see Fig. 9.1), the NL Agency communicates the decision to develop new criteria or to update existing criteria for a product group (for example, construction works) on its website and also to the interested parties that have registered for the particular product group. During the second phase, the NL Agency organises a public meeting. This meeting is arranged in order to discuss the process of criteria development to be followed, the assessment framework and the constellation of a workgroup. Once the workgroup is formed, the NL Agency, which is the secretary of the workgroup, makes a proposal for the selection of a chairperson among the parties involved. The workgroup receives the commission to prepare a draft criteria document that complies with the assessment framework. The workgroup meets several times and discusses suggestions for the criteria, and provides a justified decision on the criteria that should be and must not be included. Further on, the NL Agency prepares a draft criteria document and publishes it on the website together with the report from the initial meeting.

The third phase mainly involves the assessment of the feedback received on the draft criteria document. The workgroup decides on the modifications, after which the concept criteria document is prepared. The NL Agency publishes a report on the website that contains a summary of the feedback received and a description on how the comments were addressed within the concept criteria document. In addition, the NL Agency sends this document to all those who had provided the feedback. In the fourth phase, the concept criteria are assessed by the Ministry of Infrastructure and Environment, which examines whether or not the criteria sufficiently consider the aspects within



9.1 The process of criteria development under the Dutch SPP programme.

the assessment framework, and whether the process has been followed in the correct manner. The assessment results are communicated to the workgroup, which revises the concept criteria document. Thereafter the NL Agency describes the assessment results on the website. Ultimately, in the fifth phase, a final decision is taken on the criteria document, and subsequently it is published (NL Agency, 2010b; Melissen and Reinders, 2012).

The Dutch SPP manual, a document that serves as an aid for procurement officers and other concerned parties to implement SPP, indicates that adherence to both the manual and the criteria documents does not assure the legal and proper execution of the procurement. The contracting authorities may decide to use sustainability criteria different from those discussed in the criteria documents, in which case the contracting authority might have to conduct the legal assessment. Furthermore, it also states that 'sustainable procurement is not always about the purchase of sustainable products'. Instead, it can also aid in deciding not to purchase. This is by providing new insights that result in limiting the purchase (NL Agency, 2010a). However, Melissen and Reinders (2012) conclude that the 'set up of the current Dutch SPP programme' is in dearth of a clear vision concerning how public procurement can contribute to achieving long-term goals with regard to sustainable development. Furthermore, Melissen and Reinders (2012) critique that the setup encompasses a rigid distinction between environmental and social criteria, and that their interrelation is not adequately addressed in the actual criteria. Nevertheless, they consider it important to note that the current programme includes a number of characteristics and procedures that would facilitate the improvement of Dutch SPP. For instance, they refer to the procedure for developing criteria, which allows for comments and feedback from the stakeholders.

Moreover, it is also worth noting the political decision (in the form of a written statement to the Parliament) taken by the Dutch government on social criteria in SPP. The ministers (vide their written statement to the Netherlands Parliament) had stated that 'it is not socially acceptable for governmental bodies in the Netherlands to procure goods and services whose supply involves infringement of the fundamental rights of people in other countries, and that other levels of the government (municipalities, regions) will be asked to subscribe to the approach defined by the cabinet' (NL Government, 2009).

9.4.4 Policy context in Sweden

In Sweden, the environment ministry is responsible for GPP at the governmental level. The Swedish environmental protection agency (Swedish EPA) has the responsibility to monitor and evaluate the GPP. The Swedish environment management council (SEMCo), a government-owned corporation, has the operational responsibility for implementing the national action plan on GPP (Bergman *et al.*, 2012). SEMCo provides expertise in the area of GPP, and also develops GPP criteria for various product groups. According to SEMCo (2012), although Sweden has been ranked high in international evaluations on environmental and socially responsible procurement, there is a need for a national action plan on sustainable procurement.

Furthermore, SEMCo considers the decision to develop the national action plan on SPP as one of the priority measures for using public procurement as an effective environmental and political tool. In addition, SEMCo has also questioned the role and potential of GPP to actually contribute to fulfil the Swedish national environmental quality objectives. The Swedish national environmental quality objectives have been adopted by the Swedish Parliament for 16 areas such as reduced climate impact, a non-toxic environment, sustainable forests, among others. These objectives have been embraced in order to attain the generation goal, which is to 'hand on to the next generation a society in which the major environment problems facing Sweden have been solved' (Regeringskansliet, 2004). When SEMCo investigated the opportunities within public procurement to fulfil these 16 objectives, it made certain recommendations for strengthening sustainable procurement. These recommendations include securing credibility in the concept of sustainable procurement and considering it as a target-oriented societal economic political tool; using available practical applications to increase innovative procurement; seeking information/clarification about the environmental and climate performance of products; promoting dialogue between key players (SEMCo, 2012).

9.4.5 Policy context in New Zealand

The Australian Procurement and Construction Council (APCC) has developed an Australian and New Zealand government framework for sustainable procurement (APCC, 2007). This joint framework for sustainable procurement provides a set of principles to facilitate the public sector in Australia and New Zealand to integrate sustainability considerations into the procurement of goods, services and construction. Furthermore, APCC's guiding principles are supported by best practice implementation activities (see Table 9.1), which agencies can adhere to when developing their sustainable procurement strategies, guidance material and training tools.

Moreover, the New Zealand government had appointed an advisory group to investigate and report on topics crucial to New Zealand's success in achieving greener and faster growth (Green Growth Advisory Group Secretariat, 2011). In its report, the advisory group has made certain recommendations in the context of the construction sector. The advisory group has recommended that the government should designate construction as a

Table 9.1 APCC's four guiding principles and supporting implementation activities

Principles	Certain examples of implementation activities
Avoid unnecessary consumption and manage demand	Assess the need for procurement, and whenever possible reduce consumption; consider alternatives to product procurement such as reuse, refurbish the product to extend its durability; consider incorporation of management systems to monitor and report consumption levels; consider alternatives to acquisition such as introducing service options to meet a need.
Identify products and services with lower environmental impacts across their life cycle compared with competing services and products	Verify the sustainability credentials of a supplier/contractor and product, and ensure that decisions on sustainable products and services are evidence based.
Promote a viable Australian and New Zealand market for sustainable products and services by supporting businesses that demonstrate innovation in sustainability	Establish procurement processes that promote innovation and facilitate the commercialisation of environmental initiatives created under contract; support long-term partnerships with suppliers and contractors that adopt sustainable practices.
Support suppliers to government who adopt socially responsible and ethical practices	Require suppliers and contractors to demonstrate a commitment to ethical practices, and to consider relevant government employment policy objectives that relate to particular community sectors.

green growth sector in relation to public procurement. Furthermore, it has suggested that the highest priority should be given to green procurement in the reconstruction of Christchurch (one of New Zealand's largest cities, damaged by several earthquakes during 2011). The advisory group's report also notes the significant barriers to GPP in New Zealand, which include the tendencies of procurement decision makers and their suppliers to prioritise cost and immediate affordability above other aspects, and a lack of understanding about environmental criteria in certain areas. The report also states that the public sector agencies do not have sufficient incentive or resources to choose eco-efficient alternatives in their building, technology and vehicle procurement. Nonetheless, what needs to be investigated is the extent to which the APCC principles have been followed in the rebuilding of Christchurch. Furthermore, such an investigation should also provide an understanding of the practicalities and challenges involved in implementing various activities prescribed under the four principles.

9.4.6 Policy context in Korea

The Korean government has introduced the *Act on encouragement of purchase of green products* (2004). The purpose of this act is to prevent waste of resources and environmental pollution, and contribute to sustainable development by purchase of eco-efficient products. In accordance with this act, the minister for the environment formulates basic plans for encouraging procurement of eco-efficient products, following consultation with the heads of the relevant central administrative agency every five years. The public institutions subject to this act include state agencies, local governments, and certain designated public organisations. The heads of public institutions are required to compile procurement records of eco-efficient products pursuant to implementation plans and submit reports to the minister for the environment. According to the act, the minister for the environment can partially entrust the tasks under the act to relevant specialised institutions such as the Korean Environmental Industry and Technology Institute.

An OECD paper on Korea's low carbon green growth strategy states that the Korean government is also operating the public procurement system for minimum green standard products called 'Minimum Green Standard'. The Minimum Green Standard is composed of environmental standards such as energy consumption efficiency, recycling and others. Only the products that meet these requirements are allowed to have business transaction on the Korea online procurement system. The OECD paper highlights that this standard provides a strong incentive for the producers of eco-efficient products. In addition, in several cases, government procurement serves as one of the direct incentives contributing to the innovator's entry into the market. Once innovative products are judged to be eco-efficient, government becomes an early adopter of the product. Thus government procurement enhances public confidence in relevant business areas, and thereby stimulates the market (Kang *et al.*, 2012).

In contrast to some of the other OECD countries that have progressed to SPP, Korea is yet to take that leap. The focus of the current *Act on encouragement of purchase of green products* (2004) is on environmental labelling of products. However, if Korea chooses to shift to SPP in the near future, there is a need for a 'paradigm shift'. This holds true as well for most of the other countries that have recently extended GPP to SPP.

9.5 The need for a paradigm shift

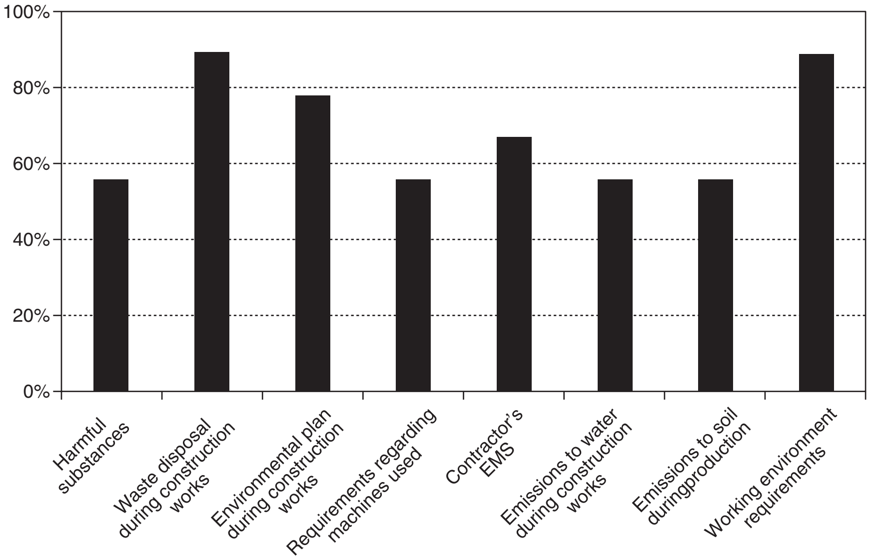
Hall and Howe (2010) highlight that a paradigm shift indicates a 'complete revolution in the mindset of scientific community', and involves the consideration of newly discovered realities against established views. Although the five countries discussed in Section 9.4 have different levels of achievement and varied approaches, it can be said that they are all moving in the same direction. The introduction of SPP in certain countries should have stimulated a profound change in the implementation of GPP, implying, first and foremost, that the scope of GPP must have expanded. However, certain studies have shown that SPP, in practice, has not triggered the movement beyond the environmental criteria (Meehan and Bryde, 2011; Melissen and Reinders, 2012). These studies indicate that the paradigm shift can generally be discussed for all the five countries and also other states that have adopted GPP and SPP. Moreover, GPP is one of the several policy instruments to have gradually become established in developing economies such as China (Geng and Doberstein, 2008). Though relatively late, China has adopted GPP policy and has started with its implementation (Qiao and Wang, 2010). Furthermore, the national policy framework for public procurement in China provides a promising basis for promoting SPP (Philipps *et al.*, 2011). In India, the draft public procurement bill states that environmental criteria of a product may be adopted as one of the criteria for evaluation of tender. Furthermore, the government has formulated a committee to develop guidelines on SPP (Kumar, 2012). If such growing momentum for SPP is to introduce the changes that are 'worthy of the name paradigm shift' (Hall and Howe, 2010), then several issues have to be addressed. We put forth certain issues for consideration, particularly within the context of GPP in the construction sector.

9.6 Implementing GPP/SPP in the construction sector

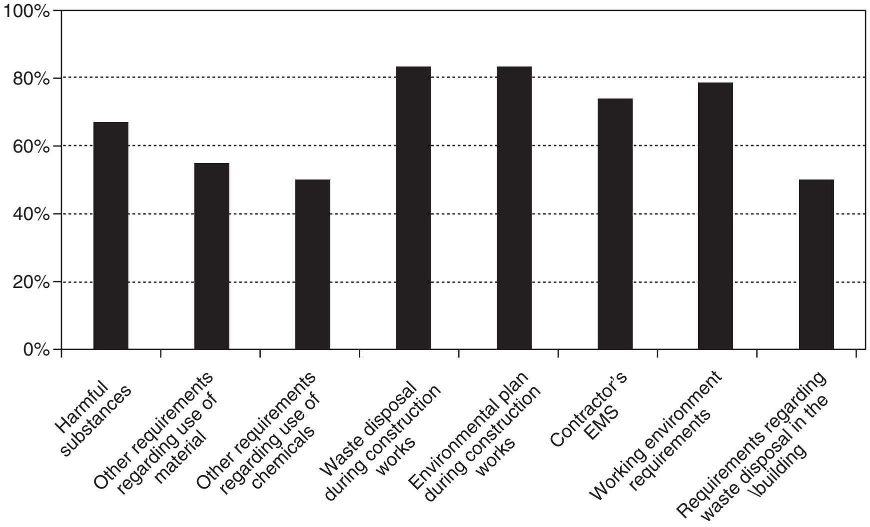
9.6.1 GPP/SPP criteria in the construction sector

A Swedish study has identified the types of environmental requirements applied in civil engineering and building projects in Sweden. Those concerning waste disposal during construction, environmental planning during construction, working environment and the contractor's EMS had a high frequency among both civil engineering and building projects (see Figs 9.2 and 9.3). The study revealed that expert judgement was the frequently adopted method for formulating the environmental requirements and criteria. In addition, tools such as life cycle assessment (LCA) and environmental impact assessment (EIA) were used to formulate environmental requirements in the cases where project-specific requirements were applied (Varnäs *et al.*, 2009a). The use of such tools to develop GPP criteria is discussed later in this chapter.

The fundamental concept of GPP relies on establishing environmental criteria for products and services based on an in-depth knowledge of their life cycle impacts (Evans *et al.*, 2010). Moreover, the establishment of



9.2 Percentage of the civil engineering projects where each of the eight most frequent environmental requirements had been applied (source: Varnäs *et al.*, 2009a).



9.3 Percentage of the building projects where each of the eight most frequent environmental requirements had been applied (source: Varnäs *et al.*, 2009a).

criteria requires the identification of key aspects. Ireland's national action plan on GPP entitled Green Tenders (2012) describes the key aspects in the construction sector, which can include design, energy, materials, ecology and site utilities. In the design aspect, procurement procedures for consultancy and engineering design services should include assessment criteria and demonstration of consultants' environmental design experience. Furthermore, energy efficient design strategies have been recommended to be incorporated at the early design stage. The national action plan has suggested that the energy aspect should aim at reducing the energy demand and maximising energy efficiency. In addition, it should also focus on securing energy supplies that are less carbon-intensive. In terms of the materials aspect, the national action plan has emphasised the assessment of all materials used in construction for environmental impacts over the appropriate appraisal period for the project. Moreover, procurers are required to ensure that the environmental advantages claimed by the material suppliers can be verified. Aspects of ecology relevant to GPP in the construction sector include, for instance, habitat protection. Site utilities include transport, water and others. The national action plan has recommended taking the necessary steps to protect or enhance site utilities. Therefore, the plan has directed that the relevant environmental criteria should consider the appropriate management of site utilities to optimise use.

Although all these aspects play an important role in understanding how GPP can be incorporated in the construction sector, the comprehensive information on the key environmental impacts of proposed projects is also crucial in adopting the GPP approach. Such key environmental impacts can include those resulting from the consumption of energy for construction purpose and heating and others, emissions caused by the transportation of construction materials and products, as well as consumption of natural resources used in the project (EC, 2008). The information regarding these impacts is used in determining the environmental criteria.

In the EU, the GPP criteria are categorised as 'core' and 'comprehensive' criteria. The core criteria are designed to facilitate easy implementation of GPP, focusing on the key areas of environmental performance of a product, and are intended to be applied with low administrative costs and minimum verification efforts. The comprehensive GPP criteria take into account higher levels of environmental performance and may require additional administrative costs, intended for use by contracting authorities that seek to go further in promoting environmental and innovation goals (CEC, 2008). Table 9.2 shows a few examples of core and comprehensive criteria in the construction sector, which are based on the recommendations provided in the Commission's GPP toolkit. In this GPP toolkit, environmental criteria have been proposed for the design, construction, use and disposal phase of buildings (EC, 2008).

Table 9.2 Examples of core and comprehensive criteria in the construction sector

Core criteria		Comprehensive criteria	
Specifications	Award Criteria (additional points for)	Specifications	Award Criteria (additional points for)
Energy performance			
Overall energy demand of the building should be [X]% lower than the maximum defined in [relevant legislation].	Lowest energy consumption compared with that demanded in the specifications.	Overall energy demand of the building should be [X]% lower than the maximum defined in [relevant legislation]. A minimum of [X]% of energy demand must be provided by localised renewable energy sources.	Lowest energy consumption compared with that demanded in the specifications. Innovative energy efficient building services.
Building materials			
Exclusion of certain materials that contain hazardous substances. Using timber from legal sources.	Use of construction materials/products complying with certain environmental criteria.	Exclusion of certain materials that contain hazardous substances. Using timber from legal sources. Silicon-blasting agents not to be used.	Use of construction materials/products complying with certain environmental criteria. Competition around insulation properties of the proposed insulation materials.

In certain cases, the criteria have also been developed by the member states. For instance, in the Netherlands, the criteria for SPP of construction works have been developed by the NL Agency. The construction works include civil engineering structures such as permanent bridge, weir, culvert, tunnel, etc. The NL Agency has made recommendations for the preparatory (points for consideration) and specification (criteria) stages of the public procurement process. In the preparatory stage for a tender call, it recommends incorporating sustainability by considering whether the purchase is truly necessary and if a more sustainable alternative might be available. It also recommends that sustainability aspects of the project should be considered at an early stage, prior to the commencement of the procurement phase. Furthermore, the specification stage should entail the formulation of criteria for supplier qualification, description of minimum requirement pertaining to supply and service, award criteria, as well as contract stipulating the contracting provisions (NL Agency, 2010c). Ensuring sustainability by questioning the necessity of a purchase and the availability of a sustainable alternative is primarily an effort to minimise the consumption of resources. Such an effort is a significant part of GPP/SPP.

However, the study conducted by Vermande and van der Heijden (2011) shows that the minimisation of using resources has received less attention in the EU construction sector. According to their study, only Italy and Slovakia have regulations for the use of recyclable materials, while Cyprus, the Netherlands, Poland, Romania and Slovenia have plans to introduce regulations. In relation to resource minimisation, Sutton and Preece (1998) have recommended procurers to apply the principle of 'dematerialisation', which generally refers to the 'absolute or relative reduction in the quantity of material used and/or the quantity of waste generated in the production of a unit of economic output' (Cleveland and Ruth, 1998).

Bernardini and Galli (1993) have highlighted that the environment is both a major driver and beneficiary of dematerialisation. However, they have also discussed certain issues that require careful examination. One such issue includes the lack of clarity concerning the possibility of dematerialisation being counterbalanced by an equivalent increase in the number of products used. An example they cited was the contribution of microelectronics to the decrease in the size and material content of television sets, which, however, had led to a strong decrease in prices, thereby favouring the diffusion of multiple sets in households.

Moreover, Cleveland and Ruth (1998) have agreed with those analysts who have argued that it is necessary to question the 'gross generalisation' regarding material use, particularly the assurance that technical change and substitution leads to decreased material intensity and reduced environmental impact. Furthermore, they have recommended that such questions need to be addressed in the 'definitive movement' towards the direction of dematerialisation. This is an indication that the development of GPP criteria including those related to resource minimisation must address the long-term implications of the criteria and also question whether they are broad enough to achieve sustainable development. For instance, in terms of the GPP criteria concerning energy efficiency, it is relevant to discuss the study of Brookes (2000). He had argued that there is no appropriate reason behind preferentially choosing energy, from among all the resources available, for efficiency maximisation. He indicated that focusing on maximisation of energy efficiency is not a proxy for enhancing social benefit or reducing environmental damage. Furthermore, in the context of action to address global warming, he suggested that it is the level of emission of harmful gases that needs to be abated. Hence, merely concentrating on energy efficiency improvement will be a blunt approach that is not being aimed directly at reducing consumption of environmentally unfavourable fuels or energy sources. According to him, the least damaging course is to determine targets, enact the restrictive measures needed to curtail consumption, and then leave it to consumers (intermediate and final) to optimise the allocation of all resources available to them given the constraints including the enacted ones.

Nevertheless, it must be acknowledged that the EU GPP criteria related to energy performance requirements in the construction sector are not restricted to energy efficiency. In addition, they also highlight renewable energy sources. Under the comprehensive GPP criteria, the technical specifications include two options, of which the first one highlights the requirement that a certain percentage of the energy demand needs to be provided by renewable energy sources. However, under the core criteria, the requirement of renewable energy sources has been included in the award criteria and not under technical specifications. In the context of such content-based distinction between the approaches within the procurement procedure, it is important to refer to the arguments of the advocacy group ClientEarth. According to ClientEarth, such content-based distinction is fuelled, partly, by an apprehension that the incorporation of sustainability objectives into technical specifications and award criteria might get out of control, and that public procurement might lead to overloading the tenderers with additional requirements that are essentially irrelevant to the procurement itself. They recommend that there is a need to move from a content-based to a role-based distinction between technical specifications, award criteria and contract performance conditions. This indicates that these three approaches in the procurement procedure need to be distinguished on the basis of their role in the procedure and not their content. Furthermore, ClientEarth emphasises that it is necessary to remove the lists and segments of detail about the types of criteria that may or may not be technical specifications or award criteria. Furthermore, they highlight that the contracting authorities should have discretion to determine whether they include sustainability considerations as technical specification, award criteria or contract performance conditions (ClientEarth, 2012).

9.6.2 Using life cycle assessment (LCA) and environmental impact assessment (EIA) to implement GPP/SPP

In the procurement stage of the road planning process in the Netherlands, contractors are required (since April 2012) to use a particular LCA model developed by the Dutch Ministry of Infrastructure and Environment (Kluts and Miliutenko, 2012). This demand put forward by the planning authority to the potential contractors can be considered as an environmental requirement. Moreover, LCA can be used to identify environmental criteria for GPP. For instance, Tarantini *et al.* (2011) have conducted a case study on windows to define GPP criteria (see Table 9.3). They used LCA to identify the key environmental impacts of the building element (windows) and the key responsible processes. Their study revealed that LCA facilitates the highlighting of some considerations that can be used to develop a structured approach for GPP of construction products. They also indicated that an appropriate evaluation of the environmental impact of building materials and components must consider not only the impact of their production stage, but also how their technical characteristics contribute to the overall environmental performance of the building in its use and end-of-life phases. For instance, the improvement of the technical characteristics of the window (air tightness, thermal transmittance coefficient) to limit the operational energy losses is the most relevant criterion to be included in a tender.

In addition, Tarantini *et al.* (2011) have discussed that in a GPP procedure, specific criteria should be developed at the level of building element such as external walls, windows, roofs, taking into account only the technical characteristics that influence the environmental performance in the use phase, leaving aside the selected products and materials. Furthermore, the GPP criteria for construction materials and products that are part of this element should refer to the environmental impact of their production. In this way, it will be possible to aggregate GPP criteria at different construction product scales (materials, products, components, elements) and address the environmental impact of building materials at different levels.

Furthermore, the use of EIA has also been discussed. Sadler and McCabe (2002) define EIA as 'a systematic process to identify, predict and evaluate the environmental effects of proposed actions and projects'. EIA has been identified as a policy instrument that can signal to the developer a potential conflict, and also facilitate discussions of eco-efficient solutions that offset negative environmental impacts (Glasson *et al.*, 2005). Varnäs *et al.* (2009b) indicate how the EIA process facilitated GPP in a construction project in Sweden. The technical specifications for this project were prepared simultaneously with the work on the EIA. The EIA process identified and assessed the environmental impacts of the different production methods to be adopted in the project. Furthermore, the information regarding these

Table 9.3 Approach to select environmental criteria for GPP of windows (adapted from Tarantini *et al.*, 2011)

Responsible process	Key environmental impacts	Selected GPP criteria
Energy losses in use phase	• Greenhouse effect • Acidification • Photo-oxidant formation • Primary energy consumption	• Maximum value for thermal transmittance coefficient, air tightness • Improvement of thermal transmittance coefficient (award criteria)
Double glazing production		• Use of best available techniques (BAT) for glass production • Prolonged warranty for window (award criterion)
Frame production (Al) Frame production (PVC) Frame production (wood)	• Greenhouse effect • Acidification • Photo-oxidant formation • Primary energy consumption • Waste production (Al, PVC) • Hazardous chemicals (PVC, wood)	• List and weight % of window materials • Use of BAT (Al, PVC production) *Award criteria* • Limit on classified chemicals (PVC) • Declaration of recycled content (Al, PVC) • Identification and marking of plastic parts (>50 g) • Use of wood from legal sources and from sustainably managed forests • Limit on formaldehyde in wood panels
Painting process in production phase	Photo-oxidant formation	Use of low VOCs paints
Windows end-of-life	Waste production	Take-back system for windows

impacts was used to describe the environmental requirements in the tender documents. The tender documents also described the steps that the contractor should take to comply with the environmental requirements and monitoring measures.

One of the benefits of using EIA to implement GPP can be related to the rule concerning the 'link to the subject matter of the contract'. The subject matter of the contract is an important part of the procurement process as it sets out the scope of the content of the procurement contract. The European Court of Justice (ECJ) has held that award criteria stated in public procurement contracts must be linked to the subject matter of the contract. In certain cases, the ECJ has clarified that so-called 'horizontal criteria' (used to promote social, environmental, and other societal objectives) must be related to the subject matter of the contract. Furthermore, horizontal criteria that satisfy the 'linked with the subject matter' rule might be perceived as being located at the intersection between the functional and horizontal (societal, environmental) objectives of the contract (ClientEarth, 2011b).

Arrowsmith and Kunzlik (2009) have illustrated the relationship between functional and horizontal objectives to argue for the equal status of the latter and other procurement policies (such as value for money). They consider the specific functions of public authorities as organised into vertical 'silos', and the horizontal objectives as cross-cutting policies, which are not necessarily arising from the particular function of a given body but may nonetheless be advanced through the way in which it conducts its activities. They cite the example of the construction sector, where the public authority might wish to favour bidders who can demonstrate that their overseas facility does not employ child labour.

Furthermore, ClientEarth (2012) argue that it is important to affirm the 'link to the subject matter' rule in order to address the apprehension (that the incorporation of sustainability objectives into technical specifications might impose additional irrelevant requirements on the tenderers), which necessitates the adoption of content-based distinction for the approaches within the procurement procedure. ClientEarth identify this rule as an appropriate constraint on the scope of sustainable procurement objectives. In addition, they recommend that the rule should be interpreted broadly to include, for instance, aspects embedded in a product or service due to choices made in the production phase, but not necessarily visible in that product or service. In particular, there has been a resistance from the Commission to accept technical specifications that relate to production processes. Therefore, ClientEarth (2011c) have debated that where a contracting authority sees a horizontal objective related to the production characteristic category as a pre-requisite, then they should be able to include it as a technical specification. Moreover, they also emphasise that technical specifications must be precise. For instance, it will not be sufficient to require 'low environmental impact' without defining what that means.

If horizontal criteria are equated to GPP/SPP criteria, then the application of an instrument such as EIA could be one approach in the construction sector to strengthen the link between the GPP/SPP criteria and the subject matter of the contract. This indicates that EIA might facilitate in locating that 'intersection' where functional and horizontal objectives can meet. Moreover, in many cases, EIA is one of the policy instruments that aid the decision-making process. In other words, along with other documents and plans pertinent to the proposed activity, EIA report contributes to the approval/disapproval of the procurement of construction projects. Hence, from that point, EIA plays an important role in procurement. Therefore, there is the potential to extend the influence of EIA to where the actual procurement of services and products are to be made for the project, which indicates that the coordination between GPP/SPP and EIA needs to be improved. Opportunities to improve such coordination have been identified (Uttam *et al.*, 2012).

9.7 Key concerns for progress towards SPP

As GPP is expanding towards SPP in several countries, it requires a paradigm shift in the way its expansion is planned. In the context of the construction sector, there are several issues to consider. The predominant focus in GPP on certain measures such as maximisation of energy efficiency needs to be investigated for its future consequences. The conflicts within such employed measures need to be addressed. In terms of energy efficiency, it can be done in the light of concerns raised in some of the previous studies (e.g., Brookes, 2000). However, SPP should also strive to move beyond energy efficiency and emphasise the procurement of renewable energy.

Kunzlik (2009) draws some conclusions regarding the extent to which the public authorities in the EU are allowed to favour the procurement of renewable energy. He argues that even if the Commission accepts that the supply of renewable energy can be specified in a contract, it does so whilst simultaneously maintaining its position against the permissibility of requirements related to production processes and methods that do not affect consumption characteristics. However, at the consumption stage, electricity from renewable sources and that from fossil fuels are indistinguishable in terms of their polluting effects. It is only at the production stage that the electricity from renewable sources is less polluting. Thus Kunzlik argues that the distinction between production processes and methods affecting consumption characteristics and those which do not is obscuring the true position.

Moreover, such issues need to be addressed if SPP has to move beyond usually adopted criteria (as in GPP) and incorporate social and economic concerns into the procurement decisions. For instance, the employment conditions of those manufacturing the product do not necessarily impact on the physical characteristics or function of the end product but are important from a sustainability perspective. In addition, strengthening the coordination between SPP and policy instruments such as EIA could be one approach to establish the relation between the sustainability criteria and the functional objectives of the procured products or services. Furthermore, if such sustainability criteria are included as technical specifications, the bidder is required to demonstrate, prior to the contract being awarded, the ability to provide goods and services compliant with the criteria stipulated. On the contrary, specific conditions, which may be included in the contract to specify how the contract is to be performed, are a less reliable mechanism for ensuring that the conditions specified are actually complied with. Thus it is highly questionable whether environmental and social considerations related to the supply chain of the procured goods or services can be linked to the performance of a contract between the contracting authority and the product or service provider (ClientEarth, 2011c). Therefore, achieving clarity on such procedural issues would enable better implementation of SPP.

There is a need to assess the degree of inconsistency between the contracting authorities' views on environmental requirements in the final selection of contractors, and the perception the potential contractors have on the same (Michelsen and De Boer, 2009). Meehan and Bryde (2011) argue that the contractor's compliance with tender requirements is not necessarily driven by the need to share 'sustainability values'. We also underscore 'values', which we believe is crucial for SPP. There have been various efforts to identify values that are deemed necessary to sustainability. For instance, the Earth Charter Initiative has put forth four general-level values: respect and care for the community of life; ecological integrity; social and economic justice; and democracy, nonviolence and peace (Earth Charter International Secretariat, 2000). 'Yet these different efforts are broadly consistent with the conception of values as abstract ideals that define or direct us to goals and provides standards against which the behaviour of individuals and societies can be judged' (Leiserowitz *et al.*, 2006). Furthermore, value concerns cannot be separated from procedural complexity; they are intertwined (Campbell, 2006). Therefore, if SPP is to truly embed sustainability in procurement decisions, then explicit consideration should be given to the values that stimulated its establishment.

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10

Assessing the environmental impact of conventional and 'green' cement production

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Abstract: With the current focus on sustainability, it is necessary to evaluate cement's environmental impact properly, especially when developing new 'green' concrete types. Therefore, this chapter investigates the available literature on every process involved during the production of cement and its alternatives. A detailed study of ordinary Portland cement's environmental impacts is followed by an assessment of improvement potentials which can be achieved with the use of supplementary cementitious materials. Finally, the environmental impacts of alternative binders such as sulfoaluminate or magnesia cements as well as alkali activated binders are studied.

Key words: CO_2, alternative binders, supplementary cementitious materials.

10.1 Introduction

Cement production has undergone tremendous developments since its beginnings some 2,000 years ago. While the use of cement in concrete has a very long history (Malinowsky, 1991), the industrial production of cements started in the middle of the 19th century, first with shaft kilns, which were later replaced by rotary kilns as standard equipment worldwide. Today's annual global cement production has reached 2.8 billion tonnes, and is expected to increase to some 4 billion tonnes per year in 2050 (Schneider *et al.*, 2011). Major growth is foreseen in countries such as China and India as well as in regions like the Middle East and Northern Africa. At the same time, the cement industry is facing challenges such as cost increases in energy supply (Lund, 2007), requirements to reduce CO_2 emissions, and the supply of raw materials in sufficient qualities and amounts (WBCSD, 2008).

In this chapter, the environmental impact of cement production will be described. The chapter will first focus on the most common cementitious product: ordinary Portland cement (OPC) and will then evaluate the main perspective in terms of reduction of cement production's environmental impacts. To do so, we will describe the improvement perspective in the cement sector as well as the alternative products that could, at least partially, replace OPC.

10.2 Environmental impact of ordinary Portland cement

Portland cement is produced by first co-grinding a mixture of about 80% limestone and 20% clays. This is then calcined and subsequently burnt at temperatures reaching 1,450°C. During the high temperature processing, the kiln conveys nodules of calcium and silicon oxides held together by a calcium aluminate melt. Their reactions lead to a material that is quenched and is referred to as clinker. It is the grinding of this clinker, together with about 5% calcium sulphate (anhydrite or dihydrate) down to a powder with a specific surface of 300–1,000 m^2/kg that produces Portland cement. A comprehensive cement manufacturing process can be found on the website of the European Cement Association (CEMBUREAU, 2010). Details of the cement manufacturing process for a few selected countries around the world can be found in Bastier *et al.* (2000), JCR (2000), Kaantee *et al.* (2004) and Sogut *et al.* (2009).

10.2.1 Description of the production process

Raw materials such as limestone, marl or chalk which provide calcium carbonate ($CaCO_3$) are extracted from naturally occurring calcareous deposits. Small amounts of 'corrective' materials such as iron ore, bauxite, shale, clay or sand are also needed to provide alumina (Al_2O_3) and silica (SiO_2) to adapt the chemical composition of the raw mix to the process and product requirements. These raw materials are then finely ground which increases the homogeneity of the raw mix and accelerates the clinkering reactions. To further reduce the natural chemical variation in the various raw materials and reduce the clinker variability, it is also necessary to blend and homogenize the raw material efficiently. This is done in continuous blending silos.

Finally, raw meal can go through the heating process. The raw meal is first passed through a series of vertical cyclones which preheat the matter with swirling hot kiln exhaust gases moving in the opposite direction. In these cyclones, thermal energy is recovered from the hot flue gases, and the raw meal is preheated before it enters the kiln. Depending on the raw material moisture content, a kiln may have up to six stages of cyclones with increasing heat recovery at each extra stage. After the preheater, modern plants have a precalciner, where limestone is decomposed to lime and carbon dioxide. Here, the chemical decomposition of limestone typically emits 60–65% of total emissions. Fuel combustion generates the rest, 65% of which occur in the precalciner. Gartner has highlighted the fact that the decarbonation process is the most energy-consuming process during the chemical reaction (Gartner, 2004).

The precalcined meal then enters the kiln. Fuel is fired directly into the kiln to reach temperatures of up to 1,450°C. As the kiln rotates, about 3–5 times per minute, the material slides and tumbles down through progressively hotter zones towards the flame. The intense heat causes chemical and physical reactions that partially melt the meal into clinker. More details can be found in Sorrentino (2011). The clinker is then discharged in red-hot form from the end of the kiln and passed through different types of coolers to partially recover the thermal energy and lower the clinker handling temperature.

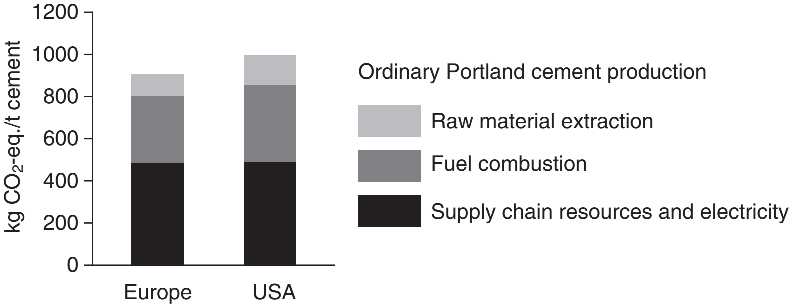
The different types of kilns are briefly explained below but more details can be found, for instance, in Szabó *et al.* (2003):

- Wet rotary kilns are used when the water content of the raw material is within 15–25%. This will make the meal more homogeneous for the kiln, leading to less electrical energy use for the grinding. However, overall energy consumption will be higher to evaporate water in the slurry. This process is still in use in some countries. However, many countries are shifting from wet kilns to dry kilns to reduce the overall energy consumption.
- Semi-wet rotary kilns are used when the wet raw material is processed in a filter after homogenizing to reduce moisture content. It is an improved version of the wet process. This is mainly used for retrofitting existing wet kilns.
- In semi-dry rotary kilns, waste heat recovered from the kiln is used to remove moisture content. Then the dried meal is fed into the kiln.
- Dry kilns with preheater include kilns with 4–6 multistage cyclone preheaters. As one part of the calcination already takes place in the preheater, it is possible to reduce the length of the kiln which will reduce the energy consumption.
- In dry kilns with preheater and precalciner, an additional combustion chamber is installed between the preheater and the kiln. This precalciner chamber consumes about 60% of the fuel used in the kiln, and 80–90% of the calcination takes place here. This reduces energy consumption by 8–11% compared to kilns without precalciner.
- Finally, a number of shaft kilns can be found in China and India. In India their share is 10%, while in China it is over 80% of the capacities. Their usual size is between 20 and 200 tonnes/day, and many of them are operated manually. Clinker quality is highly dependent on the homogenization of pellets and fuel, and on the air supply. Inadequate air supply or uneven air distribution makes combustion incomplete, resulting in low quality clinker and high CO and VOC emissions.

From the kiln, the hot clinker (1,500°C) falls onto a grate cooler where it is cooled to 170°C by incoming combustion air, thereby minimizing energy loss from the system (Zeman, 2009). The final step of the cement manufacturing process is the milling. In this step, the clinker is ground together with additives in a cement mill. All cement types contain around 4–5% gypsum to control the setting time of the product. Chemical compounds are used to improve the particle comminution during the grinding of materials (grinding aids). The most commonly used grinding aids include propylene glycol, triethanolamine, triethanolamine acetate and tri-isopropylamine. The mechanism of action of grinding aids is not known precisely, even if recent progress has been made (Mishra *et al*., 2013). Their efficiency varies with the type of grinder. The toxicity associated with the use of such chemicals must be taken into account (Bensted and Smith, 2009). The final product is then homogenized, stored in cement silos and dispatched from there to either a packing station (for bagged cement) or to a silo truck.

10.2.2 Main environmental impacts

Many environmental studies have detailed the impacts of the different processes involved in cement manufacturing (Valderrama *et al*., 2012; Josa *et al*., 2004, 2007; von Bahr *et al*., 2003; Boesch and Hellweg, 2010). Recent studies show that calcination has a predominant role in the environmental impacts of cement production (Cagiao *et al*., 2011; Chen *et al*., 2010a). Figure 10.1 shows that processes involved for the preparation of raw materials and those involved after the calcination (milling) represent no more than 20% of the impact associated with climate change. Emissions during heating are shared between 60% coming directly from the chemical decarbonation of limestone and 40% coming from fuel combustion. Similar features are observed for European and American cement production plants.



10.1 Life cycle impact assessment (LCIA) results of ordinary Portland cement production in terms of climate change (kg CO_2-eq) for Europe and USA (data from Boesch and Hellweg, 2010).

Global scale

Over the last decades, the emphasis has clearly shifted towards a global focus on climate change. A recent study has gathered the global impacts of cement production (Van den Heede and De Belie, 2012). Table 10.1 presents a summary of values found in the literature (Humphreys and Mahasenan, 2002; Hendriks *et al*., 2011; Josa *et al*., 2004; Flower and Sanjayan, 2007; Chen *et al*., 2010a; ATILH, 2002; Van Oss and Padovani, 2003; Huntzinger and Eatmon, 2009; Febelcem 2006; Gartner, 2004) for cement-related CO_2 emissions. Gartner distinguishes CO_2 emitted during the calcination process (raw material CO_2: RM-CO_2) and the CO_2 associated with energy use (energy-bound CO_2: EB-CO_2) (Gartner, 2004). Regarding EB-CO_2, the efficiency of the cement kiln plays an important role. It is directly linked to the technology used (Table 10.2). Under optimum conditions, heat consumption can be reduced to less than 2.9 GJ/ton clinker. A typical modern rotary cement kiln with a specific heat consumption of 3.1 GJ/ton clinker emits approximately 0.31 kg EB-CO_2, while this amount equals about 0.60 kg/kg clinker for an inefficient long rotary kiln burning wet raw materials with an extra heat consumption of around 0.6 GJ/ton clinker (Damtoft *et al*., 2008).

Possibilities to reduce RM-CO_2 emissions are rather limited. Partially replacing the traditional raw materials by blast-furnace slag (BFS) or class C fly ash (FA) with a higher calcium content is one option. In practice, replacement levels of about 10% are commonly reported (Habert *et al*., 2010). For a limestone replacement of 10%, the total CO_2 reductions can in theory be as high as 25% (Damtoft *et al*., 2008).

Table 10.1 Summary of CO_2, SO_2, NO_x and CKD emissions for Portland cement production expressed in g/kg cement* or g/kg clinker**

CO_2			SO_2			NO_X			CKD / PM-10		
Ref	Values		Ref	Values		Ref	Values		Ref	Values	
[1]	870	g/kg*	[3]	0.40–0.60	g/kg*	[3]	2.4	g/kg*	[3]	0.1–10	g/kg*
[2]	810	g/kg*	[5]	0.82	g/kg*	[5]	1.2	g/kg*	[5]	0.49	g/kg*
[3]	800	g/kg*	[6]	0.58	g/kg*	[6]	1.5	g/kg*	[6]	0.04	g/kg*
[4]	820	g/kg*	[7]	0.27	g/kg**	[7]	1–4	g/kg**	[7]	200	g/kg**
[5]	690	g/kg*	[8]	0.54	g/kg**	[9]	2.5	g/kg**	[9]	0.1–0.3	g/kg**
[6]	810	g/kg*							[10]	150–200	g/kg**
[7]	900	g/kg*									
[10]	895	g/kg*									
Mean	814	g/kg*	Mean	0.5	g/kg*	Mean	2.5	g/kg*	Mean	25	g/kg*

Adapted from Van den Heede and De Belie, 2012. [1] Humphreys and Mahasenan, 2002; [2] Hendriks *et al.* 2011; [3] Josa *et al.*, 2004; [4] Flower and Sanjayan, 2007; [5] Chen *et al.*, 2010a; [6] ATILH, 2002; [7] Van Oss and Padovani, 2003; [8] Huntzinger and Eatmon, 2009; [9] Febelcem, 2006; [10] Gartner, 2004.

Table 10.2 Specific thermal energy consumption in a clinker manufacturing process

Kiln process	Thermal energy consumption ($GJ / t_{clinker}$)
Wet process	5.85–6.28
Long dry process	4.60
Shaft kiln	3.70–6.60
1-stage cyclone preheater	4.18
2-stage cyclone preheater	3.77
4-stage cyclone preheater	3.55
4-stage cyclone preheater plus calciner	3.14
6-stage cyclone preheater plus calciner	<2.93

Regional scale

Regional environmental impacts include SO_2 and NO_x emissions which contribute to acid rain. Table 10.1 includes an overview of the estimated SO_2 and NO_x emisions for Portland cement according to the literature (Josa *et al.*, 2004; Van Oss and Padovani, 2003). The majority of SO_2 emitted is derived from the fuel combustion and the processing of raw materials in the kilns. However, the majority of the SO_2 leaves the kiln with the clinker (EPA, 1994) as it is absorbed due to the high alkalinity of clinker (Houghton *et al.*, 1996). The generation of NO_x and CO are mainly an output from fuel usage during clinker production. Their emissions are highly dependent on temperature and oxygen availability, and rotary kilns produce much more NO_x and less CO because of their higher operating temperature and stable ventilation compared to shaft kilns (Lei *et al.*, 2011).

Local scale

Cement kiln dust (CKD) emissions are the main contributors to the local impact. The size of CKD (0.05–5 µm) is within the size range of respirable particles (EPA, 1994). Since the diameter is smaller than 10 µm, CKD is classified as PM10. According to the Environmental Protection Agency (EPA) (1999), these fine particulates of unburned and partially burned raw materials present in the combustion gases of the cement kiln are considered as a potential hazardous waste due to their caustic and irritative nature. As mentioned in Table 10.2, the amount of CKD generated per kg of clinker produced equals about 15–20% (by mass) (Van Oss and Padovani, 2003). Nowadays, both the environmental and health risks associated with CKD can be reduced significantly by means of mineral carbonation. Sequestering carbon in CKD stabilizes the waste. The reduction in pH reduces health risks and the generation of harmful leachate (Huntzinger and Eatmon, 2009). In addition, the utilization of CKD for sequestration appears to have an advantage on the global scale, since about 7% of the carbon emissions can be captured this way. On the local scale, attention should be paid to the chromium content of cement. For instance, the sale of cement containing more than 2 ppm of soluble Cr(VI) when hydrated is prohibited by European Directive 2003/53/EC (EU, 2003). Hexavalent chromium or Cr(VI) is not stable. When dissolved, Cr(VI) can penetrate the unprotected skin and be transformed into Cr(III), which combines with epidermal proteins to form the allergen that causes sensitivity in certain individuals. The Cr(VI) content can originate from (i) raw materials and fuel entering the system, (ii) magnesia-chrome refractory blocks, (iii) wear metal from crushers containing chromium alloys and (iv) additions of gypsum, pozzolans, ground granulated BFS, mineral components, CKD and set regulators (Hills and Johansen, 2007).

Finally, a number of additional pollutants such as polychlorinated dibenzo-p-dioxins, dibenzofurans and heavy metals can potentially be released (Schumacher *et al*., 2003; Abad *et al*., 2004). Recent studies that evaluate the potential health risk for populations living in the neighbourhood of a cement plant shows that a seasonal pattern was observed with higher values recorded during the colder periods, but that the carcinogenic and non-carcinogenic risks derived from human exposure to metals and PCDD/Fs were within the ranges considered acceptable by international regulatory organisms (Rovira *et al*., 2011). Furthermore, the intensive use of alternative fuels, such as sewage sludge or municipal solid wastes which would otherwise be disposed of somehow/somewhere or refuse-derived fuels, allows a significant decrease of PCDD/Fs levels as well as in some metal concentrations (Rovira *et al*., 2010).

10.2.3 Future improvements

Alternative fuels

The use of alternative fuels for clinker production is certainly of high importance for the cement manufacturer but also for society as a whole. Alternative fuel utilization began in the mid-1980s. Starting in calciner lines, up to almost 100% alternative fuel firing at the precalciner stage was very quickly achieved. Alternative fuels are mainly used tyres, animal residues, sewage sludges, waste oil and lumpy materials. The last are solid recovered fuels retrieved from industry waste streams, and to a growing extent also from municipal sources. Waste-derived fuels consist also of shredded paper, plastics, foils, textiles and rubber.

While in some kilns up to 100% substitution rates have been achieved, in others, local waste markets and permitting conditions do not allow for higher rates of alternative fuels. In any case, their utilization requires the adaptation of the combustion process. Modern multi-channel burners designed for the use of alternative fuels allow a control of the flame shape to optimize the burning behaviour of the fuels and the burning conditions for the clinker (Wirthwein and Emberger, 2010). In a conventional preheater kiln (without precalciner), it is only possible to burn fuels in the kiln inlet with substitution rates of up to 25–30%. This is different in precalciner kilns, as usually up to 65% of the total fuel energy input is fired into the calciner and a minimum of 35% through the main kiln burner. As a consequence, when alternative fuels are used in the precalciner, it does not change the nature of the fuels introduced in the kiln and therefore does not change the kiln performance. Most operators first increase the alternative fuel substitution in the precalciner. After this, they start to increase the proportion of alternative fuels in the sintering zone firing.

In fact, when alternative fuels are used in the cement kiln, these alternative fuels are mixed with the raw meal and can have an influence on the clinker properties. A recent paper made a review of the chemical consequences of the minor elements added to the clinker by the use of alternative fuel (Sorrentino, 2011). For instance, sulphur and phosphorous have a large impact on belite content and C_4AF formation (Moudilou *et al.*, 2007; Fukuda *et al.*, 2008; Herfort *et al.*, 2010). Halogen and chloride modify the burnability of the raw meal (Maki, 2006).

A recent study has evaluated the perspective of alternative fuel use and has shown that in developed countries a ratio of 40–60% of alternative fuel in 2050 can be achieved, while in the developing countries this ratio will be around 25–35% (WBCSD, 2009). Technically, much higher substitution rates are possible. In some European countries, the average substitution rate is over 50% for the cement industry and up to 98% as yearly average for single cement plants. As fuel-related CO_2 emissions are about 40% of total emissions from cement manufacture, the CO_2 reduction potential from alternative fuel use can be significant.

Although, technically, cement kilns could use up to 100% of alternative fuels, there are some practical limitations. The physical and chemical properties of most alternative fuels differ significantly from those of conventional fuels. While some (such as meat-and-bone meal) can be easily used by the cement industry, many others can cause technical challenges. These are related to, for example, low calorific value, high moisture content, or high concentration of chlorine or other trace substances. For example, volatile metals (e.g., mercury, cadmium, thallium) must be managed carefully, and proper removal of cement kiln dust from the system is necessary. This means pre-treatment is often needed to ensure a more uniform composition and optimum combustion. However, the achievement of higher substitution rates has stronger political and legal barriers than technical ones:

- Waste management legislation significantly impacts availability: higher fuel substitution only takes place if local or regional waste legislation restricts landfilling or dedicated incineration, and allows controlled waste collection and treatment of alternative fuels.
- Local waste collection networks must be adequate.
- Alternative fuel costs are likely to increase with high CO_2 costs. It may then become increasingly difficult for the cement industry to source significant quantities of biomass at acceptable prices.
- The level of social acceptance of co-processing waste fuels in cement plants can strongly affect local uptake. People are often concerned about harmful emissions from co-processing, even though emissions levels from well-managed cement plants are lower with alternative fuel use (Rovira *et al.*, 2010). In addition, alternative fuel use has the potential to increase thermal energy consumption, for example when pretreatment is required as outlined above.

Energy efficiency

Energy demand in clinker production has been significantly reduced over the last few decades. The theoretical minimum primary energy consumption (heat) for the chemical and mineralogical reactions is approximately 1.6–1.85 GJ/t (Klein and Hoening, 2006). However, there are technical reasons why this will not be reached, for example unavoidable conductive heat loss through kiln/calciner surfaces. A critical review on energy use and savings in the cement industries can be found in Madlool *et al.* (2011). The main reason is that a significant decrease in specific power consumption can only be achieved through major retrofits, which need high investment cost with low payback potentials and strengthened environmental requirements which can increase power consumption (e.g., dust emissions limits require more power for dust separation regardless of the technology applied). As a consequence, the best available technique (BAT) levels for new plants and major upgrades are 2,900–3,300 MJ/t clinker, based on dry process kilns with multistage preheaters and precalciners (European Commission, 2010).

As the change in cement kiln size is difficult, waste heat recovery may play an important role. Currently, a large quantity of low temperature waste heat (below 350°C), approximately 30% of the total heat consumption of the system, is still not recovered and could be a promising low investment cost solution (Jintao *et al.*, 2009).

Carbon capture and storage

The use of pure oxygen instead of air can in theory result in a very significant improvement in thermal efficiency, because it reduces the volume of the exhaust gases (and their associated heat losses) by a factor of about three. It also leads to exhaust gases that are essentially a simple mixture of CO_2 and water vapour, which could easily be separated by condensation, the resulting pure CO_2 then being readily transportable or directly injectable into underground aquifers or other such potential disposal sinks. This type of approach is currently under consideration by the electric power generating industries for a new generation of coal-burning power plants, and the cement industry could in theory try to apply the same approach. However, the electrical energy required to produce pure oxygen from air with current technology is about 420 kWh/t-O_2 (ECRA, 2009). If we consider that the minimum O_2 requirement is 1 mol per mol of exhaust CO_2, this energy already represents 10–15% of the energy needed to produce clinker. Based on this consideration, oxygen enrichment would not actually save a lot of energy or CO_2 generation in cement manufacture. This situation will evidently improve as the primary energy efficiency of electric power generation plant and air separation plants improves, but this is likely to be a slow process.

Cement production

Finally, it should not be forgotten that the main impact of cement is due to its tremendous demand. In fact, 1 kg of cement only emits 0.6–0.8 kg of CO_2 which is negligible compared to other material production emissions such as aluminium or insulation materials. However, the impact of the cement industry is more important than other energy-intensive industries because of the volume of cement production. A recent study showed that the main driver for CO_2 emission reduction in the French context from 1990 to 2005 has been the reduction in cement production and not so much any technology improvement (Habert *et al.*, 2010). On the one hand, this is because investment costs are important, and on the other hand, technology improvements in the clinker production technology are limited compared to the mitigation objective.

The evolution of cement production is linked with economic activity and the levels of industrialization and infrastructure development within the country. These parameters can be expressed as an intensity of cement use that refers to the amount of cement used per unit of GDP (kg/unit of GDP). Note that a unit of GDP is here adjusted to 1,000 constant dollars (base year: 2000) and expressed in term of purchasing power parities (PPP) which are the rates of currency conversion that eliminate the differences in price levels between countries. Cement intensities differ between countries according to economic growth (GDP) and economic structure. Various studies have demonstrated that this intensity follows an inverted U-shaped curve (Vuuren *et al.*, 1999; Scheubel and Nachtwey, 1997; Lafarge, 2006; Szabó *et al.*, 2003). The intensity of cement demand will then decline in developed countries and increase in many developing countries (Taylor *et al.*, 2006).

In Western European OECD countries, the intensity of cement use is currently estimated at 21 kg of cement per unit of GDP (1000 USD, PPP 2000) and will be around 17 kg of cement per unit of GDP at the 2050 horizon (Taylor *et al.*, 2006). To give a comparison, the intensity of cement demand in China is today around 131 kg of cement by unit of GDP (1000 USD, PPP 2000) and is expected to be reduced up to the intensity of Western European countries by the 2050 horizon (Taylor *et al.*, 2006).

With assumptions on GDP evolution and on cement demand intensity, it is then possible to evaluate the evolution of cement production. A model described as VLEEM 2 (very long term energy model) has been used to make assumptions about future cement production (Chateau, 2005). It expects a strong increase in cement production in developing countries (Szabó *et al.*, 2003) and a limited increase in developed OECD countries in 2050 (Chateau, 2005). Other evaluations, following a business as usual scenario (BAU scenario) expect an increase in cement production in Western European countries until 2020 and stagnation thereafter (Szabó *et al.*, 2003). In a recent study by the IEA (2007), global cement production is expected to rise from the current situation of 3–4 Gt (between 3.86 and 4.38) in 2050 and then remain stable. These evolutions seem irrespective of fuel and CO_2 emission prices (Pardo *et al.*, 2011), which means that if cement production reduction is essential, it must be achieved through engineering solutions that could allow the same commodities to be built with less cement.

10.3 Supplementary cementitious materials (SCMs)

During cement hydration, a large amount of portlandite precipitates. This usually provides a chemical sink for CO_2, which protects steel from corrosion by maintaining a high pH in the interstitial fluid solution. Calcium hydroxide can also be effectively used to react with materials containing amorphous silica such as fly-ash, slag, silica fume, calcined clays or natural pozzolans. These pozzolanic reactions are very similar to those in Roman cements and allow replacing part of the clinker in cement by various materials that are referred to as supplementary cementitious materials (SCMs). In fact, it has to be noted that the use of pozzolanic additions in construction predates the use of SCMs in cement. Mixing calcined clays with slaked lime has been reported in construction from 7,000 BC in the Galilei area

(Israel) (Malinowsky, 1991) and in Hadrian's Wall in Britain (Guleç and Tulun, 1997).

In this section, a brief description of the different SCMs is presented and their interest in terms of environmental performance is emphasized. Generally speaking, SCMs are composed of amorphous aluminosilicate material which will react with calcium hydroxide to form calcium-silico aluminate phases, C-A-S-H (Papadakis *et al.*, 1992). This SCM can have a natural or an industrial origin.

10.3.1 Description of SCMs

Main non-renewable natural SCMs

Among SCMs with a natural origin, we can distinguish natural pozzolans which do not need to be treated to react with calcium hydroxide, and those which need an activation to react with calcium hydroxide and that we could call artificial pozzolans. The so-called natural pozzolans, used as SCMs for the production of pozzolanic cements, are pyroclastic rocks rich in siliceous or siliceous and aluminous volcanic glass. The origin of the pozzolanic activity lies in this high content of reactive silica (Massazza, 1993). Common silicate minerals are feldspar, mica, hornblende, pyroxene and quartz or olivine depending on the volcanic rock's chemical composition, but most of these minerals are easily alterable and the high porosity and specific surface area of pyroclastic rock enhance the alteration rate. Therefore, it is very common to find secondary minerals resulting from alteration of primary minerals and devitrification of the volcanic glass. Clays, zeolites, calcite and various amphiboles are classic secondary minerals. The effect of these secondary minerals on the pozzolanic activity of the natural pozzolans has already been studied (Türkmenoğlu and Tankut, 2002; Shi and Day, 2001; Perraki *et al.*, 2003; He *et al.*, 1995a). It is mainly accepted that a good pozzolanic material has low quantities of alteration minerals such as clays and zeolites (Habert *et al.*, 2008). Several techniques have been used to enhance the reactivity of pozzolanic materials and remove unreactive ones, which include calcination (Costa and Massazza, 1977), acid treatment (Alexander, 1955) and prolonged grinding (Alexander, 1960).

Among artificial pozzolans, we can distinguish volcanic rocks with an enhanced pozzolanic activity due to various treatments (Habert *et al.*, 2008) and clays submitted to a calcination process (Habert *et al.*, 2009). The pozzolanic activity of calcined clays is very much dependent on the loss of structural water which favours the creation of an amorphous structure (Ambroise *et al.*, 1987). Numerous studies have shown that there exists a specific optimal activation temperature for each clay mineral (Ambroise *et al.*, 1987; He *et al.*, 1995a, 1995b, 1996, 2000; Saad Morsy *et al.*, 1997; Kakali *et al.*, 2001; Bich, 2005). Our results confirm the existence of an optimum for each clay mineral: around 700°C for kaolinite, 750°C for palygorskite, 800°C for montmorillonite and 850°C for illite (Habert *et al.*, 2009). Muscovite and phlogopite show no pozzolanic activity.

Main industrial SCMs

The main industrial SCMs are fly ash, blast furnace slags and to a lesser extent silica fume.

FA is a by-product from the coal power industry. When coal is burning, all inorganic particles which do not burn will be released in the exhaust gas. For environmental concerns, this ash is removed from exhaust gas through an electromagnetic process. FA is then dried and stocked, before being used as a cement additive. FA consists mainly of SiO_2, but can also contain significant quantities of Al_2O_3. The amount of CaO is limited but highly variable depending on the origin of the fly ash. The ASTM C618 standard differentiates high calcium class C fly ash and low calcium class F ash. The availability of FA is quite impressive over the globe. Jahren (2007) estimates that total volume available in 2020 could be close to 2 billion tons per year, representing half of the cement needs. However, it has to be noted that these global evaluations do not take into account the local accessibility needs as this by-product cannot be transported over long distances.

Granulated blast furnace slags (GBFS) are co-produced with iron in a blast furnace. After the blast furnace processing, GBFS need to be vitrified, in order to develop binding properties suitable for its application as cement substitute (granulation process). The by-product is then ground to get a similar grain size as clinker. GBFS contain more CaO but significantly less Al_2O_3 than FA. Their high CaO content allows for the production of C-S-H with a lower C/S ratio than clinker but with binding properties. Therefore GBFS are defined as a material with latent hydraulic properties because they can be used as a hydraulic binder as soon as they are in a basic environment. The availability of GBFS is much lower than FA. We can expect between 30 to 70 Mt available per year (Jahren, 2007), which can then not in any case replace massively the clinker.

Silica fume (SF) is a by-product from the production of the silicon metal with high pozzolanic activity. It consists nearly exclusively of SiO_2 of very fine particle size. This by-product contributes to concrete improvement through two effects. The first is its high pozzolanic activity which consumes very efficiently portlandite as it is calculated that 15% of SF would completely consume the existing portlandite (Lothenbach *et al.*, 2011). The second effect is a size effect as SF is nearly ten times smaller than clinker particles; it improves the granular skeleton and creates a denser concrete microstructure, enhancing both strength and durability (Lachemi *et al.*, 1998; Müller, 2004; Song *et al.*, 2010). Due to its very high efficiency and its very small size inducing rheological constraints as soon as it is used in large amounts, SF often replaces 5% of clinker. Its availability is quite limited as its global production is estimated to just 1 million tons (Khatib, 2009). It could increase to 1.5 Mt in 2020 (Jahren, 2007), but it will always be an expansive SCM, used more to improve concrete properties in high strength concrete rather than as a SCM used for clinker substitution.

Main waste

Finally, even if the distinction between the previous industrial SCMs is somewhat artificial, we can distinguish SCMs coming from the waste treatment industry. Among them, the two main ones in terms of quantity and efficiency are: rice husk ashes and municipal solid waste incineration fly ash.

Rice husk ash (RHA) is a highly reactive pozzolan (Malhotra and Mehta, 1996) obtained when rice husks are calcinated below the crystallization temperature at 780°C (Yu *et al.*, 1999). RHA-based concrete has high strength and high durability performance (Anwar *et al.*, 2000; Sousa Coutinho, 2003; Zain *et al.*, 2011). Since each ton of rice generates 40 kg of RHA (Zerbino *et al.*, 2011), this means that the annual world rice production of almost 600 Mt can generate almost 20 Mt of RHA. Usually after calcination, the ashes are ground using a ball mill; however, Zerbino *et al.* (2011) mentioned that also unground RHA can be used to replace 15% of Portland cement with similar mechanical and durability properties. The use of ashes obtained from the calcination of other vegetable species as pozzolans in concrete has already been reported by several authors (Elinwa and Mahmood, 2002; Elinwa and Ejeh, 2004; Akram *et al.*, 2009). But as rice is the principal production in many developing countries where the cement needs are drastically increasing, it is probably the most promising vegetable ash. Note that 20 Mt is similar to GBFS and can absolutely not be compared to the 4000 Mt of cement that will be needed in 2050.

Municipal solid waste (MSW) ash is the by-product of the combustion of municipal solid waste during incineration. Two widely used processes of incinerating MSW are the refuse-derived fuel process and the mass-burning process. The refuse-derived fuel process consists of first separating metals and glass from the MSW. The MSW is then shredded and incinerated, and the generated heat is recovered to produce electricity. The mass-burning process consists of burning the MSW as it is received in the plant without waste separation or shredding. Major portions of the MSW are then transformed physically and chemically as a result of incineration. The by-product of the incineration process is ash. Ash is typically 1–30% by wet weight and 5–15% by volume of the wet MSW, depending on the nature of the incineration plant in various countries. Two types of ashes are produced as a result of the incineration process: bottom ash and fly ash. Out of the total MSW ash, bottom ash constitutes 75–80% of the total combined ash stream and is difficult to use as SCM (Li *et al.*, 2012). But MSW fly ash is used as SCM in cement (Siddique, 2010). It is a grey to black amorphous, glass-like material, which contains high levels of several toxic metals such as lead and cadmium and organic compounds (such as dioxins). These toxic metals should be encapsulated in the cement matrix (Ubbriaco and Calabrese, 1998; Qian *et al.*, 2008). The quality of MSW ash depends greatly on (i) the nature of the waste; (ii) type of combustion unit; and (iii) nature of the air pollution control device. Major elements in MSW ash are silica, calcium and iron. In addition, the ash has high chlorine, sodium and potassium contents. The main difficulties of using this source of SCM are its variability in chemical composition and its potential toxicity.

10.3.2 Environmental impact of SCMs

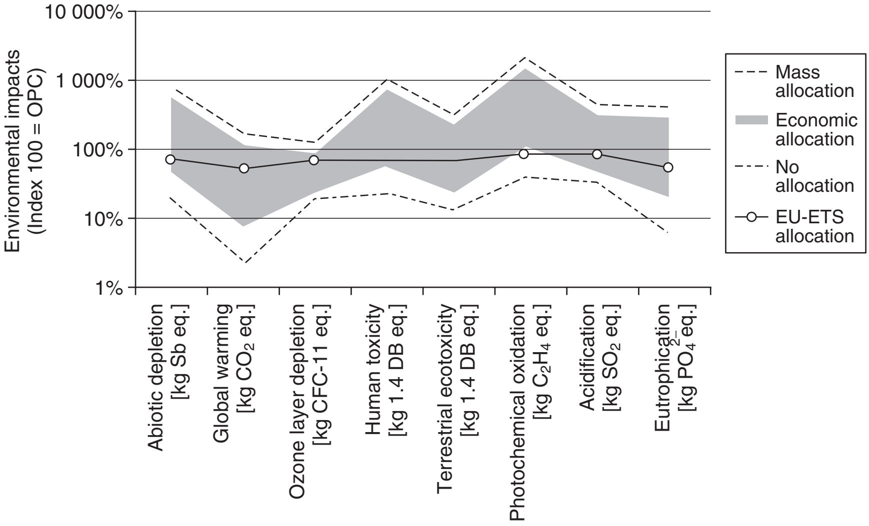
Reducing the amount of clinker in cement by the use of SCMs necessarily reduces the energy needed to produce cement as well as the amount of CO_2 released. This option also allows a cost reduction in the cement manufacturing process by saving burning costs. In this section we will just mention the aspects associated with the environmental evaluation of these alternative cementitious materials. According to ISO standards (ISO, 2006), when a production system produces more than one product, it is necessary to attribute an environmental burden to each product. This is the case for most of the SCMs and specifically to fly ash and blast furnace slags which are by-products from other industries. The question of the value of the environmental load for SCM has been emphasized by many authors and is not solved. The current practice is to consider them as waste and with that assumption to assign a null environmental impact to them, but this situation will evolve towards a practice where the different industrial sectors will have to share these environmental loads. In particular, in Europe a recent European Union Directive (EU, 2008) notes that: a waste may be regarded as by-product if the following conditions are met:

> 'Condition a) further use of the substance or object is certain;
>
> Condition b) the substance or object is produced as an integral part of a production process;
>
> Condition c) the substance or object can be used directly without any further processing other than normal industrial practice;
>
> Condition d) further use is lawful, i.e. the substance or object fulfils all relevant product, environmental and health protection requirements for the specific use and will not lead to overall adverse environmental or human health impacts.'

This Directive is very relevant to the use of SCMs such as blast furnace slags and fly ash, which now have to be regarded as a by-product because they fulfil the above mentioned conditions. In fact, their further use is certain which fulfils condition (a) of the European Directive (EU, 2008), as in some part of Europe such as France, the production of GBFS is used entirely in the cement industry and the use of FA in the cement industry is equal to 130% of its yearly production. GBFS are made from the extraction of iron from iron ore in a blast furnace, whereas it is not possible to produce iron without producing GBFS. FA is the unburnt particulates (mainly siliceous components) that are released in exhaust gas when coal is burnt in coal power plants. For sanitary reasons, these gases have to be cleaned from ashes, which are removed, and concentrates to form FA. Thus both materials are produced as an integral part of a production process and thus fulfil condition (b). GBFS are vitrified with water and ground. Fly ashes are only dried. Thus, they can be used directly without any further processing other than normal industrial practice, which fulfils condition (c). Finally, GBFS and FA that are used in the cement industry are only those which fulfil existing standards that consider their suitability in terms of mechanical performance, risk for concrete durability and risk for the environment such as NF EN 450-1 standard for FA (CEN, 2007) and EN 197-1 for GBFS (CEN, 2001). Thus, condition (d) is fulfilled. Therefore, in Europe these two materials can no longer be considered as waste but instead as by-products. Hence, the question is what is the environmental cost of these by-products?

Different scenarios can be envisaged. Dividing the impact by the relative mass value of the different products (steel/slags) induces too high an environmental load for by-products and a division in relation to the relative economic value of the products seems more in accordance with the perception of what could be the environmental load of these SCMs (Chen *et al.*, 2010b). The main problem with the economic allocation is the question of price variability as illustrated in Fig. 10.2, where blast furnace slag varies between 40 and 90€/t, and steel price varies between 150 and 1500€/t. For further reading on this subject, see Van den Heede and De Belie (2012).

It should also be mentioned that other allocation methods are proposed such as the one based on the European Union greenhouse gas emission trading system (EU-ETS) which aims at limiting CO_2, NO_x and perfluorocarbon emissions of energy intensive industries through greenhouse gas emissions trading (EU, 2009). It is interesting to note that most of the industrial activities that are submitted to the EU-ETS are either part of the cement industry or related to it through the fact that the cement industry is using their waste. Table 10.3 shows the different industrial activities from Annex 1 of the Directive, and provides a reference for each waste used in the cement industry. The allocation method is then calculated so that the economic gains and losses are the same for all of the industries involved in



10.2 Environmental impact of a mass of GBFS (1.11 kg) equivalent to the replacement of 1 kg of cement CEM I, for some of the CML indicators with different allocation procedures (adapted from Chen *et al.*, 2010b and Habert, 2013).

the trading of by-products, and underlines the overall environmental benefit of the exchanges (Habert, 2013). Results presented in Fig. 10.2 are coherent with economic allocation and show a reasonable environmental cost for the by-product which is independent from price variation.

Finally, for natural SCMs, such as pozzolan or calcined clays, there is no allocation problem and their environmental impact results only from the energy used during their processing. These impacts are always very low compared to cement.

10.3.3 Phase assemblage and sustainable cementitious materials

Finally, it must be emphasized that the most promising use of SCM is probably when they are used in tertiary blends. For instance, limestone filler can react to some extent with the aluminate phases of Portland cement and this can give a slight strength increase (Lothenbach *et al.*, 2008a, 2008b). However, the range of substitutions in which this can be exploited is relatively limited. In particular, at high cement replacement levels, most of the limestone can be considered as non-reactive, although it may affect hydration rates depending on its fineness.

However, in the case of limestone addition to fly-ash cements, it is observed that the extent of reactivity of the limestone is increased because of the alumina contained in the fly-ash (De Weerdt *et al.*, 2011). These

Table 10.3 Industrial activities submitted to the European trading scheme on greenhouse gas emissions and their relations with the cement industry. For each industrial activity, the type of waste used in the cement industry is indicated

Activities	Greenhouse gases	Waste used in cement industry	Reference
Combustion of fuels in installations with a total rated thermal input exceeding 20 MW (except in installations for the incineration of hazardous or municipal waste)	CO_2	Fly ash + refused derived fuel ashes	Chang *et al*., 1999; Barbosa *et al*., 2011
Refining of mineral oil	CO_2	Refuse-derived fuels	Karstensen, 2008
Production of coke	CO_2	Linked to coke combustion (fly ash)	Sheng *et al*., 2007
Metal ore (including sulphide ore) roasting or sintering, including pelletization	CO_2	Tailings	Yi *et al*., 2009
Production of pig iron or steel (primary or secondary fusion) including continuous casting, with a capacity exceeding 2.5 t/hour	CO_2	Granulated blast furnace slag, steel slags	Kourounis *et al*., 2007; Schneider *et al*., 2011
Production or processing of ferrous metals (including ferro-alloys) where combustion units with a total rated thermal input exceeding 20 MW are operated. Processing includes, *inter alia*, rolling mills, re-heaters, annealing furnaces, smitheries, foundries, coating and pickling	CO_2	SiMn slag and Mn oxide filter cakes	Frias and Rodriguez, 2008
Production of primary aluminium	CO_2 + PFc	Dross / sludge / red mud	Pera *et al*., 1997; Ewais *et al*., 2009

Table 10.3 Continued

Activities	Greenhouse gases	Waste used in cement industry	Reference
Production of secondary aluminium where combustion units with a total rated thermal input exceeding 20 MW are operated	CO_2	Non-metallic products and salts	Shinzato and Hypolito, 2005
Production or processing of non-ferrous metals, including production of alloys, refining, foundry casting, etc., where combustion units with a total rated thermal input (including fuels used as reducing agents) exceeding 20 MW are operated	CO_2	Slags	Shi *et al.*, 2008
Production of cement clinker in rotary kilns with a production capacity exceeding 500 t/day or in other furnaces with a production capacity exceeding 50 t/day	CO_2	Cement industry	
Production of lime or calcination of dolomite or magnesite in rotary kilns or in other furnaces with a production capacity exceeding 50 t/day	CO_2	Cement industry	
Manufacture of glass including glass fibre with a melting capacity exceeding 20 t/day	CO_2	Waste glass	Shi and Zheng, 2007; Asokan *et al.*, 2009

Table 10.3 Continued

Activities	Greenhouse gases	Waste used in cement industry	Reference
Manufacture of ceramic products by firing, in particular roofing tiles, bricks, refractory bricks, tiles, stoneware or porcelain, with a production capacity exceeding 75 t/day	CO_2	Fired bricks waste, waste gypsum	Escalante-García *et al.*, 2009; Pereira-de-Oliveira *et al.*, 2012
Manufacture of mineral wool insulation material using glass, rock or slag with a melting capacity exceeding 20 t/day	CO_2	Rock wool waste	Chen *et al.*, 2011
Drying or calcination of gypsum or production of plaster boards and other gypsum products, where combustion units with a total rated thermal input exceeding 20 MW are operated	CO_2	Linked to cement industry: calcium sulphoaluminate clinker	Kuryatnyk *et al.*, 2010
Production of pulp from timber or other fibrous materials	CO_2	Wood saw dust, wood fibres	Toledo Filho *et al.*, 2000; Turgut, 2007
Production of paper or cardboard with a production capacity exceeding 20 t/day	CO_2	Paper sludge	Pera and Amrouz, 1998
Production of carbon black involving the carbonization of organic substances such as oils, tars, cracker and distillation residues, where combustion units with a total rated thermal input exceeding 20 MW are operated	CO_2	Carbon black	Chan and Wu, 2000
Production of nitric, adipic, glyoxal or glyoxylic acids	$CO_2 + NO_x$	No known use in cement industry	

Table 10.3 Continued

Activities	Greenhouse gases	Waste used in cement industry	Reference
Production of ammonia	CO_2	No known use in cement industry	
Production of bulk organic chemicals by cracking, reforming, partial or full oxidation or by similar processes, with a production capacity exceeding 100 t/day	CO_2	No known use in cement industry	
Production of hydrogen (H_2) and synthesis gas by reforming or partial oxidation with a production capacity exceeding 25 t/day	CO_2	No known use in cement industry	
Production of soda ash (Na_2CO_3) and sodium bicarbonate ($NaHCO_3$)	CO_2	No known use in cement industry	
Capture, transport and storage of greenhouse gases in installation under Directive 2009/31/EC	CO_2	Linked to cement industry	ECRA, 2009
Aviation: Flights which depart from or arrive in an aerodrome situated in the territory of a member state to which the Treaty applies	CO_2	No known use in cement industry	

Adapted from Habert, 2013.

blended cements, also referred to as type M cements in Europe, still have margin for expansion. In class M cements, it is possible to include different SCMs. In particular, one could include a reactive and a non (poorly) reactive SCM. An example is the use of limestone filler with metakaolin. Recent studies show that 45% of substitution by 30% of metakaolin and 15% of limestone gives better mechanical properties at 7 and 28 days than the 100% PC reference because calcium carbonate reacts with alumina from the metakaolin, forming supplementary AFm phases and stabilizing ettringite (Antoni *et al*., 2012). These ternary substitutions are promising as very high rate of substitution can be achieved while still maintaining early strength properties. The early strength is actually one challenge in this area. Another is that by increasing the number of components the quality control requirements also increase.

10.4 Alternative binders

In this section we examine various cements currently being considered as potential alternatives to the more traditional cements discussed above. Their nature and degree of development vary significantly. For example, alkali-activated alumina silicates have been used for decades, but have not really achieved significant market penetration. Nevertheless, this subject is being intensely investigated, judging by the very high number of publications and citations earned in this field. In contrast, other systems are based on recent discoveries and new concepts, while one of these alternatives has reached the level of pilot testing in a full-scale cement kiln. Most of these solutions have also been discussed in other recent reviews, and readers are referred to these as complementary information (Gartner, 2004; Damtoft *et al*., 2008; Shi *et al*., 2011; Pacheco-Torgal *et al*., 2012).

10.4.1 Alkali-activated aluminosilicates

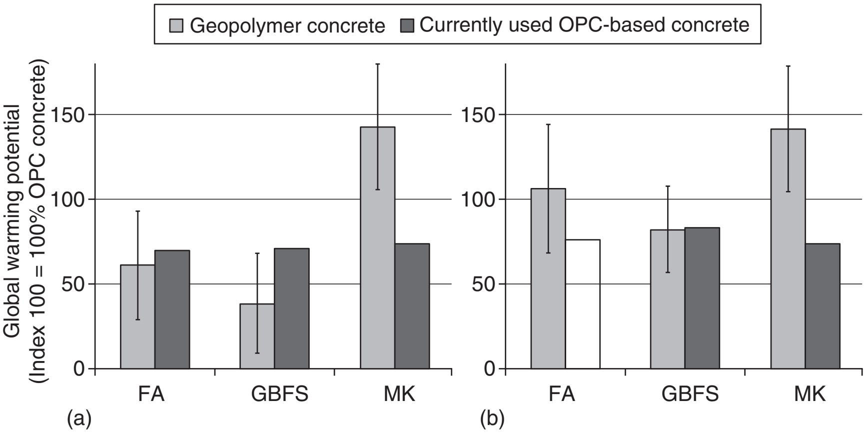
Aluminosilicates either do not react with water, or do so too slowly. However, provided these materials have a high amorphous content when they are placed in an alkaline medium, they will hydrolyse and condense, forming new inorganic polymers that can develop load-bearing capacity. In cement, these materials benefit from the natural alkalinity of the system and of the portlandite reserve to fulfil these reactions. Without Portland cement, aluminosilicates can be activated by adding strong bases. As summarized by Gartner, this has important consequences in terms of the availability of alumina and silicate species for reaction (Gartner, 2011; Sharp *et al*., 2010). The reactivity is, however, strongly influenced by the structure of these alumino-silicates, and much research has been devoted to this issue (e.g., Duxson *et al*., 2007a). Alkali-activated aluminosilicates can be classified into two groups depending on their calcium content.

1. High calcium content raw materials are mainly prepared with blast furnace slag that can react at room temperature producing a main hydration product called C-A-S-H. This is similar to the C-S-H from Portland cement but contains substantial substitution in linking tetrahedral, giving longer chains that may also be cross-linked depending on the activator used.

2. Those containing low amounts of calcium are sometimes also referred to as geopolymers. These have been used in a number of applications (Davidovits, 1982; Davidovits and Boutterin, 1982, 1983) and have been the subject of much research and interest in the last decade.

Geopolymers are formed by a two-stage reaction in which suitable aluminosilicate starting materials are mixed with a highly alkaline medium in which reactive silicate and aluminate groups dissolve. The aluminates and silicates dissolve and poly-condense into short-range ordered and cross-linked chains that form a cementitious gel and provides the binder phase in these materials (van Deventer *et al.*, 2007). Suitable geopolymer starting materials must possess high quantities of reactive silicate and aluminate groups. Such structures have a negative charge associated with the aluminium ion which is balanced by Na^+ or K^+ cations from the alkali activation solution, forming N-A-S-H of K-A-S-H gel depending on the counterion present. Alkali-activated systems are being commercially used to form concretes by Zeobond in Australia who aim to provide a practical and realistic solution by creating a sustainable alternative to the manufacture of cement. The most readily available raw materials containing aluminium and silicon are coal fly ash and blast furnace slag from the steel industry, and these are the materials that Zeobond use to produce low carbon emission geopolymer binders.

Although geopolymers are presented by many authors as a solution for 'green' concrete (Davidovits, 1999; Duxson *et al.*, 2007b), only few studies have quantified the environmental impact of geopolymers (Weil *et al.*, 2009; Habert *et al.*, 2011; McLellan *et al.*, 2011; Yang *et al.*, 2013). Most of them reported that geopolymers have a lower environmental impact than concrete. A distinction should be made depending on the aluminosilicate precursor used. A review of the available literature shows that GBFS-based alkali-activated cement has a lower environmental impact (Fig. 10.3(a)), followed by FA and metakaolin (MK) (Habert *et al.*, 2011). However, most of the time, FA and GBFS are considered as waste, and no consideration on the question of the appropriate environmental load of these mineral additions is made. In fact, when an economic allocation to fly ash or slag is considered, the environmental interest of geopolymer is considerably reduced compared to a blended cement (Fig. 10.3(b)). The other difficulty to address the environmental impact of geopolymer is the choice of the reference comparison with OPC. In practice, geopolymers are often compared to 100% OPC concrete, while more and more concrete is now produced with blended cement (20–30% SCMs). Secondly, concrete strength can vary a lot for the same amount of cement (Purnell and Black, 2012) and will therefore have nearly the same environmental impact for very different compressive strength. Depending on the chosen mix design,



10.3 Comparison of the global warming potential for alkali-activated concretes made with FA, granulated blast furnace slags (GBFS) and metakaolin (MK). The OPC global warming impact is set as a reference at 100%. The current concrete cement is made with 70% OPC and 30% mineral addition (FA, GBFS or MK). (a) FA and GBFS are considered as waste and no impact allocation is used. (b) Economic allocation is used for FA and GBFS. Data from geopolymer concrete mix designs are from Habert *et al.* (2011).

geopolymers can be compared with concrete made with much more cement per cubic meter than necessary (e.g., 290 kg of OPC + 50 kg of FA per cubic meter to reach 24 MPa; Yang *et al.*, 2013). Habert and co-authors proposed comparing cement and geopolymer concrete using the Féret equation (Habert *et al.*, 2011). It is known that the compressive strength of the OPC concrete is controlled at a first approximation by the water-to-cement ratio of the paste (De Larrard, 1999). The Féret equation use this relation to express compressive strength as:

$$f_c \approx K.Rc_{28}.\left(\frac{V_{cement}}{V_{paste}}\right)^2 \qquad [10.1]$$

where f_c is the compressive strength, K is a parameter that characterizes the aggregate quality, Rc_{28} is the specific mechanical strength of cement, V_{cement} is the volume of cement and V_{paste} is the volume of paste which includes air, water and cement. As a consequence they compare the geopolymer concrete with an OPC concrete which had the same paste volume. It allows calculation of a cement volume which can effectively provide the same strength as the geopolymer studied. This study showed that the benefit was not as important as argued in previous studies (Habert *et al.*, 2011). Recently, a very detailed study confirmed this result and could show that even for FA-based geopolymers, the CO_2 footprint was very close to that of OPC concrete (Turner and Collins, 2013). The authors showed that only 20% of improvement could be achieved. It seems, therefore, quite difficult to have a clear opinion on the environmental impact of geopolymers even if the current traditional way of producing geopolymer with the use of sodium silicate does not seem to provide a clear and significant improvement.

Finally, even if all studies seem to show that geopolymer has a lower embedded CO_2 than clinker-based concrete, other environmental impact categories seems to be less advantageous (Habert *et al.*, 2011). As most of the environmental impact from the geopolymers come from the activator when it is a sodium silicate solution, it is interesting to note Provis' argument that environmental data used for sodium silicate in all publications come from the only existing source (Fawer *et al.*, 1999), which uses old data from 1999 assessed on a production process which is not conventionally used in all the world (Provis, 2012); even though the recent study of Turner and Collins (2013) re-examined the LCA of sodium silicate production and found similar results.

10.4.2 Calcium sulfoaluminate cements

Calcium sulfoaluminate cements are well-known alternatives to OPC. They have essentially been developed in China in the 1970s. Designed by the China Building Materials Academy (CBMA), they were intended for the manufacturing of self-stress concrete pipes due to their swelling properties. Sulfoaluminate cements contain the phases belite (C2S), yeelimite or tetracalcium trialuminate sulfate ($C_4A_3\overline{S}$), and gypsum ($C\overline{S}H_2$) as their main constituents (Chatterjee, 2002; Glasser and Zhang, 2001; Wang and Glasser, 1996). When CSA cement hydrates, ettringite ($C_6A\overline{S}_3H_{32}$) is formed according to the following reactions (Odler, 2000):

In the absence of calcium hydroxide:

$$C_4A_3\overline{S} + 2C\overline{S}H_2 + 36H \rightarrow C_6A\overline{S}_3H_{32} + 2AH_3 \qquad [10.2]$$

In presence of calcium hydroxide:

$$C_4A_3\overline{S} + 8C\overline{S}H_2 + 6CH + 74H \rightarrow 3C_6A\overline{S}_3H_{32} \qquad [10.3]$$

It is known that the microstructure of ettringite is strongly dependent on the presence of lime (Mehta, 1973). Ettringite produced by the reaction in Eq. [10.3] is expansive and this property is exploited in special applications such as shrinkage-resistant and self-stressing cement (Su *et al.*, 1992). Ettringite formed in the absence of lime by the reaction in Eq. [10.2] is non-expansive and generates high early strength in cementitious systems (Beretka *et al.*, 1997).

These classical calcium sulfoaluminate cements based on yeelimite ($C_4A_3\overline{S}$) with lesser amounts of C_2S and C_4AF, should give lower raw materials-derived CO_2 emissions than OPC. In fact, considering that the thermal energy is mainly used for calcining limestone, the reduction of limestone content in the kiln feed creates a similar decrease in the energy needed for calcination. So the composition of CSA permits a production process that might be considered to be more 'environmentally friendly', than that of OPC, because:

- the quantity of limestone in the raw materials of the kiln is reduced, hence both fuel derived and raw materials-derived CO_2 emissions are reduced;
- the maximum synthesis temperatures are at least 100°C lower than those required (1,400–1,500°C) for OPC;
- the clinker is easier to grind (i.e., less energy is required for grinding) (Glasser and Zhang, 2001);
- industrial waste and secondary products can be reused more easily as raw materials (Ambroise and Péra, 2008; Zhang *et al.*, 1999).

CSA cements are thus receiving increasing attention (Gartner, 2004). Compared to alite, which releases 0.578 g CO_2 per g of the cementing phase when made from calcite and silica, calcium sulfoaluminate clinker releases only 0.216 g CO_2 per g of cementing phase when made from limestone, alumina and anhydrite. However, its production has remained stable at around 1.2–1.3 millions tonnes since 2004 in China (Diao, 2008). One of the main reasons is that more aluminium is needed in the raw material to produce CSA, which makes it more expensive to produce than OPC, and thus has limited it to various 'niche' applications which make use of special properties such as rapid hardening or self-stressing (depending on the cement formulation).

Recently, Lafarge developed an interesting approach (Li and Gartner, 2006). The clinker proposed by Lafarge and registered as Aerther® is richer in belite than classical CSA, which allows the use of less expensive raw materials. It consists of a combination of various known chemical reactions of cementitious systems, but overall leads to very different reactivity from Portland cement. Publically available information on this subject can be found in Gartner (2011), Sharp *et al.* (2010), Wang (2010) and Morin *et al.* (2011). In the first day, strength development comes from formation of ettringite ($C_6A\overline{S}_3H_{32}$) and amorphous aluminium hydroxide (AH_3) from the reaction of yeelimite ($C_4A_3\overline{S}$) with anhydrite ($C\overline{S}$) in the presence of water. Once the anhydrite is exhausted, further reaction of yeelimite yields monosulfoaluminate ($C_4A\overline{S}H_{12}$) and AH_3. Interestingly, belite seems to react with AH_3 to yield strätlingite ($C_2A\overline{S}H_8$), which occurs until about 14 days. Until this point in time, no C-S-H or CH are formed, so that the main hydrates are very different from Portland cement systems. After this, further reaction of belite does yield C-S-H and CH. It is claimed that despite these various changes in solid phase assemblage, no detrimental dimensional changes take place. Moreover, the testing of this cement is now the subject of an EU project that will include full-scale production trials (http://www.aether-cement.eu/). This material is claimed to deliver similar performance to Portland cement but is associated with a 20–30% reduction in CO_2 emissions.

10.4.3 Celitement

Celitement is a hydraulic cementitious binder developed by Karlsruhe Institute of Technology. The raw materials are similar to those used in the manufacture of conventional cementitious binders, CaO from limestone and various types of silicates that are blended at Ca:Si molar ratios between 0.5 and 2. The raw materials and water are transformed into calcium silicate hydrates using an autoclave at temperatures between 150°C and 300°C. The hydrothermal product is then blended with a second silicate component and milling activates the phases and controls product properties. This produces celitements, which are hydraulically active calcium hydrosilicates. Additives are used to control hydration reactions and the properties of the final product. The molar ratio of Ca to Si can be much less than 2 and therefore the calcium carbonate used and the corresponding CO_2 emissions are significantly reduced (Stemmermann *et al.*, 2010). The CO_2 emissions associated with celitement are reported to be approximately 50% those of normal CEM I Portland cement and compositions are reported to give low permeability mortars with compressive strengths up to 80 MPa after 28 days curing. A test plant with a capacity of 100 kg per day became operational in 2011 and, although significant barriers remain, this represents an exciting potential technology for the manufacture of low carbon cement and concrete.

10.4.4 Cements from magnesium silicates

Magnesium cements have been studied for a long time and are based on the atmospheric carbonation of a magnesium hydrate to form fibres which will provide the compressive strength (Sorel, 1867). However, it was not until a decade ago that they significantly developed, when Harrison patented reactive MGo cements (Harrison, 2003) and the production is now 14 million tonnes (compared to the 2.6 billion tonnes of Portland cement) (USGS, 2012). Extensive studies on chemistry and production of magnesia cement can be found in Shand (2006). Nesquehonite ($MgCO_3$, $3H_2O$), dypingite ($4MgCO_3Mg(OH)_2$, $5H_2O$) and hydromagnesite ($4MgCO_3Mg(OH)_2$, $4H_2O$) are the main hydrated and carbonated products. However, the main constraint on the development of these binders has been the production of magnesium hydroxide which is the main reactive phase that will then be hydrated and carbonated during setting.

In practice, magnesium hydroxides are often produced through the decarbonation of magnesium carbonate which then releases as much CO_2 as calcium carbonated (used by the cement industry). However, it should be remembered that the setting reaction for magnesium cement absorbs much more CO_2 than the hydraulic OPC reaction. It is then an interesting reaction but not free of CO_2.

However, an interesting approach is being developed by Novacem, a spin out company from Imperial College London, to manufacture the magnesium hydroxide. This process uses magnesium silicates such as olivine and serpentine as the basic raw materials for binder production. These magnesium silicates are carbonated with atmospheric CO_2 (under mild temperature and pressure conditions, 170°C/15 MPa) in order to produce the magnesium carbonates which are then decarbonated (at 700°C) for magnesium hydroxide production. Unlike previous processes, the CO_2 released during decarbonation is not a fossil CO_2 which means that the production process is roughly CO_2 neutral. The further carbonation which occurs during setting thus allows Novacem to claim that they have developed a CO_2 negative cement. Furthermore, worldwide reserves of magnesium silicates exceed 10,000 billion tonnes (Lackner, 2003), much of which is potentially extractable using open pit surface mining in a similar way and at a similar cost to limestone. Novacem aims to develop cementitious binders with performance and cost similar to Portland cement, but with significantly reduced carbon footprint.

The use of magnesium silicates combined with a low temperature production process and with the addition of hydrated magnesium carbonates enables production of a low (or negative) carbon cementitious binder. The Novacem cement is a mix of MgO, pozzolans and hydrated magnesium carbonates. There are several potentially suitable hydrated carbonates that can be used and these include artinite ($MgCO_3 \cdot Mg(OH)_2 \cdot 3H_2O$), hydromagnesite ($4MgCO_3 \cdot Mg(OH)_2 \cdot 4H_2O$), dypingite ($4MgCO_3 \cdot Mg(OH)_2 \cdot 5H_2O$), barringtonite ($MgCO_3 \cdot 2H_2O$), nesquehonite ($MgCO_3 \cdot 3H_2O$) and lansfordite ($MgCO_3 \cdot 5H_2O$), and these are produced using specialized reactor technology. The addition of hydrated magnesium carbonates controls strength development by changing the MgO hydration mechanism and the physical properties of the hydration products. Their addition also decreases the carbon footprint due to the absorption of 300–500 kg CO_2 per tonne of carbonate used. Current efforts are concentrated on the construction and optimization of a reactor system that can be increased to industrial scale and Novacem cements are not currently commercially available. However, this does represent an alternative high volume cement binder technology with potential to produce low carbon concrete products.

10.5 Balancing function and environmental impact

In all the previous sections we have been interested in evaluating the environmental impact of cement and discussed options to reduce it. However, it might be interesting to argue that in some specific case, it could be more environmentally efficient to increase the amount of cement per cubic meter and therefore to produce a material with a higher environmental impact than a reference, if it can induce, at a structural scale, a final reduction of the environmental impacts. Two brief examples are highlighted here.

10.5.1 High environmental impact for the material but high efficiency

In general, to achieve sustainability targets for the cement industry, one can argue for an improvement in cement production efficiency, a replacement of a large amount of clinker by supplementary cementitious materials, or a replacement of cement with other binders, such as sulfoaluminate clinker, geopolymers, MgO cements (Gartner and Macphee, 2011). We could also develop structures built with other less CO_2-intensive materials such as wood (Bouhaya *et al.*, 2009). However, another option is to reduce the concrete volume needed for a given construction process by enhancing the concrete performance. It means that by increasing the mechanical strength, the CO_2 emissions per cubic meter of concrete produced is increased, but the amount of concrete needed to build a given structural element is decreased.

Dimensional approaches have shown differences between structures that carry external loads and structures that carry only their own weight (Habert and Roussel, 2009), proving that for the second case, greater strength allows for a reduction in concrete volume that compensates for the CO_2 increase due to increased strength, leading to a net decrease in CO_2 emissions. In general, this dimensional approach has limitations, but studies of real structures have confirmed the hypothesis (Purnell, 2012). Another study shows that choosing a high-performance bridge construction solution to cross a four-lane divided highway with a two-lane road is always more environmentally friendly than a traditional concrete bridge solution, regardless of the observed environmental impact and the geographic context (Habert *et al.*, 2012). The study shows that the use of high performance concrete allows for a 50% reduction in greenhouse gas emissions for production of the building materials used in the bridge. A previous study applied to high rise buildings showed only a 5% reduction of the concrete volume required when 40 MPa concrete is used instead of 30 MPa concrete for vertical structural elements (Tae *et al.*, 2011). This difference is in agreement with calculations made by Habert and Roussel (2009), who showed that larger savings are reached for structures that must only carry their own loads and not for structures where external loads are dominant (which is the case for vertical elements of a building) (Habert and Roussel, 2009). In terms of sustainable construction perspectives, the design of structures using high strength concrete should therefore definitively be used for elements that must carry only their own loads in order to reach significant cement savings during the construction phase. However, high strength concrete has other advantages in terms of durability. The reduced porosity of high strength concrete reduces the concrete carbonation, which increases the time before the steel corrodes.

10.5.2 High environmental impact for the material but high durability

Addressing the question of higher durability of high performance concrete has been raised since the development of high strength concrete (e.g., Aïtcin, 2000), but few studies have quantified this improvement using the life cycle assessment methodology. Among them, an evaluation of an innovative rehabilitation system for bridges showed that this system, which uses a new UHPFRC with a large amount of limestone filler, had a similar impact as traditional rehabilitation systems without considering the service life of the rehabilitation. Furthermore, if this bridge is in use for more than 30 years, this rehabilitation system would represent less than 60% of the impact of a C30/37 concrete solution which would have needed more maintenance (Habert *et al.*, 2013).

10.6 Conclusions and future trends

The components of good concrete are extremely simple: cement, sand, aggregate, mineral additions, admixtures and water. However, as stated by Neville, the components of bad concrete are the same and the difference lies in the know-how (Neville, 1996). The first step towards a sustainable use of resources and a reduction in cement consumption is thus to make durable concrete. Today, this is not so much an issue for research as of implementation in practice.

The second step towards reducing the environmental impact of cement is to improve cement plant efficiency. In this chapter, we have seen that it is possible, especially in developing countries where the majority of cement production is located and where there are still cement plants with wet rotary kilns. However, an increase in energy efficiency requires a large capital investment, which will then probably not be done, except for waste recovery systems. Furthermore, an increase in alternative fuel use has a great potential as soon as cement plants have preheater and the waste supply chain is organized. Another improvement to the environmental impact of cement production can be achieved with clinker substitution. This solution is very efficient from a technological, economic and environmental point of view, as no drastic change in the production process is needed.

The third step is the development of new low carbon binders. Many possibilities are currently under development: magnesia and sulfoaluminate cements or alkali-activated binders. Any of these alternatives has to involve materials that are widely available around the earth if the objective is a significant replacement of OPC production, and in that sense Novacem cements are potentially interesting.

Finally, the future will probably consist of a variety of solutions, and not all of them will come from the cement composition. In fact, it has been highlighted at the end of this chapter that efficient saving potentials can be achieved with materials that are intrinsically less environmentally friendly than common concrete but which are used in such small amounts or for such a long period of time that these cements can ultimately reduce the environmental impact of a concrete structure. An appropriate use of the appropriate material at the right place is then probably the way forward.

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11

Life cycle assessment (LCA) of concrete made using recycled concrete or natural aggregates

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Abstract: This chapter is devoted to comparative environmental assessment of structural concrete made with recycled concrete aggregates and structural concrete made with natural aggregates. The results of life cycle assessment of concretes with three different types of aggregate: natural gravel, natural crushed and recycled concrete aggregate are presented. The influence of the transport phase and CO_2 uptake during the life cycle of concrete structures are analyzed. Conclusions on eco-efficiency of recycled aggregate concrete compared to natural aggregate concrete and recommendations for future research are given.

Key words: recycled concrete aggregate, natural aggregate, recycled aggregate concrete, life cycle assessment, transport, CO_2 uptake.

11.1 Introduction

Concrete is the most widely used construction material today. It is estimated that roughly 25 billion tons of concrete are produced globally each year, or over 3.8 tons per person per year (WBCSD, 2009). Twice as much concrete is used in construction around the world as all the other construction materials added together. From the total amount of waste generated in Europe each year, about 40% is construction and demolition (C&D) waste (WBCSD, 2009). According to its share in the global utilization of various construction materials, concrete waste makes up the major part of C&D waste.

Concrete is very durable construction material that can last for hundreds of years in some applications. The specific amount of harmful impacts embodied in a concrete unit is, in comparison with other construction materials, relatively small. However, due to the high global production and utilization of concrete, the final negative environmental impact of concrete structures is significant: large consumption of natural resources (aggregates for cement and concrete and energy, fossil fuels especially) and large amounts of waste generated.

Demolished concrete can be recycled, although it can not be recycled back into its original constituent materials or original whole form. Rather, concrete is crushed into aggregate called recycled concrete aggregate (RCA) for use in new applications. At present, RCA is mostly used in unbound applications for road base, back fillings, etc. But it can also be used, under certain conditions, as partial or full replacement of coarse natural aggregate in new structural concrete.

Recycling of demolished concrete into aggregate tends to produce an environmental benefit by preserving natural aggregate resources and by waste reduction, i.e., preserving the landfill space. Also, there is potential for reducing the transportation burdens, since concrete can often be recycled on demolition sites or close to urban areas where it will be reused. On the other hand, the recycling process itself and the eventual higher cement demand in structural concrete made with RCA, bring new environmental burdens. An environmental assessment of a full life cycle is needed to provide an answer to a question: can recycling really decrease the environmental burdens of concrete?

Environmental assessment of structural concrete made with recycled concrete aggregate and comparison of environmental impacts of such concrete and conventional concrete made with natural aggregates is presented in this chapter. The well recognized and standardized methodology – life cycle assessment – is applied.

11.2 Life cycle assessment (LCA) of recycled aggregate concrete (RAC)

In this chapter, application of the life cycle assessment methodology in the environmental assessment of recycled aggregate concrete for structural use is presented. Firstly, basic information on the properties of this construction material is given.

11.2.1 RAC

Recycled aggregate concrete (RAC) is concrete made with recycled concrete aggregate (RCA) instead of natural aggregate (NA). RCA are aggregates obtained by recycling of clean concrete waste where content of other building waste must be very low – less than a few percent. For example, British standard BS 8500-2 (BSI, 2006), defines RCA as recycled aggregate with maximum masonry/fines content of 5%, maximum lightweight material/asphalt content of 0.5% and maximum other foreign materials content of 1%.

The replacement of NA with RCA can be total (100%) or partial (<100%). However, the use of fine RCA below 2 mm is uncommon in recycled aggregate concrete because of the high water demand of the fine material smaller than 150 μm, which lowers the strength and increases the concrete shrinkage significantly. This high water absorption and high cohesion of fine RCA also makes concrete quality control very difficult. Therefore some standards and specifications forbid the use of fine RCA in recycled aggregate concrete for structural use (DAfStb, 2004; BSI, 2006).

Recycled concrete aggregates are produced in stationary recycling plants similar to those used for natural crushed aggregate production. Processing usually includes two-stage crushing (primary with jaw crushers and secondary with impact crushers), removing the contaminants and screening. After primary crushing, the residual reinforcement is removed by large electromagnets. All types of contaminants, such as dirt, plaster, gypsum and other building waste must be carefully removed by water cleaning or air sifting. RCA can also be processed in mobile recycling plants. They are typically used for demolition sites with large amount of homogeneous waste which is going to be reused on site (rebuilding of roads and highways, large industrial facilities). In mobile recycling plants, processing is limited to one-stage crushing, magnetic separation and screening.

When demolished concrete is crushed, a certain amount of mortar and cement paste from the original concrete remains attached to stone particles in the recycled aggregate. The presence of adhered mortar on the surface of crushed concrete aggregate generally degrades the quality of the recycled aggregate and consequently the fresh and hardened properties of concrete made from it. Compared to natural aggregates, recycled concrete aggregates have (Marinković *et al.*, 2012):

- decreased density up to 10%,
- higher porosity and higher water absorption: for coarse RCA it ranges from 2% to 9% and for fine RCA from 5.5% to 13%,
- increased Los Angeles abrasion loss up to 70%.

Besides, since the origin of RCA is usually unknown, care should be taken about its chemical properties, such as the content of chlorides, sulphates and other salts and alkali silica reactivity.

Standard methods used for the mix proportioning of natural aggregate concrete (NAC) can be used for the design of RAC mixes too. However, RCA has a high water absorption which affects the workability of the RAC mix. At higher levels of coarse aggregate replacement, the water demand of the RAC will increase for a given workability and either a small increase in cement content may be necessary, or alternatively the use of water-reducing admixtures to maintain target strength requirements. The increase in water demand will depend on the RCA properties, i.e. its absorption.

Over the last 10 years, a significant volume of research in the area of RAC properties has been performed. Published test results vary in relatively wide limits and sometimes are even contrary. In his Masters thesis on a comparative analysis of the latest experimental research on the use of RCA in structural concrete, where he compared 103 studies during last ten years, Pryce-Jenkins (2011) found:

- compressive strength is typically lower by 5–20%; commonly there is little effect below 30% replacement,
- tensile strength is typically lower by 0–30%; commonly there is little effect below 50% replacement,
- modulus of elasticity is typically reduced by 15–30%, although limit values reported are 5% and 45%; commonly there is little effect below 20% replacement,
- increased shrinkage typically by 10–20%,
- increased creep by 25–50%,
- increased water absorption typically by 40–50%,
- no effect on carbonation or even improved resistance of RAC compared to NAC,
- same or slightly decreased (up to 10%) freezing and thawing resistance,
- increased chloride penetration typically by 50–70%.

All results are given for RAC with 100% coarse aggregate replacement, compared to reference natural aggregate concrete. Properties of RAC with fine RCA are generally lower than properties of RAC with coarse RCA, especially absorption and shrinkage.

It should be emphasized that the published test data on durability-related properties of RAC are rather limited. It is obvious that RAC properties are generally lower than the properties of corresponding NAC, mostly due to the lower quality of RCA. However, this reduction is not so high as to prevent RCA from being used as aggregate in low-to-medium strength structural concrete.

11.2.2 LCA of concrete

As was already stated, concrete has a large impact on the environment because of its enormous production and utilization. That is why the environmental assessment of concrete is of great importance in the efforts towards a sustainable society. Recycling of demolished concrete is not an 'environmentally friendly' option by default. The impacts from RCA production and from the production of concrete with such an aggregate must be evaluated as for any other product or service. Today, the environmental impacts of products and processes are assessed for their whole life cycle:

from the raw materials acquisition, material production and construction, use phase and the end-of-life phase. Up to now, many different methodologies for environmental assessment have been developed, but the most acknowledged (and standardized) is the life cycle assessment (LCA). According to ISO standards (ISO, 2006), LCA consists of four steps: goal and scope definition, creating the life cycle inventory (LCI), assessing the environmental impacts (LCIA) and interpreting the results. All these steps are normally performed when assessing the environmental impact of concrete and concrete structures. In the first step, special attention should be paid to the choice of functional unit – it must reflect all functional aspects of concrete: workability, strength and durability (Van de Heede and De Belie, 2012). In the second step, allocation rules must be followed if by-products from other industrial processes are used for concrete production or in the case of concrete recycling. For assessing the environmental impacts (LCIA step), problem-oriented methods such as the CML baseline method (Guinée *et al.*, 2002) are commonly used.

Research carried out so far in this area has shown, without exception, that cement production is the largest contributor to all environmental impacts of concrete due to large CO_2 emissions during the calcination process and the burning of fossil fuels. This is why most of the research was focused on lowering the CO_2 emissions from cement production, which can be done in several ways (only the most investigated options are listed here):

- by replacing part of the cement clinker with supplementary cementitious materials that are by-products or waste from another industrial process, such as fly ash or blast furnace slag;
- by total replacement of Portland cement with alkali-activated binders, popularly called geopolymers: metakaolin, fly ash and blast furnace slag are commonly used as prime materials for alkali activation (Van Deventer *et al.*, 2012);
- by applying the high strength concrete instead of 'ordinary' concrete;
- by carefully considering the concrete mix design in detail.

Replacing natural aggregates with recycled concrete aggregates can also lead to decrease of concrete environmental impact but through preservation of natural resources and waste disposal minimizing.

However, the results of the environmental assessment of concrete applying LCA have so far shown the following:

- Replacing part of the cement clinker with fly ash (FA) or blast furnace slag (BFS) does not necessarily lead to a decrease in CO_2 emissions and hence to environmental impact decrease. Since FA and BFS are no longer considered as merely waste, but as useful by-products, they carry a part of the environmental load of the primary product production (coal-fired power generation in the case of FA and steel production in the case of BFS). If mass allocation is applied, the FA and BFS environmental burdens become higher than the burden of ordinary Portland cement and this can certainly discourage producers from implementing these materials as cement clinker replacement. That's why the economic allocation is recommended, as it results in much (an order of magnitude) lower impacts of FA and BFS (Van de Heede and De Belie, 2012).

- Significant reduction in CO_2 emissions can be achieved with alkali-activated instead of ordinary Portland cement (OPC) binders. While for OPC binders, CO_2 emission ranges from 300 to 900 kg per ton (depending on the OPC content), for geopolymer cement it ranges from 100 to 300 kg per ton (Van Deventer *et al.*, 2012). However, other research (Habert *et al.*, 2011) showed that 'geopolymer' concrete compared to OPC concrete had a higher environmental impact regarding impact categories other than global warming, due to the heavy impacts of the sodium silicate solution production. Further research is needed in this area.
- Applying a high strength concrete instead of 'ordinary' concrete results in higher cement content per cubic meter of concrete, but also in a lower total concrete volume (smaller size of elements) and longer service life of the structure (better durability related properties). Habert *et al.* (2012) showed in their specific case study of one concrete bridge structure in France that the use of high strength concrete (with compressive strength of 60 and 80 MPa) allowed for a 50% reduction in greenhouse gas emissions for production of the building materials compared to the use of 'ordinary' concrete. However, the application of high strength concrete for the structures with low-to-middle spans and loads (typical low-rise residential and office buildings, etc.) is not technically and economically justified.
- The proper choice of concrete mix recipe can bring considerable environmental benefits, for example by using a concrete with lower workability when it is possible, employing a superplasticizer, using an adequate aggregate type and using blended cements with higher strength. Purnell and Black (2012) showed that applying a superplasticizer can reduce the overall concrete CO_2 emissions by 26%. Since a given workability can be achieved at reduced water content, the cement content is accordingly reduced for the same water-to-cement ratio (strength). This saving can be achieved because the CO_2 emissions from superplasticizer production are negligible.

For detailed information on LCA aspects of concrete see also FIB TG3.6 (2008), Hájek *et al.* (2011) and Marinković (2013). Here attention is paid to replacing the natural with recycled concrete aggregates as one of the possible options for decreasing the concrete environmental impact. In the following, the case study on the environmental assessment of ready-mixed RAC and NAC production in Serbia is presented.

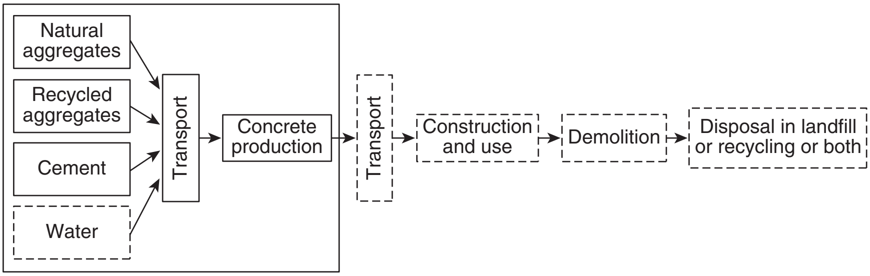
11.2.3 Case study: LCA of natural and RAC production in Serbia

This study was performed with the aim of assessing the environmental impact of the RAC production and to compare it to the environmental impact of the NAC production in Serbia. The environmental assessment was carried out using LCA, and particularly for the impact assessment, the CML baseline method (Guinée *et al.*, 2002) was used.

Four different types of ready-mixed concretes were analyzed:

- two types of natural aggregate concrete: NAC-GA made entirely from river aggregate (river sand and gravel) and NAC-CA made entirely from crushed aggregate;
- two types of recycled aggregate concrete: RAC50 made from river sand and 50% replacement of natural coarse with recycled coarse aggregate and RAC100 made from river sand and 100% replacement of natural coarse with recycled coarse aggregate; in other words, in both types of RAC the fine aggregate was natural, river sand.

The analysis was limited to a 'cradle-to-gate' level (extraction and production of constituent materials and production of concrete) and the system boundaries are shown in Fig. 11.1. Recycling was assumed to be performed in mobile recycling plant on the demolition site and for each campaign of 2,500 tones of waste, the mobile plant (20 t) is transported a distance of 200 km. The recovery rate was assumed to be 60% (Nagataki *et al.*, 2004), which means that from 1 kg of demolished concrete, 0.6 kg of coarse RCA can be obtained, and the rest is disposed of in landfill. Therefore, RCA



11.1 Life cycle of a concrete structure and system boundaries in the case study.

production included the recycling process itself, transportation of mobile recycling plant to demolition site and landfilling of the recycling waste which cannot be used as RCA. The subsequent life cycle phases (construction, use and end-of-life) were omitted since they strongly depend on the type of concrete structure to be made. However, the results of the 'cradle-to-gate' analysis can then be used as input data for the complete LCA of the specific concrete structure.

To enable the choice of functional unit only in mass units (in this case one ton or cubic meter of concrete), it is necessary that all analyzed different types of concrete fulfil the same functional requirements. This means that they must have the same strength (mechanical properties), workability and durability. For that reason, the mix proportions of natural and recycled aggregate concretes were determined so that all types of concrete have the same compressive strength and workability. Besides, it was assumed that the durability of the analyzed concretes was similar if they were exposed to non- or low-aggressive conditions.

LCI data for aggregate, cement and concrete were site-specific, obtained from local suppliers whose products were used for concrete mix (Marinković *et al.*, 2008). Emissions data for diesel production and transportation, natural gas distribution and transport that could not be collected for local conditions were taken from the GEMIS database (Öko-Institut, 2008).

The environmental impact categories included in this work were: global warming (GW), eutrophication (E), acidification (A) and photochemical oxidant creation (POC) – summer smog. Furthermore, cumulated energy requirements were calculated and expressed as 'energy use'. They were calculated using Excel-based software made for life cycle inventory and life cycle impact calculations. As already mentioned, for category indicators calculation, the CML methodology (Guinée *et al.*, 2002) was used. Regarding the RCA production, the cut-off rule was applied, meaning that no impacts from parent NAC production and all impacts from the recycling process were allocated to the RAC production (Marinković, 2013).

Transport distances were estimated for the construction site located in Belgrade, the capital of Serbia. For this case, the typical transportation distances and types are as shown in Table 11.1. Regarding the RCA transport distance (15 km), it was in fact assumed that both demolition site (which was a source of demolished concrete) and construction site were located in Belgrade.

Tests on concrete properties

The following component materials were used for all four types of concrete:

Table 11.1 Transport distances and types

Material	Route		Transport distance (km)	Transport type
	From	To		
River aggregate	Place of extraction	Concrete plant	100	Ship 10,000 t
Crushed aggregate	Quarry	Concrete plant	100	Truck 28 t
Cement	Cement factory	Concrete plant	150	Truck 28 t
Recycled aggregate	Recycling plant[a] (demolition site)	Concrete plant	15	Truck 28 t
Waste from recycling	Demolition site	Landfill	30	Truck 28 t
Mobile recycling plant[b]		Demolition site	200	Truck 28 t

[a] Recycling is performed in mobile plant on the demolition site.
[b] For each campaign of 2,500 t, the mobile plant (20 t) is transported a distance of 200 km.

- Portland cement CEM I 42.5 R,
- two types of fine aggregate (size 0/4 mm) – from Morava river for NAC-GA, RAC50 and RAC100 and from Kovilovaca quarry for NAC-CA,
- three types of coarse aggregate (size 4/8 mm, 8/16 mm and 16/31.5 mm): gravel from Morava river for NAC-GA, crushed aggregate from Kovilovaca quarry for NAC-CA and recycled concrete aggregate for RAC50 and RAC100,
- water.

Kovilovaca quarry is located approximately 100 km from Belgrade.

Coarse recycled aggregate was obtained from demolished concrete bridge structures and waste laboratory specimens, where the properties of parent concretes were not known. Mixing of different original concretes was an attempt to reproduce the situation at a real recycling plant, where RCA is produced from different quality parent concretes originating from various demolition sites. Crushing and screening into three fractions was performed in a mobile recycling plant, the only one that is operating in Serbia at the moment. The results of testing the RCA properties are shown in Table 11.2. Due to careful and selective demolition of the bridge structure, concrete rubble was obtained with practically no impurities and hence the properties of the RCA were satisfactory, similar to the properties of the gravel, which are also shown in Table 11.2 for comparison. The only property that is

Table 11.2 Coarse recycled concrete aggregate and gravel properties

Property	Unit	Type of aggregate	Fraction		
			4/8	8/16	16/31.5
Crushing resistance (in cylinder)	Mass loss (%)	RCA	23.0	26.1	32.7
		NA – gravel	21.0	24.0	28.9
Loose bulk density	(kg/m^3)	RCA	1,132	1,260	1,520
		NA – gravel	1,406	1,463	1,814
Volumetric coefficient	(%)	RCA	29.0	20.7	28.6
		NA – gravel	27.4	27.1	29.7
Fines content	(%)	RCA	0.38	0.38	0.0
		NA – gravel	0.23	0.08	0.12
Water absorption after 30 minutes	(%)	RCA	4.1	4.0	3.7
		NA – gravel	0.8	0.6	0.5

significantly different is water absorption, which is typically much higher for RCA (3.7–4.1% in this case) than for NA (0.5–0.8% in this case) and special measures must be undertaken to provide the required workability of RAC.

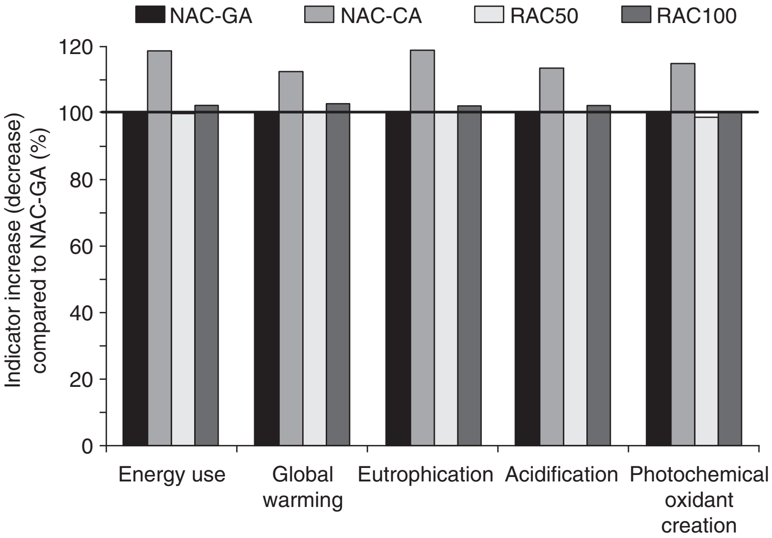
The target values of selected properties for all concrete types were: compressive strength at 28 days equal to 42 MPa and slump measured after 30 minutes equal to 8 ± 2 cm. A series of trial mixes for each concrete type were tested to obtain the required properties, and final mix proportions and properties are shown in Table 11.3. Both natural and RCA in oven dried condition were used in all concrete mixes. Because of the high water absorption value of RCA, it is necessary to add a certain amount of water to saturate it, before or during mixing. In this case, the additional water calculated on the basis of RCA water absorption after 30 minutes was added during mixing. However, the water-to-cement ratio shown in Table 11.3 refers to 'free' water content, excluding the amount of additional water. No water-reducing admixtures were used.

From Table 11.3 two important facts should be noticed. Firstly, it was not possible to obtain the same workability for the NAC with crushed aggregate as for the NAC with gravel, without increasing the amount of water. The increase in water content led to the increase in cement content to preserve the same compressive strength. That is why almost 9% more cement was needed for the NAC with crushed than for the NAC with gravel aggregate. The obvious solution to this problem was to apply the water-reducing admixtures, but this was not the intention here for the sake of fair

Table 11.3 Mix proportions and properties of NAC and RAC

Type of concrete	Cement	Water	Aggregate			w/c[a]	a/c[b]	Density	Slump after 30 minutes	Compressive strength at 28 days
			Fine	Coarse						
				Natural	Recycled					
	(kg/m^3)	(kg/m^3)	(kg/m^3)	(kg/m^3)		/	/	(kg/m^3)	(cm)	(MPa)
NAC-GA	354	185	600	1,164	–	0.524	4.983	2,400	8.0	43.7
NAC-CA	384	201	592	1,165	–	0.524	4.576	2,354	8.8	41.5
RAC50	354	185+20[c]	598	555	555	0.524	4.825	2,342	9.7	44.2
RAC100	365	180+38[c]	576	–	1,071	0.511	4.512	2,310	9.7	42.5

[a] Water-to-cement ratio.
[b] Aggregate-to-cement ratio.
[c] Additional water amount.



11.2 Category indicator results increase (decrease) compared to NAC-GA.

comparison. Secondly, while it was possible to obtain RAC50 with the same cement content as for NAC-GA, for RAC100 a slightly larger cement amount (about 3%) was needed.

Results and discussion

Figure 11.2 shows the impact category indicator increase or decrease (in percentages) of NAC-CA, RAC50 and RAC100 over the impact category indicators of NAC-GA. In this figure all category indicators of NAC-GA are presented as a 100% value, while category indicators of other concrete types are calculated as percentages of increase or decrease compared to NAC-GA.

Natural aggregate concrete with crushed aggregate (NAC-CA) has the largest indicators for all impact categories, while the natural aggregate concrete with gravel aggregate (NAC-GA) has the lowest indicators. The increase in all category indicators of NAC-CA compared to NAC-GA ranges from 12% to 18%. There are two reasons for this result: higher cement content in NAC-CA compared to NAC-GA and different transportation types. In the NAC-CA case, the assumed transportation type was truck, a much more polluting vehicle than a ship, which was assumed in the NAC-GA case. However, this result should be considered as realistic for Serbia, since in this study the attempt was made to reproduce the real transport distances and types in Serbia.

The indicators for all impact categories of recycled aggregate concrete with 50% replacement ratio (RAC50) are practically the same compared to indicators of NAC-GA (increase or decrease below 2%, which can be considered as negligible). Compared to NAC-GA, the increase in category indicators of recycled aggregate concrete with 100% replacement ratio (RAC100) is slightly larger – below 3%, but this can also be considered as negligible.

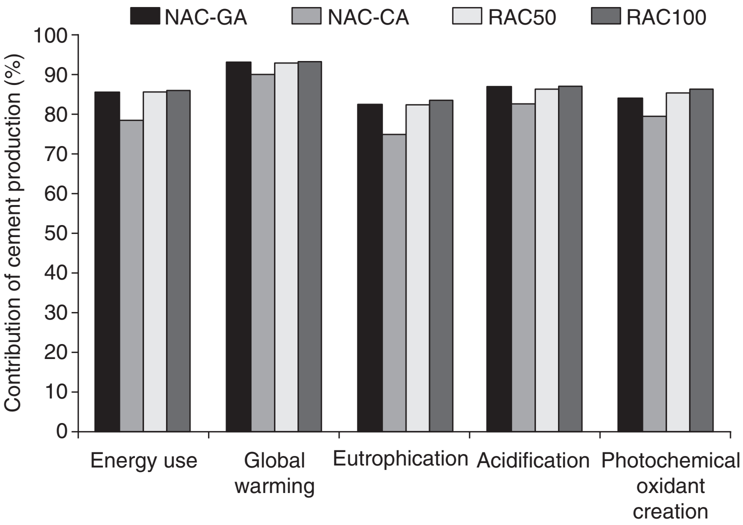
It should be noted here that in the case of RAC50 and RAC100 concretes, there is a clear benefit in terms of waste reduction and minimization of natural mineral resources depletion. For example, in the RAC100 case, the landfilling of 1,071 kg of concrete waste and extraction of 1,071 kg of natural aggregate are avoided, per 1 m^3 of RAC.

This case study was based on Serbian LCI data and typical conditions in Serbia. Within these limits, it was concluded that RCA application in structural concrete can bring environmental benefits over gravel aggregate, and certainly over crushed aggregate, but this depends on the assumed transport distances and types of natural and recycled aggregate.

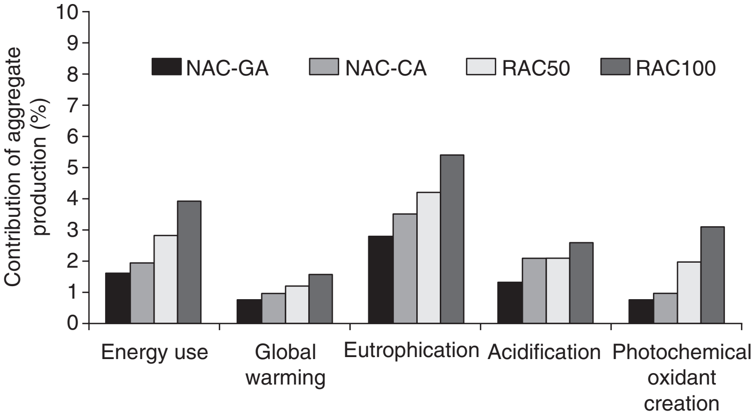
11.3 Influence of different phases in the production process for natural and recycled concrete

Figures 11.3–11.6 show the contribution to total impacts of different phases in the concrete production process, for all concrete types and calculated category indicators. The contribution of the cement production phase varies from 75% to almost 94%, depending on the category indicator and the concrete type (Fig. 11.3). The largest contribution of cement production is for RAC100 (global warming), while the lowest is for NAC-CA (eutrophication). Despite the fact that NAC-CA concrete had the largest amount of cement, the contribution of cement production is slightly smaller compared to other concrete types. This is a consequence of the fact that the contribution of transport for this concrete is significantly larger compared to other concrete types. This figure shows a well-known fact, that cement production is by far the largest contributor to all category indicators and for all concrete types. The main reason for this is the large CO_2 emissions during the calcination process in the clinker production and the use of fossil fuels.

The contribution of the aggregate production phase is small for all category indicators and concrete types (Fig. 11.4). It varies from 0.8% to 5.4% depending on the category indicator and the concrete type. However, this contribution is higher for concretes with RCA than for concretes with NA. Energy use for natural river, natural crushed and recycled concrete aggregate production is 23.8 MJ, 30.2 MJ and 62.1 MJ per tonne, respectively. More energy is consumed for the RCA production than for the production of natural aggregates and, hence, category indicators based on emissions



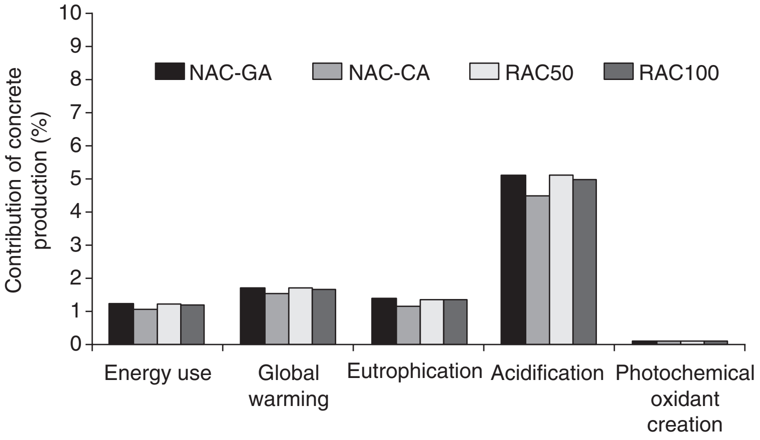
11.3 Contribution of cement production phase to category indicator results.



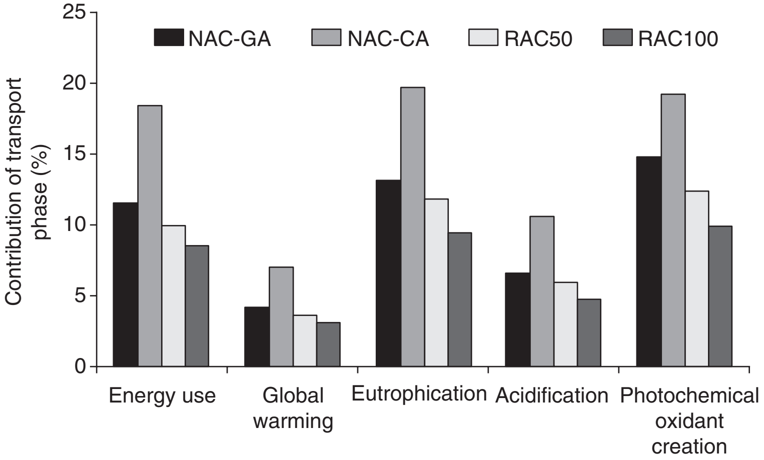
11.4 Contribution of aggregate production phase to category indicator results.

are higher in this case. Although the recycling process is similar to the crushed aggregate production process, LCI data for RCA include not only recycling, but also transport of mobile plant to demolition site and landfilling of recycling waste.

The contribution of the concrete production phase is also small and varies from 0.1% to 5.1%, depending mostly on the category indicator (Fig. 11.5). Regardless of the concrete type, for all category indicators this contribution



11.5 Contribution of concrete production phase to category indicator results.



11.6 Contribution of transport phase to category indicator results.

is below 2% of total impact, except for acidification, in which case it is about 5%.

Transportation is the second greatest source of impacts. The contribution of the transport phase ranges from 3% to 20% (Fig. 11.6). It is largest for NAC-CA (about 20% of total impact) because the assumption was made that crushed aggregate is transported a distance of 100 km by truck, which is a much more polluting vehicle than a ship.

These results also show that neglecting the transport phase, which is often omitted when performing LCA of construction materials, can lead to the wrong conclusions. LCI data for cement production are well investigated and established so far. Since the contribution of aggregate and concrete production is below 5% each, even if there are some uncertainties regarding the quality of data, this will not affect the results significantly. Therefore, the results of the LCA studies depend mainly on the assumed transport distances and types.

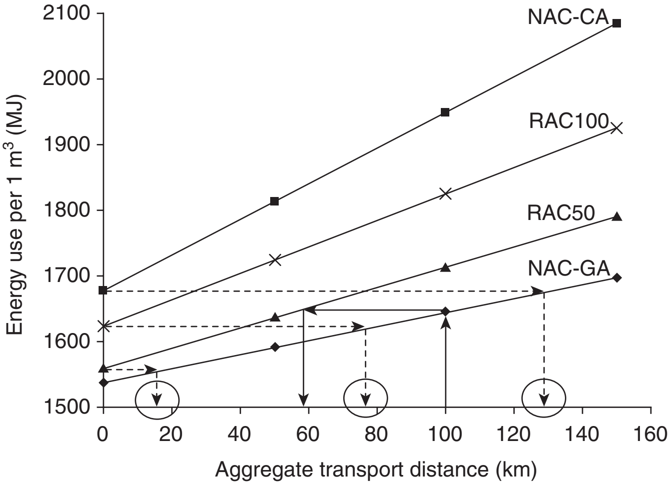
11.4 Research on the use of natural and recycled aggregates in concrete

Published research in the area of life cycle assessment of RAC for structural use is rather limited (Braunschweig *et al.*, 2011; Weil *et al.*, 2006; Evangelista and de Brito, 2007). Braunschweig *et al.* (2011) have found that environmental impacts of high quality NAC and RAC with 25% of recycled concrete aggregate are similar, as long as the increase of cement amount in RAC is small (up to a few percent). They have compared the following impacts: energy use, climate change (global warming), acidification, respiratory effects, land use and gravel use. Weil *et al.* (2006) compared NAC to RAC with 35% and 50% of recycled concrete aggregate and different cement content. Their conclusion is similar to that of Braunschweig *et al.*: for the same cement content, the energy use and global warming potential of NAC and RAC are similar. Their findings about the participation of various phases in concrete production are very similar to those obtained in this study: the cement production contribution ranges between 75% to almost 94% of the total impact, depending on the impact category indicator, where the highest contribution is in the case of global warming potential. Since the contribution of the aggregate production phase is very small compared to that, different energy consumption in production of various aggregate types does not affect the total results by more than a few percent.

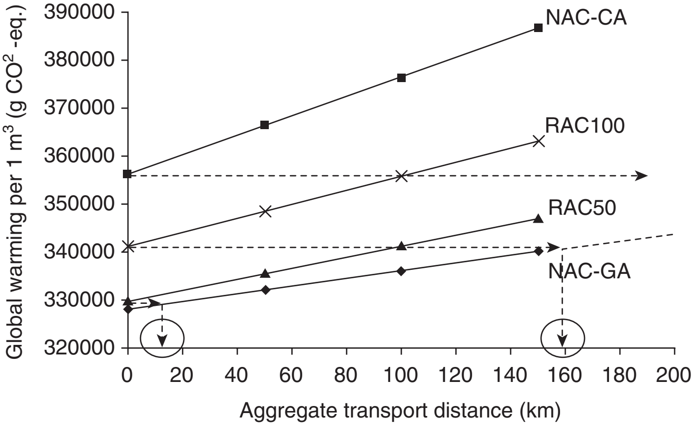
11.5 Analysis of the influence of the transport phase

To evaluate the influence of the aggregate transport distance on the environmental impacts of concrete production, the transport distance of natural and recycled aggregates from the place of extraction or recycling plant (in this case, demolition site) to the concrete plant is varied from 0 km to 150 (200) km. Results for each impact category are shown in Figs 11.7–11.11.

Concrete with gravel aggregate (NAC-GA) was chosen as the reference concrete as its impacts were lowest. Keeping all other parameters constant and varying only the aggregate transport distance, it is possible to determine the 'limit' transport distance of natural gravel aggregate. This is defined as gravel aggregate transport distance below which the environmental impact of RAC50, RAC100 and NAC-CA is higher than the environmental impact



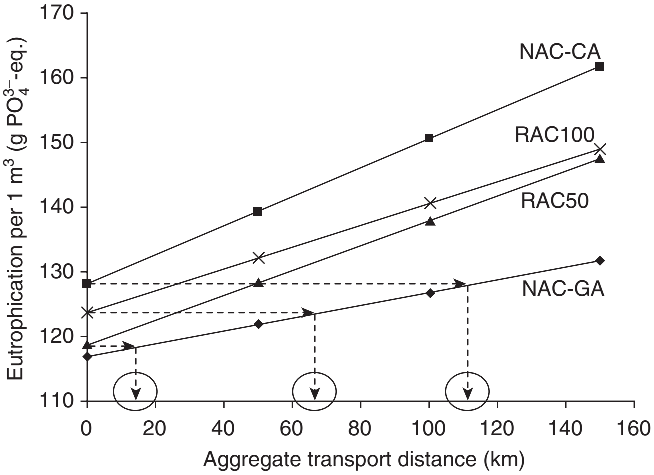
11.7 Relationship between energy use and aggregate transport distance.



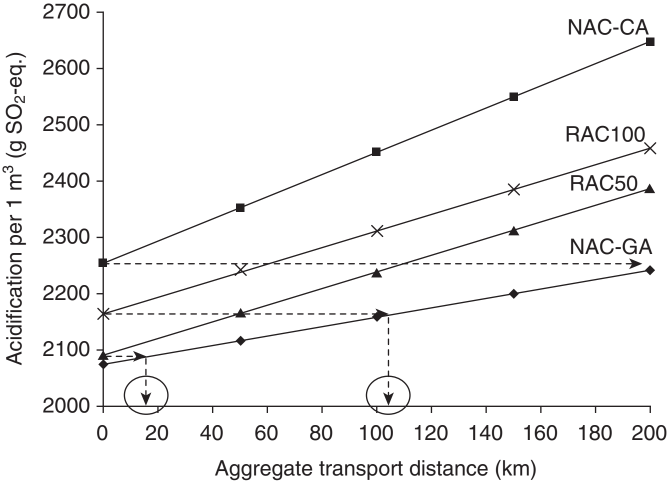
11.8 Relationship between global warming and aggregate transport distance.

of NAC-GA, regardless of RCA and CA transport distance. Figures 11.7–11.11 show the relationship between the transport distance of aggregate (natural and recycled) and calculated environmental impacts.

For example, it can be seen from Fig. 11.7 that the limit distances for gravel aggregate are about 20 km, 80 km and 130 km in the case of RAC50, RAC100 and NAC-CA, respectively. For gravel transport distances below these values, energy use for RAC50, RAC100 and NAC-CA production is higher than energy use for NAC-GA production. For gravel transport

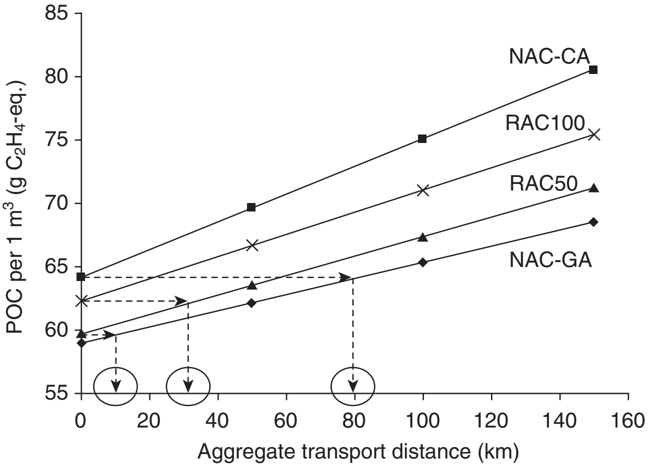


11.9 Relationship between eutrophication and aggregate transport distance.



11.10 Relationship between acidification and aggregate transport distance.

distances larger than those values, RAC50, RAC100 and NAC-CA energy use can be equal or lower than NAC-GA energy use. For instance, if the transport distance of gravel aggregate is 100 km, the required RCA transport distance is about 60 km in the case of RAC50, to get the same energy use. It should be noted here that in the case of global warming and acidification, the limit gravel aggregate transport distance for NAC-CA concrete is over 200 km (Figs 11.8 and 11.10).



11.11 Relationship between POC and aggregate transport distance.

11.6 Analysis of the influence of CO_2 uptake during the life cycle of concrete

One important issue often neglected in the environmental assessment of concrete structures is the carbonation of concrete over its life cycle. This process has been much investigated previously, primarily as a concrete durability aspect as it affects the corrosion initiation of embedded steel reinforcement. However, it is rarely taken into account in the LCA models of concrete.

11.6.1 Chemistry of the carbonation process

It is well known that cement production is responsible for large CO_2 emissions, approximately 0.8–1.0 ton of CO_2 per ton of cement. About half of this amount is emitted from the calcination process of limestone, and the other half comes from the burning of the fossil fuels in the clinker kiln, since a high temperature is needed for clinker production. The calcination process is a chemical reaction in which limestone (it mainly contains calcium carbonate) is converted to calcium oxide and carbon dioxide at high temperature:

$$CaCO_3 \Rightarrow CaO + CO_2 \qquad [11.1]$$

When exposed to air, concrete will over time reabsorb CO_2 from the atmosphere in a process called carbonation. It is the reverse chemical process to calcination, in which atmospheric CO_2 diffuses into concrete to react with calcium oxide and form calcium carbonate again. Carbonation is an inherent property of all cement-based materials. Therefore, over the life cycle, part of the CO_2 released from the calcination process will be reabsorbed or uptaken by the concrete structure (Fig. 11.12). The question is whether this is relevant for the proper environmental assessment of concrete.

11.6.2 Factors controlling concrete carbonation

Carbonation is a slow, mostly diffusion controlled process, which starts from the concrete surface and slowly penetrates into the interior of the concrete. The part of the concrete that is carbonated, measured from the surface, is called the carbonation depth and is calculated according to:

$$d_c = k\sqrt{t} \tag{11.2}$$

where:

d_c is carbonation depth, k is carbonation rate coefficient, and t is time.

The CO_2 reabsorption or uptake per m^3 of carbonated concrete can be calculated according to (Kjellsen *et al.*, 2005):

$$\mathrm{CO_2 uptake} = 0.75 \times \mathrm{C} \times \mathrm{CaO} \times \frac{\mathrm{M_{CO_2}}}{\mathrm{M_{CaO}}} (\mathrm{kg/m^3}) \tag{11.3}$$

where:

C is mass of Portland cement clinker per m^3, CaO is mass fraction of CaO in cement clinker, usually 65%, and M_{CO_2}, M_{CaO} are molar mass of CO_2 and CaO, respectively.

In deriving Eq. [11.3] the assumption was made that 75% of the original CaO of the cement is carbonated.

Factors that affect the carbonation rate are numerous (Kjellsen *et al.*, 2005), but include:

- quality (porosity) of concrete,
- exposure conditions of the surface area (relative humidity and temperature of environment, coatings, under or above ground, etc.),
- the type of cement/binder,
- the concentration of CO_2 in the atmosphere, etc.

Table 11.4 shows the carbonation rate coefficient for different concrete compressive strength and exposure conditions suggested by Lagerblad (2005), if the concrete is made with CEM I cement. For concrete made from cement with various types of mineral additions (fly ash, limestone, silica fume and blast furnace slag), Lagerblad (2005) proposed correction factors

Cement production

Service life

Crushing after demolition – RCA

11.12 CO$_2$ balance over concrete life cycle.

to be multiplied by the carbonation rate coefficient given in Table 11.4. It can be seen from this table that, for the same concrete strength, concrete exposed to indoors conditions carbonates faster than concrete under outdoors exposed conditions. This is because the optimal relative humidity for carbonation is from 60% to 80% and indoors structures are usually exposed to that humidity range (Kjellsen *et al.*, 2005). Also, concrete below ground

Table 11.4 Carbonation rate coefficient for concrete made with CEM I (Lagerblad, 2005)

Exposure conditions	Concrete compressive strength			
	<15 MPa $\text{mm}/\sqrt{\text{year}}$	15–20 MPa $\text{mm}/\sqrt{\text{year}}$	23–35 MPa $\text{mm}/\sqrt{\text{year}}$	>35 MPa $\text{mm}/\sqrt{\text{year}}$
Exposed	5.0	2.5	1.5	1.0
Sheltered	10.0	6.0	4.0	2.5
Indoors	15.0	9.0	6.0	3.5
Wet	2.0	1.0	0.75	0.5
Buried	3.0	1.5	1.0	0.75

carbonates more slowly than concrete above ground. For the same exposure conditions, higher strength concrete carbonates less than concrete with lower strength.

11.6.3 Relevance of the post-use phase (secondary life of the concrete structure)

Total CO_2 that is reabsorbed over the life cycle depends significantly on the post-use applications of the demolished concrete structure or its so-called secondary life. If demolished concrete is crushed into recycled concrete aggregate (RCA) and stockpiled for a certain period of time or used in new concrete construction, the CO_2 uptake will be higher than CO_2 uptake during service life only. The exposed surface area relative to the volume of RCA is greatly increased compared to a concrete structure itself, or landfilled concrete waste, which increases its capacity to reabsorb CO_2. However, the amount of captured CO_2 depends on the RCA application: whether it is used in below-ground applications (for sub-base and base of road structures, where it is commonly used) or as aggregate in new concrete. In the latter case, CO_2 uptake is significantly smaller and comparable to CO_2 uptake during the primary service life of the structure.

When applying LCA, care must be taken on the allocation of CO_2 uptake in the post-use phase. As with impacts from recycling, the amount of reabsorbed CO_2 in this phase should be allocated between the product that generates waste and the product which receives it, following allocation rules (cut-off, mass or economic allocation).

11.6.4 CO_2 uptake results from research

Two extensive investigations on CO_2 uptake of concrete were carried out in Europe for Nordic countries (Kjellsen *et al*., 2005) and in the USA

(Gajda, 2001). Approaches and results of these and two other studies (Collins, 2010; Dodoo *et al*., 2009) are shown in Table 11.5.

The Kjellsen *et al*. (2005) and Gajda (2001) studies were based on the survey of the volume of concrete produced, its applications, typical thickness, recycling practices, etc. in Nordic countries and the USA, respectively. The studies by Dodoo *et al*. (2009) and Collins (2010) were based on data for specific concrete structures. Only in the Gajda (2001) study was the post-use of structures not taken into account, while in the other studies analyzed, the secondary life was considered, although with different post-use RCA applications. There are also certain differences in the carbonation model, especially in the assumed percentage of CaO available to carbonation and the value of carbonation coefficient. In Collins' (2010) study, CO_2 equivalents (including other greenhouse gases) were calculated instead of CO_2 amounts. Since the author reported only the total CO_2 equivalents from cement manufacture, the figures given in Table 11.5 are calculated based on the assumption that emissions from calcination are approximately equal to one half of the total emissions from cement production.

Despite these differences, one conclusion is obvious: CO_2 uptake of concrete (compared to CO_2 emission from calcination) during service life is much smaller than CO_2 uptake during secondary life, if demolished concrete is crushed into RCA – compare 8.6% from the Gajda study to 33–57% in the Kjellsen *et al*. study or even 86% in the Collins study. Dodoo *et al*. (2009) reported that at the end of concrete building service life, CO_2 uptake was 23% of the calcination emissions, while after the concrete was crushed after demolition and exposed to air for 4 months, the absorption of CO_2 was almost twice as large – 43% of the calcination emissions. The absorption of CO_2 during the concrete bridge service life in Collins' study is only about 3% of the calcination emissions. But if the concrete is, after demolition, crushed into RCA and used in the construction of a new bridge for another 30 years, CO_2 absorption is as high as 55% and 86% of the calcination emissions, depending on RCA application.

Therefore, if demolished concrete is crushed into RCA and used in any unbound application, below or under ground, CO_2 uptake of concrete will be significantly increased compared to that from service life only. The main reason is, as explained before, the many times larger surface area of RCA than of the concrete structure. Based on analyzed research, CO_2 absorption during service life of a concrete structure is about 20% of the calcination emissions at maximum, or about 10% of the total CO_2 emissions from the cement production. If the concrete structure has a secondary life – unbound RCA application in new construction, CO_2 uptake can reach 80% of the calcination emissions, or 40% of the total CO_2 emissions from the cement production. This should certainly not be neglected in the LCA model of concrete. However, further more detailed research is needed in this area.

Table 11.5 Results of different studies on CO_2 uptake of concrete

	Kjellsen *et al.* (2005)	Gajda (2001)	Dodoo *et al.* (2009)	Collins (2010)
Type of concrete structure	Statistical data Nordic countries	Statistical data USA	4-storey concrete framed apartment building, Sweden	Bridge structure Australia
Service life	70 years	100 years	100 years	100 years
Post-use (secondary life)	30 years	–	4 months	30 years
Post-use application	RCA primarily used in unbound below ground applications (90%)	–	RCA exposed to air at stockpile	1. RCA used for embankment protection 2. RCA used for road sub-base
% of demolished concrete which is recycled	Iceland – 0% Norway, Sweden – 70% Denmark – 90%	–	90%	100%
Carbonation model				
%CaO available for carbonation	75%	21%	75%	75%
Carbonation coefficient (mm/$\sqrt{}$year)	see Table 11.4	6.9 – strength class 21 MPa 5.4 – strength class 28 MPa 3.8 – strength class 35 MPa	See Table 11.4, but for surfaces covered with paint or wallpaper decreased by 40%	3.9 – concrete with CEM I 1.5 – buried RCA
Ratio of captured CO_2 to CO_2 emitted from calcination (%)	33–57% depending on the country	7.6%	43%	1.55%[a] 2.86%[a]

[a] These figures are given for CO_2 equivalents which the authors calculated in their work, under the assumption that emissions from calcination are approximately equal to one half of the total emissions from cement production.

11.7 Conclusions and future trends

A case study on environmental assessment of four concrete types with different natural and recycled concrete aggregate was performed using the LCA methodology. The results of this specific study, along with other researchers' results, showed that the contribution of the aggregate production phase to total impacts of concrete is rather small – about 5% at maximum, depending on impact category. Although the production of recycled concrete aggregate is more energy intensive than natural aggregate production, it does not affect the results significantly.

The environmental impacts from RAC and NAC (with gravel aggregate) production are very similar under two conditions:

- the increase in cement content of recycled aggregate concrete compared to natural aggregate concrete is small, up to a few percent at most; this is possible in RAC with 100% coarse aggregate replacement if the RCA is of good quality, and certainly achievable in RAC with 50% coarse aggregate replacement.
- the transport distance of RCA is smaller than the transport distance of NA; for example, in this study the ratio of transport distances of RCA to NA was assumed to be 15 km:100 km, which means that the recycling plant must be located much closer to the concrete plant than the place of NA extraction, if similar environmental impacts are expected.

The importance of the cement content and transport distance and type is best seen with the example of natural aggregate concrete with crushed aggregates in this case study. This type of concrete had the largest impacts, due to the increased cement content and assumed transport distance and type. The influence of the transport phase was analyzed and the limit transport distance of gravel aggregate was calculated for different category indicators.

However, utilization of RAC brings environmental benefits through saving of natural aggregate resources and landfill space. These benefits cannot be expressed with specific category indicators, since most of the proposed methodologies do not include solid waste production/landfill capacity as an impact category, or consider sand and stone as abiotic resources that can be depleted.

Another benefit that can be gained with recycling is CO_2 reabsorption of concrete during the secondary life of a structure. Results of published research show that CO_2 uptake in this phase can be significant, depending on the post-use of the concrete structure. If demolished concrete is crushed into RCA and applied in new construction in unbound form for another 30 years, CO_2 uptake can reach 40% of total CO_2 emissions from cement manufacture.

Future research should be aimed towards development of special indicators for natural bulk resources depletion and landfill space depletion. Also, CO_2 uptake of concrete over the structure's primary and secondary life should be included in LCA models of concrete. Further, more detailed research is needed in this area.

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12

Life cycle assessment (LCA) of building thermal insulation materials

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Abstract: In this chapter thermal insulation materials and types of plaster and their properties are described. The impact of the selected thermal insulation materials and plaster on the environment is assessed using LCA analysis. A method of assessing the ecological and economic benefits resulting from thermal insulation of the external walls of buildings is proposed. On this basis, ecological and economic payback periods for thermal insulation are defined as well as the ecological efficiency of thermal insulation. The conducted analyses conclude that thermal insulation of the external walls of buildings is environmentally favourable.

Key words: thermal insulation materials, thermal insulation plaster, life cycle assessment, ecological and economic benefits of thermal insulation.

12.1 Introduction

The building sector both consumes a significant amount of energy and is also responsible for the emission of pollution to the atmosphere (Garcìa-Casals, 2006). Research is being undertaken to reduce the impact of building on the environment (Yan *et al.*, 2010; Dylewski and Adamczyk, 2011; Dombaycı *et al.*, 2006; Bolattürk, 2006; Dombaycı, 2007; Ucar and Balo, 2010). According to the McKinsey curve charting the reduction of greenhouse gas emissions in Poland, thermal insulation of the existing commercial and residential buildings generates important ecological benefits by achieving CO_2 reduction at low cost (Żmijewski and Sokołowski, 2010, p. 14). Thermal insulation materials and/or thermal insulation plaster are used to insulate buildings. The selection of a suitable paint to cover the façade of buildings can also influence the energy savings in summer as well as in winter (Azemati *et al.*, 2012).

Thermal insulation of buildings is a significant factor in maintaining the thermal comfort of the building's users, particularly if we take extreme temperatures in winter and summer into consideration. The insulation reduces undesirable losses of warmth (in winter) or excessive heat (in summer) and decreases energy demand for heating and cooling. Thermal

Table 12.1 The division of thermal insulation materials by Kobyliński and Szymański (1963)

Classification	Example materials
Organic materials, consisting of different parts of plants or porous plastic masses	Peat materials (peat powder, peat boards); hardboards; chip and cement or chip and magnesite boards; boards made of straw and reed; boards made of linen and hemp harls; boards made from wood, cork and pine bark; mats and boards of sheep wool; earthwork materials made from cellulose – ecofibre
Non-organic materials, obtained from natural resources	Cinder wool and its products (not used nowadays); rock wool and its products; yarn, wool and glass wool and its products; cellular glass; asbestos products (removed from usage); termalite products; products from foamed polyurethane (PIR and PUR foam); products from foamed polystyrene (expanded and extruded)

Source: based on Kobyliński and Szymański (1963).

insulation in walls and roofs reduces overall need for air conditioning as well as the power required for air conditioning when it is used, further decreasing annual energy costs. Proper insulation of buildings also brings additional benefits by reducing pollution emissions, including CO_2. Building energy-efficient houses is consistent with modern energy policies promoted by European Union Directives which encourage the reduction of energy consumption in buildings as much as possible (Directive 2012/27/EU, Directive 2002/91/EU, Directive 2006/32/EU, Directive 2005/32/EU).

The range of economic and ecological savings resulting from use of thermal insulation depends on: the type of building, climate conditions at the location, and the thickness and positioning of the insulating material. When selecting an insulating material, it is important to take the following into consideration:

- thermal conductivity,
- diffusion (penetration) of water vapour,
- class of flammability,
- resistance to chemical factors,
- resistance to biological factors,
- mechanical endurance (ability to transfer loads),
- impact on the environment.

This chapter assesses the ecological and economic benefits of thermal insulation materials, considers their different classifications and properties, and assesses their impact using LCA analysis.

Table 12.2 The division of thermal insulation materials by Steidl (2010)

Classification	Example materials
Biodegradable materials	Flax fibers – peat materials; hardboards; chip and cement or chip and magnesite boards; boards made of straw and reed; boards made of linen and hemp harls; mats and boards of sheep wool; earthwork materials made from cellulose – ecofibre
Vacuum materials of VIP type	Core is made of microsilica; core made of polystyrene; core made of polyurethane; core made of the combination of silica, the oxide of titanium and carbon
Insulating materials obtaining solar energy of solar type	Three-glass glazing with a filling of krypton; granule of silica aerogel located between two joint sheets of glass; cellular structures or capillary ones with glass plaster

Source: based on Steidl (2010).

12.2 Thermal insulation materials and their properties

In the literature, different classifications of thermal insulation materials are presented. Table 12.1 illustrates such a classification scheme (Kobyliński and Szymański, 1963).

Steidl (2010) has proposed a new division of thermal insulation materials (see Table 12.2): biodegradable materials (e.g., materials from linen fibres plus synthetic fibres and starch; FLACHSHAUS, 2012), vacuum materials of VIP type (vacuum insulated panel), and insulating materials harvesting solar energy of solar type.

Insulating materials from linen fibres represent a wide range of products (http://www.hellotrade.com/flachshaus/). Their advantage lies in their inherent biodegradability and renewable source materials, for example in one case, 80% linen fibres, 10% potato starch and 10% sodium octaborat, a flame retardant. Products from linen fibres have an E classification according to EN 13501-1. They are resistant to decay, insects and mould. These products demand low energy for production purposes and have no associated health risks. They regulate humidity and are suitable for recycling. A typical thermal conductivity coefficient is $\lambda = 0.038\,\text{W/mK}$, with density 30–50 kg/m^3.

In recent years there has been intensive development of insulating systems from vacuum panels (Caps *et al.*, 2001; Simmler and Brunner, 2005; Baetens *et al.*, 2010). The most significant feature of VIPs is their 5–10 times higher thermal resistance in comparison to conventional thermal insulation (DOWCORNING, 2012). Vacuum insulations have a core formed from a nano-structural material (pores of 100 nm), e.g. microsilica, polystyrene, polyurethane or a combination of silica, oxide of titanium, and carbon. The low heat conductivity of this insulation is determined by the value of the gas pressure in the nano-structural material. The lowest values of thermal conductivity (between 0.004 and 0.001 W/mK at temperature) are obtained at 100 Pa. Thermal conductivity increases with gas internal pressure to about 0.020 W/mK under atmospheric conditions. The core is covered by thin metal foils and metallized polymer laminates. VIPs have a long lifetime, ranging between 30 and 50 years (ECBCS, 2005). The main disadvantage of VIPs is their high price (about €200/m^2 for a thickness of 6 cm in material costs alone) (Pär, 2012).

Transparent insulations restrict heat loss from buildings whilst at the same time harvesting energy from the environment. According to Pogorzelski (2005), there are three systems in use:

- triple glazing with krypton filling;
- granule of silica aerogel located between two sheets of glass;
- cellular or capillary structures with glass plaster.

The second system reduces light transmission by about 30% compared to a traditional double-glazed window (Buratti and Moretti, 2011). The coefficient of thermal conductivity for transparent insulation is: $\lambda = 0.01$ W/mK, and density of aerogel 0.004 g/cc.

Phase change insulations increase the thermal capacity of the building. Current building construction technologies, and those relating to office buildings in particular, use materials that have good thermal insulation but restricted thermal capacity. This is possible by using so-called phase change materials (PCMs). The heat capacities of these materials result from a phase change (melting-solidifying) which occurs with changing room temperature. Using phase change materials is justified only in climates where temperatures are either higher or lower than those defining thermal comfort. PCM can be used on external walls in combination with transparent insulation (Mehling and Cabeza, 2008; Zalba *et al.*, 2003; Pasupathy and Velraj, 2008; Tyagi and Buddhi, 2007; Kenisarin and Mahkamov, 2007). Thermal conductivity coefficients of these materials are about: $\lambda = 0.01$ W/mK, with density 870–1,000 kg/m^3.

Bjørn (2011) proposed an alternative classification scheme (see Table 12.3). It concludes that no single thermal insulation material is suitable for all applications. Future potential materials include nano-insulation, dynamic insulation, and the load-bearing insulation material NanoCon (Bjørn, 2011).

Energy-efficient building practices use natural building materials which can be found close to the construction site. This reduces the environmental impact of transportation (Eriksson *et al.*, 1996).

Table 12.3 The division of thermal insulation materials by Bjørn (2011)

Classification	Example materials
Traditional thermal building insulation	Mineral wool; expanded polystyrene (EPS); extruded polystyrene (XPS); cellulose; cork; polyurethane (PUR)
State-of-the-art thermal building insulation	Vacuum insulation panels (VIP); gas-filled panels (GFP); aerogels; phase change materials (PCM)
Possible future thermal building insulation	Vacuum insulation materials (VIM); gas insulation materials (GIM); nano-insulation materials (NIM); dynamic insulation materials (DIM); concrete and applications of NIMs; NanoCon

Source: based on Bjørn (2011).

Table 12.4 Technical parameters for Bims granule and EPS plaster

Kind of plaster →	Bims granule plaster	EPS plaster
Appearance and consistency	White powder	Grey powder
Density of dry mass	334 +– 10% (kg/m^3)	200 +– 10% (kg/m^3)
Thermal conductivity coefficient (after 120 days)	0.068 (W/mK)	0.070 (W/mK)
Resistance to compressing	CS II 1.84 (N/mm^2)	≥0.5 (N/mm^2)
Adhesion	0.46 (N/mm^2)	≥0.08 (N/mm^2)
Moisture absorption	W1 0.204 (kg/m^2) min 0.5	W1 ≤0.2 (kg/m^2) min 0.5
Diffusion coefficient (after 120 days)	4.80	≤10
Consumption for 1 cm of thickness	4 (kg/m^2)	2.27 (kg/m^2)

Source: based on ThermoPlast (2008) and Schwepa (2012).

The conditions required for heat-insulating plaster are the following: thermal conductivity coefficient lower than 0.21 W/mK and density in the loose state lower than 700 kg/m^3 (Jasiczak, 2007). Table 12.4 presents properties of two types of thermally insulating plaster. EPS insulating plaster has been commercially available for many years, whereas Bims granule plaster is new to the market.

At present, polystyrene and mineral wool are the most frequently used for insulating external walls of buildings. With the light-wet method (jointless insulating system) façade polystyrene self-extinguishing boards of EPS 70 or façade boards of hard mineral wool are used. It is also possible to use other thermally insulating materials (see Table 12.5); however, their application is dependent on the construction of the wall itself.

Table 12.5 Technical parameters of thermal insulation materials

Kind of thermal insulation material →	Extruded polystyrene XPS (according to norm EN 13164:2008)	EPS polystyrene (according to norm EN 13163:2008)	Mineral wool (according to norm EN 13162:2008)	Polyurethane foam (according to norm EN 13165:2008)	Ecofibre (technical approval of ETA-05/0186)
Volume density (declared value) (kg/m^3)	28–32	14–19	100–170	30–60	32–65
Thermal conductivity coefficient (declared value) (W/mK)	0.030–0.040	0.031–0.042	0.036–0.045	0.023–0.035	0.040–0.043
Absorption (water absorptiveness)	Standard volume absorption after long-lasting total immersion – no more than 3.0%	Standard volume absorption after long-lasting total immersion – no more than 5.0%	Volume absorption after long-lasting part immersion – no more than 3.0%	Volume absorption after long-lasting total immersion – no more than 5.0%	Volume absorption after long-lasting total immersion – no more than 5.0%
Fire resistance	Class E of reaction to fire, self-extinguishing, extinguishes after the source of fire is removed, application within temperatures 50°C up to +75°C, under the influence of higher temperature boards becomes softer, changes measurements, melts and loses its mechanical properties	Class E of reaction to fire, self-extinguishing, extinguishes after the source of fire is removed, over temperature of 80°C, can undergo deformation, over 100°C undergoes deformation, melts	Class A1 of reaction to fire, non-flammable material, mineral fibres are resistant to temperature of 600°C, over 1000°C they melt, thermal resistance of binder no more than 250°C	Class E of reaction to fire, thermal resistance from –100°C to +120°C, over 150°C there appears breaking chemical bonds, over 200°C foam gets burnt	Class B2 of reaction to fire, flame resistant material, not spreading fire, in contact with fire does not burn, in temperature of 100°C fibres slowly become carbonized

Source: based on Radziszewska-Zielina (2009).

12.3 Life cycle assessment (LCA) analysis of thermal insulation materials

Environmentally-friendly solutions within constructing consider not only energy savings, but also minimization of locally sourced material resources and reductions in pollution (Berge, 2009).

12.3.1 LCA analysis

For evaluation of the environmental impact of the successive stages of construction, LCA analysis is used. Environmental life cycle assessment (LCA) developed during the 1990s and its method have been presented by various authors (Curran, 1993; LCANET, 1996; Oritz *et al.*, 2009; Dąbrowski and Dzikuć, 2012). It has been formalized since the 1998 publication of ISO 14040-43. This was replaced in 2006 by ISO 14040 and ISO 14044 according to which LCA comprises four stages:

- goal and scope definition;
- LCI – life cycle inventory;
- LCIA – life cycle impact assessment;
- interpretation.

The methodology applied to LCA analysis is described in Dylewski and Adamczyk (2011), who applied SimaPro 7.1 which considers 21 assessment procedures. In this chapter, Ecoindicator 99 was used, which considers 11 impact categories in three damage categories: human health, ecosystem quality and consumption of raw materials. The above procedure enables weighing of parameters and determines a final parameter, Pt (Pushkar *et al.*, 2005). A value of 1 Pt represents 10^3 annual environmental load for one inhabitant in Europe. This value is calculated by dividing the whole environmental load in Europe by the number of inhabitants and multiplying by 1,000 (scale factor).

12.3.2 The results of LCA analysis

The aim of the analysis is to compare the impact on the environment of thermal insulation materials and plaster in order to identify the optimal materials (the lowest value of points – Pt). The stage of inventory analysis (LCI) includes all material-energetic transfers within determined boundaries of the systems. These refer to the obtaining of raw materials as well as the production of the thermal insulation material and/or plaster. Material-energetic transfers are presented in Table 12.6.

Tables 12.7 and 12.8, together with Fig. 12.1 illustrate the results of LCA analysis at the functional unit of 1 m^3. The assignments of particular impact

Table 12.6 The comparison of more significant material-energetic transfers within a functional unit equal to 1 m^3

Insulation material or plaster	Material-energetic transfers					
EPS plaster	Sand 57 kg	Road transport 4.0 tkm	Cement 104.6 kg	Lime 21.4 kg	Electricity 20 MJ	EPS 9.6 kg
Bims granule plaster	Bims granules 320 kg	Rail transport 2.6 tkm	Cement 4 kg	Lime 10 kg	Electricity ok. 350 MJ	–
EPS polystyrene	Oil crude 11.4 kg	Road transport 0.3 tkm	–	Natural gas 18 m^3	Electricity ok. 1.9 MJ	–
Mineral wool	Marl 157 kg	Road transport 1.6 tkm	Water 1272 kg	Natural gas 0.85 m^3	Electricity ok. 375 kJ	Coal 59 kg
Polyurethane foam	Oil crude 31.1 kg	Road transport 1.5 tkm	Calcite 10.8 kg	Natural gas 55.4 m^3	Electricity ok. 108 MJ	Coal 25 kg
Ecofibre	Recycling paper 820 kg	Road transport 2.0 tkm	Borax 60 kg	–	Electricity ok. 518 MJ	–

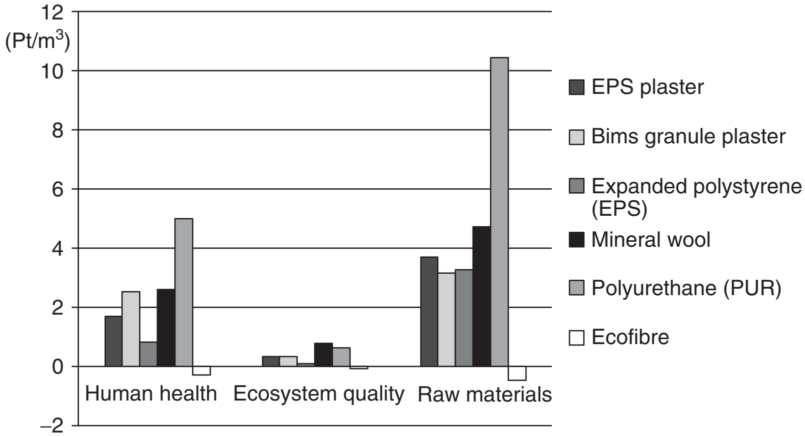
Table 12.7 The result of LCA analysis (in Pt/m^3) of thermal insulation materials divided into impact categories

Impact category	EPS plaster – 200 (kg/m^3)	Bims granule plaster – 330 (kg/m^3)	Expanded polystyrene (EPS) – 15 (kg/m^3)	Mineral wool – 120 (kg/m^3)	Polyurethane (PUR) – 45 (kg/m^3)	Ecofibre – 60 (kg/m^3)
Carcinogens	0.028	0.447	0.008	0.234	0.091	0.020
Respiratory organics	0.004	0.000	0.002	0.007	0.007	–0.001
Respiratory inorganics	1.034	1.629	0.639	1.564	4.155	–0.272
Climate change	0.604	0.473	0.166	0.773	0.743	–0.040
Radiation	0.006	0.001	0.000	0.016	0.004	0.000
Ozone layer	0.000	0.000	0.000	0.001	0.000	–0.000
Ecotoxicity	0.054	0.037	0.004	0.129	0.113	–0.016
Acidification/ Eutrophication	0.191	0.193	0.113	0.509	0.478	–0.067
Land use	0.096	0.101	0.000	0.143	0.034	0.006
Minerals	0.018	0.006	0.000	0.010	0.053	–0.001
Fossil fuels	3.665	3.148	3.273	4.723	10.383	–0.460
Total	5.699	6.035	4.205	8.108	16.062	–0.832

Table 12.8 The result of LCA analysis (in Pt/m^3) of thermal insulation materials divided into damage categories

Damage category	EPS plaster	Bims granule plaster	Expanded polystyrene (EPS)	Mineral wool	Polyurethane (PUR)	Ecofibre
Human health	1.676	2.551	0.815	2.594	5.001	–0.292
Ecosystem quality	0.340	0.329	0.117	0.782	0.625	–0.078
Raw materials	3.683	3.154	3.273	4.733	10.436	–0.461
Total	5.699	6.035	4.205	8.108	16.062	–0.832

categories to the three damage categories are as follows: human health – carcinogens, respiratory organics, respiratory inorganics, climate change, radiation and ozone layer; ecosystem quality – ecotoxicity, acidification/ eutrophication and land use; raw materials – minerals and fossil fuels. The highest impact on the environment among thermal insulation materials is attributed to polyurethane foam (16.062 Pt) (see Fig. 12.1 and Tables 12.7 and 12.8). This impact is almost twice as large as the influence of the



12.1 The result of LCA analysis of thermal insulation materials divided into damage categories.

production of mineral wool on the environment. However, a negative value of environmental impact (see cellulose fibres – ecofibre – Tables 12.7 and 12.8) shows the environmental benefits already generated at the stage of production due to the use of recycled waste paper. Among the types of plaster analysed, EPS plaster (5.699 Pt) is characterized by a slightly lower impact on the environment; moreover, it is also the plaster with lowest volume density.

12.4 The ecological benefits of thermal insulation of external walls of buildings

The ecological benefits of thermal insulation using the life cycle assessment (LCA) of products are presented in Section 12.3. The impact of the application of thermal insulation to external walls is related to the environmental cost associated with production of insulating materials versus the reduction in energy demand of the building.

12.4.1 Ecological costs, profits and benefits of thermal insulation

Ecological costs K_S (the increase of the environmental load) of a particular thermal insulation for external walls, per 1 m^2, is defined as follows:

$$K_S = d \cdot K_t \quad (\text{Pt/m}^2) \qquad [12.1]$$

where K_t is the result of LCA for 1 m^3 of thermal insulation material (Pt/m^3), and d is the thickness of thermal insulation layer (m).

Ecological profits Z_S in the use phase of the building resulting from thermal insulation are as follows (Dylewski and Adamczyk, 2012):

$$Z_S = (E_{Uo} - E_U) \cdot n/p \quad (\text{Pt/m}^2) \qquad [12.2]$$

where E_{Uo} is the result of LCA analysis of one year of thermal phase of building use, with external walls having heat transfer coefficient U_o (without thermal insulation) (Pt/year), E_U is the result of LCA analysis of one year of thermal phase of building use, with external walls having heat transfer coefficient U (with thermal insulation) (Pt/year); n is number of years of thermal insulation use, and p is surface of the building external walls (m^2).

The value of E_U (similarly E_{Uo}) can be assigned as follows:

$$E_U = D_E \cdot K_e \quad (\text{Pt/year}) \qquad [12.3]$$

where D_E is demand for heat power of the whole building (kWh/year), and K_e is result of LCA of obtaining 1 kWh of thermal energy for a particular heat source (Pt/kWh). In order to assign heat demand D_E in the phase of the building use, it is possible to apply, for example, the program Herz OZC (Herz, 2012).

Thermal insulation of the external walls of the building generates benefits for the environment O_S of the amount:

$$O_S = Z_S - K_S \quad (\text{Pt/m}^2) \qquad [12.4]$$

For the whole building, ecological benefits are then $O_S \cdot p$ (Pt).

12.4.2 Sample analyses of the ecological benefits of thermal insulation

This section considers the particular case of a building located in Poland near Zielona Góra, with surface of external walls $p = 158.7\,\text{m}^2$, usable area of 156.1 m^2 and cubature of 390 m^3. It is assumed that the use period of thermal insulation is $n = 25$ years. External walls of the building are made from cellular concrete blocks with density of 400 kg/m^3, thickness of 24 cm and thermal resistance of $R_o = 2.40\,\text{m}^2\text{K/W}$. The coefficient of heat transfer for walls without thermal insulation is $U_o = 1/(R_o + R_{si} + R_{se}) = 0.39\,\text{W/m}^2\text{K}$ (according to PN-EN ISO 6946, the resistance of heat absorption on the internal surface was taken as $R_{si} = 0.13\,\text{m}^2\text{K/W}$ and the resistance of heat absorption on the external surface as $R_{se} = 0.04\,\text{m}^2\text{K/W}$).

In Poland, according to a Regulation of the Minister of Infrastructure (RMI, 2008), the thickness of a thermal insulation layer should be selected in such a way that external walls of the building have a coefficient of heat transfer of $U \leq 0.30\,\text{W/m}^2\text{K}$. Therefore, in order to reach $U_N = 0.30\,\text{W/m}^2\text{K}$, the thickness d_N of thermal insulation should be (Laskowski, 2005):

Table 12.9 The assigned thicknesses of thermal insulation and ecological costs

Thermal insulation material or plaster→	EPS plaster	Bims granule plaster	Expanded polystyrene (EPS)	Mineral wool	Polyurethane (PUR)	Ecofibre
λ (W/mK)	0.070	0.068	0.040	0.039	0.028	0.041
d_N (m)	0.054	0.052	0.031	0.030	0.022	0.032
K_S (Pt/m^2)	0.308	0.314	0.130	0.243	0.353	–0.027

$$d_N = \lambda \cdot (1/U_N - 1/U_o) \quad \text{(m)} \qquad [12.5]$$

where λ signifies the coefficient of thermal conductivity of the thermal insulation material (W/mK).

An analysis was conducted considering the thermal insulation plaster and materials described in Section 12.3. The thicknesses of thermal insulation d_N necessary to obtain a transfer coefficient U_N were obtained using Eq. [12.5]. Next, using Eq. [12.1] together with the results of LCA analysis from Table 12.7, ecological costs K_S were calculated. The results are presented in Table 12.9.

The lowest possible thickness of thermal insulation was obtained for PUR foam. However, this material also exhibits the highest ecological costs. A slightly lower negative impact on the environment is attributed to thermally insulating types of plaster. For EPS polystyrene and mineral wool, similar thicknesses were obtained, although insulation from EPS polystyrene generates far lower ecological costs. For ecofibre, a negative value was obtained, that is, there is an ecological profit (reduction of the environmental load).

In order to assign ecological profits in the use phase of the building as a result of thermal insulation, it is first necessary to calculate the demand for heat for the building D_E, and coefficients of heat transfer U_o and U_N of the external walls. Using the Herz OZC program, the demand for heat was assigned and was set at respectively 19,192 kWh/year and 17,997 kWh/year. LCA analysis of obtaining 1 kWh of thermal energy K_e for the heat sources considered was also conducted. Three heat sources were taken into consideration: (S1) hard coal boiler, (S2) natural gas boiler and (S3) electric energy boiler. From Eq. [12.3] the results of LCA analysis associated with one year of building use E_{Uo} and E_{UN} were assigned together with ecological profits Z_S, from Eq. [12.2]. The results are compared in Table 12.10. The highest ecological profits were obtained in the case of using an electric energy boiler (S3) heat source. In each case the profits are higher than ecological costs.

Table 12.10 The assigned ecological profits according to heat source

Heat source →	Hard coal boiler (S1)	Natural gas boiler (S2)	Electric energy boiler (S3)
K_e (Pt/kWh)	0.0193	0.0123	0.0485
E_{Uo} (Pt/year)	370.406	236.062	930.812
E_{UN} (Pt/year)	347.342	221.363	872.855
Z_S (Pt/m^2)	3.633	2.316	9.130

Table 12.11 Ecological benefits (in Pt/m^2) resulting from thermal insulation of the building external walls

Heat source	EPS plaster	Bims granule plaster	Expanded polystyrene (EPS)	Mineral wool	Polyurethane (PUR)	Ecofibre
S1	3.325	3.319	3.503	3.390	3.280	3.660
S2	2.008	2.002	2.186	2.073	1.963	2.343
S3	8.822	8.816	9.000	8.887	8.777	9.157

Table 12.12 Ecological payback periods (in years) for thermal insulation

Heat source	EPS plaster	Bims granule plaster	Expanded polystyrene (EPS)	Mineral wool	Polyurethane (PUR)	Ecofibre
S1	3	3	1	2	3	0
S2	4	4	2	3	4	0
S3	1	1	1	1	1	0

In Table 12.11 ecological benefits O_S obtained from Eq. [12.4] are compared, which depend on both the thermal insulation material used and the heat source for the building. For each case ecological benefits were positive. The biggest benefits are for ecofibre thermal insulation, and the smallest for PUR foam thermal insulation, although the variation is small. The ecological payback period for thermal insulation was calculated. The results are presented in Table 12.12. Payback takes place after four years at most.

12.5 The economic benefits of thermal insulation

Thermal insulation of the external walls of a building can also be assessed economically.

12.5.1 Economic cost, profits and benefits of thermal insulation

The economic cost K_E of employing a particular variant of thermal insulation for external walls, per $1\,\text{m}^2$, can be obtained in the following way:

$$K_E = d \cdot K_m + K_w \quad (\text{PLN/m}^2) \tag{12.6}$$

where K_m is cost of $1\,\text{m}^3$ of thermal insulation material (PLN/m^3) (1.0 PLN ≈ €0.25), K_w is cost of making thermal insulation for $1\,\text{m}^2$ of the building wall surface (PLN/m^2), and d is as defined above.

Economic profits Z_E in the use phase of the building per $1\,\text{m}^2$ can be obtained from the formula (Dylewski and Adamczyk, 2011):

$$Z_E = S_n \cdot G_o \cdot (U_o - U) \quad (\text{PLN/m}^2) \tag{12.7}$$

where G_o is annual heating cost, for $1\,\text{m}^2$ of the surface of the analysed wall (PLN K/W), $S_n = \sum_{t=1}^{n} \left(\frac{1+s}{1+r} \right)^t = \frac{1-q^n}{q^{-1}-1}$, where $q = \frac{1+s}{1+r}$, r is real annual interest rate, s is real annual increase of heating costs (as a percentage), and n, U and U_o are as defined above.

Thermal insulation of the external walls of the building generates benefits O_E per $1\,\text{m}^2$ in the amount:

$$O_E = Z_E - K_E \quad (\text{PLN/m}^2) \tag{12.8}$$

Taking the whole building into consideration, economic benefits are $O_E \cdot p$ (PLN).

12.5.2 Sample analyses of the economic benefits of thermal insulation

In Table 12.13 the accepted costs K_m for $1\,\text{m}^3$ of thermal insulation materials (based on ICMarket, 2005) and thermal insulation plaster (based on ThermoPlast, 2008 and Schwepa, 2012) are presented together with the economic cost K_E obtained from Eq. [12.6]. The cost of thermal insulation

Table 12.13 Economic costs of thermal insulation

Thermal insulation material or plaster→	EPS plaster	Bims granule plaster	Expanded polystyrene (EPS)	Mineral wool	Polyurethane (PUR)	Ecofibre
K_m (PLN/m^3)	1440.00	2300.00	167.30	522.00	713.30	150.00
K_E (PLN/m^2)	197.76	239.60	35.19	45.66	45.69	34.80

K_w was taken as 30 PLN/m^2 for materials and 120 PLN/m^2 for thermal insulation types of plaster.

In order to assign economic profits Z_E, the use period of thermal insulation was taken to be the same as for ecological profits (n = 25 years), the real annual interest rate r was taken to be 5%, and the real increase of heating cost s was assumed to be 2%. Annual heating cost G_o for the analysed heat sources was defined in the following way:

- (S1) hard coal boiler, $G_o = 10.90$ PLN K/W (= $126\cdot10^{-6}\cdot24\cdot3605$), the cost of obtaining heat for heating purposes 126 PLN/MWh, (boiler efficiency of 80%, fuel calorific value 29 MJ/kg and price 769 PLN/t);
- (S2) natural gas boiler, $G_o = 30.37$ PLN K/W (= $351\cdot10^{-6}\cdot24\cdot3605$), the cost of obtaining heat for heating purposes 351 PLN/MWh, (boiler efficiency of 90%, fuel calorific value 31 MJ/m^3 and price 2.63 PLN/m^3);
- (S3) electric energy boiler, $G_o = 50.18$ PLN K/W (= $580\cdot10^{-6}\cdot24\cdot3605$), the cost of obtaining heat for heating purposes 580 PLN/MWh, (boiler efficiency of 99%, the price of electric energy 0.58 PLN/kWh).

These prices are based on GUS (2012). The number of $Sd = 3605$ of degree-days was taken (an average in Poland between 1980 and 2004; Gikas and Keenan, 2006). The figure of degree-days of a heating season is a quantitative factor defining heating energy demand of houses and public utility buildings. It is determined using climate data for a particular town (those days when the external air temperature is lower than the assumed base temperature for the day).

In Table 12.14 economic profits Z_E assigned from Eq. [12.7] are presented together with economic benefits O_E obtained from Eq. [12.8]. Thermal insulation of buildings external walls is not always profitable. The biggest economic benefits are obtained for EPS polystyrene and ecofibre.

Table 12.14 Economic profits and benefits (in PLN/m^2) resulting from thermal insulation of the building external walls

Thermal insulation material or plaster →		EPS plaster	Bims granule plaster	Expanded polystyrene (EPS)	Mineral wool	Polyurethane (PUR)	Ecofibre
Heat source	Z_E	O_E					
S1	17.19	−180.57	−222.41	−18.00	−28.47	−28.50	−17.61
S2	47.91	−149.85	−191.69	12.72	2.25	2.22	13.11
S3	79.16	−118.60	−160.44	43.97	33.50	33.47	44.36

Table 12.15 Economic payback periods (in years) for thermal insulation

Heat source	EPS plaster	Bims granule plaster	Expanded polystyrene (EPS)	Mineral wool	Polyurethane (PUR)	Ecofibre
S1	–	–	–	–	–	–
S2	–	–	17	24	24	17
S3	–	–	9	13	13	9

Table 12.16 Ecological efficiency of thermal insulation (in Pt/€)

Heat source	EPS plaster	Bims granule plaster	Expanded polystyrene (EPS)	Mineral wool	Polyurethane (PUR)	Ecofibre
S1	0.07	0.06	0.40	0.30	0.29	0.42
S2	0.04	0.03	0.25	0.18	0.17	0.27
S3	0.18	0.15	1.02	0.78	0.77	1.05

Economic payback periods were also considered. The results are presented in Table 12.15. In the analysed cases, payback of economic cost takes place after 9 years at the earliest, and more than 25 years where a hard coal boiler heat source is used. Similarly, the economic cost of using thermal insulation types of plaster is much higher than the economic profits in the use phase of the building.

Finally, the ecological efficiency of thermal insulation of external walls was obtained (the quotient of ecological benefits O_S and economic cost K_E). The results are presented in Table 12.16. For ecofibre thermal insulation, with electric energy boiler used as a heat source, it is possible to obtain a reduction in the environmental load of 1.05 Pt with an outlay of €1.

12.6 Conclusions

The use of thermal insulation materials in buildings has both economic and ecological consequences. It reduces energy demand for heat in the use phase of the building. As a result, due to the reduced energy consumption, the environmental load and the cost of heating the building decrease. However, it is necessary to bear in mind that the production and installation of the thermal insulation itself result in certain economic costs which also increase the environmental load. Some thermal insulation materials are produced from recycled materials, e.g. ecofibre, which decreases the environmental load (Adamczyk and Dylewski, 2010).

Thermal insulation of external walls is, however, beneficial for the environment. The ecological costs connected with the performance of thermal insulation are much lower than the ecological profits obtained in the use phase of the building. For the analysed examples the ecological payback period was between 0 and 4 years.

However, the economic analysis shows that the investment in thermal insulation of building external walls does not always have to be economically profitable for the investor. Profitability depends substantially on the insulating properties of the material and its price, but also on parameters of the wall without thermal insulation as well as a heat source used in the building and, as a consequence, annual heating costs.

It is also possible to assign the economically optimal thickness of thermal insulation (Dylewski and Adamczyk, 2012). Usually optimal thicknesses are bigger than the required ones for obtaining the heat transfer coefficient of $U_N = 0.30\,\mathrm{W/m^2K}$. Obviously economic benefits are then greater, but also ecological benefits become bigger.

Moreover, it is also necessary to underline that thermal insulation of the building external walls has great ecological efficiency (the ratio of ecological benefits to economic cost). In the cases analysed, values between 0.17 and 1.05 Pt/€ were obtained for thermal insulation materials and between 0.03 and 0.18 Pt/€ for thermal insulation types of plaster.

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13

Life cycle assessment (LCA) of phase change materials (PCMs) used in buildings

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Abstract: Phase change materials (PCMs) have been studied for thermal storage in buildings since the 1980s. These systems provide a higher thermal inertia to the building that, when combined with thermal insulation, can reduce the energy consumption of the building by absorbing the heat gains and reducing the heat flow. Due to the high embodied energy of PCMs, a life cycle assessment study would allow us to determine whether the reduction of the operational impact can balance out the impact produced during the manufacturing and disposal of the PCM and the encapsulation.

Key words: phase change material (PCM), thermal energy storage (TES), life cycle assessment (LCA).

13.1 Introduction to phase change materials (PCMs) and their use in buildings

The use of thermal storage technology in a building can reduce temperature fluctuations and therefore energy use. Thermal energy storage in buildings can be implemented by sensible heat management (increasing and decreasing the temperature of the building envelopes, for example), by latent heat (with the inclusion of phase change materials to increase thermal inertia), or by thermochemical storage (with chemical reactions or sorption systems) (Cabeza *et al*., 2011).

When a material stores heat at phase transition, the heat is stored as latent heat. This material is then known as a phase change material (PCM). A solid-liquid phase change process by melting and solidification can store large amounts of heat if a suitable material is selected. Upon melting, while heat is transferred to the storage material, the material still keeps its temperature constant at the melting temperature, also called the phase change temperature (Cabeza, 2011).

PCMs have been studied for thermal storage in buildings since the 1980s (Cabeza, 2011). These systems provide a higher thermal inertia to the building which, when combined with thermal insulation, can reduce the energy consumption of the building by absorbing the heat gains and reducing the heat flow. During daytime, the PCM can absorb part of the heat through the melting process, and during the night, the heat is released by solidification of the PCM, resulting in a lower heat flow from outdoors to indoors. This can be implemented in a building as a passive or active system.

In passive systems, the PCM is included in the building, usually in the building walls, and it is expected to be charged and discharged due to change in the outdoor temperature. Some examples of such a system have been developed by Shapiro *et al.* (1987), Schossig *et al.* (2005), Hawes and Feldman (1992), and Cabeza *et al.* (2007).

In active systems, the PCM is included in the building, both in the building envelope or in the climatisation system, and is charged or discharged with a heat transfer fluid. Some examples are those presented by Farid and Chen (1999), Nagano *et al.* (2000), Zalba *et al.* (2004), and de Gracia *et al.* (2013).

Many substances have been studied as potential PCMs, but only a few of them have been commercialised (Cabeza *et al.*, 2011; Mehling and Cabeza, 2008; Jeon *et al.*, 2013; Rathod and Banerjee, 2013; Zhou *et al.*, 2012; Ostermann *et al.*, 2012; Al-Abidi *et al.*, 2012; Kuznik *et al.*, 2011). The selection of the material to be used in latent heat storage is not easy. Availability and cost are usually the main drawbacks for the selection of a technically suitable material. Still today, problems such as phase separation, subcooling, corrosion, long-term stability, and low heat conductivity have not been totally solved and are under research.

Due to the high embodied energy of PCMs, an LCA study would allow us to determine whether the reduction of the operational impact can balance out the impact produced during the manufacturing and disposal of the PCM and the encapsulation. Such a study is presented here using experimental data of energy consumption from an experimental set-up located in Spain, where PCM is included as a passive system in experimental house-like cubicles.

13.2 Investigating the use of PCMs in buildings

A long-term experiment is being developed at the University of Lleida (Spain), where different forms of phase change materials (PCM) are being tested in a pilot plant (Fig. 13.1). The experimental set-up, located in Puigverd de Lleida (Spain), consisted of several identically shaped cubicles. The cubicles were designed with the help of TRNSYS using the type developed by the authors for such application, and validated in the laboratory (Ibáñez *et al.*, 2005). To be able to compare the results, all cubicles had internal dimensions of 2.4 m × 2.4 m × 2.4 m.

This area has a climate classified as Dry Mediterranean Continental, characterised by its great seasonal variations. It has low rainfall, divided

13.1 Experimental set-up in Puigverd de Lleida.

into two seasons, spring and autumn, and it has a thermometric regime with large differences between a long winter (between the spring and the last frost may take more than 160 days) and a very hot summer. The average annual rainfall is between 350 and 550 mm, and the mean annual temperatures oscillate between 12 and 14°C, with thermal amplitudes of 17–20°C. Special mention must be made of the fog, typical for the region in the months of November, December and January that can give a period of up to 55 days without sunlight. This is a very similar climate to that of the area of Madrid, albeit with a greater annual rainfall and fewer days of fog per year (Table 13.1).

The important temperature oscillations during the day and night make it very suitable for PCM operation, as the material can be melted during the day and solidified during the night. The PCM tested were designed for cooling applications (Table 13.2).

The cubicles were built using different typical construction solutions, so concrete cubicles, brick cubicles and alveolar brick cubicles are included in the overall set-up. In this study, only the brick cubicles and the alveolar brick cubicles are used. More information on the concrete cubicles can be found in Cabeza *et al.* (2007).

13.2.1 Brick cubicles

Three brick cubicles were built using typical Mediterranean construction solutions (Castell *et al.*, 2010). Each cubicle has one door at the north wall and no windows to avoid the influence of additional parameters on their thermal behaviour (Fig. 13.2). The structure consists of a concrete base measuring 3 × 3 m with reinforcing bars and four reinforced concrete pillars, one at each edge of the cubicle. The façade walls are formed by the following layers from inside to outside: superficial inside finish with plaster layer; 14 cm perforated brick layer; insulation layer; 5 cm air chamber layer; 7.5 cm hollow brick layer; outside superficial finish cement mortar layer. The roof was made using concrete precast beams and 5 cm of concrete slab. The roof insulating material (3 cm) is placed over the concrete, protected with a

Table 13.1 Normal climatic values for Lleida, Station 2. Period: 1971–2000. Altitude (m): 192. Latitude: 41°37′33″N. Longitude: 00°35′42″E. State Meteorological Agency. Spanish Ministry of Environment. http://www.aemet.es/es/portada

Month	T	TM	Tm	R	H	DR	DN	DT	DF	DH	DD	I
January	5.3	9.6	1.0	26	81	4	1	0	12	13	5	116
February	7.9	13.7	2.2	14	70	3	0	0	5	8	7	167
March	10.8	17.5	4.2	27	61	4	0	0	3	3	8	226
April	13.2	19.8	6.5	37	58	5	0	1	1	0	6	248
May	17.3	24.0	10.5	49	58	6	0	3	1	0	5	279
June	21.4	28.5	14.4	34	54	4	0	3	0	0	9	313
July	24.7	32.2	17.2	12	51	2	0	2	0	0	14	348
August	24.5	31.6	17.4	21	56	3	0	4	0	0	12	313
September	20.7	27.3	14.1	39	63	4	0	2	1	0	8	250
October	15.3	21.2	9.4	39	71	4	0	1	4	0	6	200
November	9.3	14.2	4.4	28	79	4	0	0	11	5	5	137
December	6.0	9.8	2.1	28	83	4	0	0	14	10	5	96
Year	**14.7**	**20.8**	**8.6**	**369**	**66**	**46**	**1**	**18**	**53**	**37**	**91**	**2685**

Caption

T	Monthly/annual temperature average (°C)
TM	Monthly/yearly maximum daily temperatures average (°C)
Tm	Monthly/annual minimum daily temperatures average (°C)
R	Monthly/annual precipitation average (mm)
H	Relative humidity average (%)
DR	Monthly/annual days of precipitation greater than or equal to 1 mm average
DN	Monthly/annual snow days average
DT	Monthly/annual storm days average
DF	Monthly/annual fog days average
DH	Monthly/annual frost days average
DD	Monthly/annual clear days average
I	Monthly/annual sunshine hours average

cement mortar roof with an inclination of 3% and a double asphalt membrane. The internal finish is of plaster.

In the brick cubicles different insulating solutions are compared:

1. Reference cubicle (REF) (Fig. 13.3): this cubicle has no insulation in the walls.
2. Polyurethane cubicle (PU) (Fig. 13.4): the insulation material used is 5 cm of spray foam polyurethane attached to the perforated brick layer.
3. PCM cubicle (RT27 + PU) (Fig. 13.5): the insulation used is again 5 cm of spray foam polyurethane, but here an additional layer of macroencapsulated PCM in compact storage modules (CSM) has been placed. CSM panels containing RT-27 paraffin (Table 13.2) are located between the perforated bricks and the polyurethane, in the southern and western walls and between the polyurethane and the concrete on the roof.

Table 13.2 Physical properties of the PCM used in the experimental set-up in Puigverd de Lleida Castell *et al.*, 2010

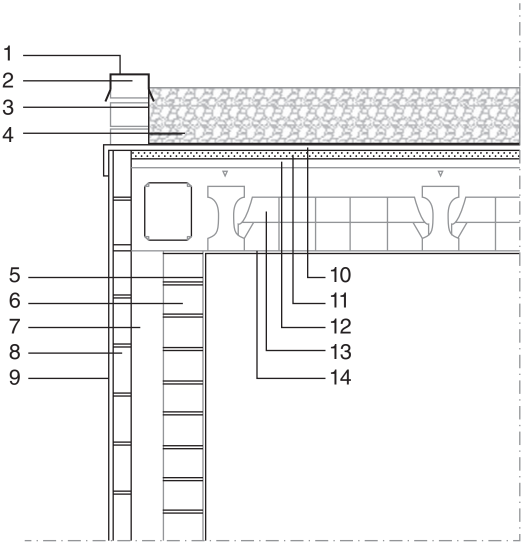
PCM	RT-27	SP-25 A8
Property		
Melting point (°C)	28	26
Congealing point (°C)	26	25
Heat storage capacity (kJ/kg)	179	180
Density (kg/L)		
Solid	0.87	1.38
Liquid	0.75	–
Specific heat capacity (kJ/kg·K)		
Liquid	1.8	2.5
Solid	2.4	–
Heat conductivity (W/m·K)	0.2	0.6

13.2 Brick cubicles: construction process.

13.2.2 Alveolar brick cubicles

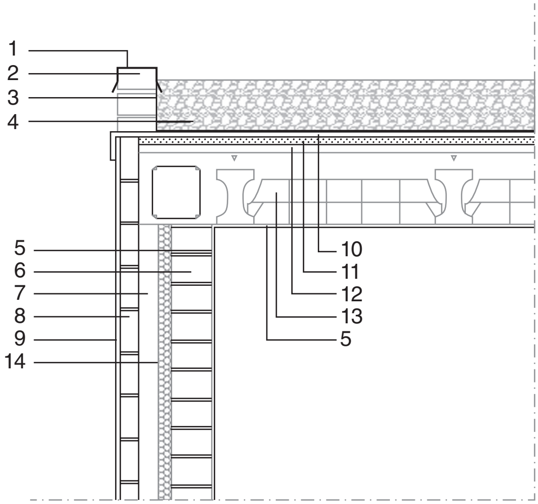
Two cubicles (ALV and ALV + PCM) were built using alveolar bricks (Fig. 13.6). Each cubicle has one door at the north wall and no windows (Castell *et al.*, 2010). The structure consists of a concrete base measuring 3 × 3 m with reinforcing bars and alveolar brick walls to support its own weight and also that of the roof. Thus, the envelope of the cubicles is composed of walls of alveolar bricks with an interior coat of plaster and exterior coat of cement mortar. None of the cubicles contains insulating materials in the walls. The roof is built with the same structure as in the conventional brick

1. Aluminium sheet
2. Perimeter brick wall
3. Waterproof sheet
4. Gravel
5. Plaster
6. Perforated brick wall
7. Air chamber 11 cm
8. Hollow brick
9. Cement mortar finish
10. Double asphaltic membrane
11. Cement mortar 3% slope
12. Roof insulation 3 cm
13. Concrete precast beams + 5 cm concrete slab



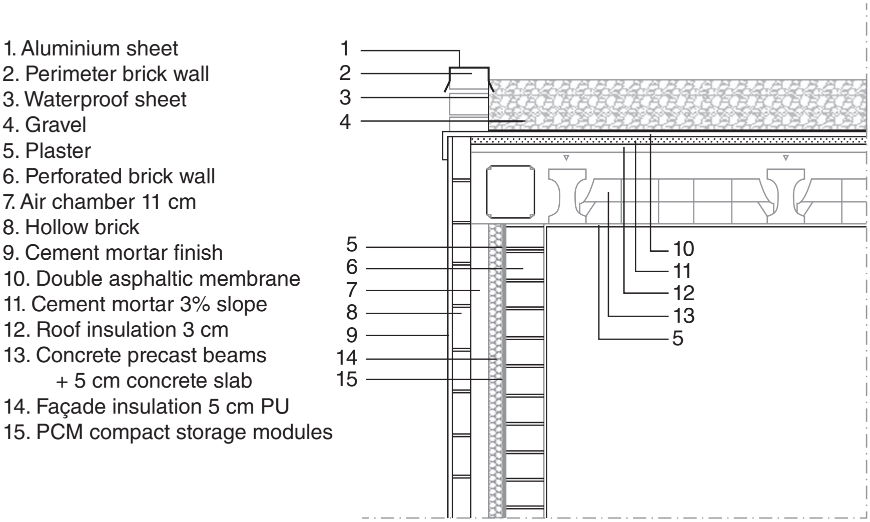
13.3 Construction system of the brick cubicles: reference cubicle.

1. Aluminium sheet
2. Perimeter brick wall
3. Waterproof sheet
4. Gravel
5. Plaster
6. Perforated brick wall
7. Air chamber 11 cm
8. Hollow brick
9. Cement mortar finish
10. Double asphaltic membrane
11. Cement mortar 3% slope
12. Roof insulation 3 cm
13. Concrete precast beams + 5 cm concrete slab
14. Façade insulation 5 cm PU



13.4 Construction system of the brick cubicles: PU cubicle.

cubicles. It was made using concrete precast beams and 5 cm of concrete slab. The roof insulating material is placed over the concrete, protected with a cement mortar roof with an inclination of 3% and a double asphalt membrane. The internal finish is of plaster.

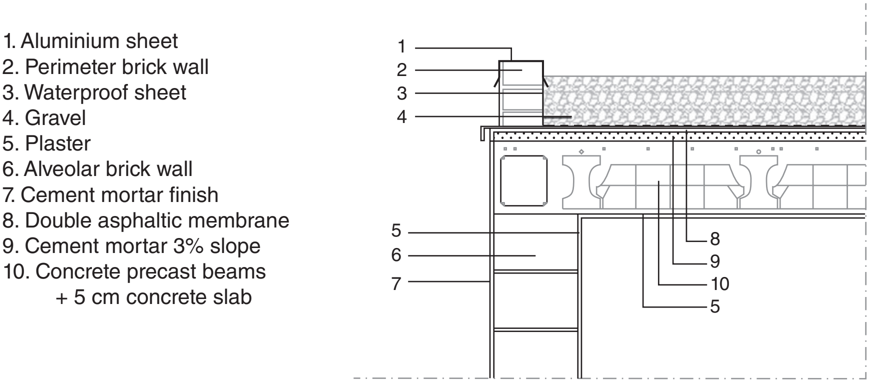


13.5 Construction system of the brick cubicles: PU + PCM cubicle.

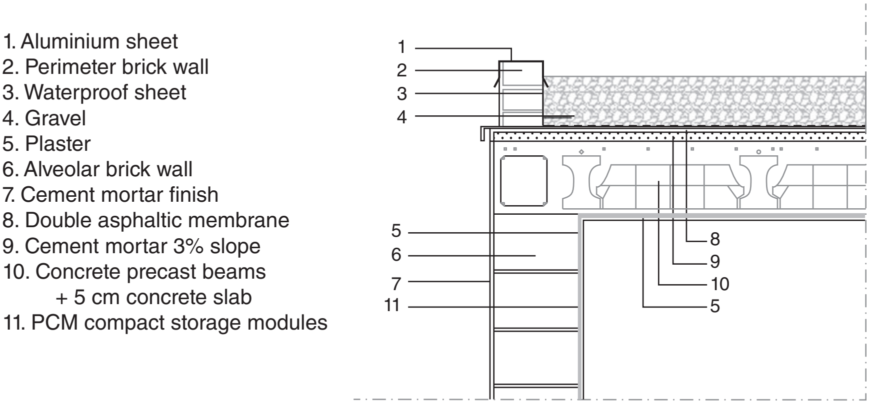
13.6 Alveolar brick cubicles: construction phase.

The differences between these two alveolar brick cubicles are:

1. Reference cubicle (Alveolar) (Fig. 13.7): the alveolar brick has a special design which provides both thermal and acoustic insulation. No additional insulation was used in this cubicle.
2. PCM cubicle (SP25 + Alveolar) (Fig. 13.8): several CSM panels containing SP-25 A8 hydrate salt (Table 13.2) are located inside the cubicle, between the alveolar brick and the plastering in order to increase the thermal inertia of the wall (in the southern and western walls and the roof).



13.7 Construction system of the alveolar brick cubicles: reference cubicle.

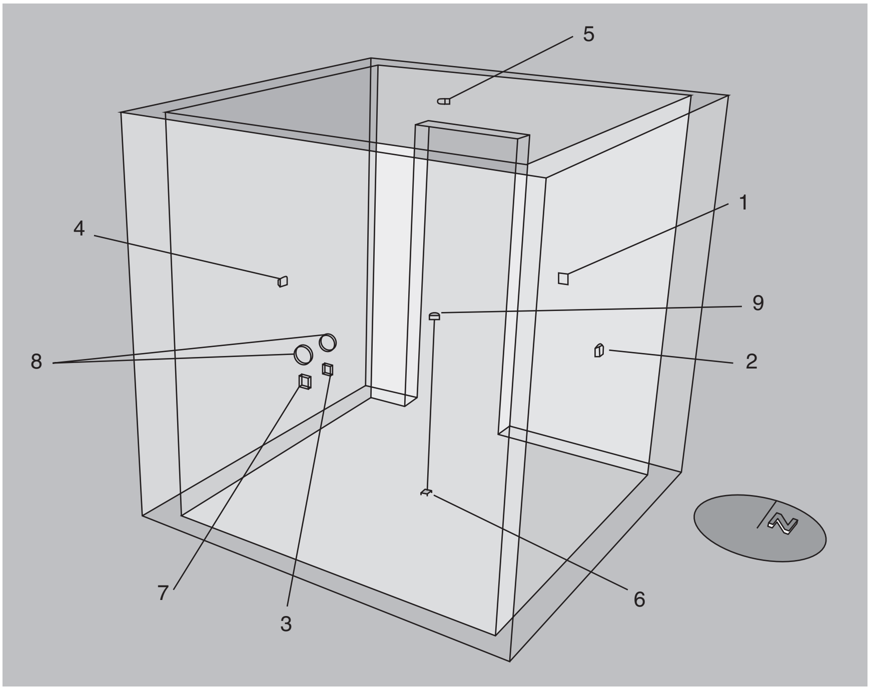


13.8 Construction system of the alveolar brick cubicles: PCM cubicle.

13.2.3 Instrumentation

The thermal behaviour of each construction system is analysed by measuring the following parameters every 5 minutes (Fig. 13.9):

- internal surface wall temperatures (1, 2, 3, 4, 5 and 6).
- external surface south wall temperature (7).
- heat flux at the south wall (internal and external surfaces) (8).
- internal ambient temperature and humidity at a height of 1.5 m (9).
- electrical consumption of the heating/cooling device.



13.9 Main measuring points in an experimental cubicle.

All cubicles were equipped with a heat pump (general inverter) and an electric oil radiator (Technofont TF09-12) for cooling and heating to simulate real conditions of a house. Their energy consumption was monitored to determine the real energy savings achieved when the cubicles remain within the comfort range. Moreover, a meteorological station was installed nearby; this meteorological station measured solar radiation, external ambient temperature and humidity and wind speed.

All temperatures were measured using Pt-100 DIN B probes, calibrated with a maximum error of ±0.3°C. The air humidity sensors are ELEKTRONIK EE21FT6AA21 with an accuracy of ±2%. To measure the electrical consumption of the heat pump, an energy meter CIRCUTOR MK-30-LCD-RS485 was used. The solar radiation was registered with a Middleton Solar solarimeter SK08 of first class, and the wind speed was measured with the instruments in the meteorological station available on site. Finally, all the data were registered using a data logger STEP DL01-CPU connected to a computer.

13.2.4 Experimental analysis

Two different experiments were performed in the experimental set-up:

- free-floating temperature, where no cooling system is used. The temperature conditions inside the cubicles are compared. The ones with PCM are expected to have a better behaviour.
- controlled temperature, where a heat pump or an electric oil radiator is used to set a constant ambient temperature inside the cubicle. The energy consumption of the cubicles is compared. The cubicles using PCM are expected to have lower energy consumptions.

For the LCA studies presented below, the data from the controlled temperature experiment were used. All the data and analysis can be found elsewhere (Cabeza *et al*., 2007; Castell *et al*., 2010; Castellón *et al*., 2009; Arce *et al*., 2012; de Gracia *et al*., 2011).

13.3 Life cycle assessment (LCA) methodology

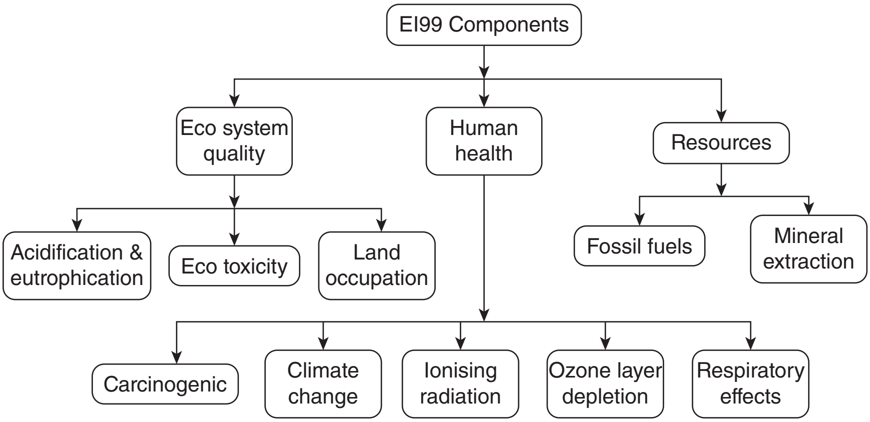
The present analysis is based on the ISO 14040-43 standard series (ISO 14040:2006, ISO 14041:1998, ISO 14042:2000, ISO 14043:2000) that are specified for LCA, which recommend four steps in order to perform an LCA study efficiently:

- definition of goal and scope
- inventory analysis
- impact assessment
- interpretation of results.

The methodology used for performing the LCA study in this chapter is based on the impact assessment methodology Eco-Indicator 99 (EI99) (PRé-Consultants, 2000) using the database EcoInvent 2009. According to this, ten midpoint impact categories are selected that are further grouped into three endpoint damage categories. Figure 13.10 shows the impact categories considered in the study.

13.4 PCM impact and selection

In this section, only the manufacturing and dismantling impacts of different PCM and encapsulation materials are considered in order to determine the most environmentally friendly PCM; therefore, three different types of PCM are compared. This is an important step for an optimal design of the application, as significant impacts can be reduced from the first design. In further sections, the operational and overall impacts are considered for a



13.10 EI99 impact and damage category groupings.

full understanding of the system. Moreover, the encapsulation material must also be considered. Its impact can be very significant in comparison with that of the PCM.

13.4.1 Goal and scope definition

The objective of this section is to evaluate the environmental impact of different PCM and their encapsulation material during the manufacturing and dismantling phases. Three PCM types are considered: paraffin, salt hydrate and ester. For the encapsulation, only one material is studied: aluminium. For the evaluation, three real PCMs with similar thermophysical properties are selected. Similar phase change temperatures and enthalpies will result in similar future applications and thermal behaviours. The selected PCMs and their properties are presented in Table 13.3.

13.4.2 Inventory analysis

To evaluate the manufacturing/disposal phase, a life cycle inventory is established (Table 13.4), which includes the different PCMs and their encapsulation material and their correlation with the materials used in the EI99 database. Since PCMs are not usually implemented in buildings, no data are usually available for their disposal in LCA databases. Here, this value has been estimated as the ratio of the disposal of PCM to its manufacturing impact for the cases of paraffin and salt hydrate (data are available for esters).

Table 13.3 Physical properties of the PCM in the impact evaluation with LCA

Property	RT-27 (Rubitherm)	SP-25 A8 (Rubitherm)	Ester
Melting point (°C)	28	26	27.6 (Feldman *et al.*, 1995)
Congealing point (°C)	26	25	26 (Feldman *et al.*, 1995)
Heat storage capacity (kJ/kg)	179	180	180 (Feldman *et al.*, 1995)
Density (kg/L)			
Solid	0.87	1.38	0.86–0.88[a]
Liquid	0.75		
Specific heat capacity (kJ/kg·K)			
Solid	1.8	2.5	1.9–2.1[a]
Liquid	2.4		
Heat conductivity (W/m·K)	0.2	0.6	0.143–0.146[a]

[a] Generic data for ester PCM.

Table 13.4 Components' relation with EI99 database for the three studied PCMs and their encapsulation

Component	Name in the EcoInvent database corresponding to the component
Hydrated salts	Calcium chloride, $CaCl_2$, at regional storage, CH, [kg] (#260)
Paraffin	Paraffin, at plant, RER, [kg] (#432)
Ester	Vegetable oil methyl ester, at esterification plant, FR, [kg] (#6592)
Aluminium	Powder coating, aluminum sheet, RER, [m^2] (#1166)
Disposal hydrated salts	Disposal, hydrated salts, to final disposal, CH, [kg] (calc.)
Disposal paraffin	Disposal, paraffin, to final disposal, CH, [kg] (calc.)
Disposal ester	Disposal, digester sludge, to incineration, future, allocation price, CH, [kg] (#6715)
Disposal aluminium	Disposal, aluminum, 0% water, to sanitary landfill, CH, [kg] (#2215)

13.4.3 Impact assessment

The life cycle assessment developed is based on the impact assessment method EcoIndicator 99 (EI99) (PRé-Consultants, 2000), extracted from the database EcoInvent 2009. The environmental impact is evaluated and expressed through ten damage categories further aggregated into three areas of protection: human health, eco system quality and natural resources. The evaluation of each impact category is given by:

Table 13.5 Life cycle inventory and impact during manufacturing and dismantling phases for the three studied PCMs and their encapsulation

Component	Mass used (kg)	Impact points (EI99)	Impact/kg material
Paraffin	99	20.570	0.208
Salt hydrate	99	5.676	0.058
Ester	99	3.247	0.033
Aluminium	61	5.081	0.083
Disposal paraffin	99	0.589	0.006
Disposal hydrated salts	99	0.163	0.002
Disposal ester	99	0.093	0.001
Disposal aluminium	61	0.154	0.003

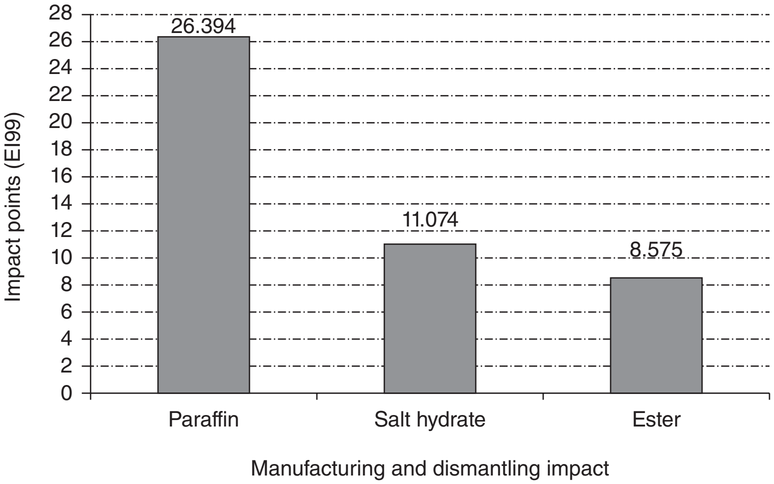
$$IMP_j = \sum_k d_{k,j} \cdot LCI_k \qquad [13.1]$$

where IMP_j is the j damage category, $d_{k,j}$ is the coefficient of damage extracted from the considered database (PRé-Consultants, 2000) associated with the component k and damage j, and finally the LCI_k is the life cycle inventory entry (i.e., kg of paraffin). For this study, 99 kg of each PCM are considered, as this will be the amount for a later overall impact assessment with experimental data (Menoufi *et al.*, 2013). Table 13.5 presents the manufacturing and dismantling impact for each PCM and its encapsulation. Results per unit of mass are also presented.

13.4.4 Interpretation of results

Figure 13.11 presents the aggregated impact for the manufacturing and dismantling phases. The dismantling impact of the PCM is not significant compared to the manufacturing phase (less than 3%). Comparing the three different types of PCM, paraffin is the one with the highest impact, being 2.4 times that of the salt hydrate and 3 times that of the ester. Therefore, natural materials, such as salt hydrates or esters, are more environmentally friendly and must be prioritised when selecting a PCM.

Moreover, the encapsulation material is also of high importance. In this study, aluminium accounts for 20% of the overall impact in the case of paraffin. This contribution is dramatically increased in the case of natural PCM: 47% for salt hydrates and 61% for esters. Therefore, the determination of an environmentally friendly material for the PCM encapsulation is also of high importance.



13.11 Manufacturing and dismantling impact results for each PCM and their encapsulation.

13.5 LCA of buildings including PCMs: case studies

13.5.1 Goal and scope definition

The main goal of this section is to evaluate the environmental impact of using PCM in building envelopes for energy savings. For such a goal, EcoInvent indicator EI99 is used, which limits the validity of the results to Europe. Moreover, the influence of climate conditions on the energy consumption of buildings limits the extrapolation of these results to similar climates.

The cases considered are two different construction systems, as explained in Section 13.2: conventional brick (Castell *et al.*, 2010; de Gracia *et al.*, 2010) and alveolar brick (Castell *et al.*, 2010, 2013).

The hypotheses used for the life cycle assessment in all the studied scenarios are:

- The lifetime of buildings is considered to be between 50 and 100 years (Gustavsson and Joelsson, 2010; Mithraratne and Vale, 2004). The expected lifetime for one cubicle is 50 years. However, a parametric analysis of lifetime is also developed and results for 75 and 100 years are presented.
- The construction of the cubicles is not taken into account, as it has little impact and can be omitted (Kellenberger and Althaus, 2009).
- The maintenance operations of the cubicles and the HVAC systems are considered equal for all the cases. Hence, it does not produce any difference in the overall global impact.

- The electricity used considers the production mix corresponding to the Spanish energy production system (#674 in the EcoInvent database).
- When no data were available in the EcoInvent database for disposal, an average ratio between the impacts of all the other components is used and their disposal is calculated.
- To evaluate the operational phase, measured energy consumption from the cubicles is used. Three different periods per year are defined:
  - Winter period: 4 months with similar heating demand to the third week of February 2009. Comfort conditions are achieved using an electrical radiator with a set point of 24°C.
  - Summer period: 4 months with similar cooling demand to the fourth week of August 2009. Comfort conditions are achieved using a heat pump with a set point of 24°C.
  - No controlled temperature: 4 months without temperature control.

13.5.2 Inventory analysis

In this step, the inventory list of all the materials used in the manufacturing and dismantling phases of the cubicles is shown and the correspondence between the experimental components and the EcoInvent database determined (Table 13.6). The energy consumption rates of the studied cubicles are also quantified (Table 13.7).

13.5.3 Impact assessment

As stated in Section 13.4.3, the life cycle assessment developed is based on the impact assessment method EcoIndicator 99 (EI99) (PRé-Consultants, 2000), extracted from the database EcoInvent 2009. A list of all the materials used in the construction of the cubicles studied and their impact is shown in Table 13.8. The impact per kg and per m^2 of façade is also given.

For the conventional brick construction system, the component with the highest impact is the bricks (around 35% of the total). Polyurethane and macroencapsulated paraffin represent 15% and 12% of the total, respectively. On the other hand, for the alveolar brick construction system, the component with the highest impact is the alveolar brick (around 65% of the total). SP25 and aluminium represent about 4% and 3.7% of the total impact, respectively. Note that not only must the addition of PCM be taken into account, but the encapsulation as well (in this case based on an aluminium sheet representing about 3–4% of the total). The impact for each damage category during the manufacturing phase is shown Table 13.9. Including PCM in the construction system increases the manufacturing and disposal impact (18.5% for the conventional brick and 8.5 for the alveolar brick).

Table 13.6 Components' relation with EI99 database

Component	Name in the EcoInvent database corresponding to the component
Brick	Brick, at plant, RER, [kg] (#495)
Alveolar brick	Brick, at plant, RER, [kg] (#495)
Base plaster	Base plaster, at plant, CH, [kg] (#536)
Cement mortar	Cement mortar, at plant, CH, [kg] (#537)
Steel bars	Section bar rolling, steel, RER, [kg] (#1170)
Concrete	Concrete, normal, at plant, CH, [m^3] (#504)
In-floor bricks	Concrete roof tile, at plant, CH, [kg] (#9244)
Asphalt	Mastic asphalt, at plant, CH, [kg] (#9245)
Polyurethane	Polyurethane, rigid foam, at plant, RER, [kg] (#1839)
Aluminium	Powder coating, aluminium sheet, RER, [m^2] (#1166)
Paraffin	Paraffin, at plant, RER, [kg] (#432)
Hydrated salts	Calcium chloride, $CaCl_2$, at regional storage, CH, [kg] (#260)
Disposal bricks	Disposal, building, brick, to final disposal, CH, [kg] (#2005)
Disposal alveolar bricks	Disposal, building, brick, to final disposal, CH, [kg] (#2005)
Disposal plaster	Disposal, building, mineral plaster, to final disposal, CH, [kg] (#2021)
Disposal mortar	Disposal, building, cement (in concrete) and mortar, to final disposal, CH, [kg] (#2007)
Disposal concrete + steel bars	Disposal, building, reinforced concrete, to final disposal, CH, [kg] (#2045)
Disposal in-floor bricks	Disposal, building, concrete, not reinforced, to final disposal, CH, [kg] (#2010)
Disposal asphalt	Disposal, asphalt, 0.1% water, to sanitary landfill, CH, [kg] (#2216)
Disposal PU	Disposal, building, polyurethane foam, to final disposal, CH, [kg] (#2040)
Disposal aluminium	Disposal, aluminium, 0% water, to sanitary landfill, CH, [kg] (#2215)
Disposal paraffin	Disposal, building, paraffin, to final disposal, RER, [kg] (calc)
Disposal hydrated salts	Disposal, building, hydrated salts, to final disposal, RER, [kg] (calc)

Table 13.7 Annual electric energy consumption (kWh) for each cubicle

Annual electric energy consumption (kWh)	REF	PU	PU + PCM	ALV	ALV + PCM
Summer period (4 months)	219	98	83	121	101
Winter period (4 months)	2393	1504	1504	1516	1516
Total for a whole year	2612	1602	1587	1637	1616

Table 13.8 Life cycle inventory and impact during manufacturing phase

REF	PU	RT27 + PU	ALV	SP25 + ALV	Component	Mass used (kg)	Impact points (EI99)	Impact/kg material	Impact/m^2 façade
X	X	X			Brick	5456	59.934	0.011	1.665
			X	X	Alveolar brick	6480	71.182	0.011	1.977
X	X	X	X	X	Base plaster	518	3.061	0.006	0.085
X	X	X	X	X	Cement mortar	608	3.193	0.005	0.089
X	X	X	X	X	Steel bars	262	2.209	0.008	0.123
X	X	X	X	X	Concrete	1240	3.263	0.003	0.181
X	X	X	X	X	In-floor bricks	1770	12.823	0.007	0.712
X	X	X	X	X	Asphalt	153	3.871	0.025	0.215
X	X	X	X	X	Polyurethane on roofs	9	3.452	0.365	0.384
	X	X			Polyurethane on walls	63	23.015	0.365	0.639
X	X	X	X	X	Aluminium	61	5.081	0.083	0.188
		X			Paraffin (PCM)	99	20.570	0.208	0.762
				X	Hydrated salt (PCM)	99	5.676	0.058	0.210
X	X	X			Disposal bricks	5456	13.251	0.002	0.368
			X	X	Disposal alveolar bricks	6480	15.738	0.002	0.437
X	X	X	X	X	Disposal plaster	518	1.071	0.002	0.025
X	X	X	X	X	Disposal mortar	608	1.523	0.003	0.042
X	X	X	X	X	Disposal concrete + steel bars	1492	4.000	0.003	0.222
X	X	X	X	X	Disposal in-floor bricks	1770	4.437	0.003	0.246
X	X	X	X	X	Disposal asphalt	153	0.252	0.002	0.014
X	X	X	X	X	Disposal PU	72	1.862	0.026	0.041
X	X	X	X	X	Disposal aluminium	61	0.154	0.003	0.006
		X			Disposal paraffin (PCM)	99	1.529	0.015	0.057
				X	Disposal hydrated salt (PCM)	99	0.014	0.000	0.001

Table 13.9 Impact results during manufacturing and each operational phase

	Cubicle	Eco system quality	Human health	Resources	Total
Manufacturing/ disposal impact	REF	5.47	47.56	47.56	100.59
	PU	6.02	55.18	80.01	141.22
	RT27 + PU	6.62	58.73	102.00	167.35
	ALV	6.05	52.704	71.57	130.32
	SP25 + ALV	7.00	56.43	77.81	141.24
Operational impact winter	REF	197.28	1944.18	1279.45	3420.91
	PU	123.97	1221.77	804.03	2149.77
	RT27 + PU				
	ALV	124.93	1231.21	810.26	2166.40
	SP25 + ALV				
Operational impact summer	REF	18.02	177.63	116.90	312.55
	PU	8.08	79.62	52.40	140.09
	RT27 + PU	6.84	67.41	44.36	118.61
	ALV	9.99	98.48	64.81	173.29
	SP25 + ALV	8.29	81.67	53.74	143.70

The operational impact for the winter period presents no differences between the cubicles due to the addition of PCM, since in cold weather conditions, temperatures do not reach the melting point of the PCM; hence no energy savings are achieved. However, during the summer period, the operational impact is reduced (15% for the conventional brick and 17% for the alveolar brick, every year). However, the rates of electrical energy consumption during the winter period are more than ten times higher than those during the summer period for the two cubicles. Thus, the effect of PCM, which is only effective under summer weather conditions, remains small. Those findings are represented by the year-round results of the operational impact of the RT27 + PU and SP25 + ALV cubicles, which are only about 1% lower than that of the PU and ALV cubicles.

13.5.4 Interpretation of results

The results of Eq. [13.1] are single score indicators representing the potential impact on the environment through different damage categories. The coefficient of damage for the natural resources damage category is expressed in MJ of surplus energy needed for future extraction. For the ecosystem quality damage category, the coefficient of damage represents the loss of species over a certain area, during a certain time (% plant, species/m^2 year). Finally, the damage to human health is expressed as the number of years of life lost and the number of years lived disabled (disability adjusted life years, DALYs).

Table 13.10 Global impact results for each cubicle

	Cubicle	Eco system quality	Human health	Resources	Total
Overall global impact	REF	220.77	2169.37	1443.91	3834.05
	PU	138.07	1356.57	936.44	2431.08
	RT27 + PU	137.43	1347.91	950.39	2435.73
	ALV	140.97	1382.39	946.64	2470.01
	SP25 + ALV	140.22	1369.31	941.81	2451.34

Table 13.10 shows the results from cradle to grave for the five studied cubicles. The addition of polyurethane reduces the global operational impact in comparison to the reference case from 3,850 to 2,431 impact points. On the other hand, the addition of PCM further reduces this value, but only by 11 points (0.5%) in comparison to the PU cubicle. Similar results are obtained for the alveolar brick construction system, where the addition of PCM results in an overall impact reduction of 19 points (0.8%).

Therefore, the benefits of including PCM in the building envelope must be increased to significantly reduce the overall impact. This could be achieved by an extended use of the PCM during the year, by extending the lifetime of the PCM, or by designing PCM systems that also work during the winter period.

13.6 Improvement in PCM use

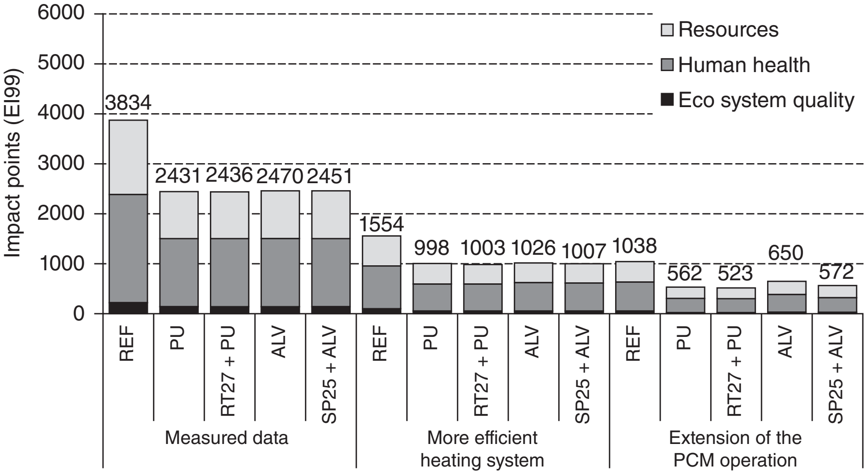
In order to determine the potential of PCM for environmental impact reduction, different scenarios with optimised use of the PCM must be analysed. In this section, two theoretical scenarios are studied.

13.6.1 More efficient heating systems

The electrical oil radiator is replaced by a heat pump as a heating system (COP considered as 3), reducing the operational impact of the cubicles during winter to one third due to its higher efficiency (Table 13.11). The benefits of including PCM are not directly increased but highlighted. The operational impact during the summer period is more significant to the global impact, and thus so are the PCM benefits. However, these benefits are still very small (between 1 and 2%, Fig. 13.12).

Table 13.11 Impact results during operational phase for improved PCM benefits

Operational Impact	Cubicle	Eco system quality	Human health	Resources	Total
More efficient heating system	REF	65.76	648.07	426.49	1140.32
	PU	41.32	407.26	268.02	716.60
	RT27 + PU				
	ALV	41.64	410.40	270.09	722.13
	SP25 + ALV				
Extension of the PCM operation	REF	54.07	532.88	350.69	937.64
	PU	24.24	238.86	157.19	420.28
	RT27 + PU	20.52	202.23	133.08	355.83
	ALV	2.98	295.45	194.43	519.86
	SP25 + ALV	24.86	245.00	161.23	431.09



13.12 Impact results for each cubicle and studied scenario for a lifetime of 50 years.

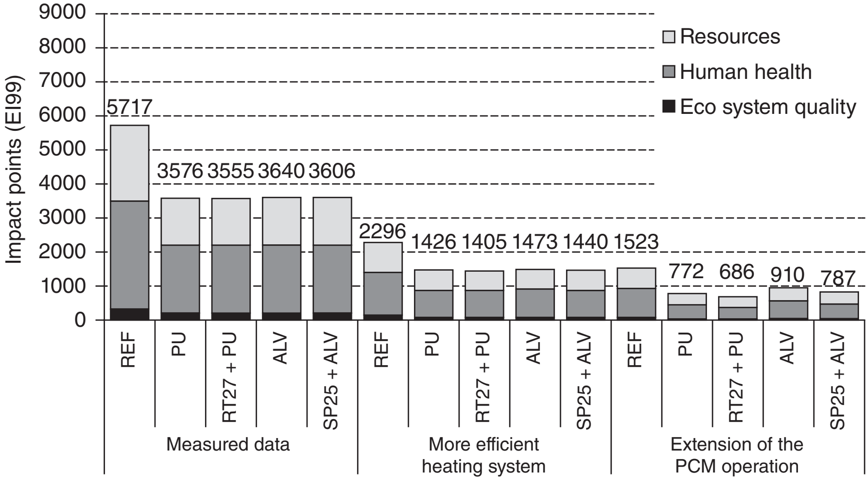
13.6.2 Extension of the PCM operation

Since the PCM considered only operates during summer periods, its use would be more effective in regions where summer weather conditions are predominant throughout the whole year (summer period of 12 months, Table 13.11). For such cases the impact reduction would increase to 10–12% (Fig. 13.12).

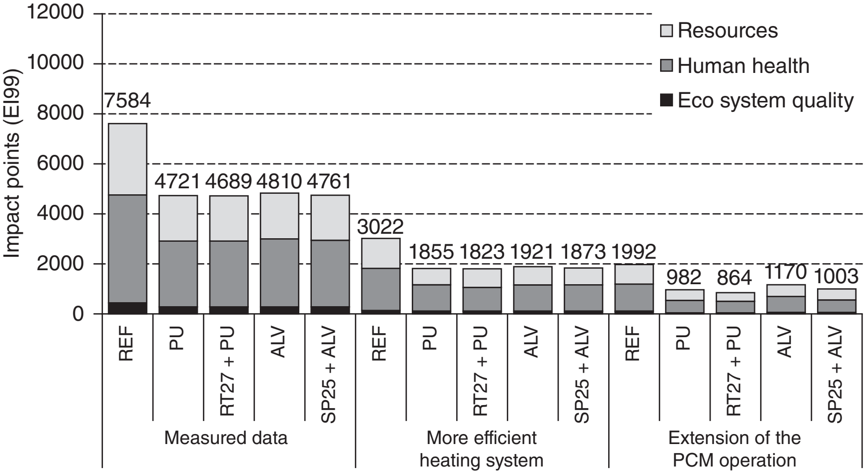
Further studies should focus on the use of PCM in applications dealing with both heating and cooling. In such a case, the benefits of the PCM would be extended to whole year operation.

13.6.3 Extending the lifetime of buildings and PCMs

The lifetime of buildings is considered to be between 50 and 100 years (Gustavsson and Joelsson, 2010; Mithraratne and Vale, 2004; Kellenberger and Althaus, 2009). Since PCM reduces the environmental impact during the operational phase, a longer lifetime of buildings and PCM will increase its benefits. Therefore, different lifetimes must be evaluated (here 75 and 100 years). This extended lifetime is considered for the three scenarios previously analysed: measured data, more efficient heating system, and extension of the PCM operation (Figs 13.13 and 13.14).



13.13 Impact results for each cubicle and studied scenario for a lifetime of 75 years.



13.14 Impact results for each cubicle and studied scenario for a lifetime of 100 years.

For the best scenario considered (extended benefits of the PCM and 100 years lifetime), the impact reduction due to the inclusion of PCM is 12% for the conventional brick and 14% for the alveolar brick. The parametric analysis of lifetime determines that the manufacturing impact of the PCM is balanced out by the impact savings during the operational phase in 61 years for the conventional brick with paraffin and in 18 years for the alveolar brick with salt hydrate.

13.7 Problems in undertaking an LCA of buildings including PCMs

Phase change materials are a suitable technology for energy consumption reduction in buildings. The environmental benefits of such savings can be determined and fully understood by a life cycle assessment. However, little information is available for PCM in LCA databases. Thus, accurate LCA studies are difficult to perform.

New LCA indicators for PCM are required for further analysis of their benefits and limitations. It has been demonstrated that incorporating PCM in building envelopes can be beneficial; however, their manufacturing impact is important and must be optimised. For such a purpose, detailed indicators must be available for different types of PCM in order to select the best one for each application. Based on state-of-the-art studies, new phase change materials are required, based on natural resources to decrease their manufacturing impact. Moreover, extended operational use of PCM is required in order to maximise their benefits.

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14

Life cycle assessment (LCA) of wood-based building materials

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Abstract: In this chapter we discuss major issues regarding life cycle assessment (LCA) and environmental performance analysis of wood-based building materials. We follow the life cycle of a wood product, beginning with a discussion of sustainable forestry and the growth of trees. We then discuss the processes of manufacturing wood-based building products, focusing on issues of adhesives and preservatives. We discuss the design and construction of buildings and infrastructure made of wood, with an emphasis on eco-design processes. We describe the system-wide material and energy flows associated with wood-based construction in a life cycle perspective, and discuss the climate benefits of using wood material from sustainably managed forests.

Key words: wood-based products, life cycle assessment, forestry, eco-design, climate change mitigation.

14.1 Introduction

14.1.1 Wood products and a vision of sustainability

A great challenge for humankind is to transition to a sustainable society. Such a society will require the use of renewable materials, coupled with large reductions in the overall use of natural resources and in environmental impacts including greenhouse gas (GHG) emissions. The built environment is a key sector in meeting this challenge, due to its large use of natural resources and primary energy and its significant impacts on the environment. In Europe, for example, the built environment accounts for about 42% of energy use and produces 35% of total GHG emissions (EC, 2007). Understanding and improving the environmental profile of the materials used in the construction sector is essential to reducing the environmental impact of the built environment.

Wood is an inherently renewable material that is produced through natural processes in forest ecosystems. The life cycle of wood building materials includes the growth of trees, the harvest and processing of woody biomass, the manufacture and assembly of wood-based products, the utilisation and maintenance of the building, and the disassembly and end-of-life management of the wood material (Yaro, 1997). When trees are harvested from a sustainably managed forest, new trees re-grow in their place, providing a renewable source of biomass feedstock. As part of a continuous cycle of material flow, use of wood products avoids the build-up of waste materials from manufacturing or disposal, as the biomass residues can be used as a source of bioenergy. Wood products have the potential to play a major role in the development of a sustainable built environment, particularly through the integration of material and energy flows in the construction sector with those in the forestry, energy, industrial and waste management sectors.

14.1.2 Life cycle assessment (LCA) of wood-based building materials

A growing concern about the environmental effects of the production and use of goods, as well as about how goods are disposed of at the end of their service life, has led to increasing interest in wood-based products made in a sustainable environmental manner. Long-term sustainable development is a key concern in many countries, giving rise to regulations regarding the impact of products during their life cycle, including the commitment to create effective reverse logistics strategies to manage post-use materials. Improved knowledge of the environmental impacts of the materials and processes associated with productive sectors including the wood-based sector is a key factor in guiding efforts towards green production processes and green markets (Bovea and Vidal, 2004).

Life cycle assessment (LCA) is a tool to assess the environmental impact of materials, products and services, and should contribute to the decision-making process towards sustainability (Baumann and Tillman, 2004). The LCA methodology has been applied to a wide range of processes and sectors. Specific to the wood products sector, numerous studies have been carried out to investigate the environmental performance of wood-based products destined for different uses such as floor coverings (Jönsson *et al.*, 1997; Petersen and Solberg, 2003; Nebel *et al.*, 2006), window frames (Richter and Gugerli, 1996; Salazar and Sowlati, 2008), particleboard (Rivela *et al.*, 2006), medium density fibreboard (Rivela *et al.*, 2007), hardboard (González-García *et al.*, 2009b), furniture (Taylor and van Langenberg, 2003), goods containers (González-García *et al.*, 2011a), paper pulp (González-García *et al.*, 2009c, 2011b), wall assemblies (Werner, 2001; Lippke and Edmonds, 2006), and packaging materials (Farreny *et al.*, 2008). Other studies have analysed complete buildings rather than building components, including single-family houses (Buchanan and Honey, 1994; Scharai-Rad and Welling, 2002; Lippke *et al.*, 2004) and apartment buildings (Börjesson and

Gustavsson, 2000; Gustavsson *et al.*, 2006b; Gustavsson and Sathre, 2006; John *et al.*, 2009; Gustavsson *et al.*, 2010). These studies have aimed to document the overall environmental performance of the wood-based products, as well as identify the processes with the highest contributions to environmental impact. Differences among LCA studies of wood-based products concern, for example, the system boundaries of the analysis (cradle-to-gate or cradle-to-grave) and the life cycle inventory data (primary or secondary data).

The LCA methodology allows not only the quantification of current environmental profiles but also the identification of improvement potentials in order to reduce future environmental impacts. LCA studies typically identify the most important contributors to the environmental impacts, which allows focused effort on reducing those impacts. End-of-life management of wood-based products is found to be an important factor in energy and GHG balances. Recovery of the post-use material for use as bioenergy is beneficial, while disposal in landfills typically causes greater impacts. Forest activities to produce roundwood (the main raw material in wood-based products) may also be an environmental hot spot due to their contribution to impact categories such as acidification, eutrophication, and formation of photochemical oxidants. The application of agrochemicals and use of forest machinery powered by fossil fuels are the main contributors in this area. Another hotspot involves activities related to processing of wood into wood-based panels (e.g., production of fibreboard) due to the use of petroleum-based resins such as urea- and phenol-formaldehyde. Nevertheless, a general conclusion of comparative studies of wood-based vs. non-wood materials is that wood products from sustainably managed forests have the potential to produce significantly less life cycle environmental impact than other common building materials such as concrete and steel (Werner and Richter, 2007; Sathre and O'Connor, 2010a).

In this chapter we discuss major issues regarding LCA and environmental performance of wood-based building materials. We follow the life cycle of a wood product, beginning with a discussion of forestry and the growth of trees. We then focus on the processes of manufacturing a wood-based building product, followed by a discussion of environmentally compatible design and building with wood. We then describe system-wide material and energy flows associated with wood-based construction, and discuss the climate benefits of using wood material from sustainably managed forests.

14.2 Forestry and wood production

The life cycle of a wood product begins with the germination of a tree seed, and continues through the growth and harvest of the tree and the manufacture and use of the product. Consideration of forest ecosystems is essential to accurately understand the eco-efficiency of wood product use. In contrast to other building materials, such as steel and concrete that are manufactured through technological processes in human-made factories, wood is produced through natural biological processes occurring in growing trees. The process of photosynthesis, powered by solar energy captured by tree leaves, produces sugars from carbon dioxide taken from the air and water taken from the soil. These sugars are converted by the trees into complex organic molecules such as cellulose, hemicellulose and lignin, which combine in a composite matrix to form wood. The wood material that is produced organically by living trees can then be harvested and processed into various types of construction products. Meaningful environmental assessment of a wood-based product generally requires that the wood be sourced from sustainable forestry. Forests managed for timber production are typically considered sustainable if the harvests remove no more wood than is grown, i.e., if the landscape-level forest inventory is not declining over time (Lippke *et al.*, 2011). Forests managed for sustainable multiple use attempt to include a balance between timber output, ecosystem services, and social values, acknowledging that not all forests can fulfil all needs.

Globally, about 31% of total land area is covered by forests, corresponding to a forest area of just over 4 billion hectares (FAO, 2010). More than half of the total forest area is in five countries: Russia, Brazil, Canada, USA and China. At the global level, forest area decreased at a rate of about 5.2 million hectares per year during the period 2000 to 2010, down from an estimated 8.3 million hectares per year during the period 1990 to 2000. This decrease in forest area is the net result of two opposing processes: deforestation, occurring at a rate of about 13 million hectares per year during the period 2000 to 2010 (down from about 16 million hectares per year in the 1990s), and afforestation and natural expansion of forests in other areas. Most of the loss of forest currently occurs in tropical regions, particularly in Africa and South America. Most of the increase in forest area occurs in the temperate and boreal zones, as well as in some emerging economies. In Europe, net forest area increased by about 700,000 hectares per year during the period 2000 to 2010, as a result of new forest planting and natural expansion of forests onto former agricultural land. In North America, forest land area has been quite stable in recent decades. In China, large-scale afforestation efforts have increased the forest area by an average of 3 million hectares per year during the period 2000 to 2010.

The quantity and quality of wood biomass produced in a forest can be significantly influenced by forest management activities. A continuum of forest management intensities is possible, from an intense plantation regime involving species selection and nutrient management to the non-management and non-use of forests (Eriksson *et al.*, 2007; Poudel *et al.*, 2012). A complete

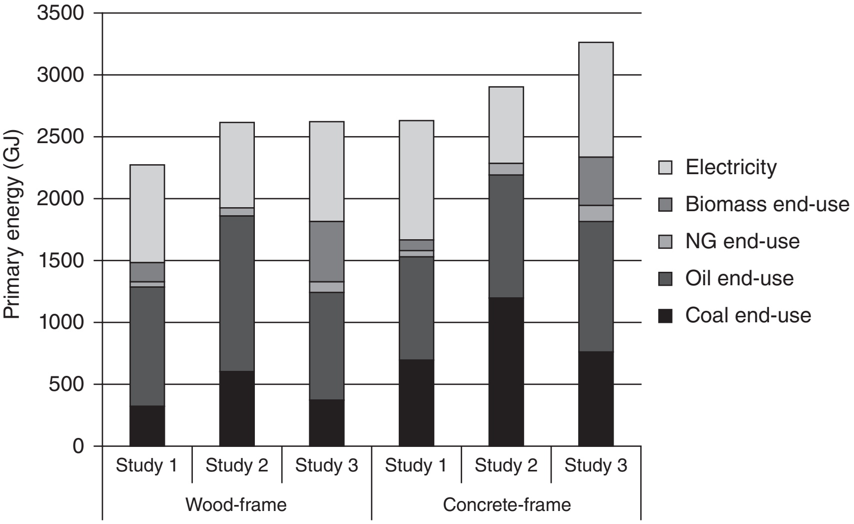
LCA of wood-based building materials will take into account the impacts resulting from forestry activities such as silvicultural operations, logging operations and secondary hauling (Berg and Lindholm, 2005; González-García *et al*., 2009a). In general, increasing the intensity of forest management results in greater production of biomass, though the return on management inputs tends to diminish as intensity increases. More intensive forest management, while producing marginally more environmental impact within the forest than less intensive management, may result in less overall impact due to the greater quantities of wood produced that can substitute for non-renewable materials and fuels (Eriksson *et al*., 2007; Sathre *et al*., 2010).

14.3 Wood product manufacture

14.3.1 Life cycle inventory data

Essential procedures in identifying and assessing the environmental impacts of wood-based product manufacturing systems include the definition of system boundaries, functional unit, and allocation methods, as well as the collection and processing of relevant life cycle inventory (LCI) data (ISO, 2006). Cradle-to-gate LCA studies cover all processes from natural resource extraction up to the factory exit gate, and exclude from assessment the product use and end-of-life management. Other studies employ a cradle-to-grave perspective including the maintenance of the product and post-use management such as recycling or disposal. The quality of LCI data is a key factor in the validity of the analysis, and adequate data must be used if the results are to be representative of the sector. LCAs of innovative products or processes will ideally use LCI data taken directly from field studies of production systems, which may be complemented with secondary data taken from databases. Variability in LCI data is inevitable, because different physical processes can be used to produce the same material, and each process has unique requirements and effects on the environment. The efficiency of industrial technologies has generally improved over time resulting in differences in energy requirements and emissions between materials processed by state-of-the-art technologies and those made in older facilities. Variation is also seen geographically, as technological innovations diffuse across countries and regions.

Data on industrial energy use can also vary depending on the methodology used to obtain the data. System boundaries of an energy analysis can range from a restrictive analysis of direct energy and material flows of a particular process, to an expansive analysis including energy and material flows of entire industrial chains and society as a whole (Boustead and Hancock, 1979). Data may be direct measurements of a particular machine



14.1 Primary energy used for production of materials for wood- and concrete-framed versions of a four-storey apartment building, using specific energy use data from three different process analyses. Study 1 is Fossdal (1995), Study 2 is Worrell *et al.* (1994) and Study 3 is Björklund and Tillman (1997) (adapted from Gustavsson and Sathre, 2004).

or factory, or may be aggregated for an entire industrial sector. As an illustration of such variability, Fig. 14.1 shows the primary energy used for producing materials for functionally equivalent versions of a four-storey apartment building made with a wood frame and a concrete frame, using specific energy use data from three different European process analyses. These results suggest that in spite of absolute differences between the analyses (due to varying system boundaries, regional differences, etc.), the *relative* energy use of wood vs. non-wood materials is more or less consistent (Gustavsson and Sathre, 2004).

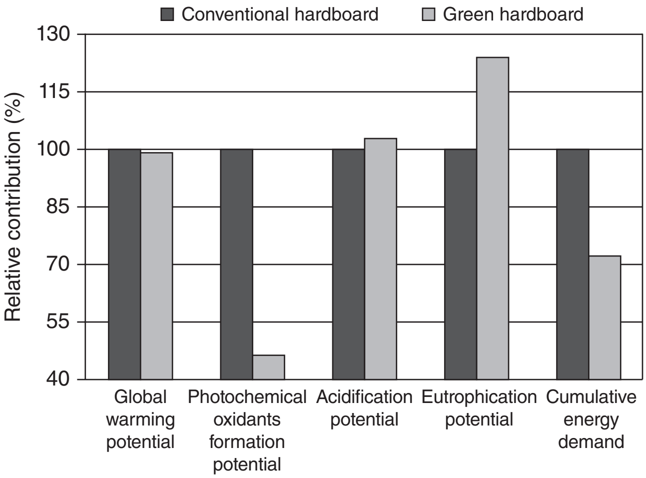
14.3.2 Wood adhesives: conventional and new green formulations

Distinctive characteristics of wood raw materials are its anisotropic and variable nature. Wood has different physical and mechanical properties in the longitudinal, radial, and tangential directions, due to different functional requirements within living trees. Furthermore, wood properties vary between tree species and climatic zones of origin, and within individual trees due to growth rings and knots. Traditionally, these factors were accommodated in building construction by conservative safety factors and by informed judgement of experienced woodworkers. More recently, these sources of performance variability are being overcome by the use of composite wood products, made by adhesively bonding together many smaller pieces of wood, thus creating a more homogeneous and predictable material. The individual wood elements can be of different sizes; examples include glue-laminated beams made from wooden boards, plywood panels made from thin wooden veneers, oriented strand board (OSB) made from strands of wood, particleboard made from flakes or particles of wood, and fibreboard made from individual wood fibres.

Composite wood products require the use of adhesives for bonding the wood elements. Typically, petroleum-based adhesives such as urea- or phenol-formaldehyde are used. The use of formaldehyde-based adhesives results in formaldehyde emissions derived from the production and end-use processes, resulting in negative environmental impacts (Imam *et al.*, 1999; US EPA, 2002). Environmental consequences of wood-based panel manufacture using conventional adhesives are documented in the literature (Rivela *et al.*, 2006, 2007; González-García *et al.*, 2009b). These studies found that the use of petroleum-based adhesive is an environmental hot spot that is responsible for pollutant emissions with significant contributions in impact categories including global warming, photochemical oxidants formation, acidification, eutrophication and toxicity. Evidently, potential environmental improvements are possible if these conventional adhesives are substituted by new green formulations.

Research in recent years has focused on reduction of adhesive use and on development of environmentally compatible adhesives. More natural and safer alternatives could be lignin-based materials (Moubarik *et al.*, 2009) such as lignosulfonates, organosolved lignin, kraft lignin, flavonoid-based tannins from certain trees (Widsten *et al.*, 2009), starch from renewable sources, or glues derived from animal tissue casein (Imam *et al.*, 1999). Widsten and Kandelbauer (2008) assessed the production of fibreboard using an adhesive based on lignin with enzymes, giving good results at lab and pilot scale. Moubarik *et al.* (2009) proposed the use of resins made from cornstarch and tannin from the quebracho tree (*Schinopsis balansae*) as an adhesive to partially substitute phenol-formaldehyde resin in plywood production. Panels with improved mechanical properties and water resistance, as well as lower formaldehyde emissions, were obtained and the environmental profile was improved.

Recent studies have documented the environmental benefits of using alternative adhesives instead of conventional petroleum-based adhesives. González-García *et al.* (2011d) studied hardboard production using a two-component bio-adhesive formulated with a wood-based phenolic material and a phenol-oxidising enzyme. Compared to conventional hardboard



14.2 Comparison of environmental profiles of conventional and green hardboard production processes.

manufacture using phenol-formaldehyde resin (González-García *et al.*, 2009b), significant environmental benefits were achieved in categories such as photochemical oxidants formation and cumulative energy demand, in a cradle-to-gate perspective (see Fig. 14.2). The highest benefits were reported in terms of photochemical oxidant formation, with reductions of up to 50%. The results indicated that the production of green hardboard using a green adhesive should be industrially viable, meeting the performance specifications of hardboard produced with conventional phenolic resin. However, special attention should be paid to the production of these adhesives, especially if the enzyme laccase is used in the composition, as the laccase production process is energy intensive which could limit the environmental benefits.

14.3.3 Wood decay and preservation

Wood is a biologically-produced material, and as part of natural material cycles can be decomposed by a variety of organisms such as fungi and insects. This characteristic contributes to the sustainability of wood products because it provides for natural recycling of the constituent materials making up the wood. However, it may also be problematic because it could lead to deterioration of the wood product while still in service. Susceptibility of wood to decomposition depends on the properties of the wood (some species are more naturally resistant to decay than others) as well as the moisture content of the wood (most decay organisms require a moist environment to live and multiply).

Several options exist for reducing deterioration of wood products in service. First, good design practices that prevent or minimise standing water on wood surfaces will reduce the moisture content of the wood, hindering the growth of decay organisms. Second, choosing wood species that are more naturally decay resistant will reduce deterioration, though this option may be limited by the tree species available. Third, surface coatings may be applied to the wood to repel water and maintain a low moisture content. Finally, the wood may be treated with chemical wood preservatives that kill the decay organisms. Two main categories of chemical treatments exist: oil-borne preservatives such as creosote and pentachlorophenol, and water-borne preservatives such as copper-based solutions (Lebow, 2010). Regulations in many countries define the allowable uses of different types of preservatives, which differ between, for example, residential and industrial applications. The landscape of chemical wood preservatives has changed significantly in the last decades towards safer materials, and continues to change. The use of arsenic in wood preservative solutions, such as the once common chromated copper arsenate (CCA), has been phased out, particularly in residential applications. In the European Union, the Biocidal Products Directive (98/8/EC) covers many common wood preservatives including CCA, resulting in restrictions on their use. Recently, the Commission Directive 2011/71/EU included creosote in this category, leading to increasing restrictions on creosote use in Europe.

From an environmental perspective, there are advantages and disadvantages of using chemical wood preservatives. By prolonging the service life of wood products, chemical preservation reduces the level of forest harvest needed to sustain a given function from wood product use. However, this comes with the burden of an increased level of toxic materials in the built environment and in the manufacturing and waste management sectors. Furthermore, opportunities for recycling of preservative treated wood are more limited than for untreated wood (Felton and de Groot, 1996). Particular concerns include worker exposure to emissions from recycling processes, and interference by preservatives with the bonding of adhesives. Energy recovery from treated wood is also restricted, although treated wood can be incinerated under suitable combustion conditions with flue gas cleaning and appropriate ash disposal.

Research is underway to develop effective wood preservation methods that do not add to the toxicity burden in the environment. For example, the acetylation process chemically modifies wood and makes it more dimensionally stable and less susceptible to biological attack, particularly by decay fungi (Rowell, 2006). In this process, acetic anhydride reacts with the free hydroxyl groups on large molecules in the wood cell walls. The hydroxyl groups are replaced with acetyl groups, and acetic acid is formed as a by-product. Although the performance of acetylated wood is generally superior to untreated wood, the current cost of the process makes it uneconomical for most applications. Another example of wood preservation through chemical modification is furfurylation, in which wood is treated with furfuryl alcohol and then heated to cause polymerisation (Lande *et al.*, 2008). The result is a cross-linked furan polymer that is chemically bonded to the wood cell wall polymers. Furfuryl alcohol is a renewable material derived from furfural, produced from hydrolysed biomass waste. Furfurylated wood is currently produced commercially on a relatively small scale by several European firms.

Another effective wood preservative with decreased toxicity concerns is borate. Borates are low cost, odourless and colourless, have very low toxicity to mammals, and are broadly effective against decay fungi and insects. However, borate compounds do not become fixed in the wood and can readily be leached out by water. Use of borate-treated wood is thus typically restricted to internal structural members and other uses where the wood will not be exposed to water or ground contact. Researchers are investigating methods for maintaining borate compounds in the wood matrix to enable long lasting wood protection in wet environments. However, the preservative properties of borates are primarily due to the tetrahydroxyborate ion formed upon exposure to water, thus complete immobilisation of the borate compound is undesirable (Obanda *et al.*, 2008). Numerous strategies have been proposed to reduce borate leaching from wood to ensure long-lasting protection, including surface treatments to hinder water uptake, formation of organo-boron compounds that bind with wood molecules, inorganic combinations of boron and metals or silicon, and polymerisation of boron-containing compounds within the wood cells. An example of the last mentioned strategy is described by Thevenon *et al.* (2009), who treated wood with boric acid and wood tannin resins and found considerable resistance against leaching and fungal attack.

14.4 Building with wood materials

14.4.1 Materials, components and buildings

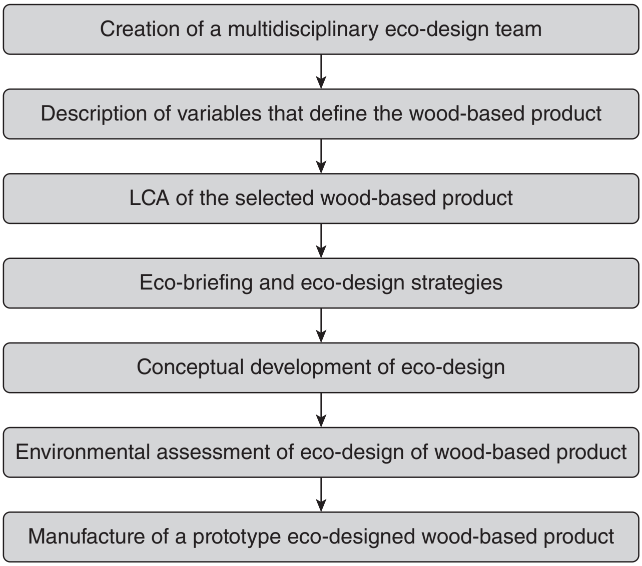
Various materials are combined to form building components such as walls, windows, floors, insulating materials, doors and furniture. In turn, these components are combined to form complete buildings. A commonly used unit by which environmental impacts are calculated is a unit mass of individual materials. For example, industrial process analyses commonly determine the primary energy required to manufacture a kilogram or tonne of material. This information can be useful input for a more elaborate analysis, but by itself is incomplete because the function of different materials cannot be directly compared. One tonne of lumber, for example, does not fulfil the same function as one tonne of steel. Similar analysis on the basis of unit volume of material suffers the same shortcoming. A more useful functional unit is to compare performance on the basis of the function provided by building components. That is, building components that provide the same function, but made of different material combinations, can be compared (Gustavsson and Sathre, 2011).

The environmental impacts of many such construction components have been quantified in recent years, generally showing that wood-based components have lower overall impacts than comparable products made of non-wood materials (Werner and Richter, 2007; Sathre and O'Connor, 2010a). For example, Richter *et al.* (1996), Kreissig *et al.* (1997), Asif *et al.* (2002), and Salazar and Sowlati (2008) all compared the environmental impacts of window frames made from different materials. LCA has also been used to compare the environmental performance of flooring materials (Jönsson *et al.*, 1997; Jönsson, 1999; Nebel *et al.*, 2006). González-García *et al.* (2012a) assessed the environmental profile of a ventilated wooden wall structure made of wood-based panels and other materials such as mineral wool and polyester resins. This study showed that the production of the wood-based panels was the main environmental hot spot due to the requirement of petroleum-based resins whose production results in toxic substance emissions as well as transport contributions. Environmental improvements were proposed by González-García *et al.* (2012a) based on the use of wood panels with lower environmental impact using wood from nearby forest plantations and renewable energy sources to fulfil the process energy requirements.

Buildings are complex systems, and a particular material may fulfil more than one function (e.g., structural support and thermal insulation), and a given building function may be fulfilled by a combination of materials. Changing one material may impact on other functions in various ways, for example sound transmission, fire protection, and the overall weight of the building and the required foundation design. Robust LCAs must ensure that these complex interactions between multiple system elements are accounted for within the functional unit. This is ideally done by comparing functionally equivalent versions of complete buildings made with different material mixes (Kotaji *et al.*, 2003). This can be based on a generic hypothetical building (Björklund and Tillman, 1997), or a case study of completed buildings (Gustavsson *et al.*, 2006b; Lippke *et al.*, 2004; John *et al.*, 2009). A general conclusion of such comparative studies is that wood-based construction systems tend to have lower environmental impacts than functionally equivalent systems using non-wood materials (Werner and Richter, 2007; Sathre and O'Connor, 2010a).

14.4.2 Eco-design in wood-based construction

The growing demand for knowledge about how products are made, where they are sourced from, what the environmental consequences of their production and use are, and how they are disposed of at the end of their service lives has provided an opportunity for the wood products sector to excel in the emerging market for green products (Bovea and Vidal, 2004). Embracing environmental strategies for optimising the life cycle of their product (including design, manufacture, use and end-of-life management), progressive manufacturers have adopted environmental accreditation with eco-labels such as Forest Stewardship Council (FSC) or Carbon Footprint (CF) as means to differentiate their products (Bovea and Vidal, 2004; Veisten, 2007). The application of sustainability criteria to the product design has received increasing attention in recent years. Eco-design, or Design for the Environment (DfE), is a concept that integrates multifaceted aspects of design and environmental considerations. The development of sustainable solutions for products or services is based on the minimisation of negative consequences under economic, environmental and social perspectives, throughout and beyond the life cycle of products (Charter and Tischner, 2001). Eco-design is a process that seeks to reduce the inherent environmental burdens associated with products. The stages of the eco-design methodology are shown in Fig. 14.3.



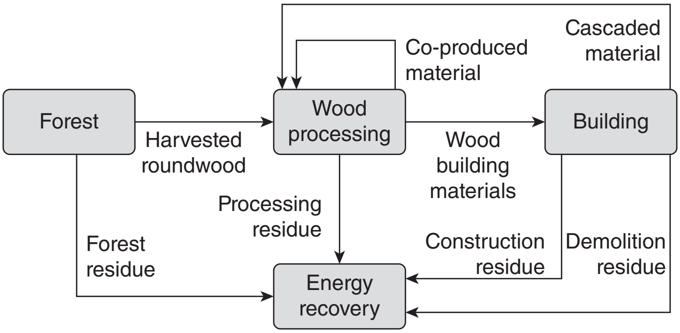
14.3 Stages in the methodology for the eco-design of a wood-based product.

An interdisciplinary design team is essential to the improvement of environmental aspects of product life cycle stages including material sourcing, processing, transport, packing, installation, use, maintenance, dismantling and end-of-life. Initial decisions made during the design phase have important consequences throughout the whole life cycle (Brezet and van Hemel, 1997). The eco-design process seeks to implement a vision for the reduction of overall environmental impacts, where the final disposition of the product is contemplated before it is even produced, with a plan for managing and minimising the waste generated throughout the whole life cycle (McDonough *et al.*, 2003; Züst and Wimmer, 2004).

An increasing number of studies can be found in the literature that combine LCA and eco-design for wood-based products such as ventilated wooden walls (González-García *et al.*, 2012a), wooden modular playgrounds (González-García *et al.*, 2012b), furniture (González-García *et al.*, 2012c), wooden containers for the food sector (González-García *et al.*, 2011a), and kitchen cabinets and office tables (González-García *et al.*, 2011c). In all of these studies, problematic environmental impacts were identified and improvement strategies were proposed in the eco-design of new products with a low environmental profile. Thus, the implementation of eco-design in the development of wood-based products helps to introduce alternatives within the production process, which allows identification of improvements and reduction of the environmental impacts with fewer iteration cycles.

14.5 Integrated energy and material flows

Integrating material and energy flows within and between the forestry, construction, energy, industry and waste management sectors (Fig. 14.4) can bring energetic, economic and environmental advantages (Sathre and Gustavsson, 2009). The energy sector is central, as it provides heat, fuels



14.4 Schematic diagram of potential biomass flows during the life cycle of wood building material.

and electricity for the other sectors and for society in general (Truong and Gustavsson, 2013). It can benefit by using by-products of the forestry and wood products sector as fuel, as well as other biomass materials that would otherwise be considered a waste product. The wood products industry has the potential to be largely self-sufficient in primary energy terms, but can benefit by providing biofuels and heat to other sectors, and receiving, for example, liquid biofuels to power forest and transport equipment. The waste management sector, which traditionally has received and disposed of materials such as construction site and demolition waste, can be a source of valuable biomass fuel to the energy sector. Thus, the closer integration of these different sectors can significantly reduce the overall life cycle impacts of a built environment based on forest resources.

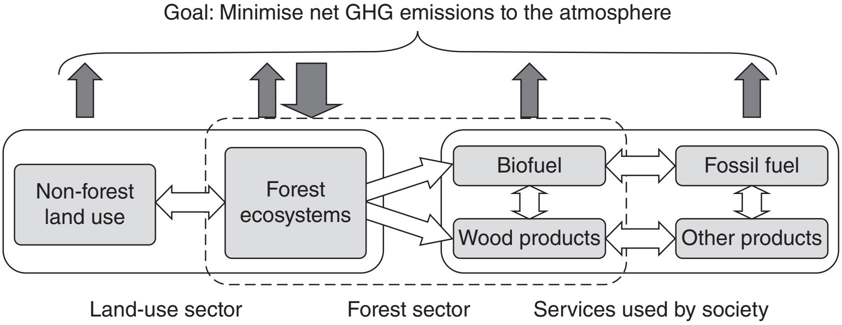
This integration of material and energy flows is already under way in many regions, and can be further optimised. The recovery and use of wood processing residues is now common in many areas, where previously such material was often disposed of as waste. The recovery of forest harvest residue for bioenergy is now done in some areas, although stumps and thinning residue are less commonly recovered (Eriksson *et al.*, 2007). Similarly, the recovery and use of wood-based construction and demolition residue takes place in some areas, but still goes unused in others. The final stage in the life cycle of a building is the demolition or disassembly of the building followed by the reuse, recycling or disposal of the materials. The percentage of demolition materials that is recoverable is variable, and depends on the practical limitations linked to the building design and whether material recovery is facilitated through deconstruction (Kibert, 2003). Systematic recovery of demolition wood is not yet practised in some areas, and demolition wood is instead landfilled. Consideration of the entire life cycle of a building material must include the fate of the material at the end of its service life.

Additional use of recovered wood material, such as reusing as lumber, reprocessing as particleboard, or pulping to form paper products, can improve the environmental performance of the material. Wood products are well suited for material cascading, which has been suggested as a strategy to increase the efficiency of resource use (Haberl and Geissler, 2000). Cascading is the sequential use of a resource for different purposes, as the resource quality degrades over time as it proceeds towards thermodynamic equilibrium. The cascade concept includes four dimensions of resource economy: resource quality, utilisation time, salvageability and consumption rate (Sirkin and ten Houten, 1994). In terms of these four characteristics, optimal utilisation of wood resources is achieved by: matching the resource quality to the task being performed, so as not to use a high-grade resource when a lower-grade one will suffice; increasing the total utility gained from a resource through prolonging the time during which it is used for various purposes; upgrading a resource through salvaging and reprocessing, where appropriate, for additional higher-grade uses; and balancing the usage rate of a resource with the capacity of forest land to regenerate lost resource quality.

A simple form of cascading is to burn a wood product at the end of its useful service life and recover the heat energy. Such a cascade chain has two links: material use and energy recovery. The efficient use of wood resources dictates that, at a minimum, the material is recovered and burned for energy recovery at the end of its useful life cycle. In some cases, particularly when forest resources are limited, it will be beneficial to employ a more complex cascade chain involving multiple material uses before final burning (Sathre and Gustavsson, 2006). In future, if more material and energy services are provided by biomass and fewer by fossil resources, wood cascading is likely to become more important by allowing more intensive use of limited biomass resources. The environmental performance of non-wood materials can also be affected by post-use management by, for example, recycling of metals and reuse of crushed concrete. Nevertheless, wood material has relatively more opportunity to improve its energetic and climatic performance through appropriate post-use management, due to its dual role of both material and fuel (Dodoo *et al.*, 2009).

14.6 Wood products and climate change

Managing forests so as to produce a yield of usable biomass, while simultaneously maintaining or increasing forest carbon stocks, is increasingly seen as a forest management strategy with large sustained mitigation benefits over the long term (IPCC, 2007) (see Fig. 14.5). The use of wood building materials instead of other materials contributes to climate change



14.5 Linkages between the forest sector and other sectors, with the overall goal of minimising net GHG emission to the atmosphere (adapted from IPCC, 2007).

mitigation through various mechanisms (Sathre and O'Connor, 2010a). A meta-analysis of 21 international studies of wood substitution found an average displacement factor of 2.1 tonnes of avoided carbon emissions per ton of carbon in wood products used in place of non-wood materials (Sathre and O'Connor, 2010b). The climate advantages of using wood products include: less fossil energy used to manufacture wood products compared with alternative materials; avoided industrial process carbon emissions such as in cement manufacturing; physical storage of carbon in forests and wood materials; use of wood by-products as biofuel to replace fossil fuels; and possible carbon sequestration in, and methane emissions from, wood products deposited in landfills. In this section we summarise the effects of each of these mechanisms.

14.6.1 Reduced fossil emissions from manufacturing

Manufacturing wood products typically requires less total energy, and in particular less fossil energy, than the manufacturing of most alternative materials. Cradle-to-gate analyses of material production, including the acquisition of raw materials (e.g., mining or forest management), transport, and processing into usable products, show that wood products need less production energy than a functionally equivalent amount of metals, concrete or bricks (Werner and Richter, 2007; Sathre and O'Connor, 2010a). Furthermore, much of the energy used in wood processing is thermal energy used for drying, for which wood processing residues are commonly used. Thus, the fossil carbon emission from wood product manufacturing is generally much lower than that of non-wood products. Composite wood products, while making more efficient use of roundwood raw materials, require a relatively higher use of fossil energy than do solid wood products. This energy, used for production of resins and additives as well as for the mechanical processing of wood fibres, is still commonly less than that needed for non-wood products. The development of green adhesives, described in Section 14.3.2, may reduce this fossil energy use.

14.6.2 Avoided industrial process emissions

Using wood products in place of cement-based products avoids the industrial process carbon emissions from cement manufacturing. CO_2 emissions are inherent to cement production, due to chemical reactions (calcination) during the transformation of raw materials into cement clinker. Avoided process emissions can be a significant part of the GHG benefits of wood products used in place of concrete and other cement-based materials (Gustavsson *et al.*, 2006b). While avoided calcination reaction emissions are well quantified, there is some uncertainty regarding the net life cycle effect of cement process emissions, due to CO_2 uptake by the carbonation reaction. Carbonation is a slow reaction that occurs over the life cycle of cement products, and involves reabsorption of part of the CO_2 that was initially emitted (Dodoo *et al.*, 2009). Nevertheless, as carbonation uptake is less than calcination emission, process emissions are avoided when substituting wood in place of cement products.

14.6.3 Carbon storage in wood products

Wood material is composed of about 50% carbon by dry weight, this carbon coming from the CO_2 removed from the atmosphere by the growing tree. In other words, wood products provide a physical storage of carbon that was previously in the atmosphere as a GHG (Lippke *et al.*, 2010). The climatic significance of carbon storage in wood products depends on the dynamics of the products pool as a whole, i.e., whether the total quantity of stored carbon is increasing, decreasing or stable. Atmospheric carbon concentration is affected by changes in the size of the wood product pool, rather than by the size of the pool itself (Gustavsson and Sathre, 2011). In the short to medium term, climate benefits can result from increasing the total carbon stock in wood products, by using more wood products or using longer-lived wood products. In the long term, as the stock of products stabilises at a higher level, wood products provide a stable pool of carbon as new wood entering the pool is balanced by old wood leaving the pool. Consideration of the long-term carbon dynamics of wood products shows that the substitution effect of avoiding fossil emissions is ultimately more significant than the carbon stored in wood products (Eriksson *et al.*, 2007; Poudel *et al.*, 2012).

14.6.4 Carbon storage in forest ecosystems

Over a complete rotation period of sustainable yield forestry, the carbon content in tree biomass remains unchanged, by definition (Lippke *et al.*, 2011). Forest soils often store more carbon than forest biomass, and soil carbon stock in managed forests generally maintains a dynamic equilibrium level over multiple rotations. Wood production in managed forests must be distinguished from the carbon balance effects of harvesting primary forests; conversion of primary (old-growth) forests to secondary, managed forests results in a loss of stored carbon from both biomass and soils, before the forest carbon stocks again reach dynamic equilibrium. The level of the new equilibrium depends on soil characteristics, forest management intensity and other factors. Afforestation, or the creation of forests on previously non-forested land, generally increases the carbon stock in biomass and soil as well as producing wood for product substitution.

14.6.5 Biofuel substitution and avoided fossil emissions

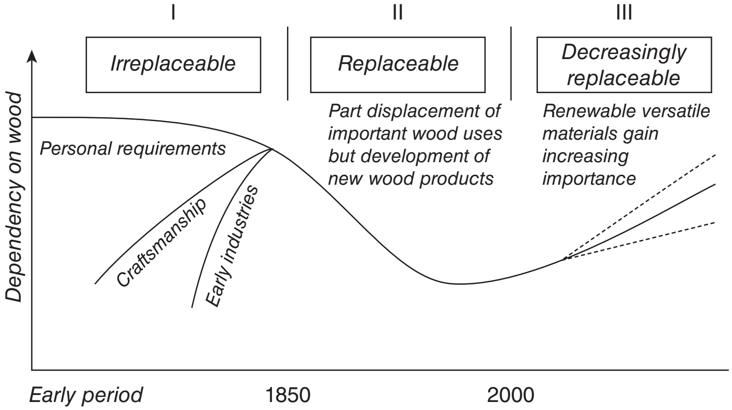
The wood contained in a finished forest product is only a part of the total biomass flow associated with the product. Substantial biomass residues are generated during forest thinning and harvest operations, during primary and secondary wood processing, and at the end of the service life of a wood product. These by-products can be used as biofuel to replace fossil fuels, thus avoiding fossil carbon emissions. The quantification of GHG benefits due to the use of residues from the wood product value chain is not straightforward; issues include the allocation of benefits to the different biomass fractions, varying carbon intensity of the fossil fuel replaced, leakage (i.e., a unit of additional biofuel does not necessarily lead to a unit reduction of fossil fuel use), potential soil carbon stock change due to removal of harvesting residues, and uncertainties about how post-use wood products will be handled by future waste management systems (Gustavsson and Sathre, 2011). Nevertheless, the recovery and combustion of the biomass by-products associated with wood products appears to be the single most significant contributor to the life cycle GHG benefits of wood product use (Sathre and O'Connor, 2010b).

14.6.6 Carbon dynamics of landfilled wood

Some wood products are currently deposited in landfills at the end of their service life. Carbon dynamics in landfills are recognised to be quite variable, and can have a significant impact on the life cycle GHG balance of the wood product (Micales and Skog, 1997). A fraction of the carbon content in landfilled wood will likely remain in (semi)permanent storage, providing climate benefits. Another fraction may decompose into methane, which has much higher global warming potential (GWP) than CO_2. However, methane gas from landfills can be partially recovered and used as a biofuel to replace fossil fuels. Thus, the landfilling option for post-use wood products carries great uncertainties, and could result in climate benefits (partial sequestration in landfills, and partial production of methane biofuel) or climate impact (emission of methane to the atmosphere). Landfilling of wood forgoes the assured opportunity of complete energy recovery from post-use wood products, and is not recommended from a resource efficiency perspective.

14.7 Wood building materials: past and future

Wood has long been a primary source of material and energy for human society (Perlin, 1989). Until recent centuries, wood was irreplaceable as the most important raw material for construction, agriculture, crafts,



14.6 Relative importance of wood material in past, present, and future (source: Sathre and Gustavsson, 2009; from Schulz, 1993).

shipbuilding, etc. More recently, however, many previous uses of wood have been replaced by materials such as concrete, metals and plastics, and by non-renewable fossil fuels such as coal, oil and natural gas. Schulz (1993) suggested that this substitution of wood by other materials and energy sources may be reversed and a new phase of increased wood use may begin due to environmental concerns and eventual supply constraints of non-renewable raw materials and fuels (Fig. 14.6).

The level of current wood use in building construction varies significantly between countries. The use of wood for constructing single-family houses is rather low in Europe, except in the Nordic countries (Gustavsson *et al.*, 2006a). Wood is commonly used in Nordic countries for single-family houses, but is less common in multi-storey apartment buildings. In contrast, wood is commonly used in North America for construction of both single-family and multi-family houses. Wood use practices in some parts of Europe are still affected by historical path dependencies. In response to large city fires during the late 19th century, several European countries introduced regulations prohibiting the use of wood frames in multi-storey buildings.

This was reversed in 1989 by a directive from the European Commission (Council Directive 89/106/EEC), which was later replaced by Regulation (EU) No. 305/2011. These regulations effectively state that any material that fulfils the functional requirements can be used for construction of multi-storey buildings. However, over two decades after the change in policy, the use of wood frames in the construction of multi-storey buildings in Europe is still low. The slow re-emergence of wood construction in Europe is largely due to the path dependency of the established non-wood construction system (Mahapatra and Gustavsson, 2008). This system consists of an inter-linked set of technologies, actors and institutions following a specific path implicitly supported by institutional, economic and social factors.

Several measures could help to overcome these hindrances and promote wood construction, including investments in knowledge creation, incentives for entry of new firms, and the promotion of collaboration between different sectors (e.g., construction and forestry) (Mahapatra *et al.*, 2012). Economic instruments to internalise the external costs of producing building materials, e.g. the social costs of GHG emissions, would improve the economic competitiveness of wood building material (Sathre and Gustavsson, 2007). The development of effective and environmentally compatible wood adhesives and preservatives would expand the life cycle opportunities for reuse and recycling of forest biomass. Increased use of wood-based building materials would be fully compatible with a broader integration of sustainable biomass resources into the material economy, as only about 20–25% of the potential harvest of forest biomass is actually built into the construction. The remaining biomass (e.g., forest, processing and construction residues) could be used for other purposes, providing economic and environmental synergies between sectors. Biorefineries may be developed to differentially extract and process the components of woody biomass such as cellulose, hemicellulose, lignin, and extractives to co-produce a range of products (Amidon *et al.*, 2008). High system-wide efficiencies can be gained through co-production at varying scales of woody biomass feedstock into, for example, district heat, electricity, and liquid and solid biofuels (Truong and Gustavsson, 2013). Such integrated biomass-based material and energy systems may contribute to fulfilling multiple societal needs, by efficiently using natural resources in a sustainable manner.

The use of wood-based building materials can contribute to a sustainable built environment based on resource-efficient systems with low environmental impact. Life cycle and system perspectives of the built environment are needed, so that all the life cycle phases – production, operation, maintenance and end-of-life – are considered and optimised as a whole, including the energy and material chains from natural resources to final services. Wood building products from sustainably managed forests are a renewable resource that can provide multiple benefits during their life cycle. In addition to their structural and architectural use within a building, the life cycle wood product chain produces significant quantities of biomass co-products that can be used as a sustainable bioenergy source to replace fossil fuels. The use of forest resources in the built environment can play an important role in a long-term strategy for sustainable development and climate change mitigation.

14.8 Sources of further information

14.8.1 Institutions and agencies

- US Forest Product Laboratory, United States. http://www.fpl.fs.fed.us/
- FPInnovations Wood Products Research Institute, Canada. http://www.fpinnovations.ca/
- SP Trä (SP Wood Technology), Sweden. http://www.sp.se/en/units/wood/
- Treteknisk (Norwegian Institute of Wood Technology), Norway. http://www.treteknisk.com/
- European Forest Institute. http://www.efi.int/
- FAO Forestry, Food and Agricultural Organisation of the United Nations. http://www.fao.org/forestry/en/
- Skogforsk (Forestry Research Institute of Sweden). http://www.skogforsk.se

14.8.2 Scientific journals

Forest Products Journal. http://www.forestprod.org/buy_publications/forest_products_journal.php

European Journal of Forest Research. http://link.springer.com/journal/10342

European Journal of Wood and Wood Products. http://link.springer.com/journal/107

Scandinavian Journal of Forest Research. http://www.tandfonline.com/loi/sfor20

Canadian Journal of Forest Research. http://www.nrcresearchpress.com/journal/cjfr

Forest Ecology and Management. http://www.journals.elsevier.com/forest-ecology-and-management/

International Journal of Life Cycle Assessment. http://link.springer.com/journal/11367

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15

The environmental impact of adhesives

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Abstract: Growth in adhesive use has contributed to environmental problems, but impact can be reduced by improving procedures and reducing use of solvents. Adhesives from established renewable resources (starch, gelatine and natural rubber) are being augmented through fermentation techniques or genetic modification. The impact of recycling must be considered from 'cradle to grave'. Some renewable resources or recycling techniques may have a greater impact than the technologies they replace. The Laws of Thermodynamics suggest that technical solutions will never be adequate if we aspire to indefinite growth within a finite global system.

Key words: adhesives, pollution and resources and limits to growth, polysaccharides, polyhydroxyalkonates, life cycle analysis.

15.1 Introduction: growth in the usage of adhesives

Adhesives have played a useful part in human culture at least since upper palaeolithic times, where they helped stick flint points to spear shafts and pigments to cave walls (Pascoe, 2005). Now the global market for adhesives (along with sealants) is said to be valued at $42 bn (Reade, 2012). An important reason why adhesive use is so widespread and growing in industry in general, and in the construction industry in particular, is because adhesives provide attractive solutions to many design engineering problems. Advantages of adhesives over other joining technologies include better stress distribution over the joint area; the ability to join dissimilar materials; the ability to join thin materials; ease of manufacture, avoiding machining operations for keyways, circlips, etc.; reduction in weight and often an improved aesthetic appearance (Watson, 2005; Institution of Structural Engineers, 1999, p. 10; Hutchinson and Hurley, 2001, p. 25).

Quite apart from traditional 'joining technology', an enormous volume of adhesives is used in composites where they essentially stick together the fibres or particles to comprise the composite materials which have transformed so many areas of engineering in the last 50 years. Adhesion technology, then, is deeply involved in contemporary industrial culture. Deeply embedded in this culture is a belief in the desirability, indeed of the necessity, of continual growth. This idea is traced back to Adam Smith (1776), the Scottish Enlightenment philosopher who argued that an expanding economy would promote the well-being of all. Nowadays, growth is often presented in quasi-religious language as necessary, not just for material prosperity, but for moral well-being – for 'political democracy, individual liberty, and social tolerance' (Delong, 2006).

Thus it seems that the aspiration of every industrial corporation is for unlimited growth, involving the production of more goods, the exploitation of more raw materials and the penetration of more markets. For example, Reade is enthusiastic about the 5.2% p.a. growth rate for adhesives in the 'dynamic' group of emerging nations, while he describes as 'modest' the 2.2% p.a. growth in the mature markets of North America (Reade, 2012). One of the aims of the European trade body, Fédération Européenne des Industries de Colles et Adhésifs, is to 'contribute to the growth of the Industry' (FEICA, 2012). The trade press lavished praise on the manufacturer of self-adhesive PVC for receipt of the 'coveted' Prix de l'Ambition in recognition of annual growth rate of over 35% over a 7-year period (Pressbox, 2007).

It is sometimes forgotten that a constant rate of growth is, in mathematical terms, exponential growth:

$$P = P_0 \exp(kt) \tag{15.1}$$

where P is the size of the economy or resource at time t and P_0 is its value at $t = 0$; k is the growth rate. It is clear that such growth cannot be sustained in a world of finite resources. It is not surprising, then, that in recent years the commitment to limitless growth has sat uneasily next to the increasing realisation that the natural environment is rapidly approaching a state of crisis.

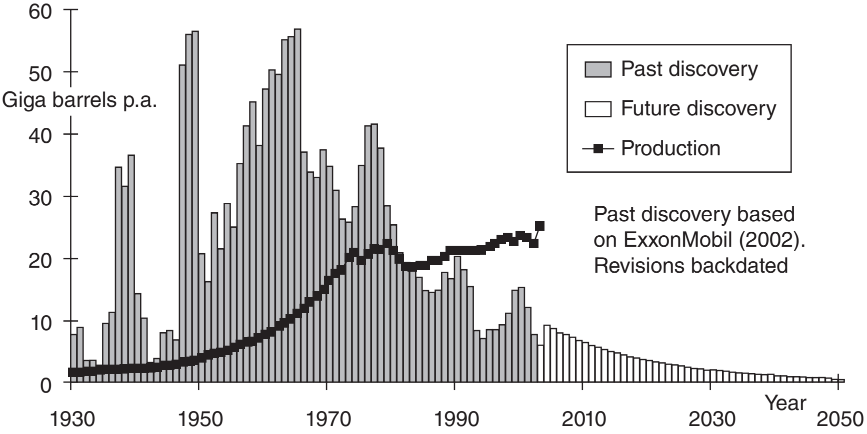
15.2 Environmental implications of the growth in adhesive use

The broad nature of the environmental problem is well known, and now widely acknowledged by the scientific community, although still contested by some popular commentators and vested interests. It is associated with the ability of the environment to provide the *resources* needed and its ability to act as a sink for waste materials, *pollutants*. Many natural resources on which our civilisation depends are not renewable, and some are being used up at a rate which is significant, when compared with the total reserves. Petroleum is perhaps the most obvious and important example of a finite resource which is being depleted rapidly (Hatfield, 1997; Leggett, 2005). Almost all materials, even renewable ones like natural rubber, require an energy input during production and processing (see Table 15.1). Because energy generation is so dependent on petroleum (and on other finite fossil fuel resources), the energy crisis is one aspect of the crisis in material

Table 15.1 Approximate energy content of some polymers (GJ/tonne)

Polymer	Energy content
Polypropylene	110
Styrene butadiene rubber	130
Polyurethane	174
Polychloroprene	120
Natural rubber	30

After Rahaman, 1994.

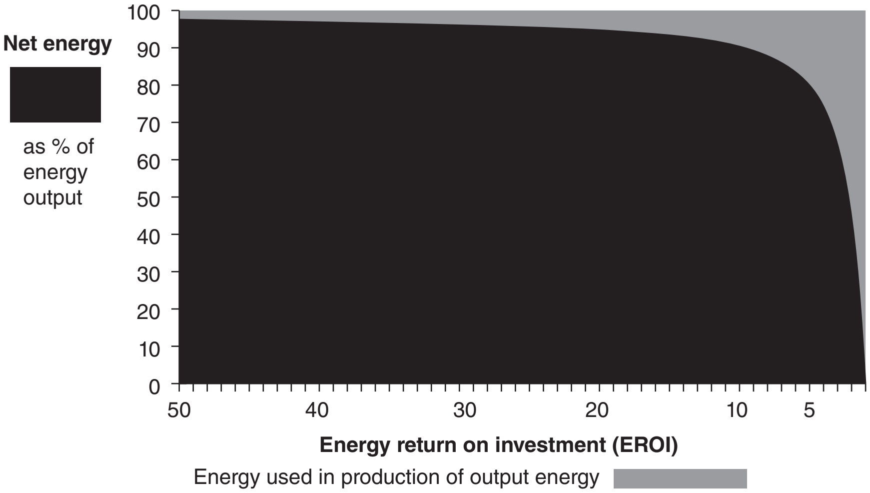


15.1 Rate of discovery and consumption of oil.

resources. Moreover, the adhesives of contemporary technology overwhelmingly come from fossil fuel sources, especially from petroleum.

Uncontested figures for petroleum reserves are not available, as commercial and political interests make them matters of intense controversy (Murray and King, 2012). However, it is clear that they are finite and being rapidly depleted, even if the precise time-scale is a matter of controversy. Figure 15.1 contrasts the rate of discovery of oil resources, which appears to have peaked, with the rate of consumption, which is still rising (Hall and Day, 2009).

It is salutary to bear in mind the enormous differences in the energy return on investment (EROI) associated with various energy sources (Hall and Day, 2009; Murphy and Hall, 2010; Gupta and Hall, 2011). EROI is the energy output from a resource (E_{out}) divided by the energy used (E_{in}) to get that energy. As 'easy' energy sources become depleted, the EROI on replacement sources tends to be much lower.



15.2 The effect of energy return on investment (EROI) on net energy (Eq. [15.4]).

Of obvious importance is the *net energy*, that is the difference between the energy output and that input to obtain it:

$$\text{Net energy} = E_{\text{out}} - E_{\text{in}} \qquad [15.2]$$

Now, because

$$\text{EROI} = E_{\text{out}}/E_{\text{in}} \qquad [15.3]$$

it follows that

$$\text{Net energy} = E_{\text{out}}\{(\text{EROI} - 1)/\text{EROI}\} \qquad [15.4]$$

This clearly shows that the important ratio of net energy to energy output (E_{out}) is a strongly non-linear function of EROI. The relationship, plotted in Fig. 15.2, shows an 'energy cliff'. For a range of high values of EROI, the net energy is quite a high proportion of E_{out}. For an EROI of 10, net energy is 90% of the energy output, but as the EROI declines further, the net energy percentage falls off rapidly (Murphy and Hall, 2010).

Reviewing the literature, Murphy and Hall (2010) found that for drilling for oil and gas in the United States, the EROI had declined from about 100:1 in the 1930s, to 30:1 in the 1970s, and to about 11:1 in 2000. They comment 'From this literature we believe that the EROI of our most important fuels is declining over time'. The EROI values quoted for 'new' energy sources tend to be low, very, very low compared with petroleum in its early days (Table 15.2). Unconventional sources, such as tar sands and shale gas from hydraulic fracturing ('fracking'), are likely to be expensive once 'the

Table 15.2 Approximate EROI of various energy resources for the United States

Resource	EROI
Tar sands	2–4
Shale oil	5
Wind turbines	18
Photovoltaic	6.8
Ethanol (sugarcane)	0.8–10
Ethanol (corn)	0.8–1.6
Biodiesel	1.3

After Murphy and Hall, 2010.

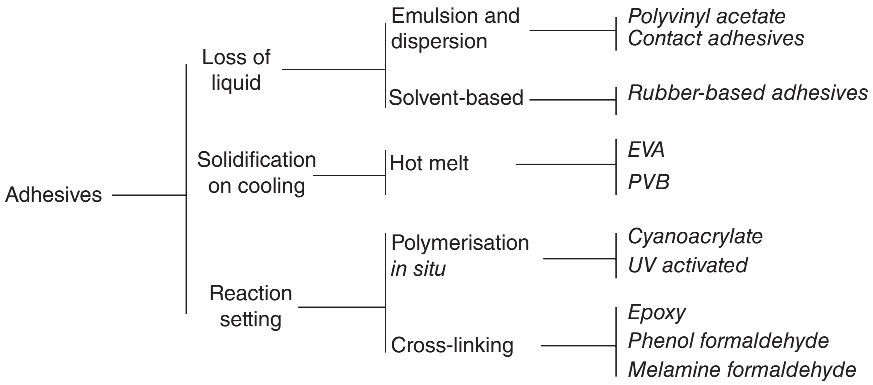
"cream" is skimmed from the sweet spots' (Guilford *et al.*, 2011). It is apparent that whatever energy sources replace oil, the cheap energy which has underpinned our growth, and has been the basis of most of our industry for the best part of a century, will not last for long.

Many consider that the Earth's capacity to carry *pollutants* is even more pressing a problem than those of depletion of material and energy sources. The most widely discussed pollution problem is the emission of greenhouse gases and consequent climate change. This has the potential not just to cause monotonic (i.e., 'linear' as contrasted with 'non-linear') increases in global temperature with extreme meteorological consequences and rises in sea levels, but non-linear chaotic changes which, by their nature, are unpredictable and which my be catastrophic (IPCC, 2001). Both the production and use of polymers and adhesives are associated with pollution.

Whether all this constitutes an environmental 'crisis' is ultimately a matter of terminology. Suffice it to say that serious scientists have considered the problem sufficiently urgent to speak of the human population being reduced to 'a few breeding pairs' (Lovelock, 2006) and of the twenty-first century as humanity's 'final century' (Rees, 2003).

15.3 Adhesives, adhesion and the environment

Adhesives comprise a diverse range of materials, the majority of which are organic polymers: indeed the majority of polymer types find applications either as adhesives as such, or as components of adhesive systems. Adhesives are applied to the surfaces to be bonded in a more or less liquid form, and then solidify. Figure 15.3 shows a simplified classification, and gives a few examples of adhesives used in the construction industry (Packham, 2005, passim; Institution of Structural Engineers, 1999, p. 13).



15.3 Classification of adhesives by setting mechanism (after Cope, 2005).

For the successful use of adhesives, it is almost always necessary to pretreat the surfaces to be bonded. This may remove cohesively weak surface layers and aid wetting of the surface by the adhesive (Institution of Structural Engineers, 1999; Hutchinson and Hurley, 2001; Packham, 2005; Hartung and Boehm, 2011). Both the adhesives themselves and the necessary pretreatments often have a significant environmental impact.

There are increasing restrictions on the release of volatile organic compounds (VOCs) to the atmosphere, as they contribute to the formation of photochemical smog, and many are implicated in the aggravation of lung diseases such as asthma. Of course, almost all organic vapours absorb infrared radiation and therefore act as greenhouse gases. A glance at Fig. 15.3 shows the enormous potential for release of such vapours in the manufacture and use of adhesives. Indeed adhesives and coatings are a major industrial source of VOCs (Metzger and Eissen, 2004).

For all these reasons, there are moves away from using organic solvents and dispersing media both in pretreatments and in adhesives and coatings such as paints. Chlorinated solvents have a long history as agents for degreasing metals. Since the recognition of their role in depletion of the ozone layer, much effort has been devoted to developing less damaging procedures, such as alkaline solutions and organic aqueous emulsions (Watts, 2005). Many adhesive systems formerly based on organic solvents are now produced as aqueous emulsions. Adhesives for rubber to metal bonding provide a good example. Similarly there has often been substitution of hot melt or of radiation-cured adhesives for solvent systems (Packham, 2005, passim). Despite all these developments, volatile emissions associated with adhesive use remain high (Metzger and Eissen, 2004).

Another focus of concern has been on the effects of very low concentrations in the environment of a range of organic compounds. Many have

Table 15.3 Some compounds of relevance to adhesive technology with reported endocrine-disrupting properties

Compounds	Comment
Polychlorinated compounds	Formed in some incineration processes
Alkylphenols	Stabilisers
Alkylphenol ethoxylates	Surfactants
Phthalates	Plasticisers
Bi-phenols (e.g., bisphenol-A)	Constituent of most epoxy resins

Source: Environmental Agency, 1998.

been implicated in producing physiological change in humans and other animals (Environment Agency, 1998; Colborn *et al*., 1996). For example, some mimic the action of hormones. There is evidence that residues of some compounds are xeno-oestrogenic and may disrupt the endocrine system (Table 15.3), depressing the human sperm count and sometimes leading to male genital deformity. Many such compounds are used as additives in adhesives, composites and coatings: because of their fundamental incompatibility with the polymer phase, they are likely to be released into the environment. This is an area of controversy, with much dispute over the effects on humans of low concentrations of such chemicals in the environment, as opposed to their effects on test animals under laboratory conditions. At one time it was expected that the European Union 'REACH' (Registration, Evaluation, Authorisation and Restriction of Chemicals) regulations (European Commission, 2006, 2012) would oblige industry to replace endocrine disruptors with safer alternatives. However, vigorous lobbying by the Chemical Industries Association (Hackitt, 2005), the European Chemicals Agency and similar organisations have succeeded in diluting the impact of REACH, and use of endocrine disruptors is still widespread (Sandra and Cobbing, 2007; The International Chemical Secretariat, 2011, 2012).

This negative environmental impact of adhesive use must be qualified by recognition of the environmental advantages brought by their increasing use. Adhesives and composites have often displaced more traditional materials resulting in more elegant, more efficient engineering solutions to design problems. While the driving force in the increased use of adhesives and composites may not have been environmental, the effect has often been to achieve a 'greener' solution. As a symbolic example, consider how the technology of carbon fibre composites, aromatic polyamides and other advanced materials enabled the Nuna series of experimental solar-powered cars, built by students at Delft University of Technology, to travel 3,000 km across Australia at average speeds of over 90 km/h (Fig. 15.4) (Nuna, 2007).

15.4 Use of adhesives and composites to reduce environmental impact: Nuna-4 solar-powered car (Photograph: Hans-Peter von Velthoven).

Thus the use of polymers and composites has often resulted in a saving in energy and materials and a reduction in toxic emissions associated with the high temperatures and complex chemistry of many metallurgical and ceramic technologies. However, environmental problems of resources and pollution remain. Let us examine to what extent these may be addressed, within the context of adhesives and composites.

15.4 Reduction of environmental impact

15.4.1 Green design principles

Environmental impact can be reduced by adopting some fairly obvious design criteria in the practice of chemistry and engineering. These are sometimes called 'green design principles'. Thus use of hazardous materials and procedures should be minimised, waste should be avoided and unavoidable pollutants controlled. Renewable resources should be preferred to depleting ones, and the dismantling and recycling at the end of life should be considered at the design stage.

To some extent these 'green design principles' are little more than platitudes, but it is of value to make them explicit. Indeed the American Chemical Society's Green Chemistry Institute has formalised them in 'Twelve Principles of Green Chemistry' and 'Twelve Principles of Green Engineering' (American Chemical Society, 1998; 2003). These general principles apply, of course, specifically to polymer chemistry and polymer engineering. We will now consider in particular the scope for reducing environmental impact by use of adhesives and composites from renewable resources and by recycling waste materials.

15.4.2 Polymers from renewable resources

Although the majority of polymers in use come from fossil fuel feedstocks, adhesives and polymers obtained from renewable animal and vegetable sources have long been in use (Finch, 2005; Henshilwood *et al.*, 2011; Suárez, 2011). Starch and gelatine-based glues and cellulosic adhesives are examples. Natural rubber is an important commercial material and adhesive and a rubber plantation is almost as effective as virgin forest in carbon dioxide absorption (Rahaman, 1994).

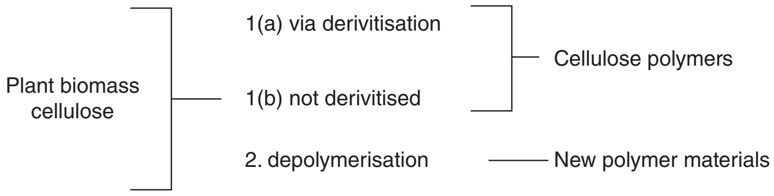
Many fascinating and impressive examples of adhesion occur in the natural world. Some molluscs, such as mussels, maintain high adhesion in the sort of hostile marine environment which constitutes a nightmare for most human adhesive technologists (Sagert *et al*., 2006; Lee *et al*., 2006; Dalsin *et al*., 2003). The gecko's ability to run up walls and across ceilings manifests the capability of strong, reversible adhesion to a wide range of surfaces. Study of such examples in recent years has led to the development of biomimetic adhesives (Suárez Bermejo, 2008; Creton and Gorb, 2007; Smith and Callow, 2006; Packham, 2010).

For example, the foot of the gecko consists of a range of fine structures which are revealed at higher and higher magnifications. The toes are covered with nanoarrays of hair-like setae (bristles) formed from stiff β-keratin. These further divide into hundreds of spatulae with nanoscale diameters. Each spatula ends in a leaf-like plate or pad which makes contact with the surface. This ultra-fine structural division is considered to be essentially linked to the gecko's adhesive ability (Autumn, 2006; Autumn and Gravish, 2008) and has inspired the development of novel adhesive systems.

For example, Lee *et al*. (2007) produced an adhesive consisting of an array of nanofabricated polymer pillars (cf. the gecko), coated with a thin layer of a synthetic polymer that was inspired by the adhesive proteins secreted by the mussel foot. This combination is reported to maintain 'its adhesive performance for over a thousand contact cycles in both dry and wet environments'.

The pressures to move towards sustainability have stimulated much interest in developing new polymers from plant sources, often using fermentation techniques or genetic modification (Perepelkin, 2005; Clark and Macquarrie, 2009; Meier, 2011). One approach is to modify biopolymers as the basis for routes to new polymer materials. Starch, cellulose and chitin are widely available renewable resources and have been used in this way. Alternatively 'biorefineries' may be used for converting biological raw materials to industrially valuable intermediates and final products (Kamm and Kamm, 2004). Clark describes ways in which everyday chemical products, currently obtained from petroleum, could be derived from renewable biomass by fermentation or controlled pyrolysis (Clark, 2007). *Acacia mangium* is a fast-growing and prominent plantation tree in Malaysia. Hoong *et al*. (2011) have shown that tannin extracted from its bark can be processed to form a natural adhesive capable of replacing synthetic phenol formaldehyde in the manufacture of plywood.

Rose and Palkovits (2011) have recently reviewed the uses and potential for cellulose-based polymers. Whereas crops grown as a source for oil and



15.5 Routes from cellulose biomass to polymer materials (After Rose and Palkovits, 2011).

sugar are in direct competition with food crops, in the authors' view lignocellulose biomass, produced by algae and fungi as well as by plants, is not. The cellulose component of such 'plant biomass' may either be processed to produce renewable polymer materials directly, or it may be depolymerised to yield glucose which, in turn, is used as a renewable feedstock (see Fig. 15.5).

For many years materials have been produced from the cellulose polymer by derivitisation. The long-established route used carbon disulphide in the production of rayon and cellophane. The use of hazardous carbon disulphide is avoided in more recently developed technologies, for example by employing urea to convert the cellulose to the carbamate. Cellulose from plant biomass may also be regenerated without derivitisation, for example by the well-known and environmentally unattractive use of Schweizer's reagent – ammoniacal copper hydroxide. Efforts in recent years to develop better direct routes to cellulose polymer materials have yet to achieve commercial viability.

An alternative approach, which has considerable potential, is the 'biorefinery' route. This uses cellulosic biomass as a route to small molecules which, themselves, are capable of further processing to produce useful products. Thermolysis yields synthesis gas (carbon monoxide and hydrogen), which, via Fischer–Tropsch synthesis, can produce a range of organic compounds. Depolymerised by hydrolysis gives glucose whence may be obtained useful organic chemicals including those important in polymer and adhesive technology. Some are shown in Fig. 15.6. Ethanol is a route to ethylene and to polymers derived from that. Lactic acid can be processed to yield acrylates as well as polyesters, polycarbonate and polyurethanes. Rose and Palkovits (2011) discuss such transformations at length.

Lactic acid, $CH_3CH(OH)COOH$, may also be polymerised to give polylactic acid, also known as polylactide. This is a polyester which may be produced as a fibre, with properties comparable with those of polyethylene terephthalate (Perepelkin, 2005) (Table 15.4). The two fibres are compared in Table 15.5 which also shows the carbon dioxide emitted and energy required in their production (Perepelkin, 2005). It is important to recognise that energy is required, and that greenhouse gases are often emitted, in the production of biopolymers, as shown here for polylactide.

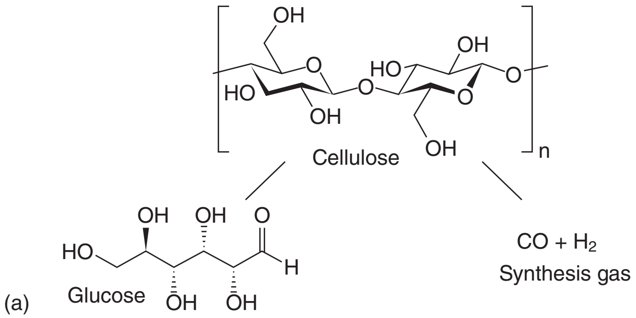
Polylactide is an example of a polyhydroxyalkanoate (PHA), many of which are produced within the cells of many different prokaryotic microorganisms. PHAs find application in hot melt adhesives (Whitehouse, 2004; Verespej, 2011), and are potential alternatives to polyesters from petroleum sources (Braunegg *et al.*, 2004). Depending on the length and structure of the alkyl chain, they have properties ranging from those of crystalline thermoplastics to elastomers. Thus polyhydroxybutyrate (PHB) is a stiff, highly crystalline material, but polyhydroxyoctanoate (PHO) is a rubbery elastomer (Table 15.6). There are other PHAs which are completely amorphous, tacky substances.

15.4.3 Recycling

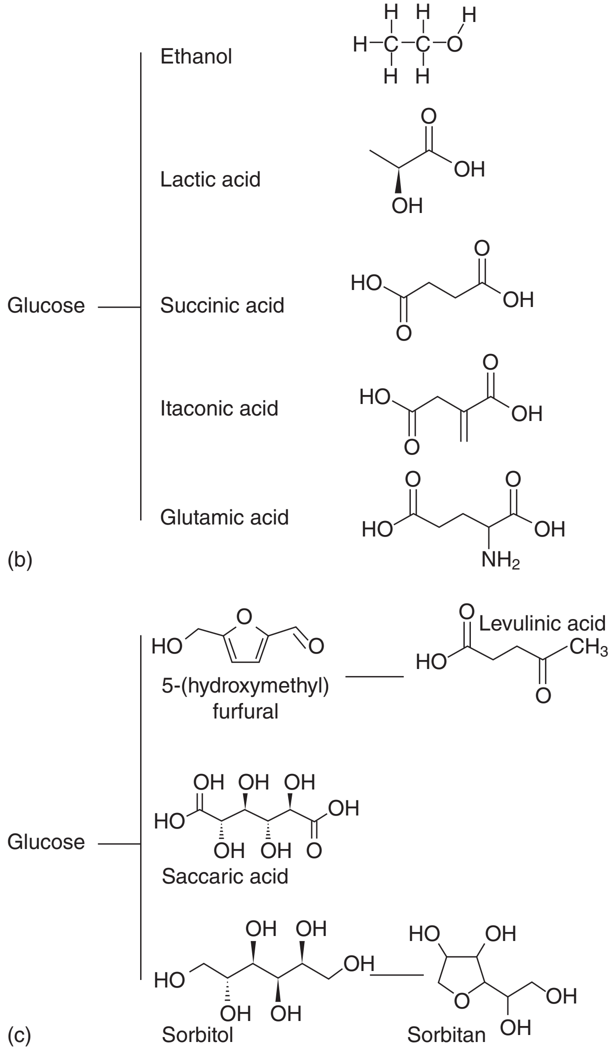
Reuse is an important means of conserving material resources. Recycling of polymers, certainly of linear polymers, is in principle possible (Hoyle and Karsa, 1997; Scheirs, 1998). In practice, successive grinding and remoulding may cause thermal degradation to an extent that the properties suffer.

The recycling of adhesives

The recycling of adhesives is even more difficult than of polymers in general, as the adhesive is usually a small proportion of a larger artefact all of which should, in principle, be recycled. Of course the use of an adhesive to join different materials can add considerable difficulty to their separation prior to recycling (Onusseit, 2006; van Halteren, 2011), especially with reaction-setting adhesives (see Fig. 15.3). Attention has been directed to developing adhesives which serve their purpose during the life of an article, but may



15.6 Chemicals from cellulose with potential for future polymer production: (a) glucose and synthesis gas; (b) biocatalytic transformation and (c) chemocatalytic transformation of glucose (adapted from Rose and Palkovits, 2011).



15.6 Continued

facilitate disassembly when component recycling is required. These have been reviewed recently by Sato (2011).

These adhesives, sometimes inelegantly called 'dismantlable adhesives', are designed to break down at elevated temperature, making the dismantling of the bonded materials easy. Some of these adhesives have blowing agents, such as encapsulated hydrocarbons, incorporated (Nishiyama *et al.*, 2003; Ishikawa *et al.*, 2005; Jonsson *et al.*, 2010); others are compounded with oxidising agents, such as peroxides, which decompose causing chemical degradation at the target temperature (Sato *et al.*, 2010; Sugimoto and Yamamoto, 2012).

Table 15.4 Comparison of polylactide and melt-spun polylactide fibres with polypropylene (PP), poly(ethyleneterephthalate) and nylon-6

	Polylactide	PP	PET	Nylon-6
Polymer				
Density (g/cm^3)	1.27	0.92	1.38	1.14
Melting point (°C)	175–180	175	265	214
Glass transition temp. (°C)	55–60	−12 to −20	90–100	40–45
Heat value (kJ/g)	19	40	23	31
Fibres				
Modulus (GPa)	4–6	2–3.5	4–8	2–4
Strength (cN/tex)	40–55	35–50	35–50	35–50
Elongation at break (%)	30–40	40–60	30–50	40–50
Elastic restoration (%) @ 10% initial deformation	64	98–100	60–65	95–98

Table 15.5 Comparison of polylactide and poly(ethyleneterephthalate) fibres

	Polylactide fibre	PET fibre
Raw material	Renewable: plant source containing hexosans	Non-renewable: petroleum
Monomer production	Biotech methods: exothermic; heat removed	Chemical engineering methods: endothermic; heat supplied
Polycondensation	In melt 200–220°C	In melt 280–300°C
Fibre melt spinning at	210–220°C	280–300°C
Energy consumption[a] (mJ/kg)	92	143
CO_2 emitted[a] (kg/kg)	4.1–6.5	8.9–12.2

[a] Entire cycle: including production of monomer, polymer and fibre.

Increasingly motor manufacturers are expected to design their vehicles so that their materials can be recovered and reused at the end of the car's life. Hutchinson *et al.* incorporated (unspecified) foaming agents into adhesives for bonding aluminium sheets and showed that by thermal activation of the foaming agent, the adhesive strength fell, which would enable bonded components in an automotive application to be separated easily (Hutchinson *et al.*, 2010).

The recycling of composite materials

The recycling of composite materials is a major challenge. Where the matrix is a linear (thermoplastic) polymer, separation of the polymer from the fibre

Table 15.6 Polyhydroxybutyrate (PHA) and polyhydroxyoctanoate (PHO) compared with polypropylene (PP)

	PHB	PHO	PP
Melting point (°C)	180	61	176
Glass transition temp. (°C)	5	–35	–10
Molecular weight (kDa)	500–2000	130	200
Density (g/cm^3)	1.18–1.25	1.02	0.905
Crystallinity (%)	70	25	70
Oxygen permeability (cm^3 m^{-2} atom^{-1} day^{-1})	45	–	1700
Water vapour transmission (g m^{-2} day^{-1})	60–70	–	–
Young's modulus (MPa)	3500	8	1700
Tensile strength (MPa)	40	9	38
Extension to break (%)	5	380	400

Source: Perepelkin, 2005; Metabolix, n.d.

or filler is in principle possible by use of a solvent, but the problems of degradation, mentioned above for a single phase polymer material will be made worse. Commonly, however, the matrix, such as a polyester or epoxy, is crosslinked, and straightforward dissolution is not a possibility. These materials are commonly regarded as not recyclable, and this is becoming a significant disadvantage as environmental regulation tightens (Correia *et al*., 2011; Oliveux *et al*., 2012).

Oliveux *et al*. discuss three methods by which thermoset composites might eventually be recycled on an industrial scale.

1. *Thermolysis* (incineration) of the resin in principle could be used as an energy source, and leaves the mineral fibres or filler, but in practice their properties are likely to have deteriorated in the process. Toxic emissions are also a problem (Correia *et al*., 2011).
2. *Chemical recycling* involves treatment of the composite with chemicals which will attack the polymer phase. Oliveux *et al*. themselves report the application of this method to glass-reinforced polyester. They used subcritical water (at temperatures between 200 and 374°C and pressures up to 22 MPa (220 bar)) to hydrolyse the polyester, and recovered both organic and inorganic phases. The ester monomer was mixed with secondary reaction products, and the glass fibres were significantly weaker than virgin material.
3. *Material recycling* involves grinding the composite and using what is produced as a filler or partial reinforcement in another product.

Correia *et al*. (2011) describe the application of such material recycling to glass-fibre reinforced plastic (GFRP) waste from the construction industry. The ground product was used to replace up to 20% of the sand in the production of concrete. Both the mechanical properties and durability of concrete were poor with high proportions of GFRP, but (the authors suggested) the concrete might find application in non-structural applications such as pavement slabs.

Mixed plastic wastes

A further problem in polymer recycling is that feedstock may consist of mixtures of different grades and compounds of the same polymer. Even worse is where the feed contains different polymer types, perhaps resulting from two polymers bonded by an adhesive. This generally leads to poor properties because of polymer–polymer incompatibility.

Several technologies are possible for the disposal of mixed plastic waste. These include mechanical recycling, feedstock recycling, incineration with energy recovery and burying, i.e. using landfill. Mechanical recycling involves separation into different types of polymer, grinding and remoulding into new artefacts. In feedstock recycling, the plastic waste is transformed by heat of chemical reaction to hydrocarbons which are then used as chemical feedstock.

Lazarevic *et al*. (2010) have recently published a review of options for the management of mixed plastic waste. They found more than 50 life cycle assessment (LCA) studies, and selected ten 'high quality' studies for their review. They compared impacts on global warming, acidification, eutrophication and energy use. Their broad conclusion was that where 'single polymer plastic waste fractions with little organic contamination are recycled and replace virgin plastic at a ratio close to 1:1, recycling is generally the environmentally preferred treatment'. Feedstock recycling and use of plastics as a fuel in cement kilns was generally preferable to municipal solid waste incineration; landfill was the least preferred choice, except for global warming potential.

Life cycle assessment (LCA), sometimes called life cycle analysis, aims to analyse the environmental impacts of a material or technology throughout its life, 'from the cradle to the grave'. Detailed reviews are available by Ding (n.d.) and Rønning and Brekke (n.d.). But, as Lazarevic *et al*. emphasise, its results are not absolute: they depend on the goals and scope chosen for the studies and on key assumptions which have to be made in order to carry out the analysis.

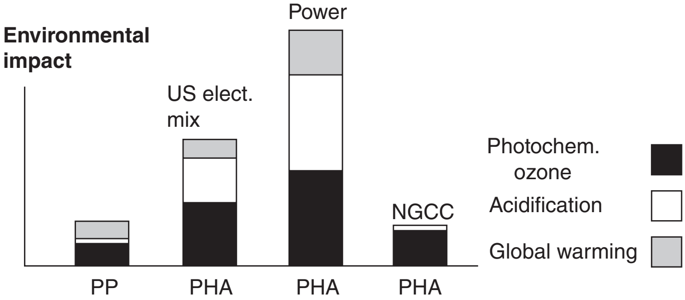
15.4.4 Life cycle assessment (LCA)

Thus, renewable substitutes for many fossil fuel-based polymers and adhesives are available, or could plausibly be developed. Recycling is not without its problems, but may be practicable, even where adhesives and composites are involved. It is, however, prudent to consider the total environmental impact of alternative materials from their production to their disposal. As discussed above, life cycle assessment provides one approach to this 'from the cradle to the grave' assessment.

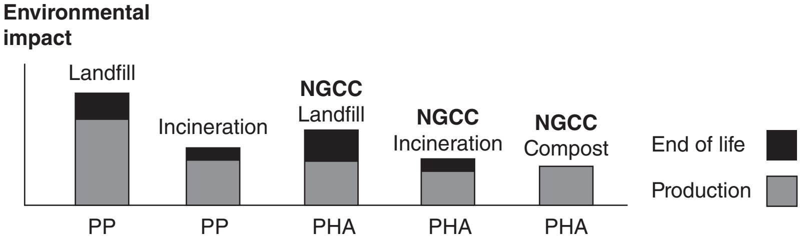
Khoo *et al.* recently published an example of the use of LCA to compare the use of polypropylene with that of renewable polyhydroxyalkanoate (PHA). Both polymers, of course, find applications in hot melt adhesives, but Khoo and her colleagues chose to compare the impact of carrier bags made from each polymer and used in Singapore, where they are based. This example of simple technology gives insight into features of significance when considering materials substitution for environmental reasons (Khoo and Tan, 2010; Khoo *et al.*, 2010).

The polypropylene was produced locally from Middle Eastern crude oil; the PHA was produced entirely in the United States, and shipped to Singapore. They considered the impacts of production and disposal separately, and quantified the impacts on global warming, acidification and production of photochemical ozone which were associated with the production of a 'standard bag' of each material. The various impacts were normalised by dividing by the normalisation value (the total impact in Singapore per person per year). The normalised values of various impacts were then *added* to give a figure to represent the total environmental impact of each case examined.

Figure 15.7 shows the impacts resulting from *production* only (sometimes referred to as '*cradle to gate*'). The polypropylene has a much lower impact than the PHA when it is assumed that the electrical energy associated with the PHA reflected the average of the generation technologies presently used in the United States. In contrast, coal-fired electricity generation would significantly increase the impact and combined cycle natural gas generation (NGCC) reduce it just below the polypropylene impact



15.7 Environmental impact of *production* of carrier bags from polypropylene and poly(hydroxyalkanoate) (after Khoo *et al.*, 2010). NGCC = natural gas combined cycle.



15.8 Environmental impact of *production* and *disposal* of carrier bags from polypropylene and poly(hydroxyalkanoate) (after Khoo and Tan, 2010). NGCC = natural gas combined cycle.

level, although the impact on global warming is still greater for the renewable PHA.

Disposal of the bags adds to the impacts. Landfill was found to have the highest impact. When incineration was considered, the energy recovered was discounted against some of the energy used in production. PHA may be composted, and the compost used as a peat substitute, thus saving greenhouse gas emissions associated with peat production. Figure 15.8 compares the impacts of the polypropylene bags with those for PHA for the environmentally favourable scenario of energy generation by natural gas combined cycle. In these circumstances the PHA is environmentally comparable with polypropylene, or preferable, if composting is used. Of course (cf. Fig. 15.7), with more polluting power generating technologies, the PHA impact is much higher than that for polypropylene.

The authors finish with the obvious, although commonly forgotten point, that multiple reuse of carrier bags will reduce the unit environmental impact much more effectively than millions of pounds of clever technology!

A recent study by Tabone *et al.* (2010) considered the environmental impact of the production of a wider range of polymers, some from petroleum and some from biological sources. The polymers chosen were compared in two ways: using metrics based on life cycle assessment and based on 'green design' principles, discussed above.

The results were complex: there was no clear-cut conclusion that biomass-based polymers were better for the environment. Ranked by 'green design' criteria, polylactic acid (PLA) and PHA come out best of the 12 materials considered. However, the LCA results put polyolefins at the top (polypropylene, HDPE and then LDPE). The biopolymers do well in terms of the effect on global warming and fossil fuel depletion, but have higher impacts on criteria such as toxicity and eutrophication, partly resulting from fertiliser and pesticide use. Different production methods for the same biopolymer were found to have significantly different impacts. The authors point out that their LCA study was limited to the *production* of the polymers.

Depending on the impacts during use and disposal, the order of merit could well change.

This 2010 paper of Tabone *et al.* attracted quite a lot of attention. Its conclusions were briefly summarised in *Chemical & Engineering News* (Anon., 2010), and later adverse criticism was published in *Environmental Science & Technology* (Carroll, 2011; Murphy *et al.*, 2011; Dale, 2011). Much of the criticism was of the details of the LCA procedures followed; Tabone *et al.* responded pointing to the difficulty of finding strictly comparable LCA input data for biopolymers (Landis *et al.*, 2011). Whatever the validity of the detailed conclusions, the paper conveys a clear warning against adopting a simple criterion – such as that renewable polymers are 'better' for the environment. The conclusions of Khoo *et al.*'s studies, discussed above, reinforce this.

15.5 A technical 'fix' for the environmental crisis

The argument of the chapter so far has been that the world is facing an environmental crisis. This crisis is concerned with the depletion of non-renewable resources, especially energy resources, and with the planet's ability to act as a sink for the wastes produced, especially for greenhouse gases emitted as a result of economic activity.

Using polymers, especially adhesives and composites, as examples, it has been argued that science and technology can do much to address these problems. Renewable energy sources and materials may well be preferred, pollution minimised, waste reduced and materials conserved by recycling. These principles of 'green design' were discussed above. The question remains as to whether, however successfully developed these technical solutions may be, their application across the whole of industry will be adequate to solve the crisis.

Before considering this question, it is crucial to recognise that there are powerful forces which seriously aggravate the crisis. Three will be mentioned. One is the pressure of population growth. Here some sort of relief may be anticipated. At present the global population is 7 billion, but according to the 2004 United Nations medium projection, it is expected to peak around 9.2 billion about 2070, but caution is necessary: the high growth scenario gives 14 billion by 2100 with steep rises thereafter (United Nations, 2004).

Another of the powerful forces is ideological (mythical in the educated usage of the word): the Western neo-liberal economic paradigm. This has become near-hegemonic: it treats inputs as if the resources concerned were infinite, and aspires to unlimited growth. This would appear to be difficult to reconcile with the First Law of Thermodynamics (Daly, 1992). The Second Law has equally serious implications: the inputs to the system are typically of low entropy (highly ordered) and the wastes typically of high entropy (disordered), as the Second Law predicts. Thus low entropy petroleum is burnt to produce high entropy carbon dioxide and water.

This economic paradigm resonates with the dominant social paradigm (DSP) with its explicit belief in progress, in the sense of continuously improving material standards of living, and implicit faith in the 'technical fix', that science and technology will infallibly provide a way out of our difficulties. This aspiration to ever-increasing material standards of living, of course, aggravates the problem of finite resources and of disposal of pollutants.

15.6 Energy demand and supply

The question as to whether technical solutions will be adequate to solve the crisis in resources can be addressed by examining likely future demand and supply. Energy, which is central to almost all technology, provides a critical example. In 2008 the global primary energy use was of the order of 514 EJ/yr (1 EJ = 1 exajoule = 10^{18} joule), according to International Energy Agency figures (Moriarty and Honnery, 2011). What will it be in the future?

Estimates of future energy demand are available, and obviously depend upon what assumptions are made about population growth and energy usage. The Intergovernmental Panel on Climate Change in 2007 reviewed a wide range of predictions of primary energy requirements, drawing attention to the assumptions on which they were based and the greenhouse gas emissions which they implied (Barker *et al.*, 2007). Estimates of future demands for global primary energy in a broadly 'business as usual' world, i.e., in the absence of policy change or major supply constraints with consequent increases in oil demand and CO_2 emissions, have been made by the Energy Information Administration and the International Energy Agency (Moriarty and Honnery, 2011). These suggest that the energy supply needed to meet the demand in 2050 is likely to be in the vicinity of 1,000 EJ/yr, and that the cumulative global primary energy requirement from 2005 to 2050 is about 30,000 EJ (Metz *et al.*, 2005).

Trainer (2010) has recently considered how an energy supply of 1,000 EJ/yr, required in 2050, might be provided in a way which was consistent with meeting safe greenhouse gas emission targets. His analysis assumes that there will be a 33% increase in energy efficiency by 2050. It emphasises that the output of many renewable generation technologies has a strong seasonal variability and an inherent intermittency. This means that there must be considerable 'redundancy' in the provision of generating plant if a steady supply is to be maintained. Trainer argues that 'It is not likely that energy-intensive economies could sustain the required multiples of present investment indicated'. Put more prosaically, his conclusion is that the energy requirement implicit in a broadly 'business as usual' world development scenario is inconsistent with a safe level of greenhouse gas emissions.

So far we have been considering a 'business as usual' scenario. Trainer (2010) further points out that the 1,000 EJ/yr primary energy requirement, which he used for the 2050 illustration, would be well below that needed to provide energy equity to the whole world. A simple calculation shows that if the expected (United Nations, 2004, medium projection) world population in 2050 of 9 billion were to consume energy at the present per capita rate of the Western world, something in the region of 4,000 EJ/yr would be needed. Some of these energy figures are compared in Table 15.7.

A similar problem with 'business as usual' world development scenarios emerges when the cumulative primary energy demand is compared with published reserves for oil, natural gas and uranium. Figures for such reserves are intrinsically imprecise, a problem compounded by governmental and commercial secrecy (Murray and King, 2012). Recognising this, Turner (2008) argued that the lower bound for ultimate resources of primary energy was about 60,000 EJ, the upper bound (including low grade coal) roughly two to three times as great. Some of Turner's figures are given in Table 15.8.

Table 15.7 Actual and projected global primary energy requirements

Scenario	Global primary energy requirement
Actual in 2008	514 EJ/yr
'Business as usual' in 2050	1,000 EJ/yr
Global energy equity in 2050	4,000 EJ/yr
'Business as usual' cumulative 2005 to 2050	30,000 EJ

(1 EJ = 1 exajoule = 10^{18} joule).

Table 15.8 Some estimates of global energy resources from Turner (2008)

Fuel	Ultimate resource (EJ)
Conventional oil	10,000
Conventional gas	10,000–20,000
Non-conventional oil	10,000–40,000
Non-conventional gas	20,000
Nuclear fission	5,000–9,500

At the time of writing (Autumn 2012) there is a frenzy in the popular media about abundant reserves of 'unconventional' oil and gas, which, it is claimed, could transform the energy situation, and even solve the energy crisis. Two considerations should temper this enthusiasm.

Estimates for unconventional hydrocarbon resources (see Table 15.8) are subject to much greater uncertainty even than those for conventional sources. Murray and King (2012) argue that US shale gas reserves have been 'substantially overstated' and point to the 'extremely large annual decline in production rates' of shale gas wells. The US Energy Information Administration (2011, p. 37) warns that the resource potential may be overestimated when 'production rates for the sweet spot [with highest known production rate] are used to infer the productive potential of an entire formation'. Moreover, the energy return on investment (EROI) for these unconventional fuels tends to be low (Johnson *et al.*, 2012) (Table 15.2) with the consequences of high energy prices and very low net energy returns (Fig. 15.2).

Secondly, there are the serious implications for climate change which would follow the use of these supposedly abundant oil and gas hydrocarbon reserves. The International Energy Agency (2012) has recently warned that 'No more than one-third of proven reserves of fossil fuels can be consumed prior to 2050 if the world is to achieve the 2°C goal [required to limit climate change to "safe" levels], unless carbon capture and storage (CCS) technology is widely deployed'. A similar point was made by the UK Committee on Climate Change (2012), in its advice to the government on gas-fired power generation. A study by the Tyndall Centre for Climate Change concluded that 'Without a meaningful cap on emissions of global GHGs, the exploitation of shale gas is likely to increase net carbon emissions' (Wood *et al.*, 2011).

15.7 The stationary state: limits to growth

Thus there are good reasons to question the plausibility of energy resources being sufficient to allow 'business as usual' economic growth for the foreseeable future. The difficulties are compounded when the need to reduce greenhouse gas emissions to a safe level is recognised. This is essentially the argument that there are 'limits to growth', which has long been made, and is implicit in the Laws of Thermodynamics. The eighteenth century moral philosopher, Adam Smith, often called the 'Father of Economics', recognised that economic growth would eventually reach limits in a 'stationary state' (Smith, 1776; Alvey, 1998). John Stuart Mill (Mill, 1848) insisted, in his influential *Principles of Political Economy* (1848, revised 1873):

> It must always have been seen, more or less distinctly, by political economists, that the increase of wealth is not boundless: that at the end of what they term the progressive state lies the stationary state, that all progress in wealth is but a postponement of this, and that each step in advance is an approach to it.

The famous Club of Rome study *Limits to Growth*, published in 1972, revived interest in the stationary state (Meadows *et al.*, 1972). It showed that, if unchecked, exponential growth in population and in resource use, together with the finite capacity for the assimilation of pollution would lead to a serious collapse in the material quality of life and human population during the first half of the present century.

The study challenged the dominant economic and social paradigms, causing intense controversy: it was widely denigrated and often misrepresented (Turner, 2008; Fisk, 2005). However, 30 years on from the original *Limits to Growth* study, it can be seen (Turner, 2008) that the predictions of its 'business as usual' scenario compare well – so far – with historic data between 1970 and 2000. Whether collapse will occur midway through the twenty-first century, in accordance with the scenario output, remains to be seen.

These philosophical and practical considerations clearly call into question the validity and continuing utility of the Dominant Social Paradigm with its belief in progress (material standards of living will continuously improve) and that scientific advance will always provide solutions to our problems. Such beliefs form a narrative which has become firmly embedded in the popular consciousness in Western industrial society; it is supported by a host of powerful vested interests, and is now increasingly adopted on a global scale. However, it is important to recognise that this narrative does not represent some sort of 'unchanging Law of Nature'. Historically speaking it is recent: its origin can be traced to the ideas of the Enlightenment in eighteenth-century Europe (Lyotard, 1988). When such narratives cease to be useful in helping the human race to understand its position in the world, they must be developed or replaced.

One such development is outlined in the paper published by the UK Government's Sustainable Development Commission entitled *Prosperity without Growth? The Transition to a Sustainable Economy* (Sustainable Development Commission, 2009; Jackson, 2009). This points to the fundamental problem with an economic system which does not adequately take into account the finite nature of many resources and the degradation of ecosystems by pollution. It argues for a much broader concept of human well-being than one which focuses exclusively on continual growth in material prosperity, and calls for a fundamental restructuring of the financial system and a basic rethinking of the economy, setting out a route to sustainability. The social transformation implicit in such a change has recently been elaborated (Jackson, 2011).

Tim Jackson, the Economics Commissioner at the Sustainable Development Commission, terms economic progress a 'myth', using the word to depict a conceptual framework which provides ways of thinking about the world. He points out that 'every society clings to a myth by which it lives; ours is the myth of economic progress … but a society that allows itself to be steered by a faulty myth risks foundering on the shores of harsh reality' (Jackson, 2004).

Such ideas, of course, challenge powerful vested interests. The Sustainable Development Commission in the UK was closed by the new Conservative-led government on 31 March 2011.

15.8 Conclusions and future trends

Since the middle of the last century, the global economy has grown by a factor of about five. Cheap and abundant oil and gas have favoured growth in general and, particularly, in the polymer and adhesives industries. These industries, then, have played a part in generating the present prosperity and the present environmental crisis of resources (especially of energy resources) and of pollution (especially of greenhouse gas emissions). By adopting the principles of 'green chemistry and engineering' much can be done by replacing the use of limited resources by use of renewable resources and to reduce polluting emissions. However, it is important that these principles are used critically: it is entirely possible, for example, that the substitution of a biopolymer for one based on petroleum will lead to an increased environmental impact. Quantitative assessment, such as detailed 'cradle to grave' life cycle analysis is needed.

So what trends might be anticipated for the future? Perhaps we will see a thorough, but critical, use of 'green chemistry and engineering' principles, with the implications for adhesive use detailed above. On the other hand, by and large the economy is still structured in a way which makes it cheaper in the short term to waste resources than to conserve them, and to use the environment (air and water) as a sink for waste disposal rather than to conserve and recycle. So there is popular political pressure in many countries facing economic problems to denigrate 'green principles', and the regulation which attends them, in favour of short-term gain, rather than longer-term stability. This is manifested in the current political enthusiasm for unconventional oil and gas sources at the expense of wind, wave and solar energy generation. If this pressure prevails, we can expect less emphasis on renewable adhesives and on reducing associated emissions. This, of course will not evade the environmental crisis, but will aggravate it.

It might perhaps seem that the thorough, but critical, use of 'green chemistry and engineering' principles would provide a resolution to the environmental crisis. It is necessary to remember that the social and economic context is one which strongly adheres to a paradigm of continuous growth. There are good reasons, including those based on the Laws of Thermodynamics, for considering that there are limits to growth and that a social paradigm which ignores these limits is in danger of 'foundering on the shores of harsh reality'. 'Every society clings to a myth by which it lives; ours is the myth of economic progress'. Are we ready to recognise that continuous growth is not an immutable Law of Nature, but a myth (according to the educated usage of the word), which may have served us well in the past but which now must be significantly modified?

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16

Life cycle assessment (LCA) of road pavement materials

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Abstract: At present the choice of materials and techniques in road construction is dictated by structural requirements and economic aspects. However, ecological factors have gained in importance due to environmental considerations in politics and society. To evaluate the environmental impact of motorways, a life cycle assessment (LCA) according to ISO 14040 was carried out for different pavement types. By investigating different case scenarios, the reduction potential of environmental impact was quantified. The biggest reduction in air pollution can be achieved by improving pavement properties (e.g., texture, stiffness and flatness) which would significantly reduce the fuel consumption of vehicles.

Key words: life cycle assessment, pavement LCA, asphalt roads, concrete roads.

16.1 Introduction

The European Union has committed itself to the sustainable reduction of climatic emissions under the Kyoto Protocol. One of the largest contributions to emissions is from the transport sector (EEA, 2012). This chapter quantifies the environmental impact of motorways. As well as pavement construction and deconstruction, the ecological impact of a motorway after wear from traffic and the effect of maintenance over a period of 30 years have been analysed systematically using LCA methodology according to ISO 14040 (ISO 14040, 2006). All input and output values for the individual processes in the production and use of pavement for a motorway section have been taken into account. These processes include the production of materials, provision of energy, manufacture of the necessary products, transport services and the employment and disposal of the individual products.

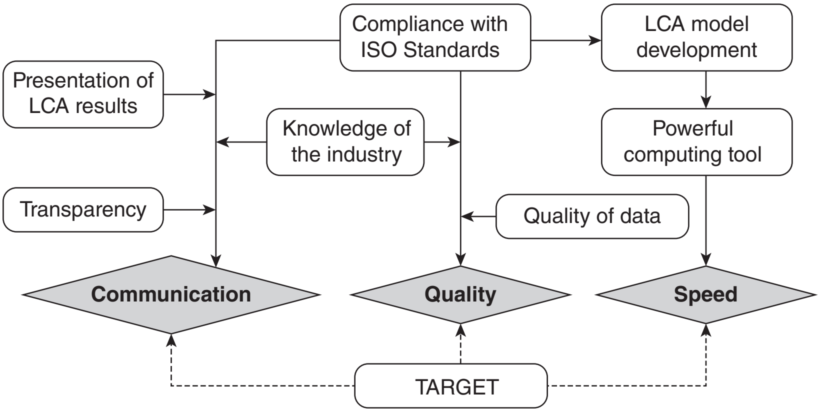
Emissions into air, water and soil were determined and, using the Dutch CML method (Guinée *et al.*, 2002), assigned to the following impact categories: global warming potential (GWP 100 years), ozone depletion potential (ODP), photochemical ozone creation potential (POCP), acidification potential (AP) and eutrophication potential (EP). The Swiss database 'EcoInvent' (EcoInvent, 2009), which provides consistent and transparent data, was used. Processes not available in the database were analysed and modelled on the basis of existing upstream processes. The data were evaluated using the LCA software 'SimaPro'. Possible reductions in environmental impact were determined by considering various scenarios.

16.2 Life cycle assessment (LCA) for roads

16.2.1 LCA according to DIN ISO 14040

Life cycle assessment is a standard method which takes into account all environmental aspects of the product life cycle (emissions into air, water and soil, waste, use of raw material and exploitation of nature). This comprehensive approach avoids the misallocation of environmental effects and provides an overview for possible impact reduction. The LCA method is described in the international standards ISO 14040 (ISO 14040, 2006) and 14044 (ISO 14044, 2006).

An LCA study comprises four phases which affect one another. In the first phase, the goal and scope of the investigations and the resulting system boundaries are defined. In the inventory analysis phase, all relevant materials and energy inputs and outputs are included in the system. In the assessment phase, the environmental effects of the system components are assigned to different impact categories. Different materials are weighted according to their damage potential and summarized in total impact indicators. In the final interpretation phase, the impacts are analysed and evaluated in order to draw conclusions or make recommendations. Figure 16.1 shows the key elements for a quality LCA study.



16.1 Key elements for a successful LCA study of pavements (Huang *et al.*, 2009).

In this study, results are presented for the LCA of the construction, use and maintenance as well as deconstruction of a motorway section 1 km in length. The study was conducted in close cooperation with industry. The data received were analysed and transformed to the EcoInvent database to guarantee high data quality. For fast processing, the LCA software 'SimaPro' was used.

16.2.2 Literature review

At present, several LCA case studies for roads exist. Some of them are summarized below. These works show the importance of LCA as decision aid. In 1995, Stripple established a dynamic life cycle inventory (LCI) model for road construction, operation and maintenance for the Swedish National Road Administration which was translated into English in 2001 (Stripple, 2001). Extraction activities as well as traffic (5,000 vehicles per day) were included, but disposal of waste and production of capital equipment were not taken into account. The study showed that the energy use of construction, operation and maintenance together was between 9.9 and 11.8% of the estimated traffic energy.

Häkkinen and Mäkelä (1996) studied the life cycle from cradle to grave of asphalt and concrete pavements. The influence of pavements on fuel consumption, as well as traffic, lighting requirements and dust formation were included. A service period of 50 years was considered. Data were gathered from industry. The study showed that the highest environmental impact stems from traffic. For asphalt roads, the main reduction potential was in asphalt production itself. The environmental impact of concrete highly depended on the cement content of concrete as well as the thickness of the concrete course. The environmental burdens of the pavements were very sensitive to the chosen maintenance operations.

In 1999, Inamura *et al*. (1999) published an LCA case study for the construction and operation of a motorway using different data sources. A hybrid I-O (input–output) model was developed and applied to an expressway construction in Japan. The study focused on carbon dioxide emissions only. It was shown that fuel combustion during the operating period produces nearly 90% of total CO_2 emissions. Schenk (2000) showed that it is feasible to use LCA to choose the most eco-efficient product for asphalt pavement treatment.

Mroueh *et al*. (2000) analysed alternative road and earthwork constructions. They established a database of environmental impact of construction materials based on literature data. Case studies were carried out using different industrial by-products, such as coal ash, crushed concrete waste and granulated blast-furnace slag. The results showed the reduction potential of using by-products as well as crushed concrete instead of natural aggregates.

The most significant environmental impacts were generated by the production of bitumen and cement as well as crushing of materials. The study showed that energy use and CO_2 emissions for each road construction alternative compared with traffic were only 0.1–0.2% for energy consumption and 0.8–1.8% for CO_2 emissions.

Birgisdottir (2005) developed an LCA model (ROAD-RES) evaluating the environmental impact of road construction and comparing different disposal methods for bottom ash from incineration of municipal solid waste. The data used stemmed from Danish producers, laboratory tests, field experience and literature. The study showed that using bottom ash in road construction has similar environmental burdens as landfilling.

In 2006, the Athena Institute published an LCA study which compared asphalt and concrete roadways in Canada over a service period of 50 years (Athena, 2006). Different data sources were used. Neither traffic, lighting in urban areas nor end-of-life scenarios were considered. The study focused on energy consumption and GWP only. For the investigated cases, concrete pavements showed significant advantages in energy consumption compared to asphalt pavements. The differences in GWP were within the 10% confidence interval and therefore insignificant.

Using Eco-indicator 99 (MHSPE, 2000), Chiu *et al.* (2008) estimated the environmental burdens of different rehabilitation options for an asphalt pavement in Taiwan. Three different recycled materials (asphalt incorporating 10–25% crushed glass, recycled hot mix asphalt and asphalt rubber) were compared with traditional hot mix asphalt. Two different life spans (the life span of one rehabilitation measure as well as a total service life of 40 years) were analysed. Over a service period of 40 years, recycled HMA as well as asphalt rubber led to 23% lower eco-burdens, while the use of glass incorporated asphalt led to 19% higher environmental impact. For all four materials, the materials as well as the heat required for asphalt production had the biggest environmental impact. Furthermore, the study showed that the planned service life needs to be taken into account when comparing materials with different durability as it changes the results significantly. According to the authors, quality assurance of pavements as well as reducing the heat requirement for asphalt production are the most effective ways to ensure sustainability in the road sector.

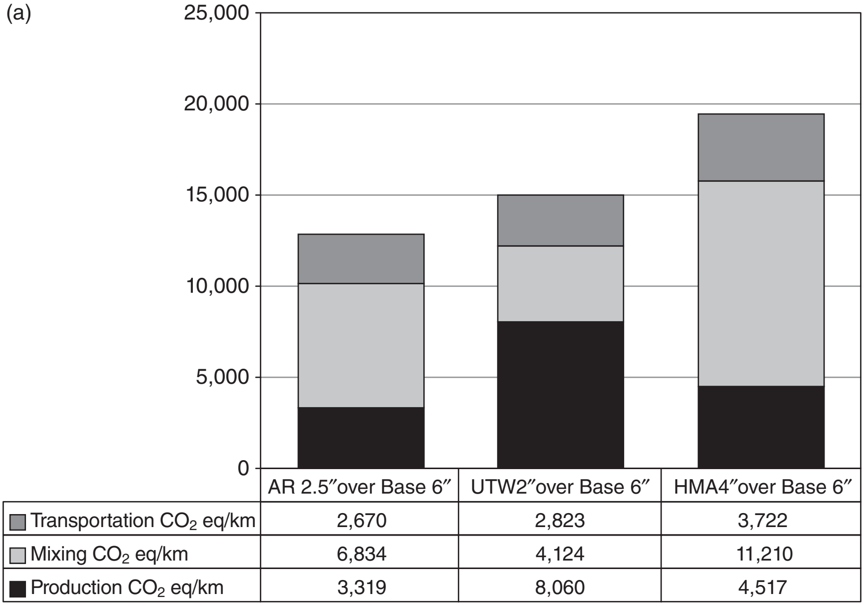
Huang *et al.* (2009) conducted an LCA for three asphalt paving projects in the UK. In addition, they developed an LCA model for pavement construction and maintenance taking different recycling rates into account. Data from UK plants and contractors were used. The production of hot mix asphalt and bitumen had the biggest environmental impact. The model and the case studies are up-to-date, in compliance with ISO standards as well as transparent. However, concrete pavements are excluded.

Sayagh *et al.* (2010) used a parametric environmental evaluation tool called ERM (elementary road modulus) to compare the construction and maintenance of three different pavement structures: one asphalt pavement, one classical concrete pavement, and one using blast furnace slag (BFS). Data from different literature sources were used. The refining of bitumen and mixing of asphalt had the biggest environmental impact for the asphalt pavement. For the BFS alternative, two cases were investigated: one considering BFS as waste from steel production and one considering BFS as a by-product. The steel and cement plant contributed the most to energy use for the reinforced concrete pavement. For BFS, the bitumen refining as well as the steel production were the main contributors. The different allocation procedures had only a small effect on the energy consumption for concrete pavement. However, the impact on BFS itself was significant. The energy consumption as by-product led to an energy consumption more than twice as high compared with the waste allocation. On the one hand, the use of BFS reduces the amount of binder which is extracted from natural resources. On the other hand, more natural aggregates are consumed. Concerning the environmental burdens, classical concrete pavement had the largest impacts in the categories global warming, acidification and eutrophication. BFS and the ‘by-product’ allocation had the largest impact in photochemical and ecotoxicity, whereas asphalt pavement had the highest toxic potential.

Weiland and Muench (2010) conducted a comparative LCA for three different replacement options for an ageing Portland cement concrete (PCC) motorway over a service period of 50 years. The options were replacement with a new PCC pavement, replacement with hot mix asphalt (HMA) pavement, as well as cracking, seating and overlaying (CSOL) the existing pavement with HMA. Different maintenance schedules were taken into account for the three options. Different data sources were used. Materials production (e.g., cement, asphalt, PCC and HMA) had the highest contribution to energy consumption, emissions, and impacts for all three options. The HMA option had the highest energy consumption mainly due to HMA production, whereas the PCC option had the highest global warming potential (GWP) mainly due to cement production itself. The CSOL was the most beneficial concerning energy consumption and GWP, and produced the lowest emissions compared to the PCC and HMA pavements. The study showed the importance of maintenance and rehabilitation works compared to the demolition of old and construction of new pavements.

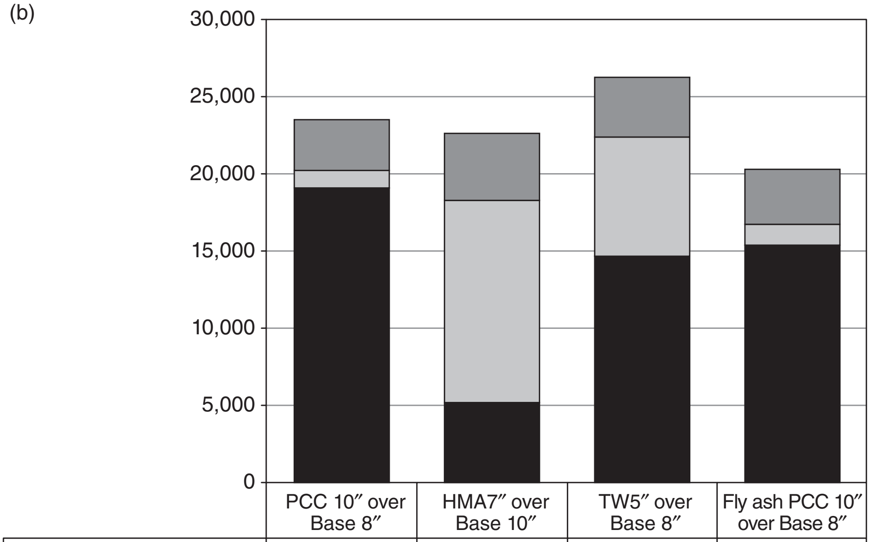
White *et al.* (2010) determined the direct CO_2-eq. impacts for seven different pavement types in the US (Fig. 16.2). Data from different sources were used. Due to the energy intensive production of the Portland cement, the biggest share on CO_2-eq. emissions for pavement types with Portland cement concrete (PCC) came from raw material production. For the production of asphalt itself, pre-heated aggregates are mixed with bitumen. As a consequence, mixing contributed the most to CO_2 emissions.

In 2011, Santero *et al*. (2011a) reviewed 15 LCA studies. According to them, a comparison between the studies is impossible since the scope of the investigations (aim, functional unit and environmental burdens considered) varied too much. In addition, different input data were used. Some studies used various databases while others gathered information from manufacturers or from other studies. Therefore, the conclusions drawn in one of the given studies are only valid for the specific case that was investigated. Hardly any reviewed study included traffic delay, fuel consumption due to pavement properties such as roughness and structure, lighting, the urban heat island effect, radiative forcing or concrete carbonation and leaching into soil. As a consequence, pavements are compared using incomplete information. To fill these gaps, the authors analysed the state of the art on possibilities to include traffic disruption, albedo, concrete carbonation, lighting, leachate as well as end-of-life scenarios in pavement LCA (Santero



	AR 2.5″over Base 6″	UTW2″over Base 6″	HMA4″over Base 6″
▩ Transportation CO_2 eq/km	2,670	2,823	3,722
░ Mixing CO_2 eq/km	6,834	4,124	11,210
■ Production CO_2 eq/km	3,319	8,060	4,517

16.2 Total annual kg CO_2 eq./km for different pavement types (White *et al*., 2010). (a) Moderate traffic volume pavement designs; (b) high traffic volume pavement designs (AR = asphalt rubber; UTW = ultra-thin whitetopping; HMA = hot mix asphalt; PCC = Portland cement concrete; TW = thin whitetopping; Fly ash PCC = PCC with fly ash (FCC); upper layer consisting of roughly 40% gravel, 39% sand, 4% fly ash from electricity generation coal combustion, 9% Portland cement and 8% water).



	PCC 10″ over Base 8″	HMA7″ over Base 10″	TW5″ over Base 8″	Fly ash PCC 10″ over Base 8″
▩ Transportation CO_2 eq/km	3,232	4,268	3,956	3,585
▢ Mixing CO_2 eq/km	1,161	13,079	7,732	1,324
■ Production CO_2 eq/km	19,105	5,242	14,620	15,394

16.2 Continued

et al., 2011b). The study showed that there is still need for further research to quantify the above-mentioned aspects properly. According to them, sufficient models need to be developed and test methods should be standardized. Their recommendations are summarized in Table 16.1.

Concerning future pavement LCAs, Santero *et al.* (2011a) state that a standardized functional unit framework needs to be developed including the function, location and design description of the pavement. Also, the system boundaries should be expanded and the data quality improved. In addition, more impact factors than energy and GWP should be analysed in order to provide a reliable decision aid.

Gschösser (2011) analysed the life cycle of national and cantonal roads in Switzerland using LCC and LCA. Different service periods (25, 50 and 75 years) were taken into account. For the use phase, only constructive maintenance works were included. Just like the present study, the data in Gschösser (2011) were provided by industrial partners or taken from the EcoInvent database. The applied maintenance strategy as well as material production processes had the biggest environmental burdens. As a consequence, the largest reduction potential was the optimization of material production and the choice of maintenance strategy. None of the investigated pavement types (asphalt, concrete and composite) was superior in all

Table 16.1 Summary of recommendations for filling the existing research gaps in pavement LCA (Santero *et al.*, 2011b)

Phase	Issue	Research recommendation
Construction	Traffic delay	• Integration of economic traffic delay models with environmental impact models • Better understanding of traffic detouring and its contribution to environmental impact
	Rolling resistance	• Completion of comprehensive empirical studies to quantify the pavement–vehicle interaction phenomenon • Development of mechanistic-based models that relate rolling resistance, fuel economy, pavement structure, and pavement roughness while accounting for exogenous variables
Use	Albedo	• Calculation of marginal impact of pavement albedo changes on electricity demand due to increased air conditioning demand from urban heat island effect (UHI) • Better understanding of localized effects of albedo increases (e.g., heat reflection and radiation on buildings) • Discussion of sensitivity of UHI impact based on climate region and urban density • Confirmation and refinement of CO_2 radiative forcing offset calculations for marginal changes in pavement albedo
	Concrete carbonation	• Continued collection of data on the rate that carbonation occurs in concrete pavements in different scenarios • Specify the impact of concrete pavement properties on the carbonation rate
	Roadway lighting	• Development of comprehensive database of lighting requirements for different pavement types • Better treatment of lighting efficiency improvements over time
	Leachate	• More conclusive tests and data on potential environmental hazards of leachate from pavements, with a focus on recycled materials and specialty applications
End-of-life	Landfilling, recycling, and remain in place scenarios	• Improved data regarding pavement end-of-life fate (i.e., rates of landfilling, recycling, or remaining in place) • Improved allocation protocols specific to pavement LCAs • Use of system expansion to more accurately quantify recycling and remain in place scenarios

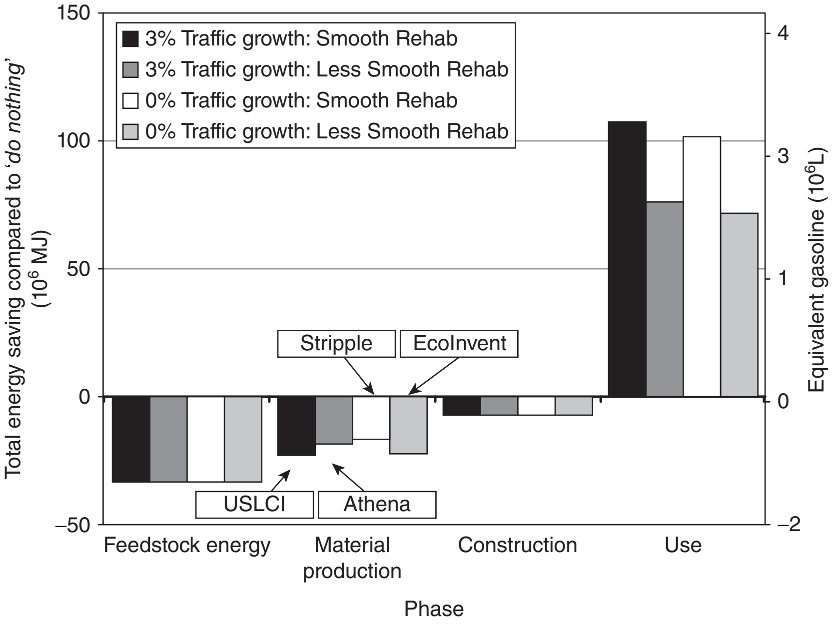
analysed categories (GWP, non-renewable cumulative energy demand and ecological scarcity indicator).

Butt *et al*. (2012) established an open LCA framework for the construction, maintenance and disposal of Swedish asphalt pavements. The framework focused on the calculation and allocation of energy used for binder and additives. Additives like polymers and waxes improve asphalt durability. This knowledge was taken into account in a case study where an unmodified and a polymer-modified asphalt pavement were compared. A further case study investigated the benefits of the use of self-healing bitumen. The data used were provided by the asphalt industry and construction companies. Asphalt production and material transportation were the main contributors to energy use and greenhouse gases (GHG). The fuel type and the electricity mix were very sensitive to the results. The study showed how the improved performance of modified asphalt as well as the use of self-healing bitumen reduced GHG emissions and energy consumption. The authors therefore concluded that better understanding of the binder (bitumen) would enable eco-efficient pavement design.

Giustozzi *et al*. (2012) analysed different maintenance and rehabilitation works. The study showed the benefits of preventive maintenance treatments on road pavements. They also developed a single tool that takes costs, performance and eco-efficiency into account. The data used are based on different sources. As a consequence, the scatter in the data is quite high. Nevertheless, the proposed approach offers an optimized decision tool that enables road authorities and municipalities a choice between different alternatives.

Wang *et al*. (2012) compared the energy and the GHG emissions from pavement maintenance and rehabilitation activities with potential savings in environmental impact due to the traffic volume, reduced rolling resistance of pavements and material used. Multiple data sources were used. The reduction of vehicle use provided the largest environmental benefits. The second most beneficial factor was the smoothness of the pavement. The third factor was the material used. Hot mix asphalt (HMA) as well as high early strength Portland cement concrete showed higher environmental impact than gap-graded rubberized HMA and calcium sulphoaluminate cement concrete. The results are shown for an asphalt roadway with high traffic volume in Fig. 16.3. The figure shows that different data sources for material production only have a minor impact compared to the use phase. Energy savings due to smooth rehabilitation are able to outweigh energy consumption due to construction and maintenance of roads.

The most extensive LCA study was carried out by Yu and Lu (2012). Their LCA study of pavement included the construction, use and end-of-life phase of three different overlay systems. Effects of pavement roughness and structure on fuel consumption were taken into account. In addition, traffic congestion due to maintenance works as well as albedo and carbonation were investigated. The study used data from various literature



16.3 Life cycle energy saving in each phase compared to 'do nothing' for a busy asphalt road (Wang *et al.*, 2012). USLCI, Athena, Stripple and Ecoinvent characterize different data sources.

sources. Material, congestion as well as usage were the main sources of energy consumption and air pollutant emissions.

Loijos *et al.* (2013) developed a general pavement LCA methodology showing the necessary concepts for comprehensive pavement LCA. This methodology was applied to the life cycle of 12 different concrete pavements representing average conditions for each major roadway classification in the US. The GHG emission of materials production, construction, use, maintenance and end of life were estimated. The service period analysed was 40 years. The study showed that cement was the main contributor to GWP in the construction phase. Materials production, transportation and end of life caused 64–80% of emissions. The overall results were most sensitive to traffic volume, and second to parameters affecting the cement production. Based on emissions and their sensitivity, the reduction of embodied emissions, as well as use-phase emissions, and end-of-life emissions were found to be the most eco-efficient measures.

This review shows that LCA is well acknowledged in the road sector to evaluate the eco-efficiency of different roadways. Most studies are consistent on the fact that fuel consumption of traffic has by far the biggest environmental impact (Stripple, 2001; Häkkinen and Mäkelä, 1996; Mroueh *et al.*, 2000; Athena, 2006; Huang *et al.*, 2009; White *et al.*, 2010; Butt *et al.*, 2012; Wang *et al.*, 2012; Yu and Lu, 2012). However, only few studies included the effect of rolling resistance, pavement roughness and stiffness on fuel combustion. All studies agree that the main contributors to pavement construction and maintenance are the material production (asphalt mixing, bitumen refining and cement production), while some studies are sensitive to transport processes.

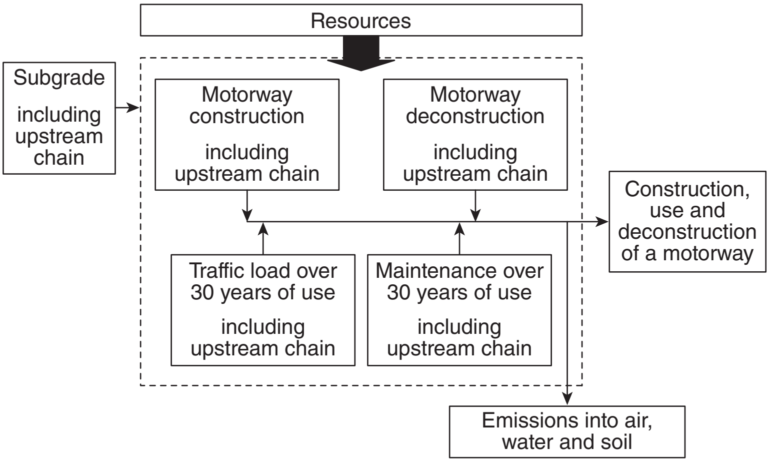
The data quality also determines the quality of an LCA study. However, some data are out of date. Also, models or databases established before the ISO 14040 was revised in 2006 may not fully comply with it. In addition, some studies use data from different sources or the underlying assumptions and calculations are unknown (Huang *et al.*, 2009). Therefore, the data quality may not be very good. Data from non-German sources may not represent Germany's average situation. Since the underlying prerequisites vary strongly, a direct comparison of the reviewed studies is not possible (Santero *et al.*, 2011a). As a consequence, the present study focuses on a 1 km long motorway section in Germany. The detailed and transparent LCI is published in Milachowski *et al.* (2010).

16.2.3 Scope of present study

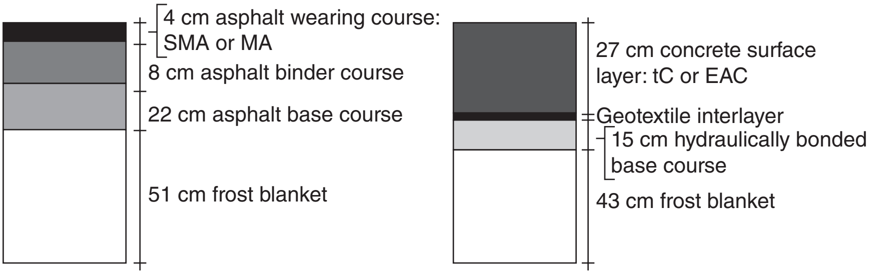
The scope of the investigations is characterized by the system boundaries and the functional unit. In ISO 14040, the system boundary is defined in a spatial context. System boundaries generally include the entire life cycle of a product, i.e. pre-manufacturing (raw material production, manufacture of parts and components), the actual manufacturing process, transport, application and disposal. Figure 16.4 shows the system boundaries of this study (dashed lines).

The functional unit was a 1 km long section of a two-lane (on each carriageway) motorway section with a pavement thickness of 85 cm. The concrete pavements investigated were constructions with an exposed aggregate concrete surface layer (EAC) and with surface texture produced by brushing (tC). Pavements with a stone mastic asphalt wearing course (SMA) and mastic asphalt (MA) were included in the study. Subgrade preparation (e.g., ground compaction) and furnishing work (e.g., road marking, safety barriers, etc.) were not part of the study. Drainage measures (drains, gullies, etc.) as well as road marking were neglected. The asphalt pavement comprising surface, binder and base courses is supported by a frost blanket (Fig. 16.5, left). The concrete motorway consists of a frost blanket followed by a hydraulically bonded base course, a geotextile interlayer and a concrete surface layer. In the present case, the concrete layer comprises two separate layers in which the surface course is either textured or is an exposed aggregate concrete (Fig. 16.5, right).

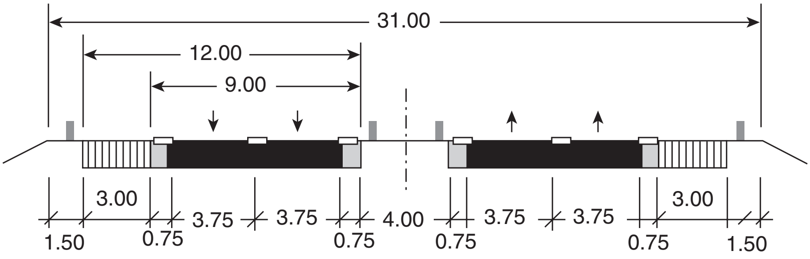
The structure of the two-lane motorway section is shown in Fig. 16.6. Each side of the motorway has a verge 1.5 m in width which is not included



16.4 System boundaries for the LCA (dashed lines). The term 'upstream chain' includes all main parameters for upstream processes with potential environmental impact (i.e., production and processing of raw materials, transportation, waste treatment, etc.).



16.5 Schematic diagram of the investigated pavement structure of an asphalt (left) and concrete (right) motorway.



16.6 Investigated standard motorway cross-section RQ 31 (FGSV, 2008), dimensions in meters.

in this study. The 3.0 m hard shoulders are separated from the inside lanes by side strips 0.75 m in width, which are also next to the outside lanes.

16.2.4 Origin of data and life cycle inventory assessment

The EcoInvent database already contained a number of materials which could be directly used for the life cycle inventory analysis of the production and service use of motorways (e.g., aggregate for the frost blanket, mixing water for the concrete, bitumen for the asphalt). Many data sets had to be adapted to the situation in question (cement, joint filler, etc.). Since data were not available on the production of asphalt, curing agents and pavement concrete, the production methods were analysed and presented in a form which enabled modelling with the available basic modules in EcoInvent (e.g., provision of electricity, electric motors, etc.). All data sets used were verified.

16.2.5 Assumptions and limitations

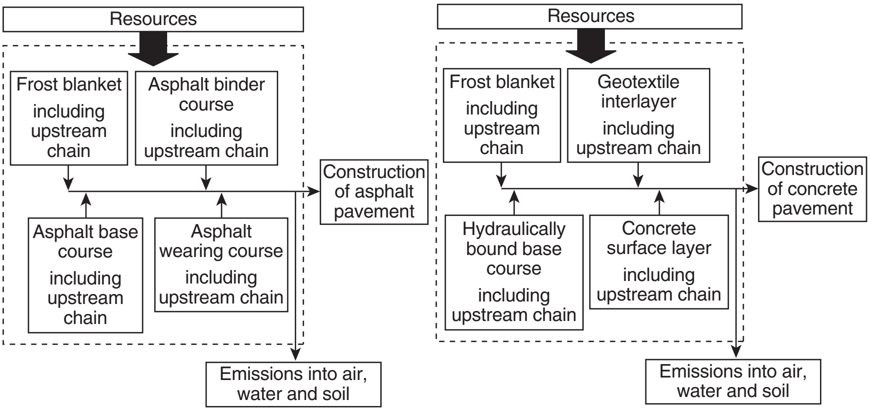
Life cycle assessment takes only ecological aspects into account, not the social and economic factors which must be considered for decision making in road infrastructure. LCA only covers standard cases, whereas the choice of a suitable and ecological construction method often depends strongly on local circumstances. In some circumstances, the potential environmental impact could be even less than in the standard case. Hence, a comparison of different construction methods should always be made in light of the further technical, economic and social requirements (cost, construction time, noise protection requirements, etc.). A further limitation of an LCA study is the focus on the current situation for the whole service life of a motorway. Changes which might occur during the real service life due to technical progress or changes in environmental impact are neglected.

16.3 LCA for motorway construction

16.3.1 Scope of investigations

For the construction of a 1 km long motorway section, the raw material extraction, material processing, transportation and material placement for the individual layers including the edge regions were analysed. Excavation processes were neglected. The system boundaries for the construction phase are shown in Fig. 16.7.

To quantify the potential for the optimization of environmental impact available in the use of different materials and optimized production, various scenarios were investigated (Table 16.2).



16.7 System boundaries for the construction of a motorway (dashed lines). Left: asphalt pavement; right: concrete pavement.

Table 16.2 Overview of investigated scenarios for the production of 1 km motorway

Asphalt constructions: pavement with mastic asphalt wearing course (MA) or stone mastic asphalt wearing course (SMA)	Concrete constructions: pavement with textured surface (tC) or exposed aggregate concrete surface layer (EAC)
Scenario A: 0% recycled material for all layers, standard asphalt production	*Scenario A:* 0% recycled material for all layers, CEM I for concrete surface layer
Scenario B: 0% recycled material for other layers 100% recycled material for frost blanket, standard asphalt production	*Scenario B:* 0% recycled material, CEM I for concrete surface layer 100% recycled material for frost blanket
Scenario C: 10% less energy use in asphalt production (optimized asphalt production)	*Scenario C:* 0% recycled material for all layers, CEM III for surface concrete layer

The use of recycled material is not considered at all in scenario A for asphalt and concrete pavements. It was not possible to determine a reliable percentage of recycled material because information on the amounts and use of recycled material in road construction vary strongly. In order to identify the potential reduction in environmental impact by reusing construction material, it was assumed in scenario B that the frost blanket consisted of 100% recycled material stemming from the upper layers of an existing motorway. However, scenario B does not comply with field practice because transport processes on site were completely neglected. The asphalt production was modelled according to data provided by the German Asphalt Organization. Thus, for the drying and heating of the aggregate for 1 m^3 asphalt, 7.08 litres of light fuel oil is required in standard scenario A. For the production of mastic asphalt, 8.06 litres of light fuel oil per m^3 are needed. The bitumen is kept warm in storage tanks for which 18.75 kWh of electricity is consumed. However, experts state that diverse measures (i.e., optimized storage, lower mix temperatures, optimized equipment) enable a further reduction in energy consumption by up to 10% which is taken into account in scenario C for the asphalt pavements (ASA Anlagentechnik, 2010; DAV, 2010; Jenny, 2009).

The concrete surface layer in standard scenario A is made with Portland cement CEM I 42.5 N. The optimization potential in the use of composite cements was quantified by replacing the Portland cement by Portland blast-furnace cement CEM III/A 42.5 R in scenario C. This cement represents the upper limit on the ground granulated blast furnace slag content of cement in German road construction.

16.3.2 Inventory analysis

Motorway construction is divided into different stages, each performed at different times. The construction work considered in this study began with the production of the frost blanket followed by the other layers in turn. The necessary operations with all materials and machines were modelled for each layer. Finishing work, such as surface texturing and curing, was also taken into account.

All input and output streams during the life of each process were determined and compiled for the environmental inventory. The frost blanket was modelled by a mixture of gravel and sand. The use of bulldozers, terrain levellers and vibration rollers was taken into account. A cold milling machine used to remove old layers was included in scenario B. The crushers required for processing the materials were included in the cold milling module. The machine specifications came directly from manufacturers. The material data were provided by manufactures and the industry association.

An overview of the main materials and machines for the production of asphalt and concrete motorway pavement is presented in Tables 16.3 and 16.4, respectively. The process chains determined from this information are presented in Milachowski *et al.* (2010), together with the data sets used and the scatter of the values.

In scenario A, the bottom concrete was mixed using 350 kg/m^3 CEM I 42.5 N and 158 l/m^3 water; the top concrete for the pavement with a textured surface was produced with 360 kg/m^3 cement and 162 l/m^3 water. The top exposed aggregate concrete contained significantly more cement in

Table 16.3 Overview of the main materials and machines for the production of asphalt motorway

Layer	Material		Machines	
Asphalt base course	36.7 kg/m^3 bitumen 2,349.0 kg/m^3 aggregate		2 pavers 4 vibration rollers	
Asphalt binder course	0.3 kg/m^2 bitumen emulsion (tack coat) 45.9 kg/m^3 polymer modified bitumen (PMB) 2,421.0 kg/m^3 aggregate		5 bitumen sprayers 2 asphalt pavers 4 vibration rollers 1 tandem roller	
Asphalt wearing course	Mastic asphalt (MA): 0.3 kg/m^2 bitumen emulsion (tack coat) 72.5 kg/m^3 low-viscosity bitumen 2,415.0 kg/m^3 aggregate 8.0 kg/m^2 chippings (scatter material)	Stone mastic asphalt (SMA): 0.3 kg/m^2 bitumen emulsion 69.4 kg/m^3 PmB 2,570.0 kg/m^3 aggregate 0.15 kg/m^2 cellulose fibres 8.0 kg/m^2 chippings	Mastic asphalt (MA): 5 bitumen sprayers 4 mastic asphalt boilers 2 pavers 2 finishing machines	Split mastic asphalt (SMA): 5 bitumen sprayers 2 feeders 2 pavers 4 rollers 4 chip spreaders

this case 430 kg/m^3 with a w/c ratio of 0.42 assumed. All concretes were mixed with an air-entraining agent (Kellenberger *et al.*, 2004). The higher requirements placed on the quality of the aggregate in the top concrete were taken into account. An average German concrete plant was assumed for the concrete production.

The distance for the transport of materials was set to 50 km and, based on field experience, the distance from the concrete plant to the construction site was taken as 20 km. These distances include fully loaded delivery and empty return. Construction site installation was only taken into account by the delivery and removal of the construction machines; a distance of 100 km was assumed.

16.3.3 Impact assessment

Figure 16.8 shows the impact indicators for motorway production with asphalt in a spider web diagram. Each axis corresponds to one of the impact categories (see Section 16.1). It was found that the use of mastic asphalt leads to about 2% higher potential environmental impact in all categories than construction with SMA. This is due to the high binder content of the asphalt surface layer and the energy consumption for paving. The use of

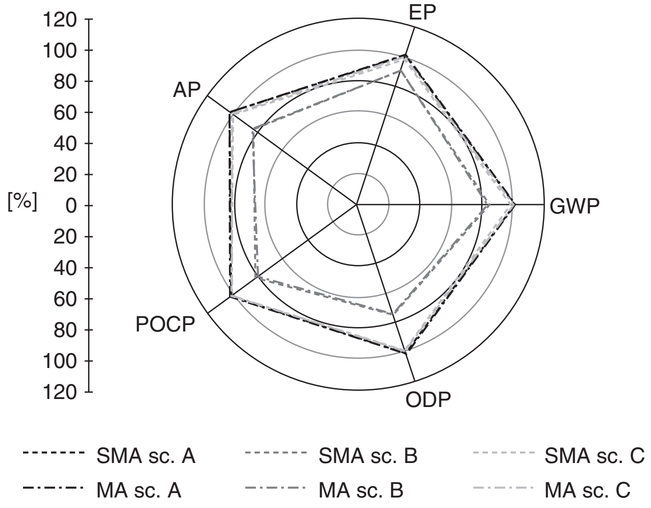
Table 16.4 Overview of the main materials and machines for the production of concrete motorway

Layer	Material		Machines	
Hydraulically bonded base course	90.0 kg/m^3 CEM II/B-S 32.5 R 1,975 kg/m^3 aggregate 110.0 l/m^3 tap water 1.6 kg/m^2 C60 B1 – N		2 pavers 6 rollers 5 bitumen sprayers	
Interlayer	0.5 kg/m^2 geotextile		–	
Concrete surface layer	Pavement concrete with textured surface (tC): 3,360 m^3/km bottom concrete 1,680 m^3/km top concrete 44.8 t/km steel for dowels and anchors 9.6 t/km curing agent 4.3 t/km joint filler	Exposed aggregate concrete (EAC): 3,360 m^3/km bottom concrete 1,680 m^3/km top concrete 44.8 t/km steel for dowels and anchors 6.0 t/km combination agent 4.8 t/km curing agent 4.3 t/km joint filler	Pavement concrete with textured surface (tC): 1 slipform paver 1 curing machine 2 groove cutters 3 joint sealing machines	Exposed aggregate concrete (EAC): 1 slipform paver 1 curing machine 2 brushing machines 2 groove cutters 3 joint sealing machines

100% recycled material in the frost blanket (scenario B) lowers the environmental impact in all categories by 9 (ODP) to 27% (EP). The reduction of energy consumption for the asphalt production leads to a reduction in environmental impact of up to 3% (scenario C).

For all categories and scenarios, the largest proportion of the potential environmental impact originates in the asphalt itself (34–89%). Due to the consumption of diesel fuel, the transportation processes and site preparation also have a large impact (17–45%). While the bitumen used for asphalt production contributes significantly to the ODP and the POCP, diesel fuel consumption mostly affects the AP and the EP. Also bitumen and all energy consumption processes (material processing, transportation, site preparation) contribute to the GWP.

The contribution of the frost blanket ranges from 11 to 31%, depending on the category, and is largest for the EP. The base layer yields by far the largest part of the potential impact. It is, depending on the category, between 38 (EP) and 57% (ODP). The amount for the binder layer lies between 17

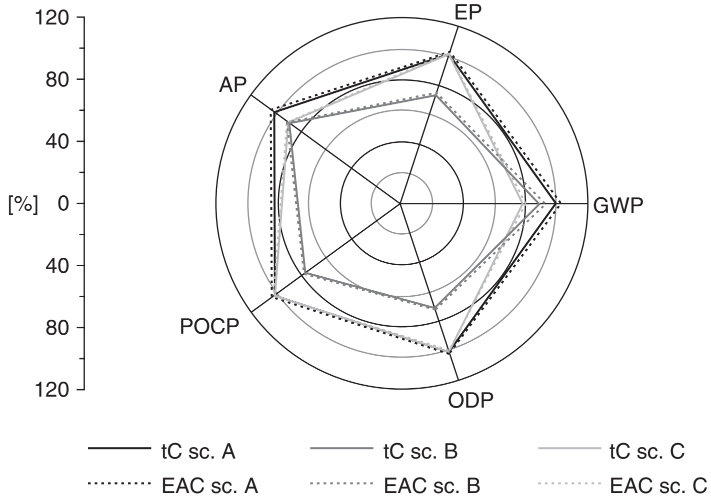


16.8 Results of assessment of environmental impact for pavement construction with asphalt. The variables are with respect to the pavement with SMA in scenario A. (MA = pavement with mastic asphalt wearing course, SMA = pavement with stone mastic asphalt wearing course).

(ODP) and 23% (POCP), depending on construction method and category. Thus the contribution from the surface layer ranges from 10 to 15%, depending on construction method and category.

The results for the environmental impact of motorway construction with concrete are shown in Fig. 16.9. The use of 100% recycled material in the frost blanket in scenario B reduces the impact by amounts between 12 (GWP) and 31% (EP), depending on the category. This is due to the neglect of transport processes in this scenario. By replacing CEM I by CEM III, GWP is reduced by amounts of 20 (pavement with textured concrete) or 21% (pavement with exposed aggregate concrete). The reduction is much less for the other categories and ranges from 0.1 to 10%. The exposed aggregate concrete pavement has, in all cases, a slightly higher environmental impact than the pavement textured with artificial grass owing to the higher cement content of the top concrete layer. Differences in curing have a negligible effect.

Depending on the category, the frost blanket contributes between 14 (GWP) and 38% (EP) to the environmental impact. The contribution from the hydraulically bonded base course is between 14 (GWP) and 23% (ODP). Between 0.4% (ODP) and 5% (AP) is from the interlayer geotextile, depending on the construction method. The concrete surface layer contributes between 42 (ODP) and 72% (GWP), again depending on construction method and impact category. Thus the largest contribution to



16.9 Results of assessment of environmental impact for the construction of 1 km concrete motorway. The variables are with respect to the tC pavement in scenario A. (tC = pavement with textured concrete surface; EAC = pavement with exposed aggregate concrete surface layer).

the potential environmental impact of the production of a concrete motorway is from, in all cases examined, the concrete itself. Dominance analysis of this material revealed that the effect of mixing water and air-entraining agent on the environment is negligible. The largest effect is due to the Portland cement and lies between 70 (EP) and 96% (GWP), whereas the potential environmental impact of the aggregate is at most 10%. Other contributions originate mainly in requirements on infrastructure and transport processes. Although the impact contributions vary with concrete composition, cement type and content, the main contribution is, in all categories, always from the cement.

An overview of the potential environmental impact for the production of 1 km motorway is presented in Table 16.5. While the environmental impact of pavement construction with concrete is 1.3–1.6 times higher than for asphalt pavements in the category GWP, the use of asphalt as construction material leads to higher potential environmental impact in the categories ODP, POCP, AP and EP.

16.3.4 Evaluation

Owing to the large variation in available information, the values chosen for the machines and fuel needed to place the mixtures of materials in

Table 16.5 Environmental impact indicators for the construction of 1 km motorway

	GWP	ODP	POCP	AP	EP
	[kg CO_2-eq.]	[kg CFC-11-eq.]	[kg C_2H_4-eq.]	[kg SO_2-eq.]	[kg PO_4^{3-}-eq.]
MA scenario A	1,694,573	0.39	413	8,191	1,232
MA scenario B	1,425,044	0.35	346	6,623	919
MA scenario C	1,647,883	0.38	406	8,069	1,221
SMA scenario A	1,667,593	0.38	406	8,067	1,221
SMA scenario. B	1,383,398	0.34	337	6,443	896
SMA scenario C	1,619,109	0.37	398	7,908	1,206
tC scenario A	2,710,311	0.13	380	6,374	1,084
tC scenario B	2,339,814	0.09	308	4,644	742
tC scenario C	2,153,620	0.13	344	6,343	1,079
EAC scenario A	2,821,219	0.13	389	6,478	1,100
EAC scenario B	2,474,597	0.09	317	4,748	758
EAC scenario C	2,227,417	0.13	350	6,447	1,094

the different layers were somewhat pessimistic. Their effect on the total potential environmental impact of the pavement was small, ranging from 2 (GWP) to at most 9% (POCP). The potential environmental impact originates essentially in the materials. In particular, the energy-intensive production of cement and asphalt is decisive, ranging between about 57 and 66% in scenario A.

While the potential for a further reduction in impact by improving the process engineering of cement production is generally considered to be exhausted, an impact reduction of up to 3% for the construction of an asphalt motorway section can be achieved by optimizing the asphalt production itself. In addition, further impact reduction may be achieved by optimization of the material itself as well as by increasing recycling rates. In the case of the concrete construction method, the LCA for the construction of a motorway section has shown that the use of CEM III/A instead of Portland cement can reduce the environmental impact by up to 21%.

The use of 100% recycled material for the frost blanket reduces the potential impact by amounts of 10 (ODP) and 31% (EP) in the cases examined. Heavy goods vehicles accounted mainly for the transport of materials. As well as reducing transport processes in general, a reduction in impact may be achieved by the partial use of transport which has less adverse effects on the environment (e.g., rail). The ODP is very much affected by the actual transport processes involved.

16.4 LCA for motorway use and maintenance

16.4.1 Scope of investigations

In this study, motorway usage encompasses use by traffic and maintenance work. Furthermore, the additional fuel consumption due to traffic congestion at construction sites was investigated. Related processes of the use phase such as road cleaning, winter services and lighting were not part of the study. In order to sufficiently model albedo and carbonation for a standard case, many assumptions are necessary (Santero *et al.*, 2011b). As a consequence, these effects were also excluded from the study. A usage period of 30 years was considered in which constructional measures to maintain a motorway were included along with typical traffic conditions, i.e. a traffic scenario with a volume of 52,000 vehicles in 24 h (42,000 passenger cars, 10,000 heavy goods vehicles). The use of the motorway section by motorcycles and buses was neglected. Traffic growth over the service life was not taken into account. In scenario A, the standard fuel consumption was taken as the European average of 0.286 kg/km diesel for heavy goods vehicles and, for cars, 0.0627 kg/km diesel or 0.0669 kg/km petrol (Spielmann *et al.*, 2004).

Traffic contributes a major proportion of the total emission of air pollutants. To quantify the reduction potential of environmental impact caused by traffic, besides scenario A, three additional scenarios were investigated.

- Scenario B: 0.5% fuel saving,
- Scenario C: 2.0% fuel saving,
- Scenario D: 10.0% fuel saving for heavy goods vehicles.

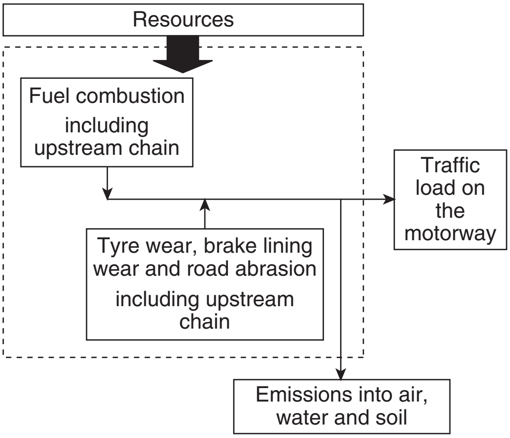
The additional scenarios chosen were inspired by literature studies which revealed a clear effect of pavement properties on fuel consumption (Ardekani and Sumitsawan, 2010; Descornet, 1990; ECRPD, 2010; EUPAVE, 2012; Hultqvist, 2010; Laganier and Lucas, 1990; Larsson and Andersson, 2004; Lengrenn and Faldner, 2010; Lundström and Finnsementti, 1998; NRC-CSTT, 2000; Sandberg, 1990; Slavik *et al.*, 2004; Taylor *et al.*, 2006; Yoshimoto *et al.*, 2010; Zaabar and Chatti, 2010; Zaniewski, 1989). Depending on pavement type, stiffness and rolling resistance (macrotexture and roughness) as well as vehicle type, fuel consumption can be lowered by up to 20%, (Table 16.6). The system boundaries for the traffic scenarios are illustrated by the dashed lines in Fig. 16.10. For diesel vehicles, only the combustion process and different types of abrasion were taken into account, whilst for passenger cars also cold start emissions and standstill evaporation were included.

As well as traffic load, construction maintenance measures were also taken into account in the period of usage. For concrete construction methods,

Table 16.6 Fuel savings due to diverse pavement features

Vehicle type	Fuel saving	Source
Passenger car	0%	Zaniewski, 1989
Heavy goods vehicle (HGV)	up to 20%[a]	Zaniewski, 1989
Passenger cars and HGV	9%	Descornet, 1990
Passenger car	5%	Laganier and Lucas, 1990
Passenger car	7–12%	Sandberg, 1990
HGV	6%	Slavik *et al*., 2004
Passenger car	2.9%	Taylor *et al*., 2006
HGV	0.8–3.1%	Taylor *et al*., 2006
Passenger car	3.5–8%	Ardekani and Sumitsawan, 2010
Passenger car	0–5%	Zaabar and Chatti, 2010
HGV	0–1.5%	Zaabar and Chatti, 2010
Passenger car	1.1%	Hultqvist, 2010
HGV	6.7%	Hultqvist, 2010
HGV	0.8–3.4%	Yoshimoto *et al*., 2010

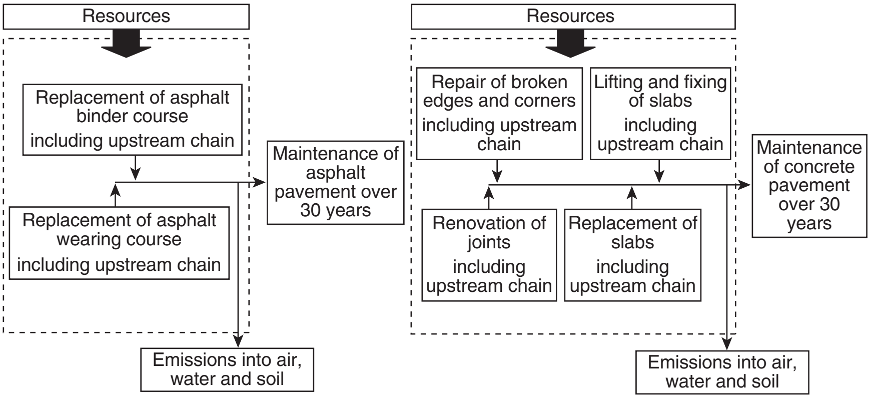
[a] not statistically significant.



16.10 System boundaries for the use of a motorway by traffic (dashed lines).

maintenance of joints, renovation of broken edges and corners, lifting and fixing of slabs as well as the replacement of complete slabs were taken into account. The system boundaries are shown in Fig. 16.11. All environmentally relevant factors including all upstream chains (raw materials production, freight, production of materials, production of machines and their use, etc.) were determined and analysed.

Since construction maintenance depends on numerous parameters which cannot all be quantified in a general LCA study, a minimum maintenance



16.11 System boundaries for motorway maintenance (dashed lines). Left: asphalt pavement; right: concrete pavement.

scenario (A) and a maximum maintenance scenario (B) based on field experience were defined. For the asphalt constructions in scenario A, the binder course was renewed once and the wearing course twice in the analysis period. In scenario B, binder and wearing course were renewed twice. For the concrete constructions in scenario A, the joints were completely renovated two times in the analysis period. Furthermore, 1% of the slabs were lifted and fixed, another 1% of the slabs were replaced and 5% of the edges and corners were repaired by injecting a two-component resin.

In scenario B, the joints were completely renovated three times in the period of use. In addition, 3% of the slabs were lifted and fixed. A further 3% of the slabs were replaced and 20% of the edges and corners were repaired. An overview of the maintenance scenarios investigated in this study is presented in Table 16.7.

Due to a lack of sufficient data, traffic disruption due to maintenance works was modelled for average German conditions and not for the specific considered maintenance works. In the case of the asphalt construction methods, the replacement of complete asphalt courses was modelled.

16.4.2 Inventory analysis

All environmentally relevant data for processes relevant to the usage of the motorway section were determined and modelled using the EcoInvent database.

In order to model traffic load, the work of Spielmann *et al.* (2004) was adapted to the current situation in Germany using information provided by the German Environmental Agency (Umweltbundesamt, 2009). Heavy goods vehicles comprised 40% vehicles with a transport weight between 3.5 and 20 t, 31% with a weight between 20 and 28 t, and 29% with a weight

Table 16.7 Overview of the maintenance scenarios for 1 km motorway

Asphalt constructions	Concrete constructions
Pavement with mastic asphalt surface wearing course (MA) or stone mastic asphalt wearing course (SMA)	Pavement with textured concrete surface (tC) or exposed aggregate concrete surface layer (EAC)
Scenario A: 2× replacement of wearing course 1× replacement of binder course	*Scenario A:* 2× complete renovation of joints 5% repair of broken edges and corners 1% lifting and fixing of slabs 1% replacement of slabs
Scenario B: 2× replacement of surface course 2× replacement of binder course	*Scenario B:* 3× complete renovation of joints 20% repair of broken edges and corners 3% lifting and fixing of slabs 3% replacement of slabs

between 28 and 40 t. A vehicle utilization of 50% was assumed for all vehicle types. Just 17.6% of the 42,000 passenger cars are operated with diesel while the rest are operated with petrol. The manufacture and servicing of the vehicles was not included in the LCA. However, the emission of dust from tyres, brake liners and road wear was included in the outputs.

In the case of the asphalt construction methods, the removal of old layers is modelled by a cold milling machine (45 t, 647 kW). The milled surface is cleaned with a brushing machine and sprayed with an adhesive. An application of 0.3 kg/m^2 unstable cationic bitumen emulsion was assumed in this study. The placement of new layers was carried out in the same manner as the original production. Basically the same materials were employed in the maintenance of the concrete pavement as in construction. A detailed description of the materials and machines considered in the assessment appears in Milachowski *et al.* (2010). Table 16.8 lists only several of the most important input parameters.

The construction machines used for motorway maintenance were based on information supplied by companies and recommendations in the list of construction appliances published by the Confederation of the German Construction Industry. A distance of 50 km was taken for the transport of old and new materials to and from the site. In analogy to the production process, only the delivery and removal of the construction machines were considered with regard to the installation of the construction site.

While minor maintenance and rehabilitation works are usually carried out at night, intensive maintenance works lead to lane closures and frequent traffic delays. Additionally, electricity due to night work was not considered

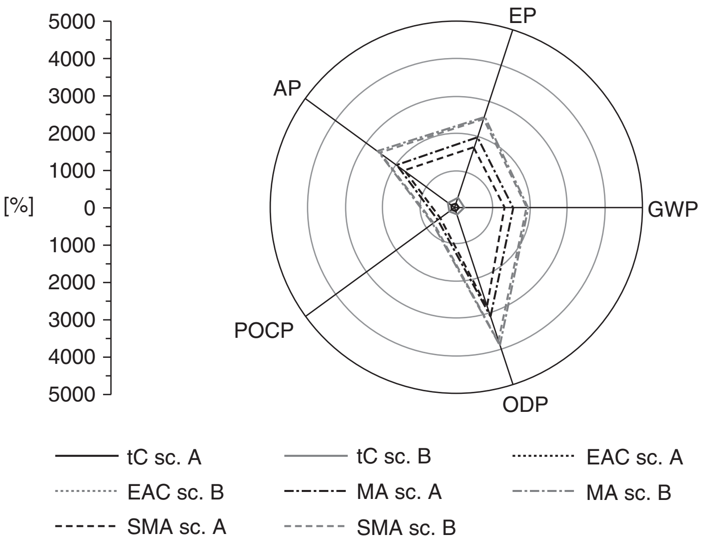
Table 16.8 Overview of the main materials and machines taken into account for construction maintenance of the concrete pavements

	Material		Machines
1× joint renovation	4.3 t/km joint filler		Compact loader, compressors, brushing machines and bitumen cauldron
1% repair of broken edges and corners	0.4 kg/m^2 adhesive bridge 1.15 kg/m^3 two-component resin		Jack hammer, shot blasting machine, small compressor and mixer
Lifting and fixing of 1% of the slabs	2.95 m^3/km repair mortar		Compressor, drill, concrete pump and vibration roller
Replacement of 1% of the slabs	Pavement concrete: 56.7 m^3/km high-early-strength concrete 0.45 t/km steel 0.11 t/km curing agent	Exposed aggregate concrete: Same materials as in production phase	Pavement concrete: concrete cutter, drill, jack, dowel and anchor machine, concrete pump, poker vibrator, smoother Exposed aggregate concrete: in addition one brushing machine

in this study. However, the additional fuel consumption and the associated environmental impacts due to traffic delay at construction sites were examined. Traffic congestion is by definition exactly when the traffic flow is less than 10 km/h for longer than one minute (AVD, 2009). Traffic disruption is generally caused by heavy traffic, road works or accidents. Within the considered service period of 30 years, only traffic disruption due to maintenance works was taken into account. Their share is between 30 and 35% of all traffic congestion on German motorways (ADAC, 2009). Therefore, a share of 32.5% was assumed. According to the German automobile society (ADAC, 2009), there were 116.7 hours of traffic congestion on a 1 km long motorway section over a period of 30 years. A higher fuel consumption of 5 l/h was considered. Thus 163,428.4 l of fuel are additionally consumed. For this purpose, the same system boundaries were chosen as for the use of the motorway section by traffic (Fig. 16.10).

16.4.3 Impact assessment

The results of the analysis for the maintenance of asphalt and concrete pavement over a service period of 30 years are shown in Fig. 16.12. All the impact categories for the maintenance measures show much less



16.12 Results of the impact assessment for the maintenance of 1 km motorway: asphalt and concrete construction methods. (tC = pavement with textured concrete surface; EAC = pavement with exposed aggregate concrete surface layer; MA = pavement with mastic asphalt surface layer, PA = pavement with porous asphalt surface layer).

environmental impact for the concrete pavement than for the asphalt pavement.

The assumed traffic load of 42,000 cars and 10,000 heavy goods vehicles per day results in an environmental impact which is up to 5,000 times higher than the impact of pavement maintenance. The additional fuel consumption due to traffic congestion caused by maintenance works is comparably low. The potential environmental impacts for the 1 km motorway section are summarized in Table 16.9.

16.4.4 Evaluation

The impact reduction potential for maintenance measures in scenario A (minimum maintenance) compared to scenario B (maximum maintenance) lies between 20 and 60% depending on the impact category. For the GWP, this means a reduction of 110–370 t CO_2-eq. In the field, keeping road pavements at high service levels through a preventive maintenance approach during the pavement service life can significantly improve their performance and reduce their deterioration rate (Giustozzi *et al.*, 2012). Furthermore, traffic congestion due to lane closures can be avoided. However,

Table 16.9 Environmental impact indicators for maintenance and use by traffic of 1 km motorway

	GWP	ODP	POCP	AP	EP
	[kg CO_2-eq.]	[kg CFC-11-eq.]	[kg C_2H_4-eq.]	[kg SO_2-eq.]	[kg PO_4^{3-}-eq.]
MA scenario A	944,116	0.21	272	5,249	723
MA scenario B	1,230,617	0.27	352	6,808	943
SMA scenario A	802,329	0.18	234	4,513	632
SMA scenario B	1,172,658	0.26	338	6,539	916
tC scenario A	60,520	0.01	46	265	36
tC scenario B	170,920	0.01	81	742	110
EAC scenario A	63,971	0.01	46	270	37
EAC scenario B	181,274	0.01	82	756	113
Traffic congestion	8,763	0.00	11	35	6
Traffic sc. A	230,904,557	29.84	167,980	1,066,521	202,078
Traffic sc. B	229,750,034	29.69	167,140	1,061,189	201,067
Traffic sc. C	226,286,466	29.24	164,620	1,045,191	198,036
Traffic sc. D	220,146,604	28.30	166,410	1,008,952	189,865

traffic delay due to maintenance works only has a minor share of the total environmental burdens caused by the operation period.

The present study showed that optimization of environmental impact potential for the construction maintenance of concrete motorways is specified mainly in the durability of the joint fillers. By reducing transport processes, the potential environmental impact could be reduced significantly for all types of pavement constructions. Impact reduction potential is also available in the optimization of the construction materials and in the precision of their application. Mixed construction methods could exploit more effectively the advantages of the different types of materials. For example, in construction with asphalt on concrete, concrete provides for the overall durability of the pavement while asphalt has a positive effect on noise reduction.

Motorway pavement maintenance for the concrete construction methods over a service period of 30 years leads to significantly lower potential environmental impacts in all categories compared with asphalt pavement. Hence, investment in durable motorway construction is rewarded in the service phase. The largest potential impact reduction lies in lowering fuel consumption since the impact is mainly due to the combustion of fossil fuel. Fuel consumption is determined by many factors. In the past, numerous investigations concentrated on the effect of road surfaces (rolling resistance, flatness, stiffness) on fuel consumption. Road surface properties such as texture, unevenness (macro- and mega-texture) and pavement stiffness can reduce fuel consumption by up to 20% (Zaniewski, 1989). Optimization potential is therefore available in pavement construction as well as in car and tyre manufacture.

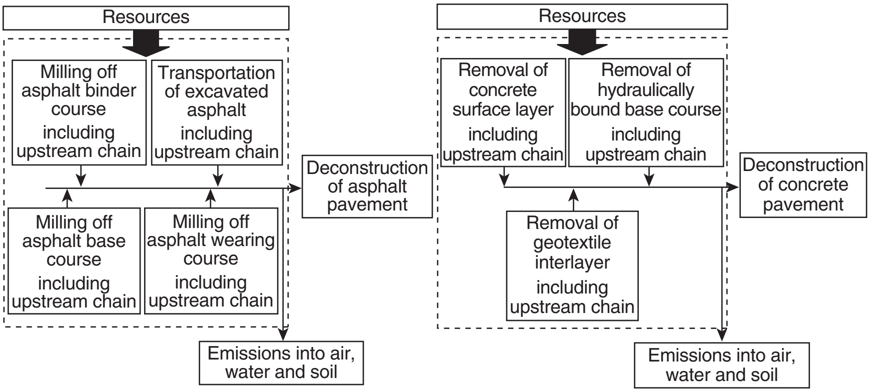
16.5 LCA for the demolition/deconstruction of motorways

16.5.1 Scope of investigations

As mentioned above, a service life period of 30 years is considered in this study. However, sub-base courses have an average service life of 80 years in practice. Therefore, for the deconstruction phase, the demolition of the frost blanket was excluded. The system boundaries for the deconstruction phase are shown in Fig. 16.13. The demolition of the existing layers was carried out with cold milling machines for all pavement constructions (scenario A). In addition, the concrete layers were removed using crawler excavators, hydraulic breakers and hydraulic shears (scenario B).

16.5.2 Inventory analysis

The environmentally relevant data needed for the deconstruction of the motorway section were provided by the industry and modelled using the EcoInvent database. For the deconstruction, five cold milling machines were taken into account (45 t and 647 kW for asphalt layers, 45 t and 708 kW for concrete and hydraulically bound layers). These machines pulverize the existing pavement to a suitable particle size in a single operation. Concrete is harder than asphalt. As a consequence, the milling speed for the removal of concrete layers is lower than for asphalt pavements. In addition, the



16.13 System boundaries for the deconstruction of a motorway (dashed lines). Left: asphalt pavement; right: concrete pavement.

cutting tools and the conveyor belts wear out more quickly. Therefore, a higher diesel and lubricating oil consumption as well as a lower service life of the cold milling machines were taken into account for concrete pavements. The motorway is deconstructed in three passes. In the case of concrete pavements, steel parts were transported to a steel mill at a distance of 200 km. For all remaining materials, a transport distance of 50 km was taken into account. In analogy to the construction and use phase, only the delivery and removal of the machines was considered with regard to the installation of the construction site.

16.5.3 Impact assessment

The results of the analysis for the deconstruction of asphalt and concrete pavement are summarized in Table 16.10. The differences between asphalt and concrete pavements concerning the investigated environmental impact categories are negligible.

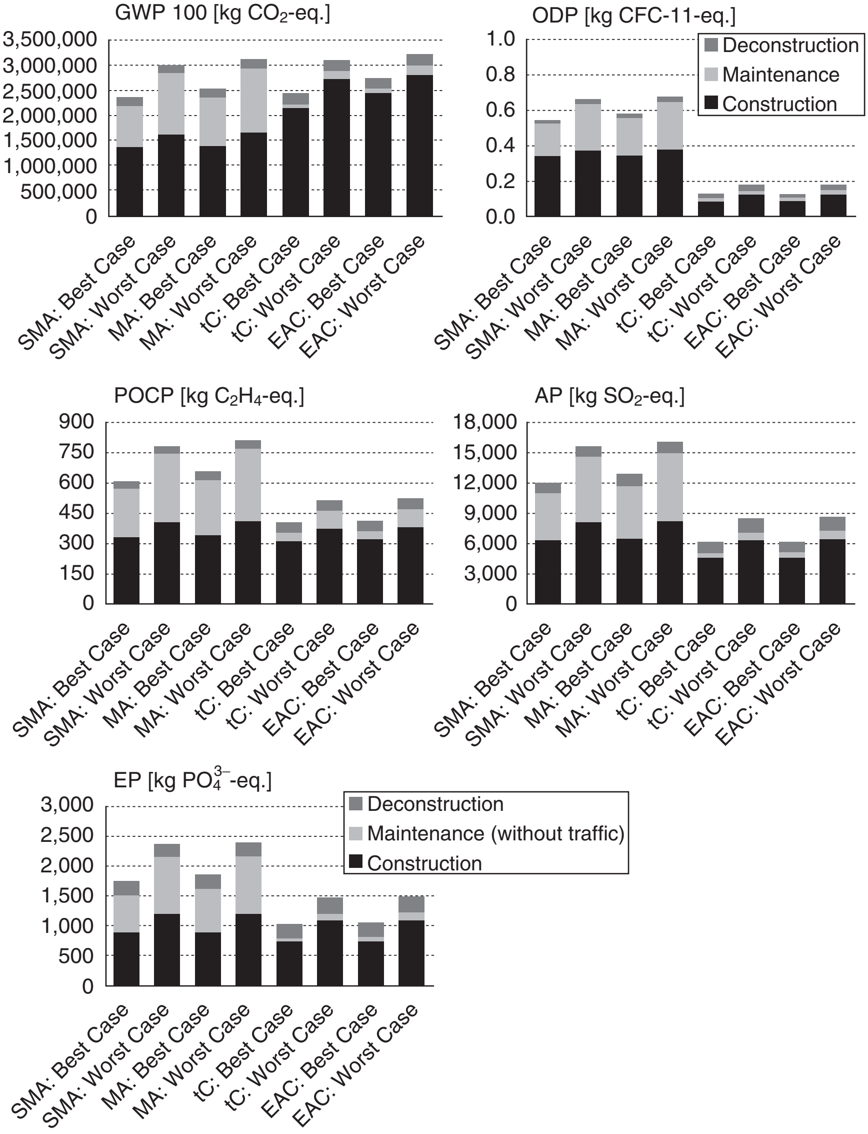
16.5.4 Evaluation

The milling of concrete layers with cold milling machines leads to higher environmental impact than removal with hydraulic breakers. However, compared to the impact of material processing in the construction phase, the environmental impact of the deconstruction is insignificant (Fig. 16.14). Optimization of environmental impact potential for the deconstruction of motorways is specified mainly in the reduction of transport processes since their share of potential environmental impact lies between 82 and 96%.

Comparison of the different pavement types with concrete and asphalt for motorway construction, maintenance and deconstruction shows that their effect on GWP is similar. For ODP, the asphalt pavements cause a potential environmental impact which is four times higher than with concrete. In the case of POCP, AP and EP, the impact is from 1.1 to 2.0 times

Table 16.10 Environmental impact indicators for the deconstruction of 1 km motorway

	GWP	ODP	POCP	AP	EP
	[kg CO_2-eq.]	[kg CFC-11-eq.]	[kg C_2H_4-eq.]	[kg SO_2-eq.]	[kg PO_4^{3-}-eq.]
MA scenario A	187,783	0.03	44	1,132	229
SMA scenario A	187,380	0.03	44	1,129	229
tC / EAC scenario A	229,031	0.03	54	1,407	287
tC / EAC scenario B	214,285	0.03	51	1,260	254



16.14 Results of the impact assessment for the construction, maintenance and deconstruction of 1 km motorway: asphalt and concrete construction methods. Potential environmental impact from traffic is not included (tC = pavement with textured concrete surface; EAC = pavement with exposed aggregate concrete surface layer; MA = pavement with mastic asphalt wearing course, SMA = pavement with stone mastic asphalt wearing course).

higher with asphalt. In contrast, the potential impact of concrete construction methods for GWP is from 1.1 to 1.4 times higher than that of the asphalt construction methods.

16.6 Conclusions and future trends

The present study shows that the environmental impact due to the material processing for motorway constructions, their use by traffic and their maintenance can be reduced, while the potentials of the construction and deconstruction processes can be neglected due to their low influence on the overall results. The potential environmental impact can be significantly reduced by optimizing the production of the construction materials. In the case of concrete motorways, a reduction in the clinker content of the cement would reduce environmental impact by up to 21%. In the case of the asphalt motorways, optimized production reduces environmental impact by up to 3%. Further reduction potentials are the use of secondary fuels and the increased reuse of reclaimed asphalt. The evaluation of a service period of 30 years shows that durable construction methods and roads with low maintenance requirements offer significant advantages. The potential environmental impact due to traffic load is 100 times more than due to construction, deconstruction and maintenance together – the largest and most effective reduction in impact is possible here. These findings are in good agreement with the literature (Chester and Horvath, 2009; Stripple, 2001; Wang *et al.*, 2012). Numerous studies have already shown the effect of pavement surface structure on fuel consumption. A reduction in fuel consumption of about 10% could be achieved by the improvement of pavement surface texture or evenness as well as pavement stiffness. Further investigations and measures on pavement optimization would lead to more effective reduction of the environmental impact of roads. A reduction of fuel consumption of 0.5% over a service period of 30 years and for a 1 km motorway section would reduce CO_2 emissions by 1,154 t CO_2-eq. A reduction of fuel consumption by 2% would lead to a reduction in CO_2 emissions (GWP) well above the impact of motorway construction and maintenance together. A reduction of 10% fuel consumption for just heavy goods vehicles would save 10,760 t CO_2-eq. Thus, construction methods aimed at lowering fuel consumption are far more eco-effective than construction methods tailored to low impact during construction and use. Reliable knowledge on how pavements interact with the environment needs to be expanded and included in pavement LCA.

Pavement LCA is a valuable tool to support decision makers to integrate sustainability into the road sector. Therefore, a reliable model should be developed combining LCA and life cycle cost analysis (LCCA). All roads are unique. As a consequence, a flexible model is needed that can be adjusted to suit each specific situation. The model ideally should use high data quality to provide a sufficient decision aid. In order to compare different pavement types and maintenance strategies, the functional unit needs to be standardized and the scope of the investigations expanded. Furthermore, the positive effect of pavement properties on fuel consumption needs to be taken into account. The impact is significant because it immediately affects every vehicle using the pavement, while vehicle industry measures can take a long time to fully prevail in the market. Therefore, further research is also needed to establish standardized test methods and reliable models that determine the influence of pavement properties on fuel consumption with certainty. This knowledge needs to be included in pavement management systems to optimize maintenance and rehabilitation strategies that support both economic and environmental goals.

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16.9 Appendix: abbreviations

AP	acidification potential
CSOL	cracking, seating and overlaying
EAC	constructions with an exposed aggregate concrete surface layer

EP	eutrophication potential
GHG	greenhouse gases
GWP	global warming potential
HMA	hot mix asphalt
LCA	life cycle assessment
MA	pavements with mastic asphalt wearing course
ODP	ozone depletion potential
PCC	Portland cement concrete
POCP	photochemical ozone creation potential
SMA	pavements with a stone mastic asphalt wearing course
tC	constructions with surface texture produced by brushing

17

Comparing the environmental impact of reinforced concrete and wooden structures

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Abstract: The topic of this chapter is the confrontation between two different construction technologies through life cycle assessment methods: wood and concrete. Nowadays, cradle-to-gate and cradle to cradle LCA analyses are used successfully in sustainable building design and construction. Our research is focused on the specific European context, trying to enhance the usefulness of the methods from a design perspective, if system boundaries are carefully defined and data sets accurately selected and adapted. According to LCA analysis, wood structures demonstrate an overall better environmental performance, but distinctions and contradictory results emerge in different categories of impact.

Key words: LCA methods, wood construction, reinforced concrete construction, system boundaries, cradle-to-gate.

17.1 Introduction

The usage of materials in the building sector should always be addressed in relation to the specific cultural and economic context in which technologies are applied. Despite globalization, and not considering some distinctive buildings – e.g., commercial buildings, skyscrapers, airports, where the selected construction technologies are related to innovative processes and linked to particular contracts – the construction industry is still a local activity, and follows the cultural peculiarities of the region. This is particularly true for small and mid-sized residential buildings. Therefore, when talking about building technology, the first distinction to be made is between advanced and developing countries, given the different production processes and the level of environmental awareness that is achieved in the different contexts. This chapter focuses on building processes applied to technologically advanced regions in Western Europe, North America and Australia.

A second issue is who evaluates the environmental impact of building technology. Manufacturers tend to promote their materials and, consequently, enhance the environmental qualities of their products. This is particularly true for two potentially competing materials like wood and concrete. Public institutions and research groups should maintain a neutral position, but this is not always the case. Who undertakes an environmental impact analysis needs to be carefully considered.

The application of life cycle assessment (LCA) methods to the built environment has grown because they provide a quantitative basis for environmentally improved designs, removing guesswork, burden shifting and greenwashing, the latter being a common form of spin in which green marketing is deceptively used to promote the perception that products are environmentally friendly when they may not be. Our comparative study is particularly significant in Southern Europe where so-called dry solutions (i.e., using wood as a building material) are slowly replacing some traditional building technologies, not only for small houses, but also for mid-sized buildings, even though the market for wooden houses is still very small compared to the USA or Canada. The percentage share of wood construction is variable (Gustavsson and Sathre, 2006): USA 90–94%, Canada 76–85%, Nordic countries 80–85%, Scotland 60%, UK 20%, Germany 10%, The Netherlands 6–7%, France 4%. According to our evaluations, the share in Italy is between 3 and 4%.

The Italian market for wood construction is particularly strong in the region of South Tyrol (formerly part of Austria), where a good percentage of the new residential stock is now represented by wood houses, keys in hand with environmental certifications (Klimahaus). In the Abruzzo region, after the earthquake of 2009, the contracts for new ready-to-use units required a certain level of sustainability and construction cost reduction: this circumstance led to the construction of a large number of wooden structures, especially X-Lam structures, and combined wood–concrete systems. The same process is now happening after the 2012 earthquake in Emilia-Romagna, where for quick reconstruction and an eco-friendly approach, dry technologies are recognized as being more convincing solutions than traditional technologies, e.g. masonry.

Wood construction for residential buildings can be roughly divided into light framed structures (balloon-frame, platform frame) and load-bearing wall structures (block-houses, X-Lam) (Fig. 17.1). Traditional concrete structures for buildings vary depending on countries; in seismic regions – e.g. in Italy – framed structures (pillars and beams) with a significant number of shear walls are usually employed (Fig. 17.2), but there are many situations in which load-bearing walls (concrete blocks or poured on site structures) are preferred.

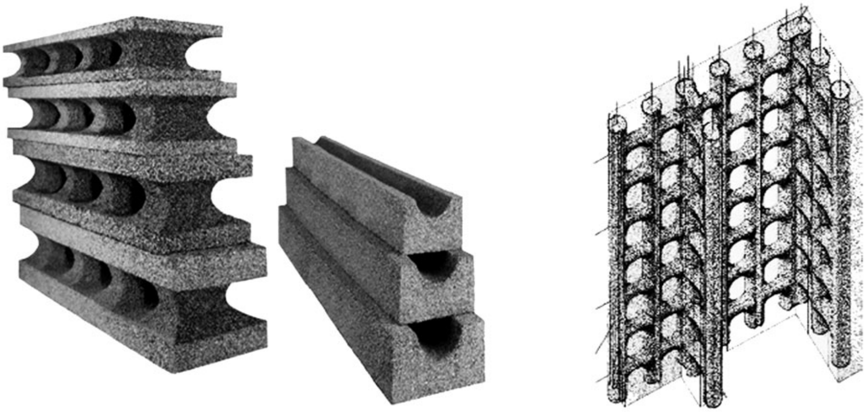
17.2 Environmental strengths and weaknesses of using wood and concrete in construction

According to Portland cement manufacturers, concrete is one of the single most environmentally friendly construction products available, offering

17.1 Construction of a concrete-framed building in the European context. Residential building in Parma, Italy, 2006.

17.2 Construction of an X-Lam building in the European context. Residential building near Verona, Italy, 2010.

durability for the residential marketplace and environmental advantages through every stage of manufacturing and use. For the US company Rastra Engineering, which pioneered high-quality building systems from recycled foam plastics and insulated concrete form (ICF), one of the most extensively tested forms available today – approved by the International Code Council (ICC) under Legacy Report ER-4203, ER-9955, and classified by



17.3 Insulated concrete forms. From Rastra.

UL under design U915 – concrete is environmentally less impactful than wood (Fig. 17.3).

The first justification of this statement, based on internal research, is that US wood, once cheap and plentiful, has recently seen an increase in imported lumber and in levels of CO_2 being put into the atmosphere. In contrast, Portland cement manufacturers have made extensive efforts to reduce the levels of CO_2 released during manufacture, progressively reducing CO_2 emissions. Besides, Portland cement production is relatively localized to a few square kilometers of quarries and mills, whereas lumber operations are far more widespread and still require considerable energy. In addition, lumber operations decrease the biodiversity of forests and may create erosion and pollution problems.

The second justification concerns the life expectancy of the house. Wood suffers from several major disadvantages: it is subject to rot, mold, insects such as termites, and is very flammable: fire departments respond to 1.5 million home fires in the US per year, many resulting in a complete loss. Wood buildings last 50–100 years, and usually require continuous replacement of deteriorated sections. On top of that, there is inherent waste in wood construction practices: most lumber gets cut shorter after it arrives at the site.

A concrete house, instead, has a much longer service life than a wood frame house, lessening the need to spend energy on reconstruction. Besides, waste from concrete structures can be recycled and reused almost indefinitely. In Europe, where the tradition is still to build in brick and concrete, the housing stock turnover is far less than in US; as a consequence, a lower percentage of gross national product (GNP) is devoted to housing.

Table 17.1 Environmental impact of a wood house, compared to a concrete block house

	Wood	Insulated concrete form
House size	2,400 sf (223 m^2)	2,400 sf (223 m^2)
Material required to construct walls	Lumber: 750 ft^3 (21.24 m^3) Acres of mature forest harvested: 2.3	Cement: 23,558 lbs (10,708 kg)
CO_2 production	CO_2 removed per acre per year: 5,200 lbs	CO_2 produced per pound of cement: 0.5 lbs
Total CO_2, Year 1	CO_2 removed: 11,818 lbs (5,372 kg) Total oxygen not created per year: 8,602 lbs	CO_2 produced: 11,779 lbs (5,354 kg)

Source: Rastra (Rastra Engineering, Compound ICF).

Finally, the analysis carried out by Rastra focuses on the negative impact of harvesting a mature forest to build the structure of a wood house versus the amount of CO_2 created in the production of enough concrete to build the walls of the same house (250–300 kg/m^2 of grouted unfinished wall). When forests are cut down to produce lumber reducing its quantity, the carbon is not gradually incorporated into the soil, as in a normally decaying forest. For a 220 m^2 wood house, it takes approximately 21 m^3 of wood to build only the walls of the house, and requires 0.93 hectares to produce the equivalent amount of wood (Table 17.1). This 0.93 hectares of mature forest would otherwise remove 5,353 kg of CO_2 per year from the atmosphere (5,756 kg/hectare).

The equivalent 220 m^2 concrete house requires 10,671 kg of Portland cement to construct the walls. During the manufacture of the required amount of Portland cement, 5,336 kg of CO_2 are released into the atmosphere – resulting in slight improvement at the end of the first year, while the loss of the cumulative impact of wood in reducing CO_2 from the atmosphere over the 50-year growth cycle of a new forest is 121,993 kg. If trees take 50 years to mature, the return on investment from using concrete is only 2,290% by the time the re-planted forest fully matures.

This analysis requires a clearer definition of the system boundaries and more transparent data sets: the reuse of wood from lumbering is not taken into account, whereas the loss of CO_2 savings equivalent to the time spent in fully regenerating the forest is not properly considered. It is also unclear how IFC blocks are considered. However, this example shows how data input can lead to subjective interpretations of environmental impacts: producers communicate the most favorable information to promote the quality of their products.

According to the National Association of Home Builders, the following are concrete's primary environmental strengths:

1. It is created from an abundance of raw materials. Portland cement, which makes up about 12% of concrete, is manufactured from limestone, clay and sand. Sources of aggregates used to make it – sand gravel and crushed stone – are plentiful. In addition, aggregate can contain recycled materials such as slag, a by-product of steel manufacturing. When using Portland cement, a portion of it can be replaced with fly ash, a by-product of coal-burning power plants, and similar materials.
2. The fuel to produce concrete increasingly comes from the waste stream. High-energy wastes such as old tires can be used as fuel in the Portland cement-making process. The process of making it also can use recycled materials.
3. Local production reduces transportation costs and fuel cement and concrete supplies are highly local or regional. At least 60% of all concrete is produced within 160 km of the construction site. Wood and steel products, on the other hand, typically have to be transported hundreds or, sometimes, a thousand kilometers or more to the job site.
4. Concrete's thermal mass yields energy savings. Many of today's concrete wall systems combine the mass of concrete with foam insulation, creating exterior wall envelopes that minimize heating and cooling costs through thermal mass, reduced air infiltration and increased thermal resistance. During the air-conditioning season, a concrete building often requires only the cooling system to be in operation at night.
5. Concrete construction requires fewer kinds of building products, such as sheathing and insulation, than wood-frame construction. Concrete is also created as needed, eliminating the waste inherent in modular products. Old concrete can be ground up and reused as coarse aggregate or pavement sub-base material.
6. Concrete does not rust, rot or burn, so housing stock built with concrete components such as wall systems can stand for generations. Concrete is less susceptible to moisture damage and can breathe. In addition, homes built with concrete also are more likely to withstand natural disasters such as hurricanes, tornados and fires than traditional wood-frame housing stock.

Although some of these statements need qualification, e.g. the fact that concrete does not rust (the reinforcing steel bars inside do), these strength factors favor the market for concrete buildings.

In contrast with these interpretations, the common assumption that wood is a better green material than concrete is primarily based on its embodied energy. The embodied energy of a material can be taken as the total primary energy consumed, or carbon released, over its life cycle.

Table 17.2 A comparative embodied energy analysis of timber, steel and concrete

Selected source	Embodied energy MJ/kg		
	Steel	Timber	Concrete
Alcorn (2003)	35.9	0.3–24.2	0.81–2
Eaton and Amato (1998)	31	13–36	0.84–1.36
Franklin Associates (1991)	44.6	14.9	–
West *et al.* (1994)	32	5.7–10	–
Berge (2003)	25	3–16	1
All database sources	6–81.8	0.3–61.3	0.07–23.9

From Hammond and Jones, 2008.

Ideally the boundaries would be set from the extraction of raw materials until the end of the product's lifetime, including energy from manufacturing, transport, energy to manufacture capital equipment, heating and lighting of factory, maintenance, disposal, etc. It has become common practice to specify the embodied energy as cradle-to-gate, which includes all energy until the product leaves the factory gate. More precisely, for construction materials the final boundary condition should be considered cradle-to-site, including the energy flows until the product has reached the point of use.

Unfortunately, the embodied energy in the cradle-to-site cycle is very variable, and always needs to be adapted to the context. A comparative embodied energy analysis for timber, steel and concrete, is introduced by Hammond and Jones (2008), using data extracted from the literature (Table 17.2). The full range of data from 250 collected sources display a large scatter for all three materials. Besides, the embodied energy for a wood house varies not only depending on the amount of material employed (light frame or bearing wall construction), but also on the type of wood. According to Randall (1999), as far as the embodied energy is concerned, the production of a standard framed wood structure for a detached house with three bedrooms requires 7,450 kWh, while the construction of the equivalent masonry structure with concrete blocks requires 12,816 kWh, which is 1.7 times higher. These data are reliable for traditional platform-frame homes: results could be slightly different for innovative designs. According to our calculations, the amount of structural wood or reinforced concrete does not differ very much in framed structures for mid-sized buildings, varying from 15 to 20 m^3 per 100 m^2 of floor area. The smaller number of concrete pillars, slender walls and slabs is compensated by their higher density, leading to a weight conversion factor, from wood to concrete, of 1:5, which differs significantly from the results of other authors (Nässén *et al*., 2012). This is probably caused by different construction technologies used in southern regions. In the other direction, bigger weight is compensated by the lower amount of embodied energy (cradle-to-site) for the production of concrete floor slabs, columns and load-bearing structures (Table 17.3).

Therefore, the real difference in the embodied energy of a building does not lie in its structure, but in the other elements, especially cladding walls. In concrete houses, cladding walls are usually made of concrete or clay blocks, whereas wood houses use light insulating materials. In fact, this is the simple reason why traditionally the embodied energy in wood houses is lower: they are light buildings. Nevertheless, there are recent examples of contemporary housing units in Europe, where the concrete structure is completed with wood cladding elements, or, conversely, wood structures are filled with high density blocks. Obviously, these characteristics lead to totally different calculations and results.

Aside from the embodied energy in singular buildings, many authors point out the convenience of using wood in countries where the production chain is already particularly strong, focusing on the side effects of its use. Pingoud *et al.* (2003) estimated the maximum wood substitution potential in new building construction in Finland, computing the total amount of materials used during one year in 11 main building parts in new construction of 9 main building types. The results indicated that nearly twice as much wood material could have been used in Finland in 1990 compared to the amount that was used at that time. Each kg of additional wood material used would have reduced the use of masonry materials by 3.6 kg and the use of metals by 0.1 kg. With this substitution, including the displacement of fossil fuels by wood residues from sawmills and construction sites, fossil CO_2 emissions would have been reduced by 1.8% of total Finnish CO_2 emissions. From a climate change mitigation perspective, it would have been beneficial to choose the less energy-intensive wood products to fulfill a given service demand.

A well-known study conducted in Sweden and Finland (Börjesson and Gustavsson, 2000) compared the net CO_2 emissions from the construction of concrete and wood-framed buildings. Two buildings were evaluated: the Wälludden project in Sweden, a four-story building containing 16 apartments with a total usable floor area of 1,190 m^2, and a building in Helsinki, a four-story apartment block built in 1997 in the ecological building area of Viikki.

Carbon accounting included emissions due to fossil fuel use in the production of building materials, the replacement of fossil fuels by biomass residues from logging, wood processing, construction and demolition, carbon stock changes in forests and buildings, and Portland cement process reactions. The results showed that wood-framed construction required less

Table 17.3 Selected database of embodied energy and carbon coefficients from the survey, Inventory of Carbon and Energy (ICE) V2.0, compiled by Hammond and Jones in 2011

Material	Energy MJ/kg	Carbon kg CO_2/kg	Density kg/m^3
Aggregate	0.083	0.0048	2,240
Concrete (1:1.5:3 e.g. *in-situ* floor slabs, structure)	1.11	0.159	2,400
Concrete (e.g., *in-situ* floor slabs) with 25% PFA RC40	0.97	0.132	–
Concrete (e.g., *in-situ* floor slabs) with 50% GGBS RC40	0.88	0.101	–
Bricks (common)	3.0	0.24	1,700
Concrete block (medium density 10 N/mm^2)	0.67	0.073	1,450
Aerated block	3.50	0.30	750
Rammed earth (no cement content)	0.45	0.023	1,460
Limestone block	0.85	–	2,180
Cement mortar (1:3)	1.33	0.208	–
Steel (general – average recycled content)	20.10	1.37	7,800
Steel (section – average recycled content)	21.50	1.42	7,800
Steel (pipe – average recycled content)	19.80	1.37	7,800
Stainless steel	56.70	6.15	7,850
Timber (general – excludes sequestration)	10.00	0.72	480–720
Glue laminated timber	12.00	0.87	–
Sawn hardwood	10.40	0.86	700–800
Woodwool board insulation	20.00	0.98	–
Wool (recycled) insulation	20.90	–	25
Clay tile	6.50	0.45	1,900
Aluminium (general and includes 33% recycled)	155	8.24	2,700
Bitumen (general)	51	0.38–0.43	–
Hardboard	16.00	1.05	600–1,000
MDF	11.00	0.72	680–760
OSB	15.00	0.96	640
Plywood	15.00	1.07	540–700
Plasterboard	6.75	0.38	800
Iron (general)	25	1.91	7,870

Full detailed survey, complete with original data, methodology and notes, available from www.bath.ac.uk/mech-eng/sert/embodied/

energy, and emitted less CO_2 to the atmosphere, than concrete-framed construction. The lifecycle emission difference between the wood-framed and concrete-framed buildings ranged from 30 to 133 kg C per m^2 of floor area. According to the authors, a net reduction of CO_2 emission could be obtained by increasing the proportion of wood-based building materials, relative to concrete materials. The benefits would be greatest if the biomass residues resulting from the production of the wood building materials were fully used in energy supply systems. Scharai-Rad and Welling (2002) also considered the utilization of demolition wood to replace fossil fuels, and found the environmental performance, including net greenhouse gas emissions, of buildings to be more favorable as the volume of recovered wood increased.

Finally, Lenzen and Treloar (2002) analyzed the wood and concrete designs of the Wälludden building described by Börjesson and Gustavsson (2000) in terms of their embodied energy, employing an environmentally extended input–output framework in a tiered hybrid life cycle assessment, and in a structural path analysis. They illustrated the complexity of the inter-industry supply chains underlying the upstream energy requirements for the building options, and demonstrated that higher order inputs are difficult to capture in a conventional process analysis. Their calculations showed that Börjesson and Gustavsson's estimates of energy requirements and greenhouse gas emissions were underestimated by a factor of about 2, and that corresponding greenhouse gas balances were positive at about 30 t C-eq. Nevertheless, they confirmed Börjesson and Gustavsson's general result, which is that emissions from concrete-framed buildings are higher.

The Consortium for Research on Renewable Industrial Materials (CORRIM) also found concrete-framed houses to use 16–17% more total energy than equivalent wood-framed houses. Because much of the energy for the wood-framed house is produced internally from wood processing, the concrete-framed houses used 2.5 and 2.8 times more non-bioenergy, respectively. CO_2 emissions were lower for the wood-framed houses due to carbon storage in wood products and reduced use of fossil fuels (Lippke *et al*., 2004).

Unfortunately, these kinds of studies apply to Northern Europe; there are no such studies in the Southern Europe context, where wood for construction is mostly imported and the beneficial effects of its use should take into consideration not only the regions of product destination but also the regions of origin.

Therefore, solid waste management appears to be the key aspect for wood's best environmental performance in the building industry, if accomplished within clear boundaries. There are recent initiatives in the US to use that waste. In North Carolina, a 10 MW landfill gas and wood waste biomass electric generating facility will be located at the Columbus County landfill at New Hope and could begin operations as early as December 2013. According to Tom Koch, member of CRE and working as a developer of the project, the operation utilizes about 150,000 tons of wood waste per year, operating 24 hours a day. A study by North Carolina State University suggests that there is more than a million tons of waste wood, including fallen trees and wood from the thinning of forests, available annually within a 20 mile radius of the landfill, seven times higher than the demands of the plant.

In conclusion, these are the observed strengths and positive side effects of using wood in construction:

1. more intensive forest management;
2. improved process efficiencies and greater use of engineered products that utilize less desirable species;
3. increased product durability through improved products;
4. recycling of demolition wastes;
5. greater use of low value wood fiber for bio-energy;
6. environmental pollution control improvements that consider LCI/LCA.

17.3 Life cycle assessment (LCA) for wood and concrete building design

Several assessment systems are needed while designing a house with a complex list of sustainability objectives. LCA is one of them, being science-based, quantitative, and integrative (O'Connor *et al*., 2012). It measures the material and energy flows to and from nature over a lifetime, and assesses the potential impact of those flows on resources, ecosystems, and human health. The assessment is usually referred to as a cradle-to-grave or cradle-to-gate evaluation. Commonly reported impact metrics include global warming (carbon footprint), acidification (acid rain), eutrophication (algal bloom), photochemical oxidant creation (summer smog), and ozone depletion (ozone hole).

Unfortunately, LCA is a comprehensive assessment process, not intended to accomplish everything on the sustainability agenda. For instance, it does not address the proper evaluation of risks, emission or exposure limits, indoor air quality or resource management. Moreover, there is always some uncertainty when evaluating life cycle environmental impacts across the complex and widespread value chain of a building, due to the difficulty of predicting future states and the impracticality of having all input data at the same level of quality. LCA should rather be looked at as a tool for estimating potential environmental impacts and comparing relative performance of alternatives. On the other hand, LCA results rarely point in the wrong direction. Of course, there is a need to produce accurate local data sets with the possibility to convert the results to an internationally comparable form.

A recent study, partly conducted under the EU LoRe-LCA project, used a LCA approach to compare the environmental impact of producing materials commonly used in construction, in addition to some more environmentally friendly alternatives. The researchers adopted simplified versions of ISO 14040:2006 and ISO 14044:2006, considering three of the impact categories which exemplify key energy and environmental problems in Europe in relation to the achievement of the EU's 20-20-20 targets: the primary energy requirements related to the production, use and disposal stages of a product, the impact on global warming, measured in kilograms CO_2-equivalents, and the water demand. The researchers assessed these impacts for 1 kg of each material. The relative quality of a material with different physical properties is based on its ability to perform a function: different weights of different materials may be able to meet the same need.

The energy-intensive manufacture of clinker, for instance, is a major contributor to the environmental impact of Portland cement products used in buildings. Switching to renewable sources of energy and improving technologies by making better use of the waste heat from the furnace or reducing the furnace temperature, could halve the emissions of CO_2 from cement manufacture by 2050. Constructing buildings with wooden structures would also lower the primary energy demand and could be almost carbon neutral, or even carbon negative if the wood was recycled and reused at the end of life.

The LoRe-LCA working group pointed out that the reuse of materials in construction, reducing the primary production, is one of the key aspects to be considered for quality evaluation (Bribián *et al.*, 2011). The construction of buildings that can be disassembled rather than demolished at the end of their life makes it easier to separate materials for reuse and recycling (e.g., bolts from wood structures instead of adhesives). Moreover, upgrading production technologies and using local resources could also limit environmental impacts. Manufacturers are urged to use EPDs (environmental product declarations – ISO Type III ecolabels) that provide standardized information based on LCA. In that regard, wood and concrete products will compete in cradle-to-cradle confrontations.

However, as stated above, LCA from cradle to cradle is extremely complicated for every kind of structure, especially for wooden ones. Buildings, which account for the largest use of wood are often unique in their size, complexity, and longevity. A residential house is built, used for a long period of time, and eventually demolished and recycled. The period of use and occupancy involves cycles of maintenance and repair (e.g., re-roofing) and a series of owners each of whom may remodel the structure to accommodate changes in desired functionality and aesthetics. As a result, the time frame between when a tree seed germinates and when a wood house is demolished could be on the order of centuries. While the structure is quite unchangeable, the other parts of the buildings can change significantly over time. Thus, the temporal distribution of events and associated environmental effects during the seed to demolition life cycle must be considered.

The Athena Institute is well known for developing LCA data on construction materials and systems and making these data accessible to building designers through user-friendly tools such as the Athena Environmental Impact Estimator (EIE) for buildings. The EIE model provides LCI measures based on the bill of materials developed for house designs.

The Canadian Wood Council used the Athena software to conduct a life cycle analysis comparison of similar wood, steel and concrete buildings. The results were published in the bulletins 'Comparing the Environmental Effects of Building Systems', and 'Life Cycle Analysis for Residential Buildings'. Wood design resulted better in the categories 'embodied energy', 'global warming potential', 'air and water toxicity' and 'weighted resource use' by 50% and in the category 'solid wastes' by 20%. The Athena Institute stressed the absolute fundamental idea of having consistent, comparable and comprehensive life cycle inventory data for individual industries and building products.

As we mentioned before, CORRIM also published the results of a study of the environmental performance of wood as a building material (Table 17.4). Typical building designs were used to construct hypothetical homes, then compared the environmental benefits of wood-framed versus steel-framed houses in a cold climate (Minneapolis, MN) and wood versus concrete in a warm, humid climate (Atlanta, GA). The study looked at environmental effects across the entire life cycle (Table 17.5), showing that using wood products instead of steel or concrete could further reduce greenhouse gas emissions from fossil fuels because more than half the energy used by wood mills comes from biomass – bark, sawdust and other residuals – a renewable source of energy.

The Portland Cement Association presented different results of an LCA conducted according to the guidelines ISO 14044 on a house modeled with two types of exterior walls: a wood-framed wall and a concrete masonry unit (CMU) wall (Marceau and VanGeem, 2008). The house, a two-story single-family building with a contemporary design, was modeled in five cities, representing a range of US climates: Lake Charles, Tucson, St. Louis, Denver, and Minneapolis. The house system boundary included the inputs and outputs of energy and material from construction, occupancy, maintenance, demolition, and disposal. The system boundary excluded capital goods, human labor, impacts caused by people, and waste treatment after

Table 17.4 The CORRIM research framework

Generic LCA model	CORRIM	Comments
	Forest growth Time frame: 25–100 + years	Nursery, planting, thinning, fertilizing, during the growth cycle. Effects on carbon sequestration/ global warming, diversity.
Raw material acquisition	Harvesting Time frame: <1 year	Logging during commercial thinning or final harvest. Effects on soil compaction and productivity, diversity, habitat, siltation, etc.
Manufacturing	Manufacturing processes Time frame: <1 year	Individual products (lumber, plywood, LVL, OSB, etc.). Assemblies of products (trusses, Glulam beams, I-joists, etc.). Effects on air and water emissions, solid waste.
	Construction of structures Time frame: <1 year	On-site or factory built components (floor, wall, roof) and finished structure. Effects on solid waste.
Use/reuse/ maintenance	Service life and use Time frame: 40–100 + years House life: 75 + years	Maintenance cycles (painting, reroofing, siding, etc.) and remodeling. Effects on energy use and associated emissions/waste, energy and emissions associated with repair/remodel products.
Recycle/waste management	Recycling and disposal Time frame: <1 year	Teardown, segregation of materials, recycle, combust for energy, landfill. Effects on energy use and substitution, air and water emissions, solid waste/carbon sequestration.

From Perez-Garcia *et al.*, 2005a,b.

disposal. The life of the houses was set to 100 years. The analysis was carried out by first assembling the relevant LCI data from published reports and commercially available databases. The LCA software tool SimaPro was used to perform the life cycle impact assessment, choosing the methods Eco-Indicator 99 (Dutch/Swiss), EDIP/UMIP 97 (Danish), and EPS 2000 (Swedish). Furthermore, three different weighting sets in Eco-Indicator 99 were used.

Table 17.5 Environmental performance indices for Athena construction

Atlanta design	Wood	Concrete	Difference	% Change
Embodied energy (GJ)	398	461	63	16%
Global warming potential (CO_2 kg)	21,367	28,004	6,637	31%
Air emission index (index scale)	4,893	6,007	1,114	23%
Water emission index (index scale)	7	7	0	0%
Solid waste (total kg)	7,442	11,269	3,827	51%

From CORRIM, The Consortium for Research on Renewable Industrial Materials, (CORRIM Fact sheet 5).

The data showed that on average the impact indicators in each category were similar for the wood and CMU houses. The most significant environmental impacts resulted from the production of electricity and natural gas and the use of electricity and natural gas in the houses by the occupants. Furthermore, the largest impacts from these uses were in the form of depletion of fossil fuel reserves (categorized as damage to natural resources) and release to the air of respiratory inorganics (categorized as damage to human health). The household use of electricity and natural gas represented approximately 97% of the negative impacts in both solutions.

Finally, a study carried out by Beijing University of Technology (BJUT) came to the opposite conclusion, showing the importance of the energy embodied in materials during the production phase, even compared with the long-term operating energy consumption. They compared the life cycle performance of similar wood, steel and concrete apartment buildings over a 50-year service life. The study focused on the production and in-use stages. The LCA considered all energy and material flows from the environment, as well as emissions to air, water and ground from the three building designs. The findings showed that the wood frame construction was about 25% more energy-efficient than either the steel or concrete frame designs across the overall life cycle. They highlighted the importance of improving building insulation levels, airtightness and other energy conservation measures to level the performances: as buildings become increasingly energy efficient, the energy required to create them becomes proportionately more significant in relation to that required to run them.

As shown by the last three case studies, the results of LCA analysis for buildings can point in significantly different directions. In order to reach persuasive information, methodology, boundary definition, and data inputs should be made very clear in all stages of the LCA process.

17.4 Using LCA to compare concrete and wood construction: a case study

Our case study is based on an innovative green residential building, designed by our working team for an architecture competition in 2009, and meant to be adaptable to different areas in Southern Europe. The units are made by the aggregation of two 3.6 × 5.4 m basic modules containing the kitchen, the living-room, and the bathroom. Other modules can be added to the basic elements according to functional needs (Fig. 17.4). This means being enlargeable and modifiable in terms of internal distribution and services. Internal and external finishes can also be customized according to the context and the taste of the user.

The construction system is a platform frame structure with joist beams. The I-joists are I-shaped wooden beams, made by two laminated wood wings glued to a core made of OSB or plywood or high-density particles. The beams use the same generating process of IPE sections, optimizing the material for its bending strength. The structure is verified for seismic loads.

The structure that was selected for the competition, named StandardED, represents a standard solution for residential units in the Italian context. The dimensions of the concrete elements are taken from standard designs according to the Eurocodes. The external walls are made of clay blocks, according to the current practice. The solutions for the envelopes are selected in order to have the same energy and acoustic performances (U-value, thermal inertia, phonoinsulating power) for both buildings; on top of that, ordinary maintenance operations were considered equivalent. In such a way the energy consumption during the usage phase of the building would be equivalent.

In order to simplify the comparative study, a functional unit, structured on three levels, was extracted from the building; the total floor area is 230 m^2, (the ground floor is 95 m^2, the first floor is 77 m^2, and the second floor 58 m^2) and the total height is 9.5 m.

17.4 Perspective view of the Joi building and its structural organization, 2009.

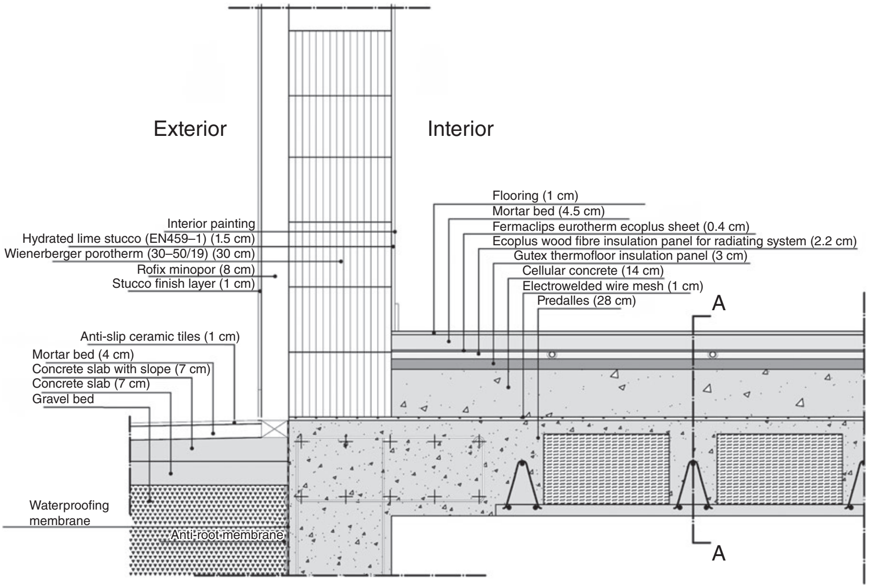
For the inventory phase, only the contribution of the construction without finishes was taken into account. In order to focus on the different environmental charges, floor and wall surfaces, stucco work and roof surfaces, were considered equivalent in both structures and, therefore, excluded. In both designs the analyzed elements were, finally: foundations, structures, floors without finishes, cladding and dividing walls without interior finishes.

The transportation distance to the authorized landfill was assumed to be 50 km. An average incidence of 100 kg/m^3 was considered for the steel bars of the concrete foundations and an average incidence of 200 kg/m^3 for the steel bars embedded in piers and beams. The reuse of scaffolding material before disposal was assumed to be 10 times. For thermal insulation, polystyrene panels were considered in the StandardED building (Figs 17.5 and 17.6), and cellulose panels in the Joi building (Figs 17.7 and 17.8).

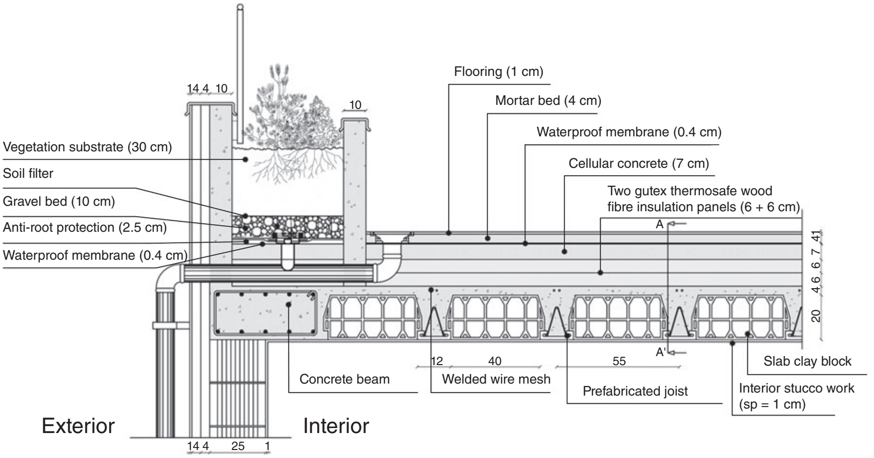
The transportation distance was:

- 100 km for concrete, low cement concrete, mortar, clay blocks and steel;
- 300 km for OSB panels and C24 wood panels;
- 200 km for coatings and thermal and acoustic insulation,
- 600 km for I-joist sections from STEICO.

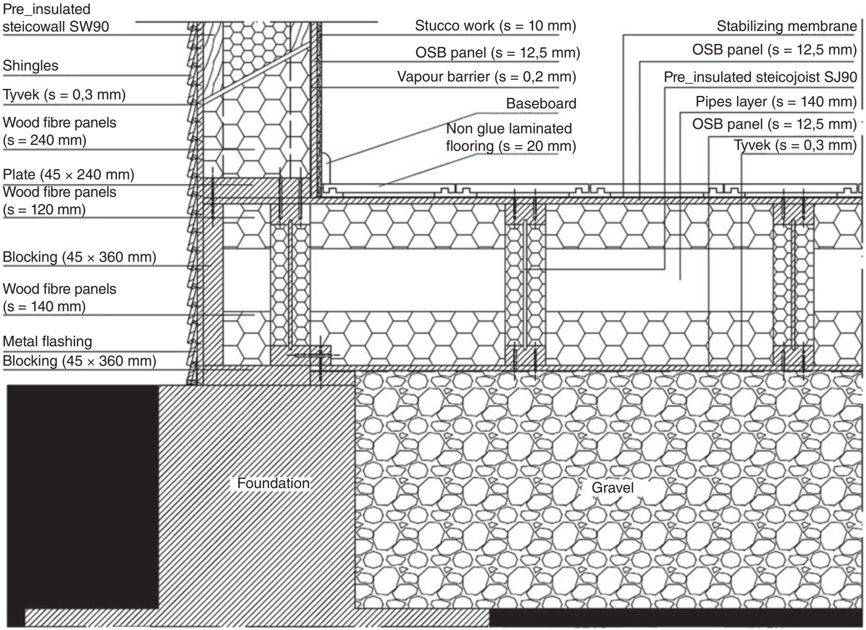
Electricity and oil consumption for StandardED were derived from a standard Italian residential building of the same size. Taking data from similar buildings, 4 months were considered for the construction phase of



17.5 Construction detail of StandardED building.

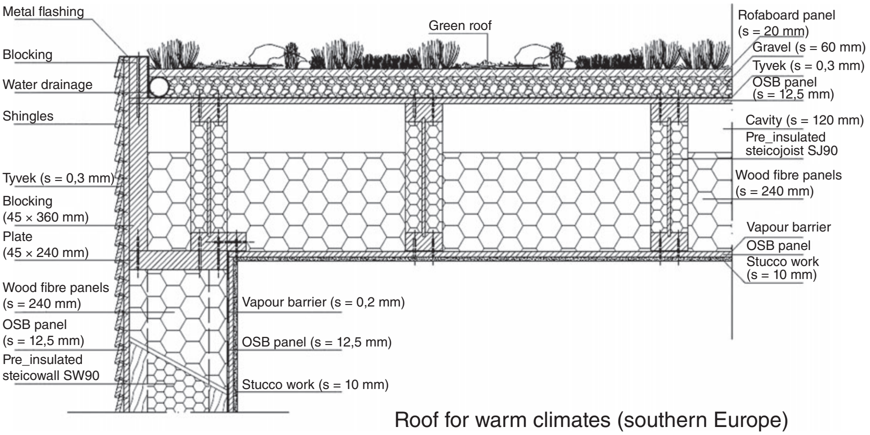


17.6 Construction detail of StandardED building.



Ground floor for warm climates (southern Europe)

17.7 Construction detail of Joi building.



17.8 Construction detail of Joi building.

StandardED. Assessing many other construction sites, the construction period for the Joi building ended up being 45% of StandardED's required time. If these last considerations seem quite simplistic, it is also true that the environmental impact and the energy consumption during the construction stage represent only 5% of the impacts and consumptions of the entire construction process, with a small margin of error.

The StandardED and Joi buildings have significantly different weights: the former weighed about 500,000 kg, the latter only 160,000 kg.

17.5 Selection and adaptation of LCA tools

The open source software openLCA (GreeDeltaTD) was used for the inventory phase and Eco-indicator 99 for the following assessment phase. Eco-indicator 99 is valid for the European context, but needs to be adapted according to personal experience. It allows the results of an LCA to be aggregated in easily comprehensible and usable parameters, called eco-indicators. These indicators consider three categories of damage: human health, ecosystem quality and resources. Each damage category is, then, subdivided into impact categories.

Eco-indicator 99 has some limitations:

- the CO_2 emissions are considered only in the impact category 'climate change'; it is important to consider them also in the category 'ecosystem quality', because the climatic changes due to the global warming not only impact on humans, but also on vegetal and other animal species;
- the characterization of 'land use' is very strong compared to the other categories of impact in 'ecosystem quality', penalizing the agricultural

use of soil, compared to the same use as a consequence of construction;
- the damage due to iron emissions are not considered in the categories 'ecotoxicity' and 'carcinogens'; the emissions due to nitrogen and phosphorus flows are not considered in 'acidification/eutrophication'; the damage due to chemical oxygen demand (COD) and biological oxygen demand (BOD) are not considered in 'carcinogens';
- the evaluation that is given according to different cultural perspectives can be an advantage, but it is based on inquiries that are inevitably influenced by single opinions and personal interests;
- water, gravel, sand, uranium and silver are not considered limited materials;
- the method does not operate economic evaluations;
- a real evaluation of energy consumption, which in some cases is very representative of the damage, is missing.
- the method is applicable only in the European context.

In order to tackle the limitations of Eco-indicator 99, some modifications were introduced, as suggested by some authors (Neri, 2007). Waters were added to the impact category 'minerals', excluding superficial waters, to take into account the fact that a bigger consumption of water (unlimited substance) always requires a bigger quantity of energy for its extraction. The characterization factor is the energy surplus (year 1990) to extract a liter of water caused by groundwater lowering of 60 m, as an effect of an increase of water consumption by five times. For waters with unspecified provenance, the characterization factor was reduced by a factor of 0.4855, because ISTAT data revealed that in Italy the extracted water from the ground was 48.55% of the total.

Gravel, sand, uranium and silver were added in the category 'minerals', as fundamental substances for the production of building materials and energy. The substances nitrogen and phosphorus, COD and BOD were added to the impact category 'acidification/eutrophication', because they produce the eutrophication of water. Iron emissions were added in 'carcinogens'. The category 'costs', which utilizes the emission cost with characterization factor 1, was also added.

Furthermore, the method 'cumulative energy demand' was added to Eco-indicator 99. It takes into account the global embodied energy of a process, considering also the extracted primary matters and the equivalent MJ that they might have produced. In such a way, it is possible to correct the fact that in Eco-indicator 99 a real energy balance is not accomplished.

The evaluation of the environmental impact with Eco-indicator 99 was performed in the egalitarian version, which outlines a long-term vision of the environmental impact. This vision takes into account the category 'human health' by 30%, the category 'ecosystem quality' by 50%, and the category 'resources' by 20%.

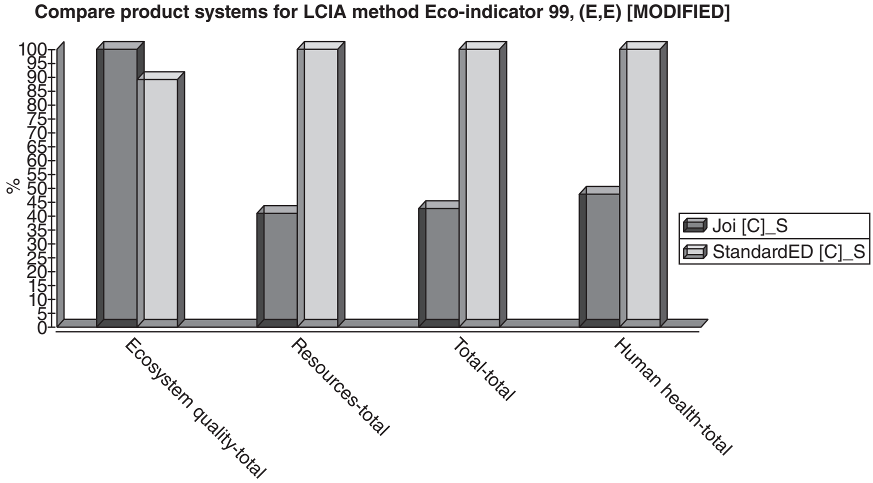
17.6 Life cycle impact assessment and interpretation

The diversity of wood structures in comparison to other materials and techniques in Europe, and the lack of an Italian database, as mentioned above, forced us to adapt the obtained data to the national context. The comparison between our study and the CORRIM research helped us to identify the production cycles that have a significant influence on the structural parts of the case study.

The following databases were tested: Ecoinvent, Swiss database, NREL, free US database by NREL, ELCD, and the database of the Joint Research Center of the European Community. These databases were imported in the software openLCA.

As far as the data used for the modeling is concerned, it was not possible to find a European production process for I-joist elements. Therefore, it was necessary to modify the composite wood I-joist processing, at plant, contained in the NREL database, and transfer it to the European context. After the sub-processes of I-joist production were analyzed according to the available data, a set of alternatives was evaluated, maintaining the same quantities and making some substitutions.

As to the results, Joi generally demonstrated a slightly bigger impact on 'ecosystem quality' (only 10%), balanced by considerably smaller impacts in the categories 'resources' and 'human health' (Fig. 17.9).



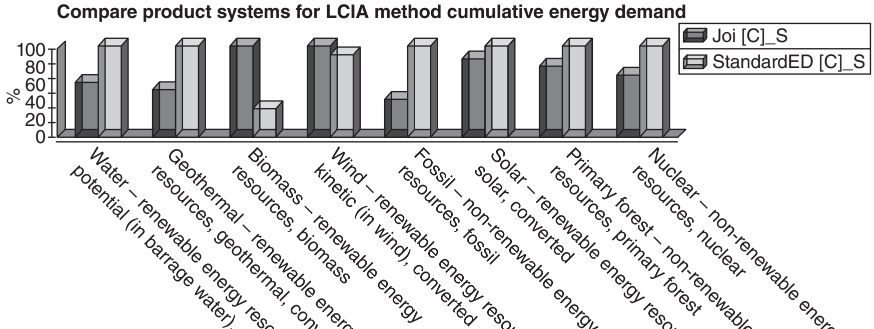
17.9 Global comparison of the damage categories.

The results of the category 'ecosystem quality' are explained by the fact that Eco-indicator 99 gives a strong characterization to 'land use', in respect to other impacts of the same damage category. Given the fact that the wood utilized for the realization of Joi is partly extracted from non-renewable forests and green areas from Eastern and Central Europe, the 'land use' category acquires a large importance in determining the impact on 'ecosystem quality'. The problem of land occupation for wood production does not concern the building sector, and it can be partially resolved with politics aiming at regenerating wood reserves. This aspect is definitively a key factor in LCA analysis.

For the other impact categories within 'ecosystem quality', Joi is approximately 50% more favorable; the same trend of 'ecosystem quality' is confirmed for the impact categories contained in 'human health'. The only comparable impact is relative to the emission of carcinogenic elements: Joi gives off cadmium and arsenic, coming from OSB and I-shape section fabrication, from the resins, from the treatment of wood and particulates, and from sawing. The same elements are present in the exit flows of concrete production.

Operating a first energy balance through the 'cumulative energy demand' method, which considers all energy contributions and not only the primary energy, the Joi building has a more favorable global energy balance (Fig. 17.10). StandardED is more convenient only for biomass energy, due to the quantity of wood in Joi, and for wind energy.

Breaking down the global process of construction into its three main parts (materials production, transportation, and construction), it is evident that the biggest contribution to environmental impacts derives from the production of materials. Transportation is, respectively, 60% and 70% of materials production; the energy during construction only 1–5%:



17.10 Comparison between StandardED and Joi energy balance.

- the production of materials for StandardED has an index of 17,500 eco-points, while the production of the materials for Joi has 6,700 eco-points;
- transportation for StandardED has been evaluated at 10,900 eco-points, transportation for Joi 4,800 eco-points;
- the energy demand for the construction site has an index of 390 eco-points for StandardED, and 195 eco-points for Joi.

Regarding the transportation processes, surprisingly wood elements are still more convenient than equivalent concrete elements even though distances are six times longer. As mentioned above, the biggest part of environmental charges and energy consumption is represented by the production phase. Globally, the production of materials for the platform frame solution is better evaluated by 60%. Resources savings are particularly favorable in material extraction savings (35%) and fossils (65%).

For the subcategories 'human health' and 'ecosystem quality' impacts and flow profiles are very similar in the whole realization process, with StandardED having a much bigger impact on the category 'land use', and similar impacts on 'carcinogenics'.

Joi's energy consumption is 45% in terms of hydroelectric energy and 170% in terms of biomass energy. The latter percentage comes from the massive use of wood and its derivatives, but it is not necessarily a weakness. On the other hand, the use of electric energy is approximately 80%, the use of fossil fuels is 45%, the use of solar energy is 80%, the use of wood from non-renewable resources (primary forest) is 70%, and the use of nuclear energy is 50%.

Analyzing the building components, relatively to 'ecosystem quality', wood floors do twice the damage of concrete slabs. As explained above, Eco-indicator 99 applies a strong characterization to the impact category 'land occupation', whose index is amplified by the large amount of wood used in the Joi building. Observing the other sub-categories, relative to 'ecotoxicity', 'acidification' and 'eutrophication', the traditional concrete technology is very impactful and not convenient compared to a wood structure. The correct management of wood resources would make wood floors even more convenient, balancing the problem related to 'land use'.

Relatively to the categories 'ecotoxicity', 'acidification/eutrophication' and 'stored ecotoxicity', most of the environmental charge comes from the resins embedded in common OSB panels, with some – albeit small – toxic effects. The substitution of the common products with less impactful but more expensive materials would obviously improve the design of the Joi building.

The foundations are the most impactful components on 'human health' for the dust and the carcinogen emissions during the preparation of concrete. This leads to the simple conclusion that reduced size of foundations always gives smaller environmental impacts.

Concerning the impact of the cladding walls, the incidence of Joi is 40–50% that of StandardED. For the category 'climate change', the charge of the proposed floor is 80%. On the other hand, the carcinogenic elements emissions are comparable in the two versions. In fact, the contributions of arsenic, cadmium and particulate are very similar in both designs. Considering the contributions of the building products, the biggest environmental damage comes from the use of common OSB panels.

With regard to the category 'resources', the overall charge of Joi cladding walls is 50% that of StandardED. Joi provokes a smaller damage in the sub-category 'mineral extraction' (20%) and for 'fossil fuels' (50%); the materials with the biggest impact are STEICO wall sections, SW90 sections, and OSB panels.

17.7 Future trends

According to Eco-indicator 99, the estimated environmental impact of the wood construction is generally smaller than the impact of the equivalent reinforced concrete structure. However, as evidenced by the method, the vast use of wood threatens the quality of the ecosystem. The use of low-quality OSB panels, like other similar low-cost materials, should be limited, as they are responsible for the damage to human health (carcinogenic agents), and to the ecosystem (acidification and eutrophication). In order to improve wood design, some alternatives should be evaluated, like high density panels or other recycled products.

According to our cost analysis, in accordance with CORRIM conclusions, wood structures are more convenient (17%); bigger savings should be obtained by speeding up the construction process. The transportation from distant places is definitely not a major problem, either environmentally or economically. The limitation of wood design in Southern Europe is primarily a cultural issue, not a consequence of material shortage.

From the designer perspective, we can introduce a short check-list of environmental actions, prior to the start of any design development or any LCA analysis aimed at making decision on materials. The following actions should be taken:

- Balance the quantity of material with the comfort expectations of the users; the utilization of a bigger quantity of materials is usually preferable in terms of thermal comfort, but generally brings bigger environmental impacts.
- Analyze the cultural background of the economic region where construction is going to take place.

- Check the origin of the products. An effective resources policy is the primary mandatory goal to favor the regeneration of the environment ('land use'), compatibly to the requirements of the building industry.
- Check energy consumption during product manufacturing and construction phase first (embodied energy). It is much lower than energy consumption during the usage phase.
- Check the impact of construction materials on ecosystems and human health. They are usually independent from the usage phase.

In conclusion, many materials can be used in buildings contemporarily, e.g. concrete and wood; the issue is not excluding one or another, rather, it is important to choose good products. For some materials, management is almost unpredictable, but material replacement (i.e., wood) should always be addressed. In addition, LCA boundaries should be made very clear. LCA analysis that takes into consideration the life expectation of buildings (cradle-to-cradle) is generally satisfying in terms of energy assessment, but often meaningless, especially if data sets are not accurate. Conversely, cradle-to-gate or cradle-to-site phases are usually sufficient to address the impacts on ecosystems.

17.8 Sources of further information and advice

- Athena Sustainable Materials Institute, Athena EcoCalculator for assemblies, 2011. Available from: http://www.athenasmi.org/tools/ecoCalculator/usersGuide.html (accessed September 2012).
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18

Assessing the sustainability of prefabricated buildings

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Abstract: This chapter aims to determine the sustainability of prefabricated buildings by studying the economic, environmental and social impact of the methods of prefabrication used in their construction. A preliminary study of prefabricated architecture is followed by an assessment of the sustainability of the main technologies used to prefabricate 161 recently constructed school buildings in Spain. This evaluation uses the integrated value model for sustainable assessment (MIVES), which is a multi-criteria decision-making method that includes value functions and has been used in workshops to define a specific sustainability assessment tool for the schools within the sample. Finally, recommendations and future trends are presented.

Key words: prefabricated buildings, sustainability, environmental impact, schools.

18.1 Introduction

All the professionals involved in architecture and construction face an important challenge to improve sustainability in the industry, i.e. the reduction of the building sector's economic, environmental and social impact. The high environmental impact of this sector is well known; in Spain, for example, construction processes and the resulting use of buildings are responsible for 32% of energy consumption, 30% of CO_2 emissions and between 30 and 40% of waste generation (Cuchí *et al.*, 2009).

But although the environmental challenges are now clear, only some new buildings reflect this new environmental awareness, and even fewer take account of other types of economic and social impact. The use of prefabricated technologies is an interesting option that should be taken into account in the design of sustainable architecture. Buildings constructed using these technologies are first produced in optimum industrial conditions, and then assembled quickly and easily on site. Hence, their life cycle processes are rationalized and, according to prefabrication institutions and companies, this type of building has less environmental impact, due to reductions in energy and water consumption, CO_2 emissions and construction waste. On the other hand, non-prefabricated building contractors claim that construction on site is cheaper and uses more local and natural materials than prefabricated technologies do; while prefabrication companies argue that their buildings are more flexible and adaptable.

This chapter studies prefabricated buildings, i.e. buildings that have been produced in a factory and are assembled on site. These edifices have been built using building systems, a set of prefabricated elements rationally organized by interdependent laws. These systems are technologies, a set of knowledge and technical means applied to a defined construction process. These differ from techniques, methods of building that are dependent on workers' skills. Prefabricated technologies are part of industrialized construction, which is produced by the industry. Prefabricated architecture and edifices will refer to those built using prefabricated and industrialized technologies and systems.

To determine whether prefabricated systems are more sustainable than on-site methods, and, if so, which prefabricated technologies are the most sustainable, the following approach is followed here: a brief historical review of prefabricated construction systems; their classification; their sustainability singularities; and an analysis of a representative sample of the main architectural sustainability assessment tools. After this preliminary study, the sustainability of the main technologies used to prefabricate 161 school buildings in Catalonia, Spain, between 2002 and 2009 is assessed.

This evaluation uses the integrated value model for sustainable assessment (Modelo Integrado de Valor para una Evaluación Sostenible – MIVES), which is a multi-criteria decision-making method; this includes value functions, and has been utilized in workshops to define a specific sustainability assessment tool for the schools within the sample. This new tool has already proven useful to assess three prefabricated technologies and one on-site. Assessment results have been used to quantify these technologies' sustainability, determine which weak points could be improved, and establish which technologies should be used to construct more sustainable edifices in the future.

18.2 A brief history of prefabricated buildings

Our prefabricated buildings have their origin during the late European colonial period and the post First World War period, when numerous temporary and emergency prefabricated edifices such as houses and schools were built (Benevolo, 1974). Notable early examples include J Paxton's partially prefabricated Crystal Palace in London of 1851 (McKean, 1994), parts of which were previously built on the ground and then assembled on the roof. Between the world wars, several prefabricated systems were developed, such as the Dorlonco system that was used to build 10,000 houses in Doncaster (White, 1965). After the Second World War, housing and educational buildings were again prefabricated, and numerous building systems were defined such as the British Consortium of Local Authorities Special Programme (CLASP) (CLASP, 1961). The majority of these systems were used to build edifices at minimal cost in short timeframes, often at the expense of quality. Most prefabricated buildings from this period go far beyond what one would consider today to be architecture, with some notable exceptions such as J. Prouvé, Le Corbusier and P. Jeanneret's emergency schools (Le Corbusier, 1995 [1946]).

Since the 1970s, architects have become more interested in new building technologies and industrialized construction. For example, several architectural offices designed buildings incorporating new materials and products from the construction industry and learning from flexible building systems such as school construction systems development (SCSD) (Chang, 1971), of which Foster Associates' school project in Wales in 1968 (Jenkins, 2002) is an example. However, at the same time, postwar prefabricated buildings, especially those built using light systems, were seen to deteriorate extensively, and increasingly failed to meet minimum waterproofing and fire safety requirements, among others. This deterioration – coupled with accidents in which the design and construction of prefabricated buildings compounded the damage caused to them – contributed to the growing unpopularity of prefabricated systems worldwide. Two examples of the latter stand out, in this respect: a fire in a residence for the elderly built by CLASP in England in 1974, and the collapse, due to a gas explosion, of a 24-story block of flats constructed with precast panels in Ronan Point, London in 1968 (Russell, 1981).

The 1980s saw a new emphasis on the added value of industrialized architecture and technologies; these included the ability to provide higher quality within minimum construction timeframes, increased possibilities for customization, and a new capacity to produce and assembly any building (Kieran and Timberlake, 2004). Richard Rogers' Lloyd's Building in London in 1986 (Powell, 1994) is an example of this new approach. During the 1990s, totally mechanized construction systems were also developed and applied, such as the Japanese T-UP system (Sakamoto and Mitsuoka, 1994).

Since the 2000s, prefabricated architecture has seen moves to reduce environmental impact. New eco-efficient technologies and buildings have been designed and built, such as the model Living Home 1.5 from Kieran Timberlake and Living Homes (Wallick, 2011), and the 30-story Ark hotel in Dongting Lake, China, built in 2012 (Jackson, 2012). Improved prefabricated systems have also been used to build emergency buildings in the aftermath of a catastrophe; for example, emergency hospitals such as the Zeppelin Mobile Clinic (Aquilino, 2011) that used light systems (ZMS, 2012).

These enhanced sustainability features initially led to increased costs in new prefabricated products and technologies, although this is no longer the case; costs may be minimized without compromising these materials' recently acquired versatility, high quality and features. Examples of this new direction include: Cattani Architects' Cité A Docks, built in France in 2010 (Derschatta, 2012); Cloud 9 Architects' Media-tic building, constructed in Barcelona in 2010 (Portilla, 2010); and Carlos Arroyo Architects' Oostcampus, a 2012 redevelopment that has transformed an old factory into the town hall and city center of Oostkamp, Belgium (Dave, 2012). New products have the following advantages, in that they: incorporate local and natural materials (Cannabric, 2010); allow the possibility of disassembling and reusing even the foundations (Techno Pieux, 2012; Krinner, Hall and container constuction, 2012); solve integral rehabilitation of façades for better environmental performance of a building (RCN, 2010); and offer reasonably priced international homes (Ideabox and Ikea, 2012). Moreover, advances in computer numerical control (CNC) technologies (Buswell *et al.*, 2007) and building information modeling (BIM) research and software tools (Sacks *et al.*, 2010) have led to improvements in future prefabricated construction.

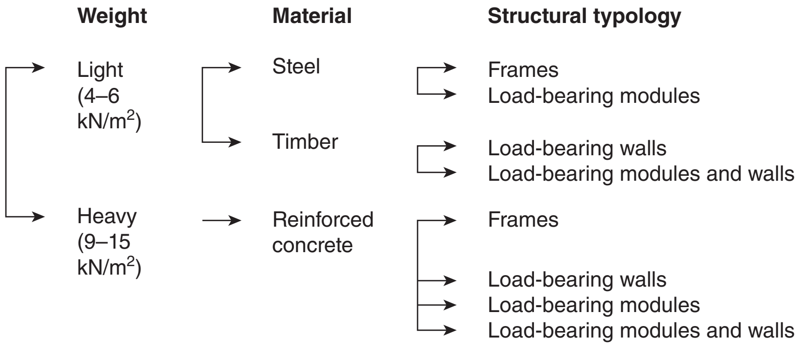
18.3 Types of prefabrication technologies

At present, these prefabricated buildings include a wide variety of buildings that have been constructed using different types of prefabricated products and systems. These products and systems can be classified according to different parameters, such as:

- industrialization;
- prefabrication;
- standardization (repetition and modulation of prefabricated elements in a building);
- automation (amount of mechanical and automatic processes without direct human intervention);
- flexibility;
- completeness (if the system employed is used to build the whole building or only parts of it); and
- structural behavior.

This last parameter deals with structural and non-structural technologies. Structural ones can also be classified according to their weight, material and structural type, as shown in Fig. 18.1.

In the case study presented in this chapter, the classification presented in Fig. 18.1 has been useful in simplifying the sustainability assessment. The assessment was carried out in three phases; firstly, the technologies used to build the 161 educational edifices in this sample were classified; secondly, the most important technologies were evaluated; and finally, the results



18.1 Classification of the structural prefabricated technologies for the case study in this chapter.

were analyzed, and conclusions drawn for the school centers that were built using these main technologies.

18.4 Assessing prefabricated buildings

18.4.1 Factors specific to prefabricated buildings in sustainability assessments

The following economic, environmental and social aspects are common to the design and manufacture of prefabricated buildings:

- Industrialized buildings have a higher initial cost, but have shorter timeframes and fewer economic and time deviations (Pons, 2009).
- There are five phases to the life cycle of prefabricated buildings: extraction, transport, construction, use, and end of life (ISO, 2006). A life cycle analysis (LCA) comparing prefabricated and non-prefabricated buildings (Wadel, 2009) found that prefabricated buildings have a lower environmental impact at the construction and the end-of-life stages, during which they consume less energy and water and produce fewer emissions. This happens for two reasons; their industrial construction occurs in optimized conditions, and the ease with which system components may be disassembled at end-of-life that allows for ready reuse and recycling of components and materials. However, the environmental impact of the transport phase depends on the distance from the point of manufacture to the point of assembly; up to a maximum distance of 150 km, a similar environmental impact to that of transporting materials for non-prefabricated construction may be expected, while systems with factories located at longer distances from the site have a higher associated environmental impact in the transport phase than non-prefabricated

methods. There are ways of minimizing this impact; the transport of completely finished prefabricated room modules, which are empty volumes, may not have a higher environmental impact than other systems if transport is well managed (e.g., if vehicles transporting modules to the assembly site can be repurposed to carry other things during their return journeys). Extraction and use phases have a variable impact which mainly depends on the material, the building typology and the final users; but it does not depend on the building prefabrication grade.

- The social impact of industrialized buildings is uncertain (Pons *et al.*, 2010). At present, prefabrication is still not well accepted by most of society, largely due to the legacy of its misuse. More positive views of its high quality and new eco-efficient prefabrication have emerged, as has recognition of the lowered risk of accidents to workers and users during building enlargements that it offers.

18.4.2 Sustainability assessment tools

Sustainability assessment tools for buildings contribute towards a more sustainable architecture because they recognize and institutionalize the importance of assessing the economic, environmental and social impacts edifices create. In enabling the use of assessments of buildings' sustainability, they provide a necessary framework for the design and construction of more sustainable buildings; consequently, they encourage research in this field (Cole, 1999). While the contribution of these tools – and the need for a more sustainable architecture – has been visible in the past decade (ICLEI, 1994), most sustainability studies, guides and evaluation tools about construction have focused on environmental impact (Todd *et al.*, 2001). Nevertheless, the number of research papers and assessment tools that also incorporate economic and/or social issues is increasing (Cole, 2005).

Table 18.1 presents 13 sustainability assessment tools for buildings, that are a representative sample of over 30 methodologies analyzed in recent reviews (Haapio and Viitaniemi, 2008; Ding, 2008; Reed *et al.*, 2009). Roughly speaking, they vary in three major ways: firstly, in terms of how often they are used, or the extent they have been adopted globally; secondly, the extent to which they incorporate environmental, economic and social indicators; and thirdly, in terms of their completeness and resulting complexity.

18.5 Case study: sustainability assessment of prefabricated school buildings

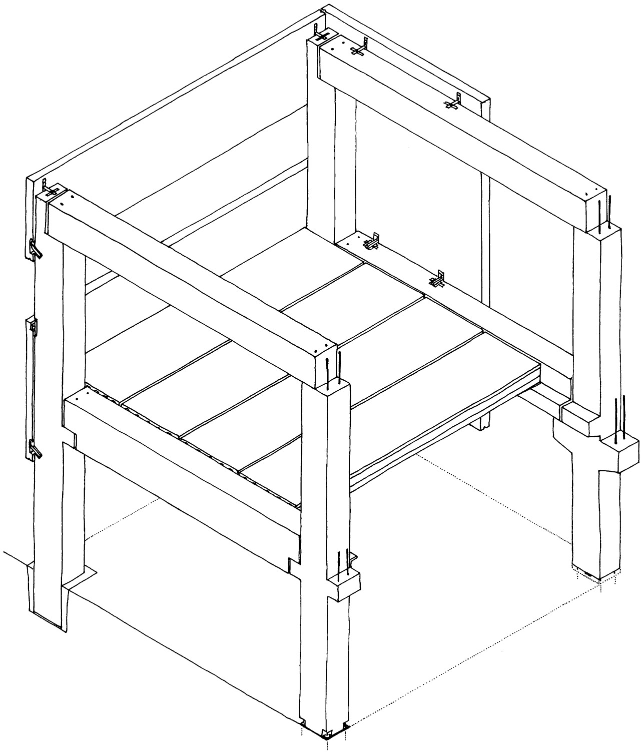
This case study represents part of a broad research project, with the objective of a better, more holistic architecture for school centers. This project

Table 18.1 Comparison of 13 sustainability assessment tools

	Name	Country	C	E	S	P	Use	Complex	Reference
1	BREEAM	UK, 1990	–	X	X	A	I	X	(BRE Global, 2011)
2	CASBEE	Japan, 2004	–	X	–	A	N	X	(CASBEE, 2008)
3	DGNB-Seal	Germany, 2008	X	X	X	A	N	X	(DGNB, 2011)
4	EcoEffect	Sweden, 2000	X	X	–	R	N	X	(Assefa *et al.,* 2010)
5	Green Globes	Canada, UK, 1996	–	X	–	A	N	–	(ECD, 2004)
6	Green Star	Australia, 2003	–	X	X	A	N	X	(GBCA, 2011)
7	HQE	France, 1996	–	X	X	A	N	X	(EPSM, 2011)
8	BEAM	Hong Kong, 1996	–	X	X	A	N	X	(BEAM, 2010)
9	LEED	USA, 2000	–	X	–	A	I	–	(USGBC, 2013)
10	PromisE	Finland, 2002	–	X	X	R	N	X	(Sitra *et al.,* 2011)
11	ProITACA	Italy, 2004	–	X	X	A	N	X	(Rocco *et al.,* 2010)
12	SABA	Jordan, 2008	X	X	X	R	B	X	(Ali and Al Nsairat, 2009)
13	VERDE	Spain, 2010	X	X	X	R	B	X	(Macías and García, 2010)

C – economic requirements (cost and timeframe); E – environmental requirements (energy consumption, water, CO_2 emissions, waste, etc.); S – social requirements (social acceptance and risk of accidents); P – sustainability assessments of prefabricated buildings; A = tool already used to assess prefabricated buildings; R = tool ready to use for prefabricated buildings; Use – I = internationally consolidated, N = nationally consolidated, B = being implemented; Complex – Tool with a complex application.

studies educational buildings from a complete point of view, taking into account environmental, historical, technological and pedagogical aspects, among others. A major outcome has been the development of a sustainability assessment tool to evaluate the building systems used to construct school centers. A sample of 386 school edifices built in the 2000s in Catalonia (DE, 2011) was taken into account to formulate this tool, which was presented recently in the article 'Integrated value model for sustainable assessment applied to technologies used to build schools in Catalonia, Spain' (Pons and Aguado, 2012).

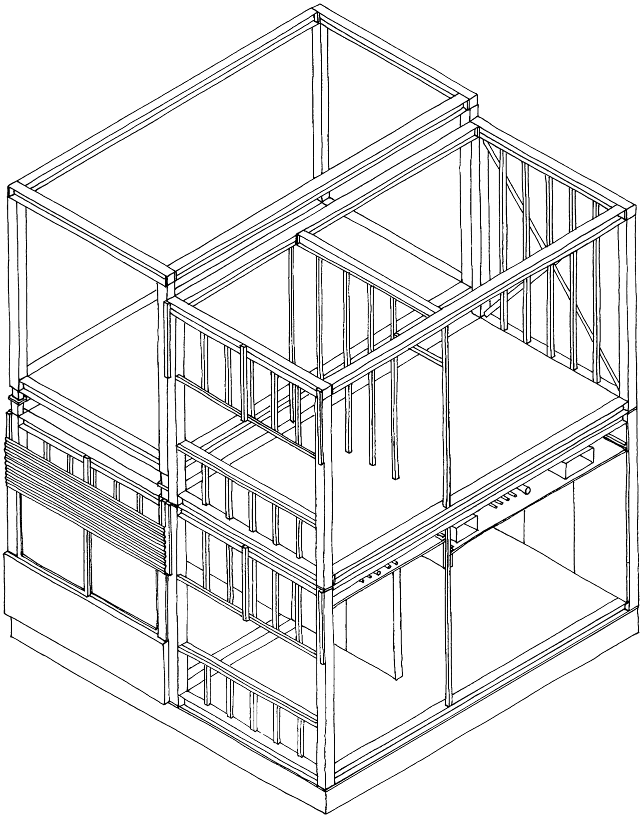


18.2 Axonometric view of a building constructed using the precast concrete technology.

In this sample, 161 schools were built mainly using prefabricated technologies and had at least their structure and façade constructed using industrialized systems. Among all the prefabricated building systems used to build these 161 schools, the three most important prefabricated systems, described below, are assessed:

1. Precast concrete technology (Fig. 18.2) has been used to build 41% of this study's prefabricated educational edifices. These centers have a reinforced or prestressed concrete unidirectional prefabricated structure (Fig. 18.3). An example of such construction is shown in Fig. 18.2; this presents a structure with 40 × 40 cm reinforced concrete columns, 60 × 40 cm prestressed concrete beams and 25 cm hollow core slabs with five centimeters poured *in situ* concrete topping layer. The façades of these buildings are normally finished with precast panels, and have a closed air chamber.
2. Prefabricated steel building systems (Fig. 18.4) have been used to build 21% of this study's industrialized schools. These schools are composed

18.3 Assembly of the precast concrete educational edifice Escola Can Coll in Torrelles de Llobregat, Spain, designed by Jordi Canyelles.

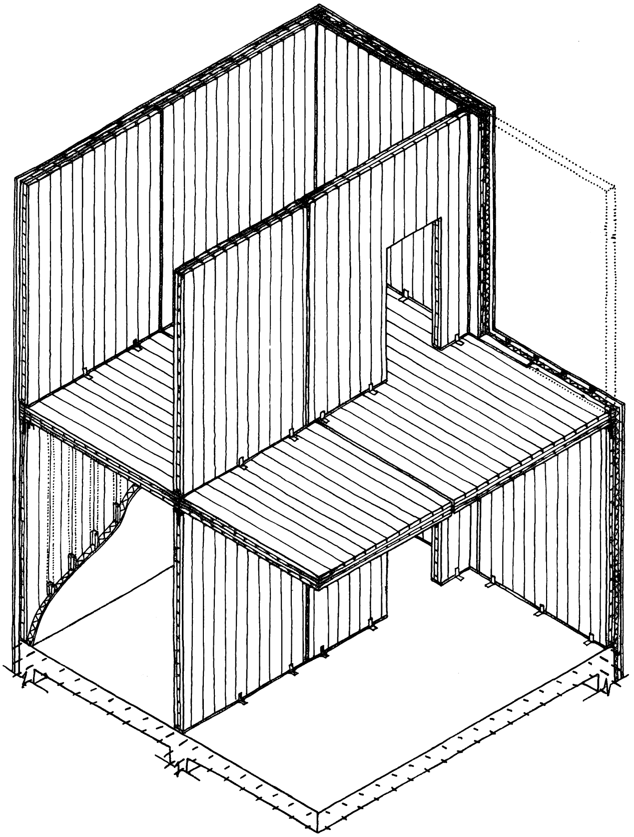


18.4 Axonometric view of an edifice built using the prefabricated steel building system.

18.5 Construction of the prefabricated steel school building Escola Garigot in Castelldefels, Spain, designed by TAC architects. Courtesy of TAC architects.

of metallic room modules having steel frames and composite slabs (Fig. 18.5). The example in Fig. 18.4 shows a structure with 14 × 14 cm hollow-section steel columns, UPN 270 steel beams, and 12 cm reinforced concrete composite slabs. The façades of such buildings may be covered with different types of panels.

3. Industrialized wood technology (Fig. 18.6) has seen limited use, despite the material's outstanding environmental behavior. Finished schools have a timber structure composed of load-bearing walls, load-bearing room modules and slabs (Fig. 18.7). For example, in Fig. 18.6 there is a structure with massive 15.8 cm laminated timber panels and massive 20.2 cm laminated timber slabs. Façades are constructed by fixing finishing boards to the timber structure.



18.6 Axonometric view of a building constructed utilizing the industrialized wood technology.

The other 225 schools in the sample were built mainly using non-prefabricated techniques and the most used construction method was the following:

4. Non-prefabricated technique: schools built using this method have a reinforced concrete structure poured on site, composed of frames and ribbed floors. An example would be a structure with 30 × 30 cm reinforced concrete columns, 45 × 30 cm reinforced concrete beams and 25 cm ribbed floor. Façades are normally finished with masonry walls and have a closed air chamber.

18.5.1 Methodology

The sustainability assessment tool used in this study case was developed using MIVES, which was combined with a simplified LCA (Pons and Wadel,

18.7 Assembly of an industrialized wood school edifice Escola Catalunya in Sant Cugat del Vallès, Spain, designed by Xavier Tragant.

2011). MIVES has already been used to develop tools for several fields, e.g. Spanish standards for structural concrete (Aguado *et al.*, 2012). MIVES was chosen because it permitted researchers to define a complete sustainability assessment tool that could: consider the particularities of the prefabricated school buildings in Spain (Pons, 2009), take into account previous analysis about sustainability tools (see Section 18.5.2), and carry out rapid assessments for precast school buildings with tight timeframes.

To develop this assessment tool, it was crucial to define the best requirements tree, incorporating all important requirements, criteria, indicators and the weighting of these different components. It was also critical to choose a suitable requirements tree, in which the amount of indicators is not excessive so that all important indicators have representative weights. To that end it was necessary to discard non-discriminatory indicators and merge similar indicators. Hence, this process was carried out by multidisciplinary experts during two seminars using an analytic hierarchy process (Saaty, 1990) and based on numerous and rigorous references (Pons, 2009; Wadel, 2009). Table 18.2 shows the requirements tree for this case study, with its three requirements, nine criteria, 17 indicators and their weights (λ_i):

- The five economic indicators assess construction and maintenance costs over 50 years, building timeframes and the probability of deviation in both. Timeframe indicators were considered to permit assessment of both prefabricated and non-prefabricated school buildings. However, timeframe indicators could not be factored in if only prefabricated

Table 18.2 Requirements, criteria, indicators tree, and weights in percentage

Requirements	Criteria	Indicators
R1. Economic (50%)	C1. Cost (52%)	I1. Production and assembly cost (30%) I2. Cost deviation probability (25%) I3. Maintenance cost (45%)
	C2. Time (48%)	I4. Production and assembly timeframe (38%) I5. Timeframe deviation probability (62%)
R2. Environmental (30%)	C3. Phase 1: extraction and fabrication of materials (30%)	I6. Water consumption (22%) I7. CO_2 emissions (40%) I8. Energy consumption (38%)
	C4. Phase 2: transport (10%)	I9. CO_2 emissions (100%)
	C5. Phase 3: building and assembly (15%)	I10. CO_2 emissions (58%) I11. Solid waste (42%)
	C6. Phase 4: use and maintenance (30%)	I12. CO_2 emissions (100%)
	C7. Phase 5: end of life (15%)	I13. Solid waste (100%)
R3. Social (20%)	C8. Adaptability to changes (35%)	I14. Neither adaptable nor disassemble building percentage (theoretical) (50%) I15. Deviation of neither adaptable nor disassemble building percentage (50%)
	C9. Users' safety (65%)	I16. Labor risk of accidents during building and assembly (40%) I17. Users' risk of accidents during building enlargements (60%)

buildings were evaluated. The cost of building usage is not considered in this study because it is not discriminatory. The end-of-life cost has not been considered either because it is already taken into account in the indicator I13.

- Environmental indicators include five LCA phases: extraction and production, transport, construction and assembly, use and maintenance over 50 years, and end of life. This requirement is based on the aforementioned simplified LCA and from this LCA five-phase study, special attention was paid to four aspects: CO_2 emissions, waste production and energy and water consumption. Water consumption during the third LCA phase was not considered; in all cases, it accounted for less than 0.01% of the whole life cycle water consumption.
- The four social indicators analyse ease of disassembly and component replacement during the building's use, as well as its capacity deviation probability and the risk of accidents during construction and assembly. The use of social indicators also enables the tool to assess both prefabricated and non-prefabricated school building construction. Other social aspects, such as the ease by which buildings may be enlarged or the safety of construction workers, were not considered as they could not meaningfully discriminate between prefabricated and non-prefabricated building systems.

For this assessment tool it was also vital to define value functions (Alarcón *et al.*, 2011) for each indicator. Value functions vary from zero to one, zero being the minimum satisfaction and one the maximum satisfaction for each indicator. Thus, in this case study, these 17 functions' adimensional values $V_i\ (x_i)$ can be aggregated, although they are the result of 17 indicators with different units. Aggregating these 17 adimensional values, the global sustainability index V was obtained:

$$V = \sum \lambda_i \cdot V_i(x_i) \qquad [18.1]$$

The 17 value functions depend on five parameters (Eq. [18.2]). These parameters define its shape and consequently how each indicator value variation corresponds to the adimensional scale. For example, if the function shape is a line, then the value variation is proportional to the indicator satisfaction in all the indicator values. Nevertheless, if the curve is S-shaped, then its function middle value variation has an adimensional variation higher than the initial and final indicator value variation.

$$V_{\text{ind}} = A + B \cdot \left[1 - e^{-ki \cdot \left(\frac{|X_{\text{alt}} - X_{\text{max}}|}{Ci} \right)^{Pi}} \right] \qquad [18.2]$$

where A is the indicator's abscissa (value X_{max}), and X_{alt} is the assessed indicator abscissa which generates a value V_{ind}, Pi is a shape factor that determines if the curve is concave, convex or S-shaped, *Ci* establishes, in functions with $Pi > 1$, the abscissa's value for the inflexion point, *ki* defines the response value to *Ci*, *B* is the value that keeps the function in the range from zero to one and is defined as:

$$B = \left[1 - e^{-ki\cdot\left(\frac{|X_{\max} - X_{\min}|}{Ci}\right)^{Pi}}\right]^{-1} \qquad [18.3]$$

Table 18.3 presents these five parameter values for each indicator. Of the 17 value functions, four decrease concavely (CcvD), five decrease linearly (SD) and eight decrease convexly (CvxD). Concave curves represent indicators for which the administration and society demand a maximum satisfaction, such as safety. On the other hand, convex functions are for indicators such as construction timeframes, or environmental indicators, for which the administration and society could accept a medium value. All functions have zero as the most satisfactory value ($X_{\min}$) and take the highest value of the evaluated systems as the least satisfactory value.

18.5.2 Discussion and results

The global sustainability index assigned to each system and their three requirements' adimensional values are the main results obtained from this assessment, which are shown in Table 18.4.

Table 18.3 Parameters for each indicator value function

I_x	Unit	$X_{\max}$	*Pi*	*Ci*	*ki*	Shape
I1	(€/m^2)	1,800	1	300	1	CvxD
I2	(%)	20	1	20	0.01	SD
I3	(€/m^2)	800	1	800	0.01	SD
I4	(months)	20	1	5	1	CvxD
I5	(months)	4	1	1,5	1	CvxD
I6	(m^3/m^2)	30,000	1	9,000	1	CvxD
I7	(kgCO$_2$/m^2)	750	1	250	1	CvxD
I8	(MJ/m^2)	7,000	1	2,000	1	CvxD
I9	(kgCO$_2$/m^2)	33	1	12	1	CvxD
I10	(kgCO$_2$/m^2)	45	3	45	0.01	CcvD
I11	(kg/m^2)	130	3	130	0.01	CcvD
I12	(kgCO$_2$/m^2)	800	1	150	1	CvxD
I13	(kg/m^2)	1,500	1	1,500	0.01	SD
I14	(%)	100	1	100	0.01	SD
I15	(%)	100	1	100	0.01	SD
I16	(points)	250	3	250	0.01	CcvD
I17	(points)	1,000	3	1,000	0.01	CcvD

Table 18.4 Global index and requirements' indexes

System	Requirements' indexes			Global index	Application	
	Economic	Environmental	Social		Only Prefabricated	Total
a) Precast concrete	0.83	0.64	0.55	0.72	67%	27%
b) Prefabricated steel	0.81	0.51	0.78	0.71	32%	13%
c) Industrialized wood	0.53	0.58	0.73	0.59	1%	0.4%
d) Non-prefabricated	0.38	0.41	0.17	0.35	–	59%

The global index grades the four building systems, prefabricated concrete, steel, wood and non-prefabricated, from more to less sustainable; with indexes 0.72, 0.71, 0.59, and 0.35, respectively. When deciding which technology should be used to construct a new educational edifice, this global index should be taken into account.

Table 18.4 also shows that these four building systems' global sustainability indexes differ from their application indexes. These last indexes were obtained by considering the accumulated percentage of the surface of the studied school edifices built since 2002. In general, this divergence exists because these buildings were not constructed with a consideration of sustainability requirements; this study is thus their first sustainability evaluation. Along the following lines, this divergence is studied for each building system:

- Precast concrete and prefabricated steel systems have a disparate application due to their limited industrial productive capacity (Pons, 2010). The 1980s marked a low point in the production of Spanish prefabrication industries, from which the sector began to recover in the 1990s. Nevertheless, from 2005 to 2008, new industries were completely overwhelmed by demand, and although more factories were set up, they were never capable of supplying this growing demand (Pons, 2009). Since 2002, up to ten consolidated groups of professionals – groups comprising architects, engineers, building contractors, and industries – took part in prefabricating schools using the precast concrete system. On the other hand, prefabricated steel productive capacity was lower, with fewer and smaller factories and only five consolidated groups of professionals participated in constructing school buildings using these steel systems. A corrected application has been developed in order to consider this differential productive capacity. To do that, the application values from Table 18.4 have been divided by the number of consolidated groups of professionals. Precast concrete and prefabricated steel corrected applications are 51% and 49% respectively, which are proportional to their assessment values.

- Until now, the industrialized wood system shows the highest divergence between application and sustainability indexes. Only three educational facilities have been constructed since this wood system was first used in school construction in 2008. An explanation for this low usage would look at logistical factors – the production center for this system is based in Austria, and the relative newness of this system means that there is a limited pool of contractors with experience of working with it.
- Non-prefabricated techniques are the most used, but carry the lowest index score. Their popularity may be explained by the more ready supply of parts and building capacity; it has also had greater political, social and technical support. It has been the least sustainable due to its high economic, environmental and social impact.

Prefabricated industries can take advantage of their requirements' indexes in order to improve their sustainability. To that end, the requirements' indexes presented in Table 18.4 are analyzed:

- Precast concrete systems have a notable economic index score, due to their competitive construction costs and timeframes, with hardly any deviations and low maintenance cost. Their environmental index is the highest-scoring, but improvements remain to be made in phases of production – such as raw materials extraction and manufacturing – that have high CO_2 emissions and energy consumption. However, their weakest aspects are their social indicators, which score lower than the other systems examined. This is due to a combination of the following: poor adaptability during the building's usage and refurbishment; difficulties in reusing or recycling component parts of a building at its end of life; and a high risk of accidents during enlargement projects due to its heavy components and the extensive use of manual labor.
- Prefabricated steel technology has a similar exemplary economic index score, for the same reasons. However, it has the worst environmental index score, due to the extraction and manufacturing of its raw materials, which carry the highest CO_2 emissions and energy consumption. The long distances from factory to site are another weak point, with an average of 900 km contributing significantly to its environmental impact. On the other hand, its social index score is the best, as schools built using this system can be easily enlarged, disassembled and reassembled; they have a quick and simple assembly process, which seldom requires manual work.

- Industrialized timber technology has the worst economic index, mainly due to cost and durability aspects; until recently, such projects required a high initial investment and an extensive maintenance programme. Its environmental index score is lower than expected, largely because of exceptionally long distances – 1,600 km – between factory and building site.

The factors outlined so far mean that these results are specific to this case study; and consequently the differences from other sustainability assessments for similar building technologies, whether theoretical or studying different production and construction environments, should be expected.

18.6 Conclusions, recommendations and future trends

This case study is a specific sustainability assessment of the main prefabricated technologies and *in situ* techniques used to build 161 and 225 Catalan schools, respectively, from 2002 to 2009. This assessment differs from others because it evaluates these technologies and their application in the construction of these schools. Hence, technologies and techniques are assessed in terms of their practical application. Two main conclusions can be drawn:

- Prefabricated systems are more sustainable than non-prefabricated ones.
- Prefabricated technologies have the lowest index score for some indicators. The most paradigmatic example is that of industrialized timber technology; these are excellent systems which have the best results in the majority of environmental evaluations (Gustavsson *et al.*, 2006), but in this particular case, the excessively long distance between the factory and the building site impacts badly upon its environmental effect score.

This study obviously recommends designing and building the most sustainable architecture using the most sustainable building technologies. To that end, it recommends all architecture and construction professionals to choose the most sustainable construction method basing their decision on a specific sustainability assessment, evaluating the different construction technologies that can be used, taking into account the particular context and condition of the building. An example of this kind of assessment is the case study presented in this chapter. This recommendation is based on the author's research group studies and their results, which conclude that technologies are not sustainable by themselves but depend on their application. If such an assessment is not possible, then it is recommended to use an existing assessment as a reference. That reference assessment must have assessed similar types of edifices with similar context and conditions. Then, some results and conclusions from that reference assessment could be interpreted and applied to a new building with caution and responsibility.

This chapter also recommends considering and evaluating both prefabricated and non-prefabricated systems for the construction of all buildings before choosing the most sustainable one.

- The most sustainable prefabricated technology will depend on the case studied, in which case the distance from factory to building site is critical.
- Depending on a project's requirements, some non-prefabricated technologies may yet be more sustainable than prefabricated ones. An obvious example of this would be an edifice built in an area without building industries located nearby, and where the required construction time is not critical. Another example would be a building in any area where the main consideration is the initial cost, and the construction time (and changes in this, as well as in overall building cost) is not taken into account.
- In the future, the author and his research team expect that industrialized technologies will be the most sustainable and commonly used, as long as their current deficiencies (e.g., in cost, or logistics of production) are rectified.

18.6.1 Future trends

Sustainability is one of the several social demands that prefabricated architecture has incorporated recently. Several new edifices built using prefabricated technologies have been designed to minimize their economic, environmental and social impact. William McDonough + Partners' Nasa Sustainability Base is an example; this was built in Moffett Field, California, in 2012 and was awarded the highest Leadership in Energy and Environmental Design (LEED) certification (McDonough and Partners, 2012).

Moreover, new prefabricated technologies have addressed their predecessors' weak points. For example, Cannapanel is a prefabricated wall system made with rammed-earth and hemp, which are natural and local materials (Cannabric, 2010). The supply of prefabricated products is also increasing; several such products include prefabricated foundations (Llorens and Pujadas, 2011) with completely finished micro piles that can be disassembled and reused (Techno Pieux, 2012; Krinner, Hall and container construction, 2012); prefabricated houses, which have reduced their price and have internationalized their market (Ideabox and Ikea, 2012; Wallick, 2011); and prefabricated solutions for integral rehabilitation of façades, that permit the reduction of a building's environmental impact (RCN, 2010). There are also numerous computer numerical control (CNC) technologies (Buswell *et al.*, 2007) and building information modeling (BIM) studies and software tools (Sacks *et al.*, 2010) providing optimizations that are specific to prefabricated construction.

18.7 Sources of further information and advice

When designing and building eco-efficient prefabricated architecture, we can rely upon prefabrication and sustainability institutions. Several institutions including Precast and Prestressed Concrete Institute (PCI) (www.pci.org), Modular Building Institute (www.modular.org) and proHolz Tirol/Holzcluster (http://www.proholz.at/) publish their reports in books, journals, documents and events. In addition to the aforementioned institutions, we may find it useful to follow the methodologies, tools and assessments carried out by the World Green Building Council (WGBC) (www.worldgbc.org) members such as LEED (new.usgbc.org) and Building Research Establishment Environmental assessment Method (BREAM) (www.bre.co.uk).

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19

Life cycle assessment (LCA) of green façades and living wall systems

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Abstract: Greening the building envelope using vegetated (green) façades is a good example of a new construction practice. Plants and partly growing building materials, in the case of living wall systems (LWS), have a number of functions that are beneficial for the built environment. However, the development of LWS is so rapid that various different materials and characteristics are available at the moment. The latter positively or negatively influence the environmental burden as discussed in this chapter. Greening the building envelope can be a sustainable option for new and retrofitted constructions, by using materials with a relatively high influence on the environmental profile, although not all benefits are yet quantifiable. The present study identifies new scientific directions to reduce the environmental costs of green constructions.

Key words: façade greening, living wall systems, LCA (life cycle assessment), sustainability, environmental burden, environmental benefits.

19.1 Introduction

19.1.1 Background

Green façades and living wall concepts (LWS) have a number of functions that are beneficial for the built environment, for example: increasing the biodiversity and ecological value, mitigation of the urban heat island effect, indoor and outdoor comfort, insulating properties, improvement of the air quality and the social and psychological well-being of city dwellers (Dunnet and Kingsbury, 2004; Perini *et al.*, 2011; Ottelé *et al.*, 2010; Ottelé, 2011). Vertical greening systems are a growing field of study which has developed rapidly especially in the last 4–5 years, so that various living wall systems and greening systems with different materials and characteristics are available at the moment (Corrado, 2010). The usage of these (extra) materials in combination with the necessary equipment for these systems (nutrients, water pumps, etc.) influence either positively or negatively the environmental burden as will be shown in this chapter.

A comparative life cycle assessment (LCA) for different greening systems is performed, in order to calculate the environmental burden profile for greened façades. A life cycle assessment is an effective way to evaluate the sustainability of a building considering the integral balance between the environmental load and possible benefits. In this study a life cycle assessment of five different greening systems and the environmental burden profile in relation to the energy savings for air conditioning and heating are discussed.

Vegetation can be seen as an additive (construction) material to increase the (multi)functionality of façades of buildings. Vertical green, also commonly referred to as a 'vertical garden', is a descriptive term that is used to refer to all forms of vegetated wall surfaces (www.greenroofs.org). Vertical green is the result of greening vertical surfaces with plants, either rooted into the ground, in the wall material itself or in modular panels attached to the façade in order to cover buildings with vegetation and can be classified into façade greening and living wall systems (Köhler, 2008; Dunnet and Kingsbury, 2004).

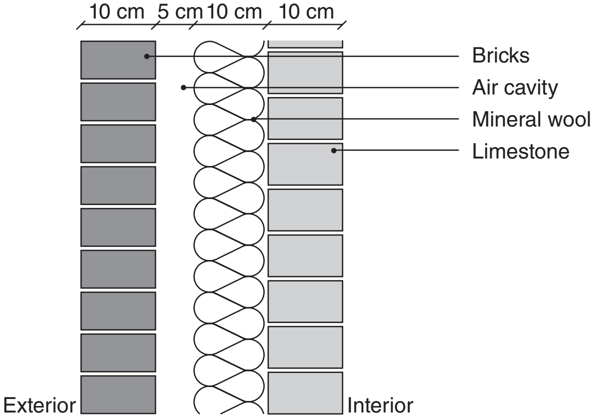
Greening the cities is not a new approach (the Hanging Gardens of Babylon are a good example), but the benefits are rarely quantified. Greening façades is a good example of combining nature and buildings (linking different functionalities) in order to address environmental issues in dense urban surroundings (van Bohemen, 2005). The main benefits of the use of green façade applications are of economical, social and environmental origin, such as greenhouse gas (emissions) reduction, adaptation to climate change, air quality improvements, energy saving by insulation, habitat provision and improved aesthetics (Krusche *et al.*, 1982; Minke and Witter, 1982; Peck *et al.*, 1999). Also sound reduction is possible by the use of vegetation (Pal *et al.*, 2000).

Vertical greening concepts can be divided into categories (green façades and living wall systems) according to their growing method (Dunnet and Kingsbury, 2004). Green façades are based on the use of climbers (evergreen or deciduous) attached directly to the building surface (as in traditional architecture), or supported by steel cables or trellis. In the first case, climbers planted in the ground at the base of the building allows a cheap façade greening to be obtained, but there could be implications for any building works that are needed to be carried out (e.g., damage and maintenance to the façade), besides the fact that climbing plants can only grow to a maximum of 25 m in height, which can take several years (Dunnet and Kingsbury, 2004). Supporting systems are sometimes necessary and planter boxes, such as prefabricated and prevegetated systems (living wall systems), attached to walls can require specific growing substrate to facilitate plant growth. Living wall systems (LWS) consist of modular panels, each of which contains its own soil or other growing medium (soil, felt, perlite, etc.) based on hydroponic culture, that is using balanced nutrient solutions to provide all or part of the plant's food and water requirements (Dunnet and Kingsbury, 2004).

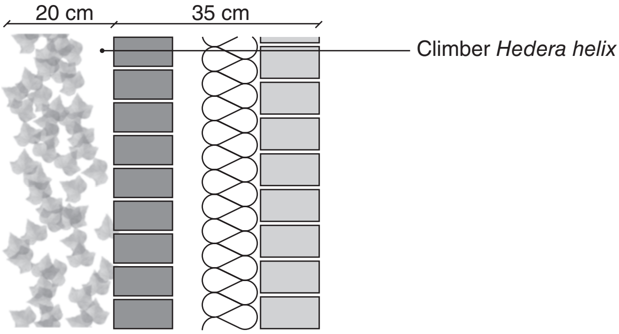
19.1.2 Aim of the research

The goal of this life cycle assessment (LCA) is to evaluate the actual and potential environmental aspects associated with constructing, maintaining and disposing of 1 m^2 façade and to determine the impact of the raw material depletion, fabrication, transportation, installation, operation, maintenance and waste for four greening systems compared to a bare façade.

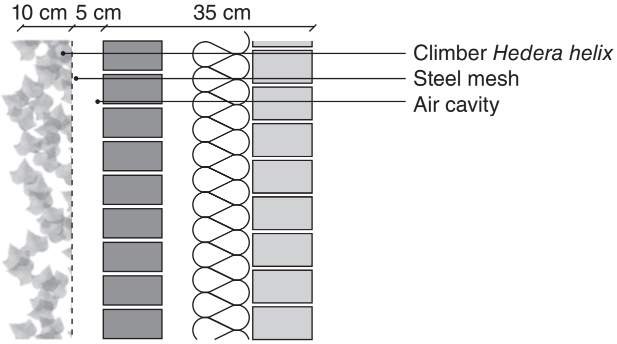
The LCA research presented here was conducted in Delft (The Netherlands) and examines a conventional bare built-up European brick façade (Fig. 19.1), a conventional façade covered with a climber planted at the base of the façade (greened directly) (Fig. 19.2), a conventional façade covered



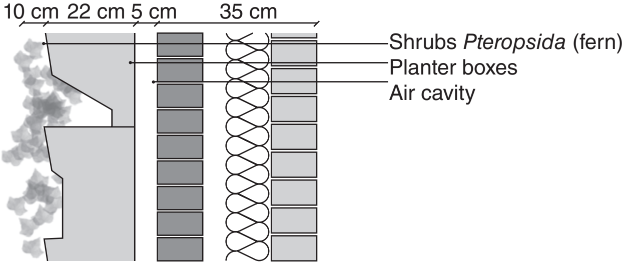
19.1 Bare wall with material layers involved.



19.2 Direct greening system with material layers involved.



19.3 Indirect greening system with material layers involved.

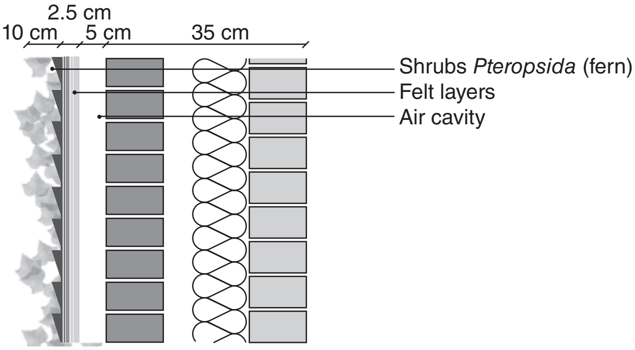


19.4 LWS based on planter boxes with material layers involved.

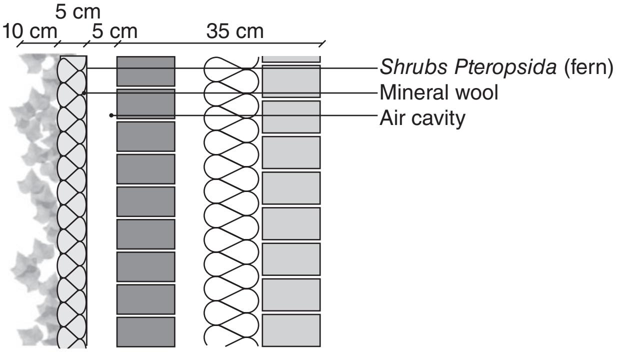
with a climber (planted also at the base of the façade) using a stainless steel framework to create a cavity between foliage and façade (Fig. 19.3, indirectly), a conventional façade covered with a living wall system (LWS) based on planter boxes filled with potting soil (Fig. 19.4), a conventional façade covered with a living wall system based on felt layers (Fig. 19.5) and a conventional façade covered with a living wall system based on mineral wool (Fig. 19.6).

To develop the LCA model, an inventory analysis was created. In this phase, information has been collected about the materials involved for the bare wall and the different greening systems. The materials needed were obtained from the project construction documents and information provided by the manufacturers.

A life cycle assessment for the five vertical greening systems investigated allows their sustainability to be evaluated in relation to the achievable environmental benefits, as a measure of ecological quality based on our knowledge of the influence on the environment. Sustainability can be defined as a general property of a material or a product that indicates



19.5 LWS based on felt layers with material layers involved.



19.6 LWS based on mineral wool with material layers involved.

whether, and to what extent, the prevailing requirements are met in specific applications. These requirements, which relate to air, water and soil loading, have influences on the well-being and health of living creatures, the use of raw materials and energy, and also consequences for the landscape, the creation of waste and the occurrence of nuisance to the surrounding environment (Hendriks *et al.*, 2000).

19.1.3 Overview of the environmental and microclimatic benefits related to green façades

The ecological and environmental benefits of the integration of vegetation on buildings, through green roofs and vertical greening, include the improvement of air quality, energy savings for the building heating and cooling, and reduction of the heat island effect (Ottelé *et al.*, 2010; Perini *et al.*, 2011; Onishi *et al.*, 2010; Ottelé, 2011). Living wall systems (LWS) and green façades have different characteristics that can influence these benefits, such as cooling and insulating properties. Relevant aspects are the thickness of the foliage (creating a stagnant air layer and shading the façade), water content, material properties and possible air cavities between the different layers.

Between the façade and the dense vertical green layer, both rooted in the soil and rooted in artificial pre-vegetated based systems (hydroponic), a stagnant air layer exists. Stagnant air has an insulating effect; green façades can therefore serve as an 'extra insulation' of the building envelope (Krusche *et al.*, 1982; Minke and Witter, 1982; Perini *et al.*, 2011). Also direct sunlight on the façade is filtered by leaves, thanks also to the phototropism effect. Of the sunlight energy that falls on a leaf, 5–30% is reflected, 5–20% is used for photosynthesis, 10–50% is transformed into heat, 20–40% is used for evapotranspiration and 5–30% is passed through the leaf (Krusche *et al.*, 1982). This blocking of the direct sunlight exposure ensures a cooling effect in warmer climates. Secondly, green façades and roofs will cool the heated air through evaporation of water (Wong *et al.*, 2010); this process is also known as evapotranspiration. As a consequence, every decrease in the internal air temperature of 0.5°C will reduce the electricity use for air-conditioning by up to 8% (Dunnet and Kingsbury, 2004).

In winter, the system works the other way round and heat radiation of the exterior walls is insulated by evergreen vegetation. In addition, the dense foliage will reduce the wind flow around the façade and thus also help to prevent the cooling down of the building.

Greening vertical surfaces has a beneficial effect on the insulating properties of buildings through exterior temperature regulation (Krusche *et al.*, 1982). The role of insulation materials and stagnated air layers is to slow down the rate of heat transfer between the inside and outside of a building, which is a function of the difference between the inside and outside temperatures. The insulation value of vertical greened surfaces can be increased in several ways (Peck *et al.*, 1999):

- By covering the building with vegetation, the summer heat is prevented from reaching the building skin, and in the winter, the internal heat is prevented from escaping.
- Since wind decreases the energy efficiency of a building by 50%, a plant layer will act as a buffer that keeps wind from moving along a building surface.
- By the materials and substrates used in the case of living wall concepts.

At the beginning of the 1980s, Krusche *et al.* (1982) estimated that the thermal transmittance of a 160 mm plant cover is $2.9\,\mathrm{W m^{-2} K^{-1}}$. Minke and Witter (1982) also suggested reducing the exterior coefficient of heat transfer. By reducing the wind speed along a green façade, the exterior surface resistance coefficient can be equalized to the interior surface resistance coefficient as demonstrated by Perini *et al*. (2011). Field measurements on a plant covered wall and a bare wall by Bartfelder and Köhler (1987) show a temperature reduction at the green façade in the range of 2–6°C compared with a bare wall. Also Eumorfopoulou and Kontoleon (2009) have reported a temperature cooling potential of plant covered walls in a Mediterranean climate; the effect was up to 10.8°C. Another recent study by Wong *et al*. (2010) on a free standing wall in Hortpark (Singapore) with vertical greening types shows a maximum reduction of 11.6°C. Alexandri and Jones (2008) simulated a temperature decrease in an urban canyon with greened façades, reducing the air temperature by 4.5°C for the Mediterranean climate and 2.6°C for the temperate climate. In research by Eumorfopoulou and Aravantinos (1998), they concluded that a planted roof contributes to the thermal protection of a building but that it cannot replace the thermal insulation layer.

19.2 Life cycle assessment (LCA) methodology

19.2.1 Basic approach to LCA

In this research, a life cycle assessment (LCA) is used to calculate the environmental impact of the production, use, maintenance and waste for five common systems of vertical greening of buildings. This is to compare the environmental burden and benefits of the green system with a bare brick wall (Fig. 19.1):

1. bare wall (brick)
2. direct façade greening system + bare wall
3. indirect façade greening + bare wall
4. living wall system (LWS) based on planter boxes filled with soil + bare wall
5. living wall system (LWS) based on felt layers + bare wall
6. living wall system (LWS) based on mineral wool + bare wall

As shown in Fig. 19.1, the conventional bare wall (option 1), also used as the basis for the greening systems analysed, is constructed by masonry, air cavity, insulation material and limestone. The direct façade greening (option 2) consists in a well-grown evergreen climber *Hedera helix*, attached directly to the building surface and planted at the base of the greened façade. The thickness of the *Hedera* foliage is ± 20 cm. The third system analysed in this study is an indirect façade greenery (option 3). This system is constituted by steel frames as support for evergreen climbing plants (*Hedera helix* ± 10 cm). The fourth investigated greened façade, a living wall system (option

4), is based on plastic modules (HDPE), filled with soil and planted with evergreen species (*Pteropsida*), working with a system for water and nutrients. The fifth system (option 5) is an LWS based on several felt layers as substrate supported by a PVC sheet and also planted with ferns (*Pteropsida*) and working with a system for water and nutrients. The sixth system (option 6) is an LWS based on mineral wool as substrate supported through a frame and also planted with ferns (*Pteropsida*) and working with a system for water and nutrients.

A number of factors are considered in the analysis: raw material depletion, fabrication, transportation, installation, operation, maintenance and waste for the façade(s) area. The transportation distances used are all to and from the city of Delft (The Netherlands).

In order to work with the LCA model, a functional unit should be defined to serve as a basis for comparison of the greening alternatives. According to the ISO 14044 standard (ISO, 2006), the functional unit is defined as the reference unit through which a system performance is quantified in an LCA. The chosen functional unit in this research is $1\,m^2$. As basis for calculating the materials and products involved in every system, a fictitious façade of $100\,m^2$ is used.

The results of the assessment are expressed as the accumulation of environmental impact over the service life; therefore the frequency of maintenance activity and the times at which replacements are needed are described. Finally, the assumptions and limitations of analysis and the data will be discussed.

19.2.2 Tools and methodology

The database used to develop the process models for this analysis is based on the Dutch National Environmental database compiled by the Dutch Institute for Building Biology and Ecology (NIBE). The complete set of environmental impact categories is known as the 'environmental profile'. The environmental profile is divided into ten categories:

- abiotic depletion (kg Sb equivalents)
- global warming (kgCO_2 equivalents)
- ozone layer depletion (kgCFC-11 equivalents)
- human toxicity (kg1.4-DB equivalents)
- fresh water aquatic ecotoxicity (kg1.4-DB equivalents)
- marine water aquatic ecotoxicity (kg1.4-DB equivalents)
- terrestrial ecotoxicity (kg1.4-DB equivalents)
- photochemical oxidation (kg C_2H_4)
- acidification (kg SO_2 equivalents)
- eutrophication (kg PO_4 equivalents)

The category marine water aquatic ecotoxicity will not be taken into account because of significant problems associated with the calculation of the impact in the method. These problems are related to the time a substance is present in the marine ecosystem and missing data for normalization (Blom *et al.*, 2010). The environmental calculation is built up by three main classes: materials, transportation and waste. For every class the environmental burden is calculated according to the ten categories described above.

19.2.3 Description of the (greening) systems as used for the assessment

For this life cycle assessment, only the components of the building façade directly related to the systems analysed will be examined. The bare wall is taken as the basis for each material inventory; in general a façade is a barrier against environmental conditions to separate the building interior and exterior, nowadays a greening system is added as an extra layer with the previous described multi-functionalities. The differences between the material quantities between the bare façade and the greened ones are the layers involved depending on the greening type: for the direct climber system, the layer consists of a single plant layer; the indirect climber system also involves a stainless steel support for the plant; the layer added for the first LWS is based on planter boxes filled with potting soil; the second LWS involves several layers for rooting, waterproofing and supporting; and the third LWS involves a layer of mineral wool for rooting and aluminium as supporting material.

The materials used for this inventory were obtained from product forms and information provided by companies (Table 19.1). The transportation distances used are all to and from the city of Delft (The Netherlands). For the conventional bare wall, the majority of the materials are local. All the plants used for the inventory come from a cultivation area 30 km away from Delft. The steel components analysed for the indirect greening system are manufactured in Rotterdam (18 km from Delft). The distance between Delft and the company of the LWS based on planter boxes is 15 km, for the LWS based on felt layers the distance is 65 km, and for the LWS based on mineral wool the distance is 70 km (Tables 19.2–19.7).

19.2.4 Assumptions made to complete data inventory needed for assessment

The analysis period to study the environmental aspects and potential impacts is based on a façade's service life of 50 years. The life expectancy of the conventional bare wall is assumed to be 50 years as well as for the façades covered directly and indirectly with climbing plants (Dunnet and

Table 19.1 Components and materials for bare wall and greening systems

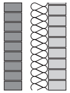
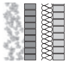
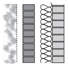
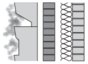
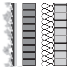
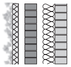
Components	1. Bare wall	2. Direct green	3. Indirect green	4. LWS planter boxes	5. LWS felt layers	6. LWS mineral wool
System preview						
Inner masonry	Limestone	Limestone	Limestone	Limestone	Limestone	Limestone
Insulation	Mineral wool	Mineral wool	Mineral wool	Mineral wool	Mineral wool	Mineral wool
Air cavity	50 mm	50 mm	50 mm	50 mm	50 mm	50 mm
Outer masonry	Brick (clay)	Brick (clay)	Brick (clay)	Brick (clay)	Brick (clay)	Brick (clay)
Air cavity	–	–	50 mm	50 mm	50 mm	50 mm
Structural support member	–	–	Steel mesh	Steel profile	Steel profile	Steel profile
Supporting system	–	–	–	HDPE boxes	PVC foam plate	Aluminium
Inner layer	–	–	–	–	White fleece	–
Growing material	–	Terrestrial soil	Terrestrial soil	Potting soil	Wool fleece	Mineral wool
Damp open foil	–	–	–	–	PE fleece	–
Outer felt layer	–	–	–	–	Black fleece	Black fleece
Irrigation system	–	–	–	PE pipes	PE pipes	PE pipes
Water demand	–	Groundwater	Groundwater	Tapwater + nutrients	Tapwater + nutrients	Tapwater + nutrients
Vegetation	–	*Hedera helix*	*Hedera helix*	*Pteropsida* (ferns)	*Pteropsida* (ferns)	*Pteropsida* (ferns)

Table 19.2 Components involved for bare wall system

Components	Material	Weight (kg/m^2)	Distances (km)	Service life (years)
Inner masonry	Limestone	147	62	50
Insulation	Mineral wool	4.3	190	50
Air cavity	Cavity	–	–	–
Outer masonry	Brick (clay)	145	80	50
Mortar	Sand + cement + water	84	15	50

Table 19.3 Components involved for direct façade greening system

Components	Material	Weight (kg/m^2)	Distances (km)	Service life (years)
Inner masonry	Limestone	147	62	50
Insulation	Mineral wool	4.3	190	50
Air cavity	Cavity	–	–	–
Outer masonry	Brick (clay)	145	80	50
Mortar	Sand + cement + water	84	15	50
Vegetation	*Hedera helix*	5.5	30	50

Table 19.4 Components involved for indirect façade greening system

Components	Material	Weight (kg/m^2)	Distances (km)	Service life (years)
Inner masonry	Limestone	147	62	50
Insulation	Mineral wool	4.3	190	50
Air cavity	Cavity	–	–	–
Outer masonry	Brick (clay)	145	80	50
Mortar	Sand + cement + water	84	15	50
Air cavity	Cavity	–	–	–
Bolts	Stainless steel	0.015	18	–
Spacer brackets	Stainless steel	0.045	18	–
Structural support member	Stainless steel mesh	1.55	18	–
Vegetation	*Hedera helix*	2.7	30	50

Table 19.5 Components involved for LWS based on planter boxes

Components	Material	Weight (kg/m^2)	Distances (km)	Service life (years)
Inner masonry	Limestone	147	62	50
Insulation	Mineral wool	4.3	190	50
Air cavity	Cavity	–	–	–
Outer masonry	Brick (clay)	145	80	50
Mortar	Sand + cement + water	84	15	50
Bolts	Steel S235	0.27	15	–
Spacer brackets	Steel S235	0.315	15	–
Air cavity	Cavity	–		–
Supporting U section	Steel S235	4.62	15	–
Planter boxes	HDPE	13.2	15	50
Growing material	Potting soil	75.6	30	50
Vegetation	*Pteropsida*	8	30	10
Watering system	PE	0.26	35	7.5
Water demand	Tap water	365	0	1

Table 19.6 Components involved for LWS based on felt layers

Components	Material	Weight (kg/m^2)	Distances (km)	Service life (years)
Inner masonry	Limestone	147	62	50
Insulation	Mineral wool	4.3	190	50
Air cavity	Cavity	–	–	–
Outer masonry	Brick (clay)	145	80	50
Mortar	Sand + cement + water	84	15	50
Bolts	Steel S235	0.13	65	–
Spacer brackets	Steel S235	0.19	65	–
Air cavity	Cavity	–	–	–
Supporting U section	Steel S235	4.62	65	–
Foam plate	PVC	7	65	10
White fleece	Polypropylene	0.3	65	10
Wool fleece	Polyamide	0.6	65	10
PE fleece	Polyethylene (LDPE)	0.045	65	10
Black fleece	Polypropylene	0.27	65	10
Vegetation	*Pteropsida*	7.5	30	3.5
Watering system	PE	0.09	35	7.5
Water demand	Tap water	1095	0	1

Table 19.7 Components involved for LWS based on mineral wool

Components	Material	Weight (kg/m^2)	Distances (km)	Service life (years)
Inner masonry	Limestone	147	62	50
Insulation	Mineral wool	4.3	190	50
Air cavity	Cavity	–	–	–
Outer masonry	Brick (clay)	145	80	50
Mortar	Sand + cement + water	84	15	50
Bolts	Steel S235	0.13	70	–
Spacer brackets	Steel S235	0.19	70	–
Air cavity	Cavity	–	–	–
Supporting U section	Steel S235	4.62	70	–
Aluminium frame	Aluminium	2.9	70	50
Mineral wool	Rockwool	4.3	70	10
Black fleece	Polypropylene	0.85	70	10
Vegetation	*Pteropsida*	7.5	30	3.5
Watering system	PE	0.09	35	7.5
Water demand	Tap water	730	0	1

Kingsbury, 2004). The replacement frequencies of plants for living wall systems are simplified as 10 years (10% replacement/year) for the one based on planter boxes and 3.5 years (30% replacement/year) for the felt layers and mineral wool-based systems.

The life expectancy for the plastic (HDPE) planter boxes is estimated to be more than 50 years. With reference to a study performed by Riedmiller and Schneider (1992), it is clearly indicated that the service life of PVC layers (structural support for the felt layers) is only 10 years, and that the whole module (inclusive felt layers) has the same life expectancy. Therefore the whole module is assumed to have a life expectancy of 10 years as well.

The automated watering system needed for the LWS concepts (planter boxes, felt layers and mineral wool) has to be replaced every 7.5 years because of crystallizing of salts. The amount of water and nutrients needed for the LWS is controlled by a computer and sensors (due to the complexity of these systems), and is therefore not included in this analysis. Besides the watering system, living wall concepts need a nutrient solution, which is not taken into account due to the small (1%) influence. The water consumption for the living wall system based on planter boxes is assumed as a quantity of 1 liter/day (average value for whole year), for the living wall system based on felt layers 3 liter/day (average value for whole year), and for the LWS based on mineral wool 2 liter/day (average value for whole year). Watering systems are not taken into account for the direct and indirect greening system due to the fact that the climbing plants are rooted into the ground (Tables 19.2–19.7).

For all the greening systems analysed in this study, the possibility of recycling and reuse is taken into account. If it is possible for the waste class, the option to recycle or reuse is chosen for the calculation. Exceptions are made if it is not possible to separate multiple-layered components for recycling processes. In this case, due to the service life and complexity, none of the materials will be reused.

19.3 Interpretation and analysis of LCA results

19.3.1 Environmental burden analysis

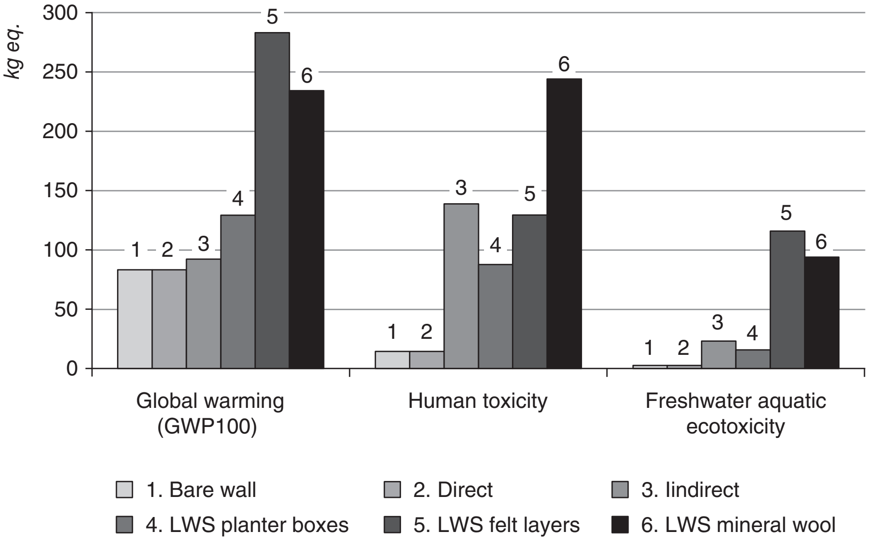
An LCA was calculated for the entire life cycle for each component of green façade alternatives. The results show that there is a significant difference between the greening systems and the bare wall, except for the direct greening system. This is caused mainly by the materials involved for the supporting systems.

Starting from the full environmental profile, only global warming, human toxicity and freshwater aquatic ecotoxicity are considered for showing the results due to the very small influence of the other six categories.

From Fig. 19.7 it is possible to deduce for global warming that the living wall systems based on felt layers and mineral wool are more than double compared to the other systems described (i.e., bare wall, direct, indirect and LWS based on planter boxes). For human toxicity, the indirect system and both living wall systems have a high impact compared to the bare wall and the direct greening system. The same trend is noticeable for the freshwater aquatic ecotoxicity, except for the LWS based on felt layers and mineral wool which is almost more than five times the indirect and LWS based on planter boxes.

The environmental burden for stainless steel in the database is based on 30% of recycled stainless steel for the production process. This percentage is a common average used in databases worldwide, but the amount of recycled stainless steel could be higher, which could lead eventually to a lower environmental burden.

The graphs in Fig. 19.8 built up for each system show the influence for the classes material, transportation and waste of the bare wall, supporting systems and vegetation. The largest difference in the analysis regards the material impact for the supporting systems. Due to this, the direct greening system has the lowest environmental burden. For this system as for the indirect one, the vegetation also has a very small impact, since it is only related with transportation (no watering and nutrient system and replacement of plants is needed). For the living wall systems based on felt layers and mineral wool, the waste class has a major impact due to the impossibility of recycling the entire module involved. Besides for the mineral wool

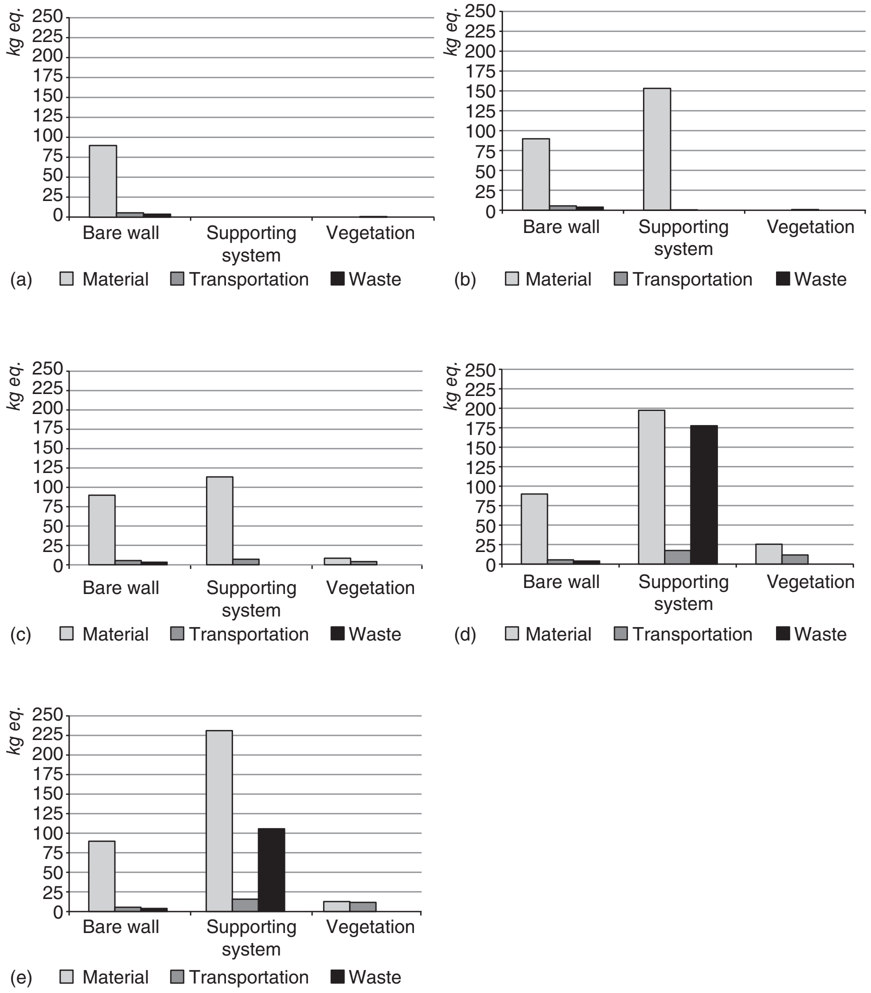


19.7 Environmental burden profile for global warming, human toxicity and freshwater aquatic ecotoxicity.

LWS, the aluminium used as supporting material contributes significantly to the burden profile.

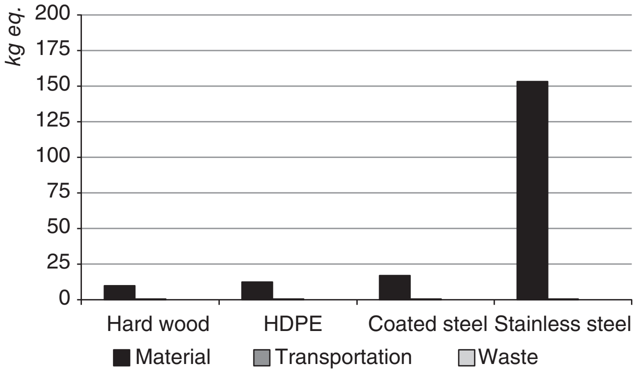
19.3.2 Designing vertical green with respect to sustainability

This study concerns the analysis of five types of greening systems (direct, indirect, LWS based on planter boxes, felt layers and mineral wool) and presents an overview of the tendency with respect to the green façades technologies. Since the development in this field is growing rapidly, especially in the last 3–4 years, many systems with different materials and characteristics are available. The different systems and materials can have an influence on the environmental burden, either positively and negatively. For example, for the indirect greening system, based on stainless steel mesh acting as support for climbing plants, other materials, such as different types of wood, plastic, aluminium and steel, can be used. Each of these materials can change the aesthetic and functional properties due to the different weight, profile thickness, durability and cost. Figure 19.9 shows (for the indirect greening system) the influence on the environmental burden profile of four different materials that can be used as supporting system. The burden profile of the stainless steel support (as used for this LCA) is roughly 10 times higher than supporting systems based on recycled plastic (HDPE), hard wood (FSC certificated) and coated steel.



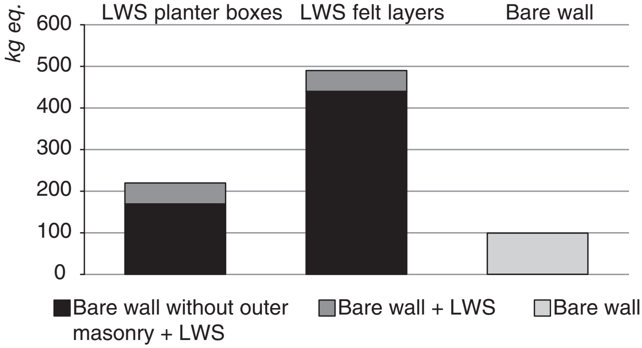
19.8 Total environmental burden profile for classes material, transportation and waste for (a) direct greening system; (b) indirect greening system, (c) LWS based on planter boxes, (d) LWS based on felt layers, (e) LWS based on mineral wool.

In the case of living wall systems, a sustainable approach can involve, in addition to the material choice, a higher integration within the building envelope by combining functionalities. In the case of the bare wall analysed in this study, it is possible, for example, to omit the outer masonry (Fig. 19.10), since the protection against the environmental parameter can be provided by a living wall system. Figure 19.11 shows the possible reduction of the environmental burden through this integration.



19.9 Environmental burden profile for different supporting materials for the indirect greening system.

19.10 Living wall systems (based on planter boxes and based on felt layers) without the exterior masonry for a total integration of the greening concept.



19.11 Comparison between the environmental profiles for two different types of living wall concepts applied to a traditional bare wall and with the omission of the external masonry layer.

19.4 Interpretation of the LCA analysis

19.4.1 Environmental burden profiles for the systems analysed

All the systems studied reveal similar dominating impact categories, though the magnitude of these differs considerably. This difference is caused mainly by the supporting material and the replacement both for plants and material. Due to this, the living wall systems based on felt layers and mineral wool have the largest impact for global warming and freshwater aquatic ecotoxicity, as it is necessary to replace the panels five times in a service life of 50 years. The human toxicity profile shows a higher impact for the four green façade types with supporting system for plants (indirect, LWS based on planter boxes, LWS based on felt layers and LWS based on mineral wool). For the systems studied, vegetation contributes to the environmental burden only because of the transport (which is dependent on the different replacement required for each system).

The indirect greening system has a high impact profile for the supporting system due to the use of stainless steel. Since stainless steel is a high-quality material, it could be possible to use it for a life span longer than 50 years. In this case, due to the foliage package that is weaving through the stainless steel mesh, it is not recommended to reuse the material after the service life. Therefore in this case also other supporting materials could be used (hard wood, HDPE, coated steel). As shown in Fig. 19.9, the choice of one of these materials for the supporting system can reduce the environmental burden profile.

The highest environmental burden that is found in the study concerns the living wall systems based on felt layers and mineral wool; because it is difficult to recycle the panel, the environmental burden of waste has a similar impact profile as the original construction. The environmental profile of the living wall systems analysed can be decreased by higher integration of the building envelope (omitting the exterior masonry), by choosing materials other than those used for the systems analysed. Choosing materials with a lower environmental impact (instead of stainless steel and aluminium) makes LWS-based systems a more sustainable option from the material point of view.

19.4.2 Environmental benefits of vertical green related to the assessment

The benefits that can be derived from the green layer are dependent on the growing rate of the plants (covering of the façade). For the direct and indirect system, the full covering of the façade by *Hedera helix* is estimated after 20 years (according to Bellomo, 2003, 0.5 m/year for vertical growing). For both living wall systems, due to the amount of plants and the several material layers involved, it is possible to calculate the benefits after installation of prefabricated modules.

For calculating the energy savings for heating, due to the increase of the insulating properties with greening systems, the additional thermal resistance is assumed to be $0.09\,\mathrm{Km^2W^{-1}}$. This assumption is used for all of the direct and indirect greening systems analysed due to the fact that there is a stagnant air layer in and behind the foliage (Perini *et al.*, 2011). For the living wall systems, the thermal transmittance of the substrate and the materials used are added.

For the benefits related to energy savings (increase of R-value), in this study a simulation model (Termo 8.0 Microsoftware) is used to calculate the influence on the environmental profile for two different types of climates, namely temperate climate (The Netherlands) and Mediterranean climate (Italy), since a green layer has a greater effect on the insulation properties for the temperate climate and the cooling potential for the Mediterranean climate. The simulated buildings used for both locations are $75\,\mathrm{m^2}$ with a volume of $296\,\mathrm{m^3}$ (three floors) freestanding and situated in an urban context with energy efficiency class B. The energy savings due to the cooling potential of the four greening systems are based on the research conducted by Alexandri and Jones (2008), regarding the temperature decrease in an urban canyon with green façades and the percentage reduction that is reached for the air-conditioning (Table 19.8). For the temperate climate, the energy saving for air conditioning is not taken into account; for the Mediterranean climate, the calculation is based on the consumption of energy for air conditioning (energy class B) in the northern part of Italy (Genoa). Energy savings from the additional insulation provided by the green façade types and for the cooling potential are found for the building and converted into unit savings to be applied across the LCA calculation.

Also the use of biomass produced (capturing of CO_2) by pruning the *Hedera helix* and by the replacement of plants (*Pteropsida*) from the living wall systems can be converted in energy (kWh). The calculation of this benefit shows a very small impact on the total environmental benefits.

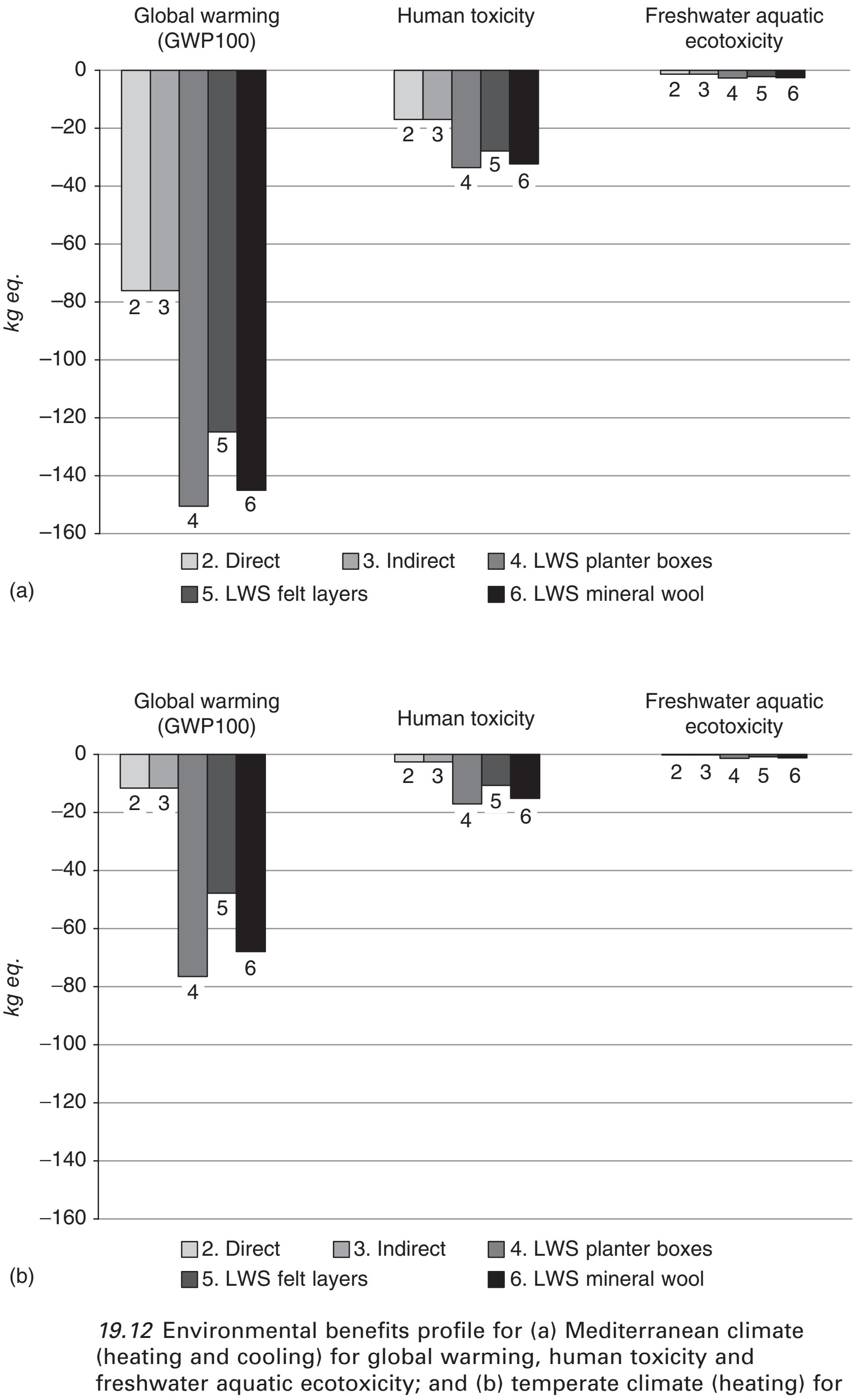
The calculation of the savings for the environmental impact as described above is based on the benefits thanks to the energy saving for heating (insulation) and cooling (only for the Mediterranean climate). The energy saving thanks to the thermal properties of the systems is calculated through subtraction of the amount of energy that can be saved due to the 'extra' insulation layer. For the direct and indirect greening systems, the energy saving for heating is estimated as 1.2% of the annual consumption. For the living wall systems based on planter boxes, felt layers and mineral wool, the saving was, respectively, 6.3%, 4% and 6.3%. The temperature decrease

Table 19.8 Energy saving (calculated with Termo 8.0 software) for heating, energy saving for cooling and temperature decrease for Mediterranean and temperate climates (based on Alexandri and Jones, 2008)

Greening system	Benefit	Temperate climate	Mediterranean climate
Direct green	Energy saving for heating	1.2%	1.2%
	Temperature decrease	2.6°C	4.5°C
	Energy saving for cooling	–	43%
Indirect green	Energy saving for heating	1.2%	1.2%
	Temperature decrease	2.6°C	4.5°C
	Energy saving for cooling	–	43%
LWS planter boxes	Energy saving for heating	6.3%	6.3%
	Temperature decrease	2.6°C	4.5°C
	Energy saving for cooling	–	43%
LWS felt layers	Energy saving for heating	4%	4%
	Temperature decrease	2.6°C	4.5°C
	Energy saving for cooling	–	43%
LWS mineral wool	Energy saving for heating	6.3%	6.3%
	Temperature decrease	2.6°C	4.5°C
	Energy saving for cooling	–	43%

thanks to a green layer is estimated to be 4.5°C (43% energy saving for air conditioning) for the Mediterranean climate and 2.6°C for the temperate climate according to Alexandri and Jones (2008).

The categories considered for the environmental benefits thanks to energy saving for heating and air conditioning are, as for the environmental burden, global warming, human toxicity and freshwater aquatic ecotoxicity. From Fig. 19.12 it can be derived that the benefits for heating for the living wall systems are more than three times the direct and indirect greening system. This is caused mainly by the contribution for the insulation properties of the materials involved. The graphs show a similar benefit profile for both direct and indirect façade greening systems as for the living wall systems; since for the direct and indirect façade greening systems only the vegetation layer (*Hedera Helix* for both systems) has an influence on the insulating properties. The profiles for the living wall systems have major



19.12 Environmental benefits profile for (a) Mediterranean climate (heating and cooling) for global warming, human toxicity and freshwater aquatic ecotoxicity; and (b) temperate climate (heating) for global warming, human toxicity and freshwater aquatic ecotoxicity.

benefits. The benefits are a little bit higher for the one based on planter boxes thanks to resistance of the soil package and for the LWS based on mineral wool; the wool layer ensures a higher insulating property (Ottelé, 2011).

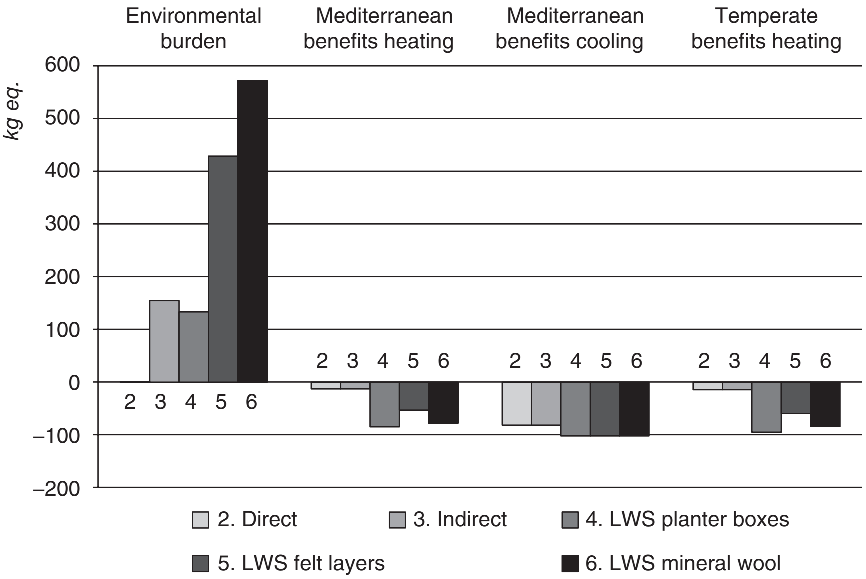
Thanks to the energy saving for air conditioning, the environmental benefits profile for Mediterranean climate is almost double the one for temperate climate. The cooling potential and the extra insulation property due to a green layer have a similar influence on the environmental profile for the living wall systems analysed. For the direct and indirect greening systems, the energy saving for air conditioning has a higher benefit than the energy saving for heating.

19.4.3 Overview

The life cycle assessment presented shows the difference between the greening systems analysed for the environmental burden and for the environmental benefits related to two types of climate (Mediterranean and temperate).

Looking to the environmental burden profiles, the indirect greening system and the living wall systems show a major impact (due to the materials used and the life span) even if, as shown, the environmental profile can be reduced by more sustainable material choice and an integrated envelope design. The environmental burden profile for the living wall system based on felt layers appears to be higher even with a different envelope design, as the durability aspect plays an important role. For the indirect greening system and the LWS based on mineral wool, the outcome can be changed thanks to a different material for the mesh or supporting system, as stainless steel and aluminium both make a high contribution to the profile. In general, the direct and indirect greening systems have a low contribution to the energy savings for heating but, for the Mediterranean climate, a higher influence was noted for the cooling properties of the plants. The materials involved for the living wall system based on planter boxes and mineral wool affect the insulation properties and cause the highest energy saving for heating.

The present study is limited to The Netherlands for calculation of the environmental burden profile with respect to transportation distances. The results about the environmental burden of this study could be projected also to other locations; since the transportation distances could be similar, for example, in many European countries (all the materials in this analysis are commonly available). A difference is found in the benefits calculation. For the Mediterranean climate, less annual energy consumption for heating is needed, so the energy saving thanks to the greening systems has a lower impact on the positive environmental profile; on the other hand, the cooling



19.13 Total environmental burden given for the five discussed greening systems (supporting systems + vegetation), benefits for heating and cooling for Mediterranean climate and benefits for heating for temperate climate.

potential of vegetation plays an important role for the indoor comfort (energy savings for air conditioning).

For the temperate climate, the environmental burden profile is higher than the energy savings for heating for all the greening systems (supporting system + vegetation), except for the direct greening system that is sustainable, considering a system sustainable when the environmental burden is lower than the environmental benefit profile (Fig. 19.13). For the Mediterranean climate, thanks to the energy savings related to air conditioning, the direct greening system is sustainable and also the living wall system based on planter boxes is almost sustainable. For the living wall system based on felt layers and mineral wool in both climate types, the environmental burden profile is higher than the benefits gained for heating and cooling. The environmental burden and the benefits for heating and cooling are calculated both for the service life of the greening systems studied.

As shown in Section 19.3.2., the choice of a different material (hard wood, HDPE, coated steel), as supporting system for the indirect greening system, can lead to a sustainable option for the Mediterranean climate.

In order to carry out this analysis the study relies in part on published data from other green roof and wall research and practice for estimating these effects. This may introduce some bias, and indicates that this work is subject to revision as increasing experience with more and better data obtained from green façade research emerges.

19.4.4 Unquantifiable categories

Other categories may be relevant in green façade applications, but were not included in this assessment either because of a lack of reliable data or incompatibility of the benefit with the tools and categories used in this study. These categories are mainly related to macro-scale ecological and environmental benefits such as:

- increased biodiversity
- human health
- improvement of air quality mainly related to reduction of fine dust levels
- reduction of the heat island effect with regard to the lower amount of heat re-radiated by greened façades and the humidity affected by the evapostranspiration caused by plants.

In this study, only micro-scale benefits are taken into account, such as the energy savings for air conditioning and heating. However, for quantifying the cooling potential, several environmental parameters are involved (humidity, temperature, wind, etc.) and due to a lack of reliable data, the calculation is based on an estimation for all the greening systems and it does not take into account the specific characteristics of each system (living wall systems evaporate a larger amount of water than the systems based on climbers).

19.5 Conclusions

The results from the LCA provide insight into the environmental impact of different greening systems. The energy benefits provided by the greening options make a noteworthy impact in the LCA and are calculated for Mediterranean and temperate climates; for the Mediterranean climate, the benefits calculated are roughly two times higher thanks to the energy savings related to the cooling potential. The materials needed to build up the (green) façades are important (environmental impact) when the energy demand of a building can be reduced or when the multifunctionality of the construction due to the integration of vegetation can be increased. From the presented LCA research, the following conclusions can be drawn.

- The direct greening system has a very small influence on the total environmental burden; for this reason, this type of greening, without any additional material involved, is always a sustainable choice for the cases examined.

- The indirect greening system analysed based on a stainless steel supporting system has a high influence on the total environmental burden; the choice of another material for the supporting system can lead to a sustainable option for the Mediterranean climate (thanks to the energy saving for heating and air conditioning).
- The LWS based on planter boxes does not have a major footprint due to the materials involved, since the materials affect positively the thermal resistance of the system. The environmental burden profile could be further improved by a higher integration within the building envelope (combining functionalities).
- The LWS based on felt layers has a high environmental burden due to the durability aspect and the materials used.
- The LWS based on mineral wool has a high environmental burden due to the materials used, although a higher energy saving could be achieved through better insulating properties.
- Greening the building envelope considering the materials involved, which, as shown, can have a high influence on the environmental profile, and taking into account all the (unquantifiable) benefits, is a sustainable option.

Despite the need for additional resources initially, the direct greening system, the indirect greening system with a supporting system based on hard wood, coated steel or HDPE, and the living wall system based on planter boxes, are the environmentally preferable choices when constructing and retrofitting a building, due to the reduction in energy demand for heating and cooling (this study can easily be applied to other construction types). However, it should be noted that this case study is limited to the façade type, climate and location of the study, but is also dependent on the assumptions that are made inside the analysis. Further research is essential for improving the analysis to confirm or refute the assumptions made in this study, especially for the unquantifiable categories (increased biodiversity, human health, the improvement of air quality, mitigation of urban heat effect).

Besides this, economic benefits are involved and could be estimated in a life cost analysis thanks to durability aspects, aesthetical value and social factors. Some systems, such as the living wall systems described, offer much more creative and aesthetic potential, but due to the material used and durability in some cases cannot be considered as sustainable. Material choice and durability aspects are important (environmental impact) when the energy demand of a building can be reduced or when the multi-functionality of the construction due to the integration of vegetation can be increased.

As suggested by Henry and Frascaria-Lacoste (2012), the adoption of LCA analysis for the labelling of green products could increase their use, since it has the potential to boost the confidence of consumers. Therefore this could lead to particular focus being placed on specific green elements, which could potentially further homogenize natural features within cities, with possible negative impact on other green benefits, such as biodiversity (Henry and Frascaria-Lacoste, 2012). However, LCA analyses could lead to deeper consideration by manufacturers of the environmental burden produced by their systems to improve the balance between benefits and burden for a more sustainable built environment.

19.6 Acknowledgements

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20

Assessing the environmental and economic impacts of cladding systems for green buildings

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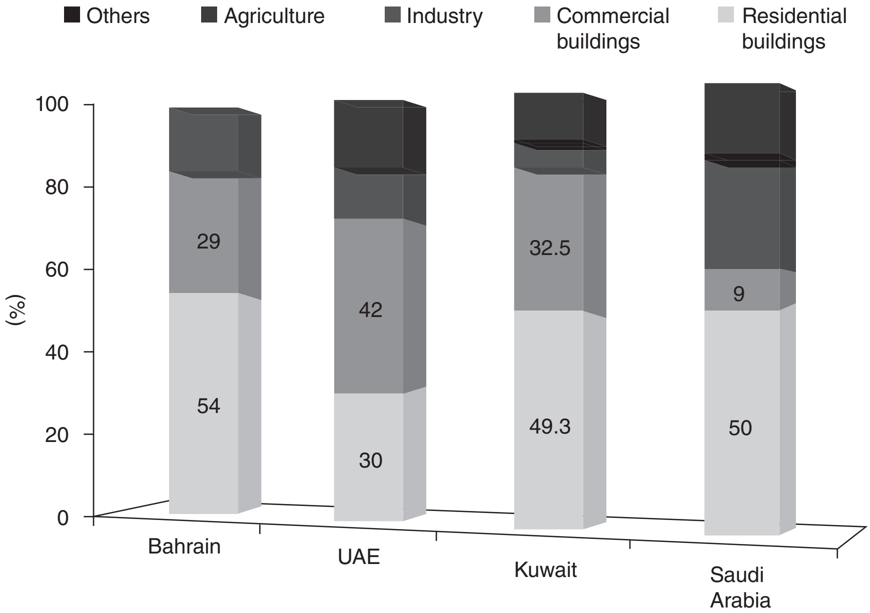
Abstract: This chapter first introduces the role of cladding systems towards a green future in the building sector. It then gives a brief description of the use of the eco-efficiency concept in the assessment of green cladding systems. This chapter presents a systematic assessment of a real case study from Bahrain. The ultimate purpose of this chapter is to provide a common understanding of the importance of carrying out an eco-efficiency assessment and how that assessment can benefit the appraisal of green cladding systems and therefore the design decisions made in developing various scales of green buildings.

Key words: cladding systems, eco-efficiency, green building sector.

20.1 Introduction

The building sector is one of the major contributors to changes in the natural environment. It consumes almost 33% of the world's natural resources, including 40% of its energy and up to 12% of its water (GBCA, 2012). These estimates do not consider embodied energy (i.e., the energy used to obtain, manufacture, use and dispose of building materials), which can represent a significant proportion of a building's lifetime energy consumption. It is estimated that the embodied energy used in the construction of buildings is about 5% of global consumption, while energy consumption for operating buildings accounts for about 45% of the total global consumption (Rigg and Lahav, 2001). The consumption of natural resources by the building sector is an important factor in the economy of many nations. Authoritative reports show such trends in many parts of the world. In the United States, for instance, about 40% of the total national energy production and almost 70% of electricity production are used in the building sector, as well as 28% in transportation – a factor which is partly influenced by urban design. The same scenario occurs in the rich oil-producing areas of the Gulf Corporation Council Countries (GCCC) as can be seen in Fig. 20.1. The building sector in Kuwait, for example, accounts for nearly 81% of the yearly electric energy consumption, whilst in Saudi Arabia this



20.1 Energy consumption in the Gulf Corporation Council Countries.

sector consumes about 59% of the total electricity consumption. In Bahrain, the smallest country within the GCCC, buildings account for 83% of the national consumption of electricity (EIA, 2010).

The building sector is also responsible for 40% of global greenhouse gas (GHG) emissions and 40% of the waste which ends up in landfill (GBCA, 2012). Firstly, building construction, raw material processing and product manufacturing overall are the largest sources of GHGs. They account for some 40% of the world GHG emissions. The building sector creates the most waste, habitat destruction and is responsible for the most pollution. Secondly, GHGs, particularly CO_2, are the main by-product of non-renewable energy consumption, and as buildings are, in total, among the largest consumers of energy, they are also the major contributor to the increase in CO_2 emissions and other GHGs. While most available data related to these contributions are for the developed world, reports show that, on the whole, these contributions are worse in developing countries such as the GCCC. According to the Intergovernmental Panel on Climate Change (2007), the GCCC have become major GHG emitters. Recent statistics show an increase of CO_2 emissions due to excessive energy consumption in different GCCC sectors, particularly the building sector. The increase in CO_2 emissions had been within the range of 30–35% between 1997 and 2006 (United Nations Statistic Division, 2007). The GCCC are found to contribute 2.5% of the global GHG emissions and therefore are major contributors to changes in the natural environment.

To tackle energy and environmental issues, two fundamental changes in patterns of resource consumption are required: first, effective measures to protect the depleted resources and second, valid policies to replace non-renewable energy with free and clean energy (Butt, 1992; Trudenu, 1991). The emergence of green building concepts represents a practical step to conserve natural resources, protect the environment and, more recently, prevent climate change. The implementation of this concept demands advances in many aspects of building design, which can be translated into environmental design, efficiency techniques, innovative renewable technologies and also the appropriate selection of construction materials that can contribute to a building's comfort, energy and economic and environmental performance. The cladding system is a clear example in which significant improvements in the functional, environmental and operational performance of green buildings can be achieved. Much literature has shown that the optimum selection of wall cladding systems has direct and indirect impacts on the appearance, function and performance of buildings. The remainder of this chapter briefly discusses the need for green buildings in the GCCC; introduces the role of cladding systems towards a green future in the building sector; and finally presents a real eco-efficiency assessment of cladding systems from Bahrain.

20.2 The need for green buildings

One of the main principles of the GCCC is to enhance the economic and environmental actions related to the adoption of policies and unifying environmental laws as well as the conservation of natural resources (GCC, 2008). Within this context, a twofold policy which aims at promoting energy regulations and sustainable developments, has been adopted. A major role has been given to the building sector, with a special focus on the important role that efficiency regulations can play in reducing energy consumption and protecting the environment.

20.2.1 Green building initiatives

On the ground, many actions have been taken in the GCCC in order to achieve sustainability in the building sector, such as the implementation of green building regulations. Most of these regulations are based on the USA's Green Building Council's (US GBC) Leadership in Energy and Environmental Design (LEED) rating system, with modifications made to account for the local environmental conditions (Radhi, 2011). In terms of green construction, many initiatives have been made in different parts of the GCCC. Examples can be seen in the Bahrain World Trade Centre in Manama (WTCA, 2010), the large-scale Masdar City in Abu Dhabi (Masdar City, 2010), the campus of King Abdullah University of Science and Technology in Saudi Arabia (KAUST, 2010) and the Energy City in Qatar (ECQ, 2010). These projects incorporate several efficiency techniques and green materials. A consideration of these huge, costly projects shows that three parties can benefit from such developments: governments and owners can save energy and protect the environment, thereby gaining a favourable image; contractors and suppliers can sell green products; and developers can use the affirmative image as a positive marketing tool. However, in his article 'The business of green', Elsheshtawy (2010) claims that some green and LEED certified buildings in the GCCC end up consuming much more energy than the evaluators predicted. Coupled with this is the economics of energy efficiency measures and green materials.

20.2.2 Cost of building green

Even the briefest scrutiny of green buildings shows that if energy efficiency is a target at the outset of the design process and material selection, then the cost of a green building is competitive. In a commercial setting, projects such as those highlighted above can result in reduced energy consumption, saved environment, improved occupant health and comfort and reduced capital costs. Many rigorous assessments show that the overall cost of these projects is no more than that of any equivalent conventional project. Increases in first cost are reported within the range from 10 to 15%. During the construction phase, the use of the green design and technologies, such as downsizing of costly mechanical, electrical and structural systems, can increase the savings in initial costs, while during the first two decades, the increases due to the use of green technologies will result in savings of at least ten times the initial investment in operating costs for utilities such as electricity (Kneifel, 2010). Other perceived cost benefits are seen in the increase of building value, improvement in the return on investment, increase in occupancy ratio and, most of the time, building rent (USGB Council, 2008).

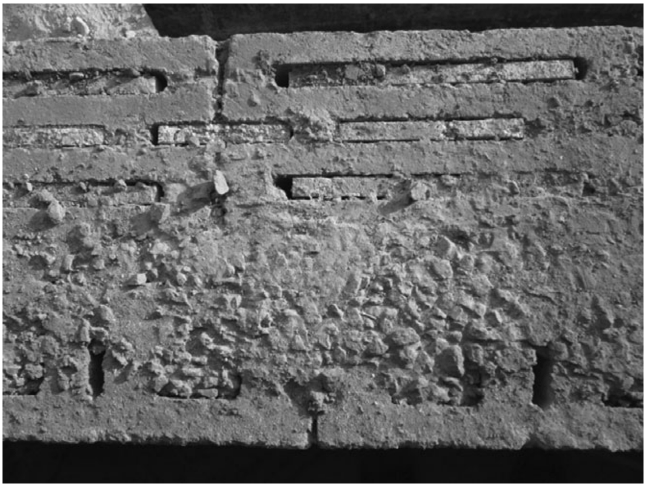
In many rental properties, owners are concerned only with the initial cost, especially in the cases where tenants are paying the bills. Governments and some owners, however, can realise the energy savings and so are willing to pay more for minimising the operation cost and reducing the environmental impact. The trade-off between economic costs and environmental benefits can stimulate governments and people on the basis that adoption of green technologies will have environmental and social benefits outside the margin of cost consideration. Although the concept of eco-efficiency in many cases does not take into account the social benefits (Kolsch *et al.*, 2008), such a concept can balance environmental design with cost-effectiveness. This balance is discussed in Section 20.3.3.

20.3 The role of cladding systems in making buildings green

Today, cladding systems are available in many forms and materials, and are often chosen based on economics and aesthetics. For any successful design of cladding systems, however, two criteria are of importance, namely design and performance criteria.

20.3.1 Design of cladding systems

Technically, the cladding system consists of two main components: first, the wall system and second, cladding layers. The wall system is generally classified as a cavity wall, barrier wall or mass wall (NIBS, 2012). The cavity wall (sometimes called the screen wall system) is the preferred method of construction in many climatic regions due primarily to its ability to achieve pressure equalisation. The barrier wall is an exterior wall system of assembly. The principal difference of this system is its ability to integrate the surfaces of the outermost exterior walls and construction joints, which can offer resistance to bulk moisture ingress. Figure 20.2 shows a mass wall which relies principally upon a combination of wall thickness and storage capacity. The most important advantage of this type is the thermal mass effect, which is long lasting, low maintenance, reusable and recyclable. Many



20.2 Mass wall systems.

20.3 Stone cladding layers.

critical differences exist among these types of walls, such as their fire safety, moisture protection, thermal performance, acoustics, material durability and maintainability, and so consequently their impact on the energy and environment also varies.

Cladding is the exterior finish layer that is installed to cover exterior walls and/or support structures, as shown in Fig. 20.3. This finish layer serves several functions, including improving the appearance, optimising thermal and environmental performance and keeping undesirable outdoor elements away. With the advances in building technology and construction materials, many alternatives of cladding systems are now available in the market. Examples are studied in the current work, namely, stucco, masonry veneer, marble, ceramic tile and the exterior insulation and finish system (EIFS). Stucco is a hard, dense, thick and non-insulating material, such as the plasterwork, that can be used for surfacing interior and exterior walls. Physically, it is similar to concrete and consists mainly of Portland cement or masonry cement with sand, lime and water. Unlike the ordinary stucco system, the EIFS (also known as synthetic stucco) is a lightweight synthetic wall cladding that includes foam plastic insulation and thin synthetic coatings. The masonry veneer is made from a mixture of Portland cement and aggregates under controlled conditions. It provides cladding and resists transferring wind and heat loads to the building support structure. The marble cladding system is a natural stone, while the ceramic tile cladding system consists of a mixture of clay and other ceramic materials. To improve the thermal and environmental performance, recycled windshield glass is often added to the ceramic mix (Brookes and Meijs, 2008).

In façade design, the use of any alternatives of cladding determines the type of wall system and vice versa (Smith, 1988). If a particular cladding is selected, therefore, the type of wall systems and support structure best suited to the functional requirements of the building are at hand (Chudley and Greeno, 2006). The mass wall system, for example, can form structural elements and/or finished cladding systems. This system is commonly associated with plaster and masonry cladding systems. Whereas, the barrier wall is used with pre-cast concrete spandrel panels and some types of metal cladding systems such as composite and solid metal plate as well as with exterior insulation and finish systems (EIFS). Such design can affect the performance of cladding systems and the building as a whole. Simultaneously, the required thermal properties and environmental characteristics of cladding systems can affect the building design. The next sub-section discusses how the design of cladding systems can affect the thermal and environmental performance of green buildings.

20.3.2 Performance of cladding systems

Green buildings are generally designed and built in an ecological and resource-efficient manner. They often respond to their local environment, and hence different building designs are available in different climatic regions. In any region, however, the ultimate objective of buildings is to provide comfortable conditions in an economic way. The building's envelope, particularly the cladding system, represents the connection between the indoor environment and the outdoor conditions. Thermally, a key function of the cladding system is to reduce the need to modify the indoor environment as little as possible in response to the environmental load from the outdoor climate (Radhi *et al.*, 2009). The cladding system fails to meet its objective due to the lack of environmental design of wall systems or the inappropriate selection of cladding materials that probably make it impossible for any specific level of comfortable environment to be achieved. Then, it is necessary to rely upon electrical and mechanical systems to achieve comfort. This reliance leads to higher cost which is translated into bigger capacity requirements for lighting and mechanical equipment and higher capital costs for such equipment as well as larger amounts of energy consumption by lighting and air-conditioning systems.

In contrast, efficient environmental design and appropriate selection of green cladding materials can result in a comfortable inside environment, reduced project initial and running costs and a building that is energy- and resource-efficient with lower operating costs than conventional buildings. Practitioners have demonstrated that the implementation of environmental design and green technologies contributes to a building's comfort, economy and energy consumption (USGB Council, 2008). The use of green cladding systems, in particular, is able to make a significant impact on the thermal and operational performance of green buildings. Reports show that when green cladding systems are taken into account at the conceptual design phase, significant improvements (25–35%) in the energy performance can be achieved (Radhi and Sharples, 2008).

In addition to their influence on building performance, the cladding system has a significant impact on the natural environment. The production of cladding materials such as pre-cast and aluminium increases atmospheric concentrations of GHGs. The environmental impact starts with the chemical reactions during the production phase, where such materials represent one of the largest sources of CO_2 emissions and other GHGs. Then, the transportation of the materials to construction sites consumes considerable amounts of primary energy and generates high levels of GHG emissions. At the installation phase these materials generate different types of waste, whilst at the operation phase some of them influence the interior and exterior spaces by producing unhealthy components in the air. Some construction materials have relatively short useful lives and, consequently, the disposal and manufacture of replacement materials occur, thereby generating more GHGs and waste. Research experts have shown that a careful selection of low environmental impact materials reduces the CO_2 emissions by up to 30%. Some cladding materials are reported to have the capacity to reduce ozone emissions and other sources of pollutants (Radhi, 2010).

After having raised the issue of design and performance of cladding systems, the next task is to show various approaches to measure the eco-efficiency of cladding systems. The main thrust of the research into eco-efficiency as described in this chapter is to enable an understanding of the outcome of carrying out an eco-efficiency assessment and how that assessment can benefit the appraisal of green cladding systems.

20.3.3 Measuring the eco-efficiency of cladding systems

The World Business Council for Sustainable Development (WBCSD, 2000) terms eco-efficiency as the synthesis of economic and environmental efficiency in parallel. Within this context, eco-efficiency in the building sector can be determined by three broad objectives:

1. Reduce natural resource consumption by minimising the use of embodied and operational energy, raw materials, water and land as well as enhancing material durability and recyclability.
2. Reduce environmental impact by minimising GHG emissions, waste disposal and water discharges as well as encouraging the use of renewable resources.

3. Increase the value of materials and systems by providing more benefits through material functionality, flexibility and modularity.

In the light of these objectives the important question is how the eco-efficiency of cladding systems can be measured. Significant scientific work has been addressing this issue by introducing suitable assessment methodologies and rating systems (Seppäläa *et al.*, 2005; Michelsen *et al.*, 2006). This is best seen in the environmental life cycle assessment (LCA) and life cycle cost (LCC) approaches developed by the international standards for LCA principles and framework – ISO 14040 (ISO14040, 2006). An assessment is performed in four phases, including goal and scope definition, inventory analysis, impact assessment and interpretation.

Two main approaches are available to classify and characterise environmental impacts. The first is the problem-oriented approach (mid-point). The second is the damage-oriented approach (end-point) (Blengini and Carlo, 2010; Hermann *et al.*, 2007; Dreyer *et al.*, 2003). A great number of methods have been developed under these two approaches (Singh *et al.*, 2009) such as the critical volumes (weighted load) and ecological scarcity (eco-points) systems in Switzerland, environmental priorities system in Sweden, Eco-indicator 99 in the Netherlands and the environmental problems system in the United States. The use of such methods makes it possible to select cladding systems and finishing materials that reach the most appropriate balance point between economic and environmental efficiency based on certain values of the project team.

20.3.4 Achieving eco-efficiency using the whole building approach

To achieve building eco-efficiency, it is necessary to apply an integrated approach with the assistance of a team of professionals across different areas. This is realised in what is called the whole building approach (NIBS, 2012; Pedhazur and Schmelkin, 1991). This approach represents a key factor in the design and construction of green buildings, especially with the advance of technology and increased complexity of construction systems. The incorporation of the whole building approach at the project's conceptual design phase enables the evaluation of a building's design, materials and systems from the perspectives of all the project team members as well as from the perspectives of owners and occupants. A principal advantage of this approach is the coordination and mutual dialogue between project team members, which represent a cornerstone for any successful projects. By applying the whole building approach, initial and other cost savings can be realised, energy efficiency evaluated and environmental impact assessed.

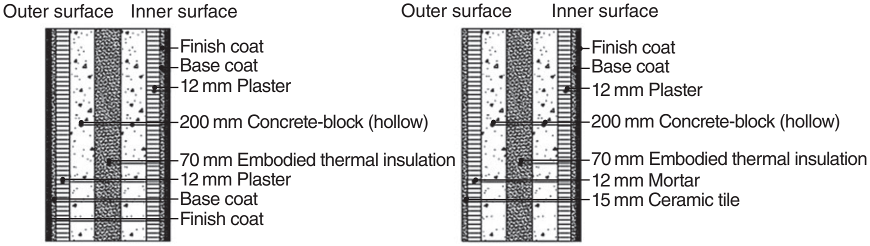
20.4 A typical Bahraini residential building.

20.4 Implementation: assessing the eco-efficiency of cladding systems in Bahrain

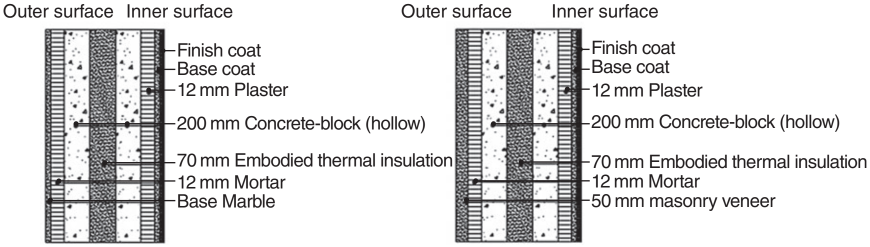
The current assessment, based on the LCA of residential buildings (Radhi and Sharples, 2013), is performed to characterise the eco-efficiency of cladding systems in Bahrain. Bahrain is chosen, as many of its building construction approaches and techniques are typical of those found in the GCCC. The production, construction, use and disposal of a 95 m^2 front façade of a typical Bahraini residential building (Fig. 20.4), formed the basis of this assessment. Local materials represent the strong focus for the investigated façade scenarios because imported materials consume more energy due to transport. To provide each scenario with the basic system's quantities per functional unit, the existing façade parameters and wall materials of the studied building are considered as a reference scenario, in addition to the operational aspects that are influenced by the building façade. Five available cladding systems are assessed under real construction and thermal scenarios with the same wall system (mass wall) illustrated in Fig. 20.5.

20.4.1 Life cycle inventory

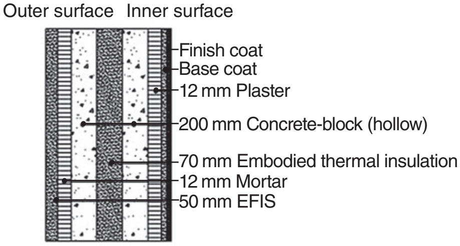
The LCA method and LCC technique are integrated to deliver a complete and detailed assessment of the overall potential impact of the examined cladding systems. An important point to note is that system



(a) Plaster cladding (b) Tile cladding



(c) Marble cladding (d) Masonry cladding



(e) EFIS cladding

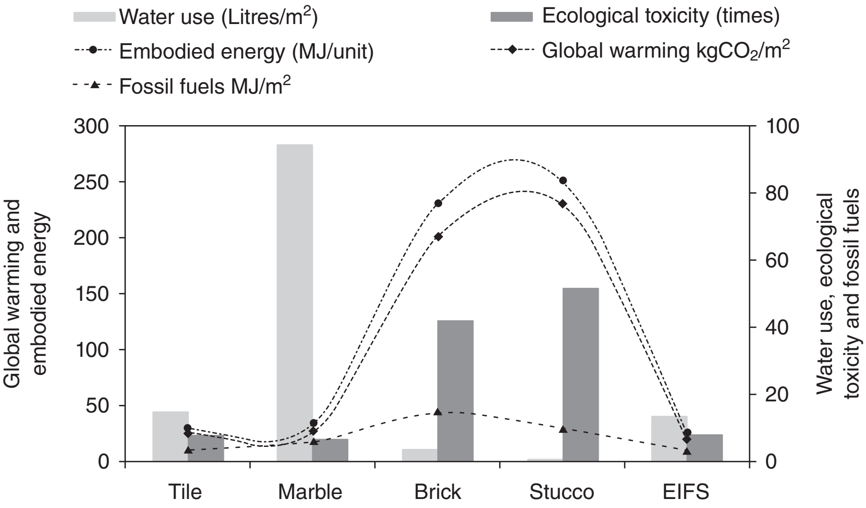
20.5 Details of different cladding systems, (a) plaster cladding, (b) tile cladding, (c) marble cladding, (d) masonry cladding, and (e) EFIS cladding.

and material selection based on a single impact could obscure other factors that might cause equal or greater damage. Therefore, the adopted LCA methodology takes a multidimensional life cycle approach, in which multiple environmental impacts are considered over the entire life of the assessed cladding systems. To balance the assessment, the LCC is performed over a 60-year life span, and is based on published data and methods outlined in Radhi (2010). Categories of expenditure typically include costs for purchase, installation, maintenance, repair and replacement. Measuring the economic performance is relatively straightforward by using real cost data collected through a field survey. The data in question are the real cost data that occur and the subsequent cost, which will occur in the future.

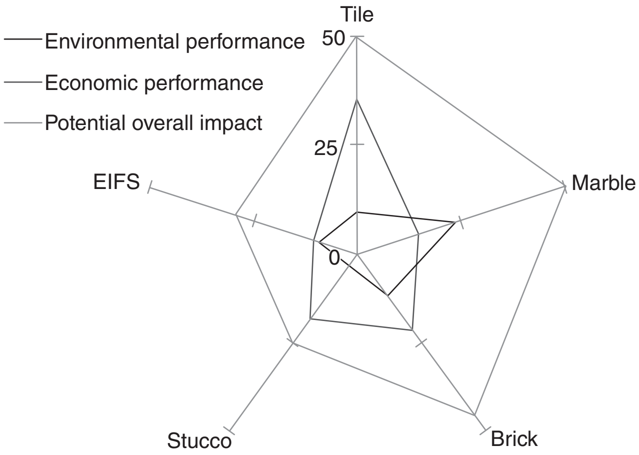
Normalisation is carried out in this study in order to present a more useful scale of measurement and to simplify comparisons of various systems. Normalisation is an optional step in impact assessment and can be described as a form of benchmarking (Kicherer *et al.*, 2007), where the flows of each environmental impact are first summed and then divided by fixed Bahraini scale impact values. This can yield measures that are placed in the context of Bahraini activity contributing to that impact. The placing of each measure in the context of its associated Bahraini impact measure makes it possible to reduce different values to the same scale and allows comparison across impacts. The resulting performance measures are, thus, expressed in non-commensurate units. For credibility, the commercially available BEES model (NIST, 2007) for building construction materials coupled with the international inventory data (Hammond and Jones, 2011) were used to compare and check. The BEES model is generally used to measure the environmental and energy performance of building products and façade materials using the life cycle assessment approach outlined in ISO 14040.

20.4.2 Environmental impact assessment

Given the desire to link environmental and economic performance through the concept of eco-efficiency, the ideal way is to base the eco-efficiency indicators on international agreement as far as possible. According to the framework of the United Nations (2004), the assessment of eco-efficiency includes various generic environmental issues such as energy consumption, global warming contribution, water use, ozone depletion and waste. From these indicators, energy consumption and CO_2 emissions, water use and ecological toxicity are of the greatest relevance for this study. Figure 20.6 compares these indicators with respect to the five studied cladding systems. Some of these systems, such as the marble cladding, have significant impacts on water use but moderate impacts on global warming and embodied energy. Other systems, such as stucco, have significant impacts on both the energy consumption and global warming but moderate impacts on water use. The others, such as the EIFS, have moderate impacts on different generic environmental issues. From the illustration, the EIFS system seems to be the best performer, followed by the ceramic tiles, marble and finally the brick. Stucco is found to be the least effective system in terms of energy consumption and ecological toxicity as well as in relation to CO_2 emissions. This can be related to the large amounts of CO_2 emissions during cement production, which is the main component of the plaster cladding system.



20.6 Environmental indicators of wall systems.



20.7 Environmental and economic indicators of wall systems.

20.4.3 Environmental versus economic impact

When the overall environmental impact of the examined systems is considered, a different scenario appears. The overall environmental performance is illustrated in Fig. 20.7. Two main observations can be highlighted: firstly, the overall environmental performance ranking of the five systems is different from single measures such as energy consumption and global warming. The EIFS cladding system is the best environmental performer, whilst the ceramic tile system is the worst performer. The difference is more than 24 points. As systems with lower scores are greener, the EIFS cladding system is greener because it contributes, on average, 0.1% of annual per capita Bahrain environmental impacts, whilst the marble contributes a larger share, 0.35%. Secondly, the environmental performance ranking is different from that of the economic performance. The illustration shows that the economic impacts of cladding systems are various and different from the environmental impacts. For example, the stucco cladding is illustrated as the best economic performer, but it is not in terms of the environmental performance. The difference in score is significant, being almost 11 points. This can also be seen in the case of the ceramic tile cladding. In contrast, the marble cladding achieves a high overall environmental performance and a low economic performance with a difference that reaches almost 21%. The EIFS cladding seems to have a balanced environmental and economic status. The same ranking occurs when both environmental and economic performance are estimated.

By using the multi-attribute decision analysis technique, environmental indicators and the economic efficiency are combined into an overall performance measure (NIST, 2007). It is important to mention that the overall performance scores in this work are not indications of absolute performance. Rather, they reflect proportional differences in performance and represent relative performance among system alternatives. By following this procedure, these scores can be changed when the number of system alternatives is increased or reduced. The potential overall performance of the studied systems shows different scenarios when compared with the environmental and economic performances. The stucco cladding seems to be the most eco-efficient system in spite of its poor environmental performance, followed by the EFIS system with a score of 29%, with the masonry veneer coming next. In contrast, the ceramic tile cladding is found to be the worst with almost 50%, in spite of its moderate economic performance.

In short, different cladding systems have different environmental and economic performances. Some cladding materials improve the environmental performance, but provide a moderate impact in terms of economic performance, and vice versa. Others positively improve the environmental performance and can optimise the economic performance. Therefore, a careful eco-efficiency assessment should be undertaken in selecting wall cladding systems. Such an assessment can benefit the appraisal of green cladding systems and therefore the decisions made in developing various scales of green buildings.

20.5 Interpretation and conclusions

Today's modern building systems, particularly the cladding system, are often selected and assessed based on aesthetics and cost rather than their environmental performance or their overall potential impact. The concept of eco-efficiency introduced in this chapter balances the environmental performance with economic aspects. This chapter presented a systematic eco-efficiency assessment of cladding systems and explored its role in progressing a green future in the building sector. The interrelation between environmental indicators and economic performance was examined by comparing various cladding systems, considering both overall environmental impact indicators and life cycle cost. The differences in environmental indicators of various cladding systems, namely, stucco, masonry veneer, marble, ceramic tile and the EIFS system, are generally significant. The ranking of these systems in terms of environmental and economic performance is different. Some of the cladding systems, such as the marble cladding, reduce energy consumption and CO_2 emissions, but lead to a small reduction in terms of the life cycle cost, and vice versa. Others, such as the EFIS system, impact positively upon the environmental indicators and can optimise the overall potential impact. This system has the ability to reduce energy consumption and CO_2 emissions; however, other aspects, such as maintenance and life expectancy, should be considered at the time of system selection.

The scope of the current study focused on the eco-efficiency of representative residential cladding systems in a developing country. Consequently, the outcome of this assessment may not be applied to buildings in countries with different economic and environmental situations. In spite of this shortcoming, this assessment approach may provide useful quantitative and qualitative information for cladding design decisions. Therefore, it is important to highlight some general notes:

1. New green building technologies, such as the exterior insulation and finish systems (EIFS), are effective cladding systems in promoting a green future in the residential building sector.
2. To improve the overall potential impact, wall cladding systems in desert climate regions, such as Bahrain, can be designed as exterior insulation and finish systems.
3. Every building is unique in both design and operation. Academic experts and practitioners benefiting from this work should consider the impact of related variables, and therefore a careful assessment must be performed during the selection process in order to achieve eco-efficiency in the building sector.

In addition to its ability to assess building cladding systems, the eco-efficiency concept can be used with various building systems, materials and innovative applications. It can yield a more precise assessment in the case of multifunctional problems in relatively short times and at relatively low cost. In the near future the concept of eco-efficiency will become more important in the context of the green built environment in order to show which design processes, building systems and renewable technologies are more favourable than other alternatives.

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21

Life cycle assessment (LCA) of windows and window materials

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Abstract: Windows are a significant component in sustainable buildings in both the impacts caused by their material life cycles and by their influence on the performance of a building over its service life. Life cycle assessment (LCA) studies have compared the impacts of different framing materials with mixed results. LCA has also been used to estimate the environmental payback of higher manufacturing impacts from producing better performing windows. Future sustainable window selection should make use of standardized LCA data for windows and utilize advanced technologies to optimize window performance.

Key words: green windows, sustainable windows, windows LCA, windows EPD, LEED windows.

21.1 Introduction

Window selection is one of the most important decisions one must make in the design and specification of a sustainable building. Windows, which typically account for 10–25% of a building's exposed surface (Recio *et al.*, 2005), provide the unique function of light transmission from the outside environment into the interior space. Windows account for much of a building's character and are considered critical to the livability of homes and productivity of commercial and institutional building occupants.

Light and heat transmission through windows also significantly change the thermal properties of a building. The addition of natural light has the ability to reduce supplemental heating and light requirements but also causes increased cooling requirements in summer months. Windows typically provide less insulation (reported as conductivity U-factors for windows which has an inverse relationship to the R-value) than the rest of the building envelope which causes additional heat loss in winter and heat gain in summer. For these reasons, window orientation and material selection is often a key part of conceptual design that seeks to minimize the operational energy use.

In addition to the noted performance characteristics of windows, the selection of the materials also causes significant upstream impacts in the material sourcing, manufacturing, and delivery of the product. Additionally, the choice of materials also has a strong influence on the longevity of the window's service life and its fate at the end of this service life. The material impacts are accentuated in the case of windows because they are typically high-value products that are subject to greater engineering and manufacturing than other building materials which leads to disproportionate impacts relative to their mass and square footage.

The interdependence of the impacts caused by windows in the various stages of their life cycle makes them a perfect candidate for life cycle assessment (LCA). LCA is a structured accounting method for considering the impacts of products to preclude burden shifting between the various life stages. Because sustainable window selection must consider the entire life cycle of the product to be effective, this chapter will draw heavily on the various LCA studies of window materials.

21.2 Modern window construction

21.2.1 Anatomy of a modern window

Modern windows are complex products that have evolved to provide the greatest possible energy performance over their service life. The most significant element in a modern window is the use of insulated glazing units (IGU), which are multiple panes of glass separated by spacer bar to create a gas filled void. This spacing increases the insulation properties of the windows beyond what is possible with a single pane. The glass panes may also receive a low-E coating which filters low-frequency radiation to further increase the insulation properties. In the case of operable windows, those that open, the glazing is mounted inside a sash which serves as a frame within a frame. In fixed windows, the glazing unit is mounted directly to the frame.

The following lists the components of a modern window and their function:

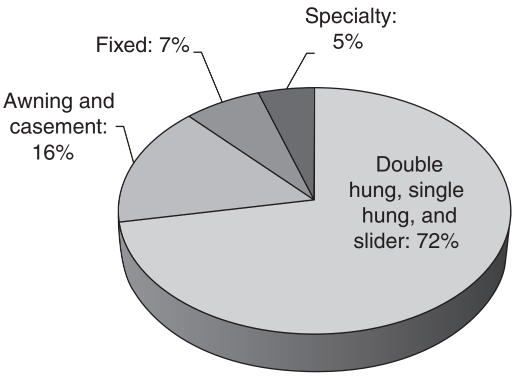
- ***Frame and sash:*** The frame and sash provide the structural integrity of the window by supporting the glazing unit(s).
- ***Double (triple) glazing:*** Two and sometimes three glass panes are used to create a gap between surfaces exposed to external temperatures and those contacting climate controlled environments.
- ***Low-E coating:*** Glass surfaces are available with low-E coatings that allow high frequency sunlight to permeate, but trap lower frequency radiant heat. There are two types of low-E coatings available: 'hard-coat' and 'sputter-coat'. Hard-coat glass is manufactured by applying molten tin to the glass surface as the glass sheets are being manufactured. Sputter-coat glass uses vacuum deposition to apply a thin metallic (often silver) coating to the glass surface as an additional manufacturing step.

- ***Gas fill:*** Air or inert gases such as argon, xenon, and krypton provide a layer of insulation between the multiple glass panes. Argon-filled units are more common in residential applications, while air-filled units are more prevalent in commercial windows.
- ***Spacer bar:*** Spacer bars are used to separate multiple panes in insulated glazing units. Spacers have typically been manufactured as a perforated aluminum rectangular tube with desiccant held inside to reduce condensation. Hard foam, plastic, and steel are also commonly used.
- ***Weatherstrip:*** Rigid and flexible polymers are used to create a weatherproof seal between the fixed frame and operable sash.

21.2.2 Window types

Several window types are available to the window consumer and include both fixed (those that may not be opened) and operable (those in which the IGU is mounted to a sash which moves relative to the frame so that the window may be opened). The most common window types include the following.

- ***Fixed:*** The most basic type of window is the fixed (also called 'picture') window. This type consists of a singular frame around a transparent glazing unit that remains in one position and cannot be opened.
- ***Awning/casement:*** The awning/casement window swings open on either the horizontal axis (awning) or vertical axis (casement). These types of windows include an external frame, an internal sash that moves with the IGU, and a hardware kit that includes a crank, extension mechanism, stop locks and hinges. This type of window also requires rubber weatherstrip to prevent weather permeation between the frame and sash.
- ***Double hung/sliding/single hung:*** A second type of operable window is called double hung (also called sliding when horizontally oriented) and contains two sashes within a frame; both sashes (or only one, in windows called 'single hung') move within the frame to slide over the other, such that half of the window may be opened without the sash extending into or out of the building. The double hung window differs from the casement in both hardware and frame, which must be deeper to accommodate the stacked sashes.
- ***Curtainwall:*** This type of fixed window is typical in commercial construction and consists of a mullion system that extends from floor to ceiling containing a glazing unit spandrel panel. The structural requirements and application limit the materials that may be used for the mullion and the type of glazing that is installed. This type of window/glazing is typically fabricated on site.



21.1 Market share of different window types used in North American residences (Ducker, 2006).

The market share of each window type in residential applications is shown in Fig. 21.1. Double-hung, single-hung, and slider windows make up 72% of all residential windows sold in 2005.

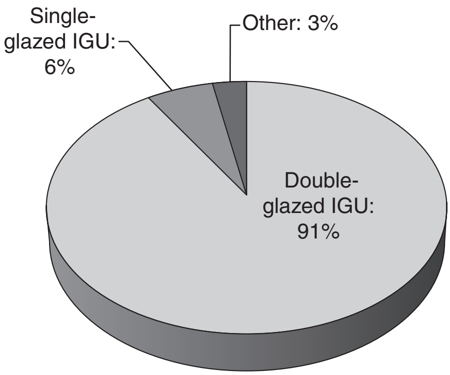
21.2.3 Insulated glazing unit

Insulated glazing units, as noted in the previous section, comprise two or more glass panes that are separated by a spacer that runs along their perimeter to create a gas-filled cavity. The entire unit is sealed with various compounds specific to the spacer bar type; the most common system is an aluminum spacer sealed with a dual-compound system (polyisobutylene and silicon). The glazing units are then set into the window frame (into the sash in operable windows) and locked into place with an additional frame piece called a glazing stop.

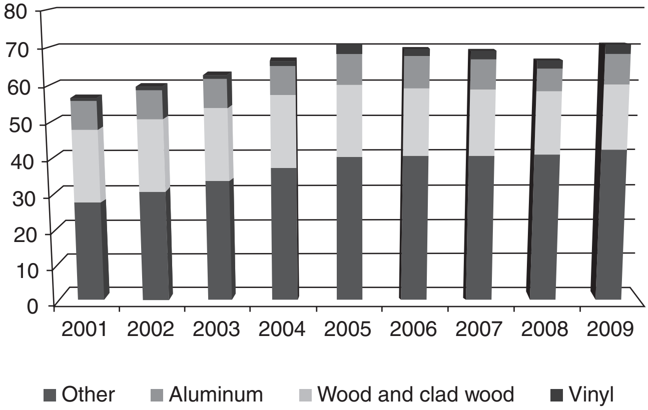
Double-glazed IGUs have significantly improved performance over traditional single pane windows to the point that nearly all windows currently manufactured are double glazed (see Fig. 21.2). The natural extension of this trend was the creation of triple- and quadruple-glazed units that have an additional insulation break between the interior and exterior surfaces. Triple- and quadruple-glazed windows are rarer as the additional glass is subject to diminishing performance returns and increases the weight and subsequent framing required.

21.2.4 Frame type

Different materials are used for window frames based on the preference given to their inherent characteristics in that application. While cost is a driving issue in both residential and commercial construction, the value



21.2 Market share of different glazing types used in North American residences (Ducker, 2006).

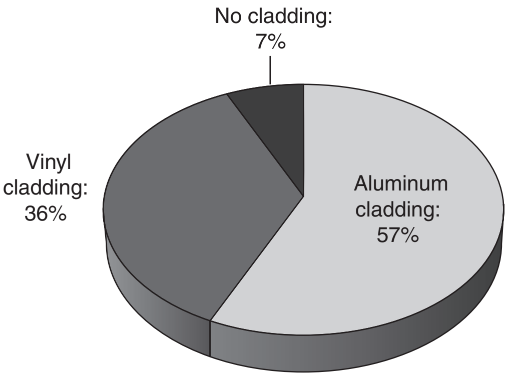


21.3 Market share of frame types used in North American residences from 2001 to 2009 in millions of units sold (Ducker, 2006).

given to aesthetics, ease and lack of maintenance, and thermal performance varies amongst purchasers.

Primary material

Traditionally, wood has been used as the primary material for window frames used in residential construction with metals dominating commercial applications. The advent of vinyl frames that typically cost less to manufacture and require less maintenance has led to their dominance in the residential marketplace. Figure 21.3 shows the relative market share of various frame types in the residential sector.



21.4 Market share of different cladding types used on wood windows (Ducker, 2006).

Cladding

Protective cladding is employed on more than 90% of all wood window frames. Aluminum and vinyl are the most popular cladding materials, accounting for 57% and 36%, respectively. Figure 21.4 illustrates the relative market share of different cladding types used on wood windows.

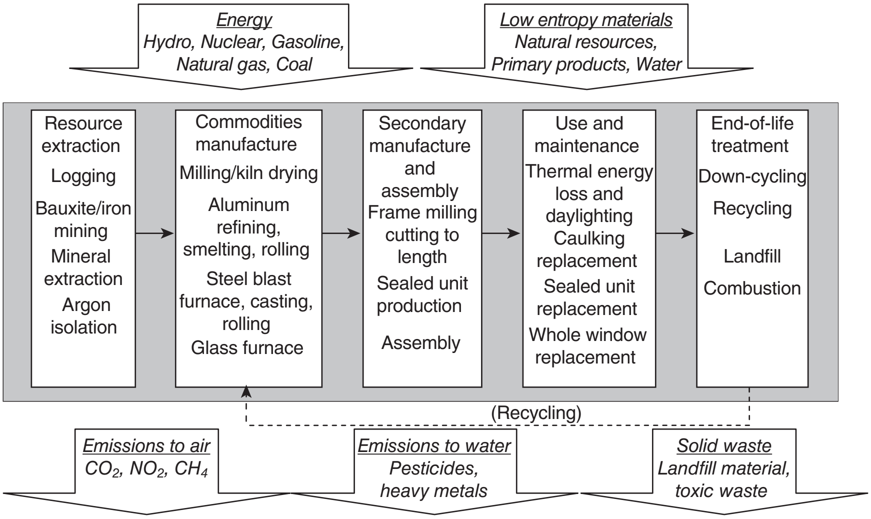
21.3 The life cycle of a window

The life cycle of a window is similar to the life cycle of any building product. The life cycle begins with the extraction of raw materials from the natural environment. The source of these raw resources reflects the materials with which the window is manufactured. Resource extraction causes impacts to local ecosystems, depletes stocks of non-renewable resources, requires energy, and causes waste. The nature and quantity of these impacts include those associated with mining ore for metal production, oil and natural gas extraction required for plastics, logging for wood used in frames, and the quarrying of silica sand to manufacture glass.

After extraction, the raw materials are then shipped to large-scale manufacturing facilities where they are converted to standardized inputs for use in window manufacturing or one of any number of other products. Logs are milled into rough lumber; natural gas is ‘cracked’ to produce ethylene and then combined with chlorine (which is the product of sodium chloride electrolysis) to produce PVCu, referred to as ‘vinyl’; iron ore and bauxite are smelted to produce steel and aluminum; and silica is melted in furnaces to produce float glass.

The basic commodities are then shipped to window manufacturers where they are extruded, milled, cut, bent, and fastened to produce the various components used in modern windows. These manufacturers may specialize in producing just the frames, glazing units, or hardware. They may simply fabricate a finished window with purchased components, or the manufacturers may be vertically integrated and manufacture most of the components in-house in addition to producing a finished window ready for installation.

After secondary manufacturing, the window is shipped to the installation site where it is installed in the structure. Careful installation is critical to the longevity and performance of a window over time. Similarly important is the ongoing maintenance (e.g., caulking) that is required to ensure the integrity of the air and water barrier around the window. Eventually, either due to failure of the window, demolition of the building, or the availability of better performing windows, the windows reach the end of their service life. At this point it is common for windows to be ‘down-cycled’ for use in greenhouses or buildings with lower performance requirements. The most common treatment for windows, however, is to send them to landfill, as recycling of the various components is hindered by the relatively low volumes of recycled material present in windows, with the potential exception of the metal portions. Figure 21.5 shows the life cycle of a window and its related product system.



21.5 The life cycle of a window (Salazar and Sowlati, 2008b).

21.4 Previous window life cycle assessment (LCA) studies

A literature review of window LCA studies (Salazar and Sowlati, 2008b) found that window LCA research has been completed on a case study basis to answer two primary questions:

- What window frame material has the lowest manufacturing impacts?
- What is the incremental environmental payback associated with additional window manufacturing inputs relative to their long-term performance and energy savings?

The first type of window LCA studies, those that consider the impacts of the window frame materials in isolation, narrows the scope for one of two reasons. First, the studies may be conducted to simply increase the available literature on the embodied impacts of various building products. The results of these studies may be used as a component to calculate whole building embodied impacts. These whole building material studies may or may not be subjected to energy simulation to consider the trade-offs of specifying different materials in the various applications.

Second, the use-phase may be excluded in window LCA studies when the various windows are assumed to have similar performance. This rationale is similar to the exclusion of common elements in many LCA studies. Since windows are complex products constructed of numerous materials, the studies are often streamlined for the sake of feasibility to highlight the relative differences between various window types. This streamlining, however, limits the ability of the various LCA studies to estimate the overall scale of impacts attributable to window selection. Additionally, window LCA studies that focus on the energy performance also typically limit the impact assessment portion to fossil energy consumption and greenhouse gas emissions and provide no information as to environmental or human health impacts commonly calculated in LCA.

Since the publication of the aforementioned literature review on windows in 2008, several window LCA studies have been published and these have been added to the literature review presented here. Table 21.1 lists the scope of the various LCAs that have been conducted on windows and glazing systems and the various life stages considered in each study.

21.4.1 Comparative LCA of window frame materials

Asif *et al*. (2002) conducted one of the first comprehensive LCA studies of window frame materials. In their study they compared the embodied impacts of aluminum, wood, vinyl, and aluminum-clad wood window frames. This study also estimated the service life of the various frame types by

Table 21.1 Published LCA studies of window materials (Salazar and Sowlati, 2008b; updated in 2013)

Study	Goal	Functional unit	Life cycle stage considered in study			
			Resource extraction and commodities manufacture	Secondary manufacture	Use, maintenance and replacement	End of life
Weir and Muneer 1998	Consider relative impacts	Double-glazed wood window	Argon, Krypton, Xenon, Wood, Aluminum, and Glass	Sealed Unit, Wood frame, Aluminum flashing, Spacer, and Hardware		
Citherlet *et al.* 2000	Compare frame materials & justify energy payback	Numerous window systems	Float Glass, Fiberglass, Argon, Aluminum, Thermoplast, Wood, Plywood, PVC	Wood, PVC, and Aluminum frames, Sealed unit, Blinds, and Finished window system	Replacement, thermal performance	
Entec 2000	Compare frame materials	Wood and PVC window frames	Wood, Paint, PVC, Steel, Aluminum, TBTO Preservative	Wood and PVC frames	Thermal performance	Landfilling
Asif *et al.* 2002	Compare frame materials	Aluminum, PVC, wood, and clad frames	Aluminum, PVC, and Wood	Wood, Aluminum clad wood, PVC, and Aluminum frames,	Maintenance and replacement	
Kiani *et al.* 2004	Justify energy payback	Fully-glazed commercial building envelope	Glass, Argon, and Krypton	Tinted and Reflective sealed unit	Thermal and lighting performance, replacement	Recycling

Recio *et al.* 2005	Compare frame materials & justify energy payback	Five window types	PVC, Steel, Glass, Aluminum, and Wood	Aluminum, Wood, and PVC frames	Thermal and permeation	Landfilling
Syrrakou *et al.* 2005	Justify energy payback	Electrochromic device	K-glass, Tungsten Oxide, Lithium Perchlorate, and Silicone	Electrochromic device	Thermal performance	
Salazar and Sowlati 2008a	Compare frame materials	Aluminum clad wood, PVC, and fiberglass windows	Aluminum, Wood, PVC, Fiberglass, Glass, and Steel	Al Clad Wood, PVC, and Fiberglass frames, Sealed units, Finished windows	Maintenance and replacement	Landfilling
Blom *et al.* 2010	Compare frame materials and maintenance regimes	Softwood, hardwood, and PVC windows	Azobe, Spruce, PVC, Alkyd in Solvent and H2O	Softwood, Hardwood, and PVC windows	Stripping, painting, replace hinges, locks, and sealant	Incineration and landfilling
Tarantini *et al.* 2011	LCA to inform green public procurement policy	Wood frame windows	Wood, Preservative, Adhesives, Glazing, Paint	Wood frame, Double glazing	Thermal performance and maintenance	Landfilling and recycling
Sinha and Kutnar 2012	Compare frame materials and thermal properties	Wood, PVC, and aluminum frame windows	Wood, PVC, Aluminum, Glazing, Chromium steel, PET, Nylon	Wood frame, Aluminum frame, and PVC frame windows	Thermal performance	Incineration and landfilling

conducting accelerated aging tests and a survey of industry experts so that the embodied effects could be normalized against the number of times a given window would be replaced.

In Asif *et al.*'s study, wood window frames were found to require the least energy to manufacture (995 MJ) but had the second shortest estimated service life (40 years). The service life of wood windows was improved with the addition of protective aluminum cladding, increasing the service life to 47 years, which was the longest of any of the windows considered in the study. This increased the embodied impacts to 1,460 MJ, an increase of roughly 50% to give the window an additional 10% service life. The aluminum window required the most energy to manufacture, 6,000 MJ, or 6 times that of the wood window, and was unaffected in all accelerated aging tests (service life 44 years). The use of recycled aluminum reduced energy consumption by greater than 90%, which was the lowest of any of the windows that were considered. The vinyl windows had roughly the same normalized impact as the virgin aluminum windows with a relatively high embodied energy (2,980 MJ) and a significantly shorter service life than all of the other window types (24 years).

A more recent study (Salazar and Sowlati, 2008a) of windows also considered various frame materials. In this study, the cradle-to-grave impacts of aluminum clad wood windows were compared to fiberglass and steel-reinforced vinyl windows. Energy performance during use was excluded as it was assumed to be similar amongst the three types. The LCA included the entire window systems (including the hardware and IGU) and found embodied impacts to be generally similar with no one window having the lowest impacts in the 15 impact categories that were considered. In terms of global warming potential, the aluminum clad wood window was lowest (341 kg CO_2eq) with fiberglass second (357 kg CO_2eq) and vinyl third (456 kg CO_2eq). These results included the normalization for service life (25 years for Al-clad wood and fiberglass and 18 years for vinyl), which when removed from consideration, the vinyl window was nearly identical to the impacts of the aluminum clad wood window.

Blom *et al.* (2010) compared softwood (spruce), hardwood (azobe), and vinyl windows with the purpose of comparing the frame materials and various maintenance regimes. They found that the wood window performed best over the life cycle when efforts were made to maximize the service life of the materials and that the wood window had less impacts than the PVC. Minimizing solvent use in window paint was found to cause little improvement in terms of overall results.

Sinha and Kutnar (2012) developed a carbon footprint of wood, PVC, and aluminum frame windows that considered both the embodied impacts of the materials and the thermal transmission of the materials that comprise each window type. Their study found that the wood window frame had the smallest embodied carbon footprint (130 kg CO_2eq) and the best thermal properties. In contrast, the aluminum window had the highest embodied carbon footprint (486 kg CO_2eq) and had the highest U-value of the three window types. The study did not combine the embodied carbon footprint with energy simulation results to produce an overall cradle-to-grave carbon footprint as did several of the studies detailed in the following section.

21.4.2 Justification of energy payback

The first LCA to consider the embodied impacts of insulated glazing units and the various gases that may be used between glass panes was conducted by Weir and Muneer (1998). Their study considered the cradle-to-grave embodied impacts of an aluminum clad wood window and specifically studied argon, xenon, and krypton gases. They found that while the use of xenon and krypton gas significantly changed the embodied impacts of the windows, the use of argon was miniscule compared to the rest of the materials (0.1 MJ for the argon compared to 1,030 MJ for the window). While no energy simulation was conducted, the addition of the argon gas fill decreased the U-factor of the window from 1.6 to 1.4 which would easily offset the increased impacts. Weir and Muneer (1998) also noted that wood window frames sequestered more than 10 times the carbon dioxide equivalents as are emitted in the various extraction and manufacturing processes (250 kg CO_2eq compared to 22 kg CO_2eq).

Recio *et al.* (2005) highlighted the significance of the use phase in their study of vinyl, wood, and aluminum single- and double-glazed windows that included energy simulation. Aluminum windows were found to have the highest manufacturing impacts and use phase energy loss. The study assumed a slightly lower R-value (higher U-factor) for wood windows compared to vinyl windows, which outweighed the manufacturing benefits of wood over vinyl. The payback period for the higher manufacturing impacts of vinyl compared to wood was roughly 19 years. In all cases, the double-glazed windows far out-performed their single-glazed counterparts.

The benefits of advanced glazing systems were further highlighted in a study by Kiani *et al.* (2004) that isolated this element in their study of curtainwall systems. Their study included thermal simulations of a commercial building in London and found that the increased energy required to manufacture reflective glass compared to tinted glass was far outweighed by the energy improvements of the reflective glass. The study found that the environmental payback period for the reflective glass was roughly 3 years.

The potential benefits of increasing the performance of glazing systems were taken one step further by Syrrakou *et al.* (2005) in which they studied the effects of an electrochromic device, a technology that uses a low-voltage switch to tint the glazing unit and thus vary light transmission characteristics. Their findings, that the environmental payback period was less than 2 years and that the unit saved more than 33 times the manufacturing energy, provide a good indication as to the power of LCA to further the push for advanced window technologies in the face of ever-increasing life cycle thinking in building design.

21.4.3 Mitigating life cycle impacts

One recent study, Tarantini *et al.* (2011) completed an LCA of a wood frame window with the purpose of informing green procurement public policy in Italy. Their study went beyond quantifying environmental impacts to provide a structured framework for mitigating impacts throughout the life cycle. Tarantini *et al.* (2011) also recognize the environmental impacts that are caused in each life stage. Table 21.2, which was originally published in Tarantini *et al.* (2011), presents these elements in addition to the pertinent legislation and technical references and thus serves as a framework for similar initiatives in other jurisdictions.

21.4.4 Findings in the literature

The window LCA studies that have been completed to date have been largely consistent in their findings. The window frame comparisons generally show wood to be of lower embodied impact than vinyl or aluminum (Asif *et al.*, 2002; Salazar and Sowlati, 2008a; Blom *et al.*, 2010; Sinha and Kutnar, 2012), but that vinyl outperforms wood over the life cycle (Recio *et al.*, 2005). The benefits of wood windows are hindered somewhat by the requirement of protective cladding to achieve the service life of materials like fiberglass and aluminum (Asif *et al.*, 2002; Salazar and Sowlati, 2008a). Vinyl windows have been shown to fall between aluminum and wood in terms of embodied impacts, but with the shortest expected service lives (Asif *et al.*, 2002). The LCA studies that include manufacturing and energy performance consistently show the performance benefits outweighing any additional manufacturing requirements (Weir and Muneer, 1998; Kiani *et al.*, 2004; Syrrakou *et al.*, 2005; Recio *et al.*, 2005). The greatest potential benefit is realized when an LCA is used to recognize environmental hotspots in the life cycle and is used to provide a structured framework for guiding policy to mitigate impacts (Tarantini *et al.*, 2011).

21.5 The influence of timing on the results of window LCA

The window LCA studies that have been conducted previously have excluded one key issue that has potentially significant impacts on the results

Table 21.2 Adopted approach to select the relevant environmental criteria for GPP of windows (from Tarantini *et al.*, 2011)

Responsible process or life cycle stage	Main environmental impacts	Mitigation strategies	Selected GPP criteria	Legislation, technical standards and reference documents
Energy losses in use phase		• Bioclimatic architecture for new buildings • Maximum value for U_w, air tightness	• Maximum value for U_w, air tightness • U_w improvement, declared g, τV value (award criteria)	• Directive 2002/91/EC • Decree 135, 11/03/08 • UNI EN 14351-1 • UNI EN 410 • UNI EN 12207 • UNI EN 1026
	• Greenhouse effect • Acidification • Photo-oxidant formation • Primary energy consumption	• Improvement of window characteristics (U_w, g, air tightness)		
Double glazing production		• Use or best practices and energy efficiency measures for glazing production • Increase the window service life	• Use of Best Available Techniques (BAT) for glass production • Prolonged warranty for window (award criterion)	Reference to BREF for glass production

Table 21.2 Continued

Responsible process or life cycle stage	Main environmental impacts	Mitigation strategies	Selected GPP criteria	Legislation, technical standards and reference documents
Frame production (Al)	• Greenhouse effect • Acidification • Photo-oxidant formation	• Increase the window service life • Use of best practices (Al, PVC production) • Increase the content of recycled materials	• List and weight % or window materials • Use of BAT (Al, PVC production) *Award criteria*:	• Reference to BREF (Al production) • Directive 67/548/EC • Directive 1999/45/EC
Frame production (PVC)	• Primary energy consumption • Waste production (Al, PVC) • Hazardous chemicals (PVC, wood)	• Limit on classified chemicals (PVC) • Identification and marking of plastic parts (>50 g)	• Limit on classified chemicals (PVC) • Declaration of recycled content (Al, PVC) • Prolonged warranty for window • Identification and marking of plastic parts (>50 g) • Use of wood from legal sources • Declared recycled content • Use of wood from sustainably managed forests	• Reference to BREF (PVC production) • Vinyl 2010 comm. • UNI EN ISO 11469 • Directive 67/548/EC • Directive 1999/45/EC • …

Frame production (wood)		• Use or wood from legal sources • Use of wood from sustainably managed forests • Limit on classified chemicals	• Limit on formaldehyde in wood panels	• Decree 10/10/2008 • Directive 67/548/EC • Directive 1999/45/EC
Painting process in production phase	Photo-oxidant formation	Use of low VOCs paints	Use of low VOCs paints	• Legislative Decree 161, 27/03/06 • Directive 161, 27/03/06 • Legislative Decree 152, 03/04/06
Windows end-of-life	Waste production	• Increase the window service life • Take-back system for windows, frame materials recycling (Al, PVC)	Take-back system for windows	UNI EN ISO 11469

in nearly every study. These studies uniformly ignore the influence of the timing of impacts and the difference between impacts that occur in the short term (e.g., manufacturing impacts) with those that occur sometime in the future (e.g., energy use during building occupancy). Impacts that occur in the future are subject to greater uncertainty as to the representativeness of LCI data and may be valued differently by stakeholders based on subjective perspectives. The concept of discounting future costs to estimate the ‘net present value’ is common in life cycle costing which closely mirrors LCA.

Since there is no universally accepted discount rate for any of the various environmental impacts considered in LCA, impacts that occur many years in the future are typically accounted identically to those that happen in the near term. Again, this is an issue that applies to all building LCA studies and is not unique to windows, but it is especially critical to properly weighing future energy savings against manufacturing impacts that is common with window LCA studies. Timing also has significant influence on the results for recycled and recyclable materials like steel aluminum as well as wood that sequesters biogenic carbon.

21.5.1 Energy payback estimation

Many of the studies considered in the previous section sought to directly calculate the energy payback period of windows that have higher manufacturing impacts. These studies apply assumptions as to the energy requirements of buildings based on energy simulation modeling. These simulations model annual seasonal energy use which is then extrapolated to each of the years that a building is estimated to be in service. In this way, the annual savings of one window system over another may be subtracted from the difference in manufacturing impacts to estimate the true life cycle difference.

The extrapolation of energy simulation results over the service life of the building implicitly assumes that no changes are made to the mechanical equipment, heating, ventilation and air conditioning (HVAC) systems for example, or to the thermal properties of the building envelope. Additionally, the environmental impacts of the energy savings are estimated based on the current state of the energy mix and delivery efficiencies as estimated by available LCI databases. This may be a fair approximation over the first several years that a building is in operation, but the representativeness of these assumptions becomes less and less certain the further into the future that one attempts to extend the analysis. As windows have service lives typically estimated in the range of 20 to 50 years, and the buildings in which they are installed are often modeled between 60 and 100 years and include multiple window replacements, the energy savings and impacts attributable to a particular window type can become quite uncertain. The window manufacturing impacts themselves are also subject to uncertainty over time in such studies that include multiple window replacements. One must only look 20 to 100 years in the past to consider the degree to which environmental impacts may change over the life cycle of a window or building.

The uncertainty of future building energy use and impacts is not to the degree that it negates the assumptions of the studies presented in the previous section that showed very short payback periods for windows that achieve significantly greater performance. For example, Weir and Muneer (1998) found negligible manufacturing impact increases for argon compared to a conductivity reduction of roughly 13%. Kiani *et al.* (2004) and Syrrakou *et al.* (2005) found payback periods for reflective glass and electrochromic devices of 2 and 3 years, respectively. On the other hand, in studies in which only slight use phase benefit is realized, comparing vinyl and wood frames in Recio *et al.* (2005) for example, the payback period may be of sufficient length that the results carry significant uncertainty. This will also likely be the case in future studies that consider window performance in buildings with already low overall use phase impacts (e.g., those that operate largely on renewable energy), which is commonly the case for buildings seeking various green building certifications.

Another important consideration in estimating environmental payback over time is the idea of a discount that some stakeholders should apply to future impacts relative to those that occur in the short term. For example, many leading climate change scholars (IPCC, 2007, Section 2.2.3) believe that the next several years are critical to prevent reaching greenhouse gas concentrations that will be irreversible or extremely costly to adapt to; this is commonly referred to as a ‘tipping point’. It should be noted that any reasonable discount rate would not completely eliminate future savings from net present impact value estimations (1% per year is a commonly applied GHG discount rate). Even so, one should always strive to reduce environmental impacts as they occur and not ignore the irreversibility that occurs when critical thresholds are reached.

21.5.2 Recycling

Many of the materials present in windows either contain recycled materials, may be recycled at the end of their service life, or both. A generally accepted method of accounting for this recycling is the use of ‘system expansion’ in which the net benefits of using recycled material relative to virgin materials are credited to the product system. In the case of wood, plastics, and float glass, this crediting is less critical because these materials generate relatively low manufacturing impacts, are not cost effective to recycle, and are subsequently recycled very little. While container glass is frequently recycled, post-consumer float glass is rarely recycled because it is considered a contaminant to glass container production. Metals, however, cause relatively high environmental impacts in manufacturing, retain all of their properties when recycled, are cost effective to recycle, and are thus recycled in significant quantities. This is particularly the case for aluminum, which is very energy intensive to smelt from raw bauxite, while recycled aluminum consumes roughly 10% of the energy of virgin sourced material (Asif *et al.*, 2002). If one were to credit an aluminum window with the benefits of recycling over virgin production, the logic being that the presence of recyclable scrap substitutes for virgin production that would have otherwise occurred, then the impacts of this window type would be reduced significantly.

The credit granted for future recycling is subject to many of the same uncertainty and discounting questions raised in the previous section. While LCI data sets reasonably estimate current aluminum production and the energy mixes that power this production, the further one projects into the future the less certain the results become. Similar to the caution that must be extended in sacrificing current impacts for future savings in terms of energy performance, one must also consider the uncertainty of modeling future metals production and the various perspectives of stakeholders that may discount future impact savings relative to those in the short term.

21.5.3 Carbon sequestration

Durable wood products such as window frames are often granted a greenhouse gas credit for their storage of carbon. The credit is granted for wood's growth that consumes carbon dioxide from the atmosphere which is accounted as a negative emission. The extraction of wood during logging means that the carbon stored in durable products is sheltered from natural disturbances such as fire and insects. The logging also 'resets' the natural growth cycle in the forest to one in which young trees consume additional carbon during growth while an unlogged forest would have otherwise reached maturity and consumed no additional carbon. For these two reasons, a sequestration credit is often granted to the wood product that cancels some or all of the fossil-based greenhouse emissions during the rest of the life cycle. This was noted in Weir and Muneer (1998) in which the wood frame sequestered more than 10 times the carbon emitted in its manufacturing.

Wood also partially decays in landfills for many years after the service life and should be accounted as an emission against the sequestration credit. The LCA practitioner must assign a time cut-off for the modeling of this decay or estimate the eventual permanent sequestration by extending the time boundary of the assessment indefinitely. Carbon sequestration as a product of forest growth and long-term decay takes many years to occur and is thus subject to the same uncertainties and discounting as energy consumption during a building's use and recycling that occurs at the end of the service life.

21.6 Use of advanced technology

Two technological advances in windows have the potential to dramatically improve the impacts of windows in the context of whole building performance. These are the optimization of window selection in overall building design and the use of electrochromic devices.

21.6.1 Integration of window specification in building design

The integration of window specification in building design seeks to optimize the performance of windows in terms of their impacts in buildings. A recent work by Su and Zhang (2010) conducted simulation studies to determine the optimum window-to-wall ratio in terms of energy use. They found that for north-facing orientations, the window wall ratio (WWR) should be minimized while in south-facing orientation an upper bound of 0.5 would be optimal. The study confirms conventional wisdom about optimal window orientation, but does provide a scientifically valid confirmation of these assumptions and demonstrates the ability of energy simulation to actually optimize window performance.

A more recent study by Ihm *et al.* (2012) integrated life cycle costing data with energy simulation for six different glazing types. They found that high performance glazed windows can be a net contributor to the overall performance of a building in mild climates. The study concludes with recommendations to Korean building codes that in energy-intensive climates and for larger windows double-glazed low-E windows be required.

21.6.2 Electrochromic and thermochromic coatings

It was noted previously that Kiani *et al.* (2004) and Syrrakou *et al.* (2005) found payback periods for reflective glass and electrochromic devices of 2 and 3 years, respectively. Several more recent studies have been conducted that further the potential of electrochromic windows in terms of LCA. Papaefthimiou *et al.* (2009) use electrochromic windows as a case study to propose alternative rating systems for windows that incorporate concepts of eco-efficiency as determined in LCA.

Ye *et al.* (2012) and Ye *et al.* (2013) explore the potential of electrochromic devices to deliver on a 'perfect window' that optimizes near infrared light transmission. By varying the transmission of near-infrared light, an optimal window performs both the functions of the two common low-E types currently available in the marketplace. This would allow the retention of long wave radiation in winter and the reflectance of intense short wavelength radiation in summer months.

21.7 Selection of environmentally friendly window materials

The attention paid to environmental criteria in building material selection and recognition that windows are a critical element in the life cycle impacts of a building has made it imperative that accurate and complete life cycle data be made available to inform window material selection. The LCA studies that have been completed on windows thus far, while agreeing on the general preference for certain materials in various applications, have also identified that a systems approach is the only meaningful way to fully capture the environmental impacts of windows. Compounding these two issues is the fact that the windows industry is a very dynamic one, with advanced technologies such as triple glazing, electrochromic and mechanical shading systems, and sophisticated building simulation seeking to shift long-held conventions in window specification and use.

21.7.1 Data for window LCA

All of the LCA studies that have been completed to date were conducted as one-off case studies. No common databases have been generated for the various components that make up a window. As a result, each window LCA has done little to build on the previous research, instead rebuilding LCI models for primary materials and processes based on either first-hand surveys or databases for common commodity materials. This is particularly troublesome in the case of windows when one recognizes the number of different materials and manufacturing steps that must take place to produce a finished window. For example, a single aluminum-clad wood window includes the following materials: wood, wood preservatives, wood glue, primer and paint, aluminum cladding and spacer bars, steel hardware and fasteners, desiccant, PIB sealant, caulk, polymer weather-strip, low-E coated float glass and argon (Salazar and Sowlati, 2008a). Additionally, the production of the window frame, hardware, glazing unit, and its fabrication may all take place at different facilities before the window is even delivered to a distributor and eventual building site. In many ways, the life cycle of a window is a microcosm of the life cycle of a building itself.

It is no wonder, then, that in the LCA of a whole building, with products such as framing materials, insulation, roofing, and siding having uniform databases from which to pick and choose representative data sets, to complete an LCA of a building that includes windows, the data must be extrapolated and taken out of context.

The problem of non-uniform LCI data for windows is not an insurmountable one. Windows are fairly modular products with the glazing units, hardware, and frames being generally interchangeable. Developing an LCI database for windows should focus on these components and seek to encompass the variety of market alternatives. This way, the scope of necessary window LCA studies can be limited from the factorial number of potential window permutations to focus on a few key variations of the various components. The availability of a common window component database would make possible the simple addition of windows in whole building LCA by allowing the data users the ability to choose window attributes 'a la carte'. Any window component database should include all of the parts identified in Section 21.2.1 and the most popular frame materials, glazing types, and operability identified in Sections 21.2.2, 21.2.3, and 21.2.4. The LCI database must also apply a uniform treatment of recycling and carbon sequestration. The various stakeholders of the window database must determine the acceptable tolerance for uncertainty of future impacts and any discounting that may or may not be applied.

21.7.2 Considering embodied and use phase impacts

With the availability of databases for the various window components that facilitate whole building LCA, one may then begin to examine the performance of windows in a proper systems context. It is well understood that numerous factors influence the impact that a window has on the performance of a building and that these factors vary based on the placement of the windows in a building, the nature of the heating and cooling systems in place, and the location and orientation of the building itself. The potential number of LCA studies for windows is thus limitless, which is the case for all envelope materials that provide an insulation function, but is particularly the case for windows which are so sensitive to climate and whose impacts are influenced by far more than their insulation properties. These points aside, a structured use phase simulation may be designed that provides useful rules of thumb for window selection and may be used in green building rating systems that must be pragmatic to be implemented.

Use phase energy is typically found by considering several representative climates, by redefining the functional unit of comparison to include a built structure with or without the windows installed in it, and by simulating the thermal load difference caused by the window. In North America, fulfilling the first criteria of building selection has been standardized by the North

American Fenestration Standard (NAFS, 2008) with performance criteria for windows for five classes of windows and describes their application as follows:

- ***Residential:*** Commonly used in one- and two-family dwellings.
- ***Light commercial:*** Commonly used in low-rise, multifamily dwellings, low-rise professional offices, libraries and low-rise motels.
- ***Commercial:*** Commonly used in lighter use industrial buildings and factories, hotels and retail sales buildings.
- ***Architectural:*** Commonly used in hospitals, schools, institutions and public buildings or high-rise buildings to meet increased loading requirements. Also commonly used in buildings where possible misuse of the fenestration product might be anticipated.

Case study buildings must first be established for each of these representative building types that are based on typical or average construction practice. The location of these buildings must then be defined and represent the variation of climate conditions that the data are expected to represent. In North America, several cities have been identified as representative of the various climates and are often used by other energy analysts when estimating energy use in buildings. The cities and the climate zone numbers are as follows:

- **Miami, Florida:** a hot and humid climate (Zone 1A)
- **Phoenix, Arizona:** a hot and dry climate with large daily temperature swings (Zone 2B)
- **Memphis, Tennessee:** a mild climate (Zone 3A)
- **Seattle, Washington:** a cool climate (Zone 4C)
- **Denver, Colorado:** a cold climate with large daily temperature swings (Zone 5B)
- **Minneapolis, Minnesota:** a cold climate (Zone 6A)

After the buildings have been defined for each of the five classes, they should be modeled in each of the six cities with and without the windows to isolate their impacts. The thermal performance results may then be normalized based on the overall area of windows present. The integration of these data with the manufacturing impacts must carefully consider the time sensitivity of the results in terms of uncertainty and discounting.

21.7.3 Integration of advanced glazing technology

These LCA results that estimate embodied effects may be combined with energy simulations based on the representative building types and climate zones. The results may similarly be normalized to determine the per unit impacts of these advanced technologies.

Advanced window technologies provide the greatest potential for designers to optimize the performance of buildings in the future. As buildings and building components continue to become more efficient, it is well understood that the law of diminishing returns begins to set in. This is clearly evident by the popularity of double-glazed windows, while those that are triple- and quadruple-glazed are quite rare. Advanced modeling would allow designers to most efficiently allocate the window purchasing budget to apply the highest cost and embodied impact windows to the areas of the building where they are of most benefit by accurately estimating the payback period in very specific applications.

The optimization may be further enhanced with the use of dynamic shading systems that will certainly become a key component in future 'smart buildings'. These systems may be pre-programmed to change the light and thermal properties of windows throughout the day based on seasonal patterns, or even to respond to conditions in real time. These applications could drastically change the performance and thus the overall life cycle impacts of windows and buildings as a whole.

One can easily imagine a scenario in which the selection of environmentally friendly windows means far more than selecting a frame material based on an anecdotal attribute such as its recyclability or renewability or a glazing unit based on its number of panes or coating. With the availability of modular material data, thermal performance estimates, and the capability to modify a window's properties on the fly, windows are likely to become synonymous with sustainable building design and occupancy.

21.8 Current developments and future trends

Several initiatives are currently underway that are likely to drive window LCA research over the coming years. These include the development of two new window product category rules (PCRs) that will allow the standardization of window LCA for the purposes of environmental product declaration (EPD) development.

The Norwegian EPD Foundation (http://www.epd-norge.no/) has recently completed a draft PCR for Windows and Doors that is the subject of public consultation. The Earthsure EPD Program (http://iere.org/earthsure.aspx) is currently developing a PCR for North American windows. These two documents, when complete, will be useful references for developing future window LCA studies.

Several trade associations represent European and North American window manufacturers and are useful sources for information as to current and future trends in the window industry. In Europe, the Federation of European Window and Curtain Walling Manufacturers' Associations (http://www.faecf.org/) and in North America, the Window and Door

Manufacturers Association (https://www.wdma.com/) represent window manufacturers.

For LCA data on windows, the authors of the studies presented in the literature review may be contacted directly. Ecoinvent (http://www.ecoinvent.ch/) also has some European data on standard glazing and framing elements. In North America, the Athena Sustainable Materials Institute (http://www.athenasmi.org/) has recently developed a beta version of a window LCA calculation tool that scales framing and glazing data independently and in accordance with the forthcoming windows product category rules.

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22

Life cycle assessment (LCA) of ultra high performance concrete (UHPC) structures

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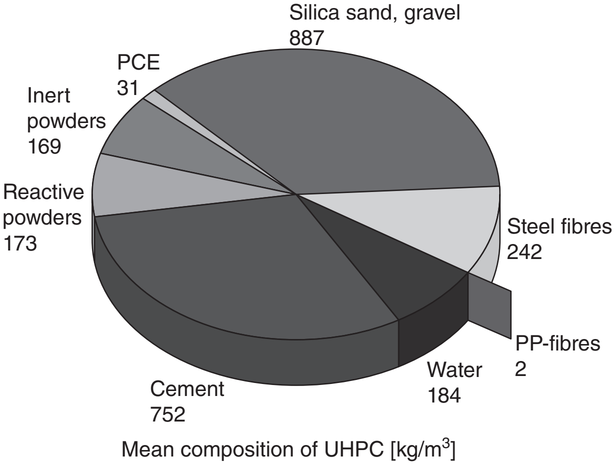
Abstract: The excellent mechanical properties of UHPC are mainly achieved by the use of a high amount of energy-intensive raw materials like Portland cement or micro steel fibres. Due to very high strength, it is possible to reduce the cross-sectional area and thus the weight of structures. On the one hand, a smaller amount of raw materials is needed, but, on the other hand, the portion of energy-intensive constituents must be increased. Hence the extent to which environmental effects of construction can be reduced by using UHPC was determined. Therefore the ecological impact was determined at a materials as well as on a structural level. The ecological impact of UHPC structures was also compared to structures made of conventional materials.

Key words: UHPC, steel fibre reinforcement, high rise building column, traffic bridge, foot bridge, micro steel fibre, superplasticizer.

22.1 Introduction

Within the last decade, different ways of reducing the ecological impact of concrete structures have been identified and adopted by the construction industry. One of these ways is the use of new materials with enhanced mechanical properties and higher durability. Ultra high performance concrete (UHPC) is just such an innovative upcoming cementitious material for the building industry. Compressive strengths of typically 200 N/mm^2 and tensile strengths around 15 N/mm^2 are achieved by the use of a high cement content (>700 kg/m^3), a low water-to-cement ratio ($w/c < 0.30$), a low maximum grain size diameter ($D_{\max} < 1$ mm) and the addition of pozzolanic filler materials like silica fume (Dehn, 2004; Fehling *et al.*, 2005). Furthermore, UHPC is highly durable (high resistance against freeze-thaw attack, high resistance against permeation of gases and liquids) even under severe conditions. In view of its excellent mechanical properties, UHPC is predestined for use in, for example, highly loaded columns, prestressed members or filigree constructions.

However, under load, this material is characterized by explosive failure following ideal elastic deformation. The fracture surfaces are extremely



22.1 Mean composition of UHPC based on a literature study by Stengel and Schießl (2008).

smooth. This type of failure is, of course, unacceptable for construction materials. In order to obtain more ductility and improved tensile properties, micro steel fibres are often added to the mix. The fibres are about 0.15 mm in diameter, between 4 and 20 mm long and have a tensile strength of about 3,000 N/mm^2. The better ductility obtained with the fibres enables exploitation of the excellent mechanical properties of UHPC. Due to the high load-bearing capacity of UHPC, it is possible to reduce member cross-sectional area and therefore the weight of the structure. On the one hand, a smaller amount of raw materials is required to build structures, but on the other hand the proportion of energy-intensive components (e.g., cement, steel fibres and superplasticizer) must be increased to guarantee the aforementioned properties and to ensure good workability. The mean composition of UHPC based on the evaluation of 75 references in the literature covering the last 10 years is presented in Fig. 22.1.

The mean cement content is roughly 750 kg/m^3. If all reactive components are included, the binder content is about 925 kg/m^3. This yields a mean water/binder ratio (w/b) of 0.20 for a mean water content of 180 kg/m^3. A high content of superplasticizer on polycarboxylate ether basis is necessary which is, on average, 30 kg/m^3, i.e. 3.4 wt.% with respect to the binder. The mean content of steel fibres used is 240 kg/m^3 or 3 vol.%. Another literature research performed in 2004 yielded similar results (Dehn, 2004). Lower cement and micro steel fibre contents were used to produce UHPC, for example, by Stengel and Schießl (2012) and Hassan *et al.* (2012). Stengel and Schießl (2012) also showed that ultra high strength of around 250 N/mm^2 can be reached at w/c ratios of 0.28 by using appropriate very fine fillers.

Over the last five years, more than 20 constructions all over the world mainly in the field of bridge engineering were erected using micro steel fibre reinforced UHPC. Some of these bridges were built to replace structures beyond repair. In 2005, according to the US National Bridge Inventory (NBI; see Federal Highway Administration, 2008), there were approximately 156,000 structurally deficient or functionally obsolete bridges (roughly 25.4%). This number is slightly decreasing to roughly 23.8% in 2011 (see Federal Highway Administration, 2013). The US Federal Highway Administration's Ultra-High Performance Concrete Research Program has been investigating the use of UHPC in highway infrastructure to promote the application of the new material. Up to now, two traffic bridges were realized in the US and five in France using micro steel fibre reinforced UHPC.

The extent to which the environmental effects of constructions can be reduced by using the new UHPC material is investigated in this study with the help of life cycle assessment methods (LCA). Four different applications of UHPC are considered and results compared to conventional building materials:

1. high rise building columns
2. single span bending beams
3. precast single span bridge girder
4. traffic and footbridges.

LCA is performed in all four cases for UHPC and conventional building materials, i.e. normal concrete or steel. The aim of the study is to answer the question whether or how the use of UHPC yields more environmentally friendly constructions when compared to conventional building materials.

22.2 Life cycle assessment (LCA) data and impact assessment method

LCA enables estimation of the potential environmental impact of, e.g., production processes. The LCA procedure is laid down in the standards DIN ISO 14040 and comprises four stages. Owing to the large amount of data, life cycle assessment is carried out using software and databases for processes and materials. In this study, the software SimaPro and the Swiss Ecoinvent database for life cycle inventory data were used. The calculation of potential environmental impact from the life cycle inventory analysis can, in principle, be performed using different methods, for example the eco-points, CML and eco-indicators methods. Depending on the particular method, one or more impact category indicators are specified. The contribution of each individual material in the system analysed to each impact category indicator is calculated using characterization factors. In this study, the estimation of impact was performed using the CML method where results were obtained for the impact categories global warming (GWP), ozone depletion in the stratosphere (ODP), summer smog, i.e. photo chemical ozone creation (POCP), acidification (AP) and eutrophication (NP). These are widely known and the so-called baseline CML impact categories (see, e.g., Van den Heede and De Belie, 2012). The life cycle inventory analysis and impact assessment were carried out using SimaPro version 7.0 software. The data required to construct the production process were retrieved from the Ecoinvent database (Ecoinvent, 2012) as well as from own data compilation.

22.3 Impact assessment of raw materials used in ultra high performance concrete (UHPC)

22.3.1 Cement

Environmental impact of cement production is calculated based on the data provided by Ecoinvent (2012). Detailed information on the production process and on all inputs can be taken from Künniger *et al.* (2001). The functional unit is the production of 1 kg of Portland cement strength class 42.5 (CEM I 42.5 R). Average fuel for clinker production is composed of 6.81×10^{-3} MJ natural gas (high pressure), 3.74×10^{-4} kg light fuel oil, 2.55×10^{-2} kg heavy fuel oil, 3.54×10^{-2} kg hard coal and 3.91×10^{-3} kg petroleum coke. In addition 5.80×10^{-2} kWh electricity (medium voltage) is considered. Besides 0.912 kg of Portland cement clinker, an input of 0.063 kg gypsum (not balanced, origin from flue gas desulphurization), 0.025 kg additional milling substances (not balanced as it is taken as waste without environmental burdens, e.g. dust from the cement rotary kiln, fly ash, silica dust, limestone) and 3.50×10^{-4} kg ethylene glycol (process material for grinding) is taken into account. A total energy for grinding and packing of 4.85×10^{-2} kWh (electricity, medium voltage) as well as transport processes corresponding to 4.40×10^{-3} tkm (lorry, 16 t) is necessary. The production of 1 kg Portland cement CEM I 42.5 R yields an environmental impact of 0.833 kg CO_2-eq. (GWP100), 2.241×10^{-8} kg CFC-11-eq (ODP), 4.211×10^{-5} kg C_2H_4-eq. (POCP), 1.138×10^{-3} kg SO_2-eq (AP) and 1.702×10^{-4} kg PO_4^{3-}-eq. (NP).

22.3.2 Fine mineral additions

Typically, mineral additions used in UHPC are silica fume (microsilica, by-product of the silicon and ferrosilicon alloy production), finely ground blast furnace slag, quartz powder and limestone powder. Silica fume is taken as a by-product without environmental burdens and therefore not balanced

Table 22.1 Environmental impact of production of 1 kg round gravel, river sand and silica sand

Material	GWP100	ODP	POCP	AP	NP
	kg CO_2-eq.	kg CFC-11-eq.	kg C_2H_4-eq.	kg SO_2-eq.	kg PO_4^{3-}-eq.
Round gravel	2.422×10^{-3}	2.653×10^{-10}	5.904×10^{-7}	1.502×10^{-5}	2.910×10^{-6}
River sand					
Silica sand	2.101×10^{-2}	2.986×10^{-9}	2.791×10^{-6}	5.813×10^{-5}	7.240×10^{-6}

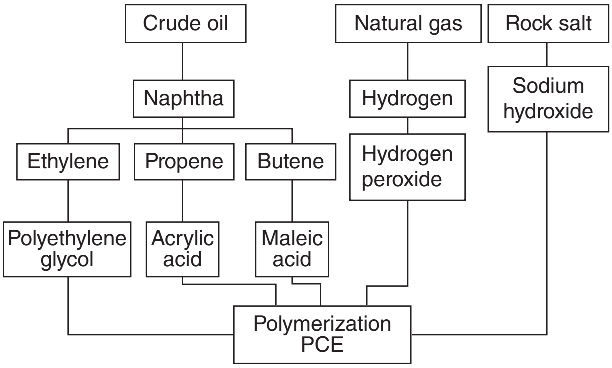
in this study. The remaining mineral additions are for sake of ease modelled with the help of an Ecoinvent data set for milled limestone (for details, see Künniger *et al.*, 2001). The functional unit is 1 kg of milled and packed limestone powder. The environmental impact is 2.313×10^{-2} kg CO_2-eq. (GWP100), 2.150×10^{-9} kg CFC-11-eq. (ODP), 4.119×10^{-6} kg C_2H_4-eq. (POCP), 1.011×10^{-4} kg SO_2-eq. (AP) and 2.142×10^{-5} kg PO_4^{3-}-eq. (NP).

22.3.3 Gravel and sand

Data for round gravel (see Künniger *et al.*, 2001), river sand (see Künniger *et al.*, 2001) and silica sand (see ÖSPAG, 2002) can be taken directly from the Ecoinvent database. The environmental impact for the three materials is given in Table 22.1.

22.3.4 Superplasticizer (polycarboxylate based)

Polycarboxylate ethers (PCE) contain groups with polyoxyalkylene, especially polyethylene or polypropylene glycol groups as well as carboxylic acid and/or carboxylic acid anhydride monomers, e.g. acrylic acid, methacrylic acid, maleic acid and its anhydride, itonic acid and its anhydride. In addition monomers based on vinyl or acrylate can contribute to the chemistry of PCE. The raw materials and the molecular chaining hierarchy of the constituents for the synthesis of PCE are shown in Fig. 22.2 in a schematic flow diagram. The constituents are represented by Ecoinvent process data for acrylic acid, maleic acid, ethylene glycol, sodium hydroxide and hydrogen peroxide. The final product, superplasticizer based on PCE, also contains water and biocides which were also represented with the help of Ecoinvent process data. The batch polymerization process requires a polymerization plant and suitable industrial buildings. The necessary infrastructure and energy for this were determined in this study.



22.2 Schematic showing the production of polycarboxylate-based superplasticizer.

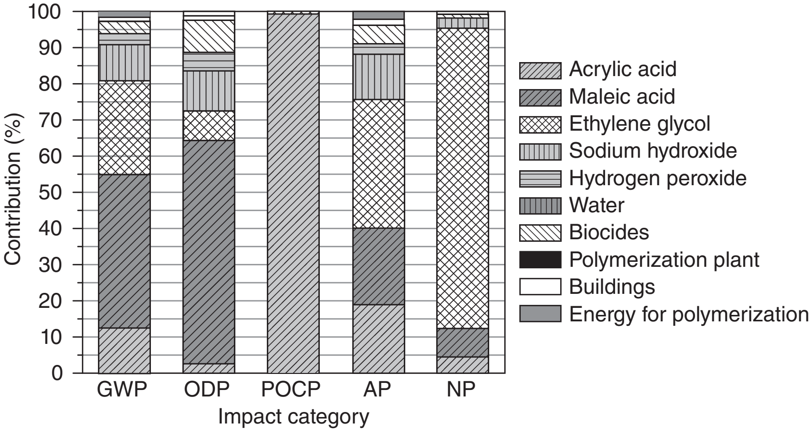
The following information on process engineering for the production of PCE is taken from Plank *et al.* (2004), Hirsch (2005), Ignatowitz (2003) and Sattler and Kasper (2000). PCE plants comprise a number of storage and supply vessels as well a chemical reactor with a capacity usually around 20 m^3. The lid of the reactor is fitted with a cross beam stirrer complete with motor. A supply system is used to charge the reactor with the constituents from above. Beforehand, some of the monomers are transformed into macromonomers in a small agitator container and then transferred with the main charge to the reactor. The main polymerization reaction is controlled by the dosage of an initiator and a heating and cooling system. The temperature of the reactor is maintained at 60–80°C during polymerization.

In this study, the reaction time was set at 5 h. After a cooling phase lasting one hour, neutralization and the addition of substances such as defoamer and biocides, the reactor can be recharged. The plant in this study weighs 95,325 kg and has electric motors with a total power of 135 kW as well as operation and production buildings with a volume of 35,000 m^3. The service life of the plant is set at 25 years. Working in three shifts, the plant synthesizes approximately three times 20 t PCE per day. This corresponds to a total production of 390×10^6 kg aqueous polycarboxylate solution for a total operating time of 156,000 h. The polymerization plant was modelled with Ecoinvent process data for an industrial machine, a general operating building and the energy mix necessary for operation. The results of the impact estimation for the production of 1 kg PCE are presented in Table 22.2.

Figure 22.3 shows the contributions of the different subprocesses to each impact category. At 42%, the production of maleic acid contributes most to GWP. This is followed by the production of ethylene glycol, at approximately 26%. Acrylic acid and sodium hydroxide each contribute between 10 and 12%, respectively. ODP is dominated by the production of maleic

Table 22.2 Environmental impact of 1 kg polycarboxylate-based superplasticizer

Impact category	Result	Unit
GWP	1.11	kg CO_2-eq.
ODP	6.09×10^{-8}	kg CFC11-eq.
POCP	1.97×10^{-2}	kg C_2H_4-eq.
AP	4.81×10^{-3}	kg SO_2-eq.
NP	1.75×10^{-3}	kg PO_4^{3-}eq.



22.3 Results of dominance analysis for the production of PCE.

acid, approximately 62%. The production of sodium hydroxide, biocides and ethylene glycol also contribute significantly, with between 8 and 11% to this indicator. A total of 99% of POCP comes from the production of acrylic acid. With 35%, ethylene glycol production has the largest effect on AP. It is followed by the production of acrylic acid, maleic acid and sodium hydroxide, which each contribute between 13 and 21% to AP. With 83%, ethylene glycol production clearly dominates NP.

22.3.5 Steel fibres

Steel wire with a diameter lower than 0.2 mm is not primarily fabricated for the production of micro steel fibres but for pneumatic tyres. Coated with brass the so-called steel cord reinforces the elastomeric matrix of pneumatic tyres (so-called belt and carcass reinforcement) (see Golis *et al.*, 1999). A typical average pneumatic radial tyre contains approximately 9% of steel cord. This results in 0.45–0.91 kg for a passenger car tyre (see Golis *et al.*, 1999). A light truck tyre contains from 2.7 kg to 3.6 kg of steel cord, while 5.5–6.8 kg are used for large truck and bus tyres (see Golis *et al.*, 1999). Taking into account the estimated 2006 world tyre production of 1,455,000,000 tyres (Freedonia, 2008) and a mean steel cord content of roughly 0.70 kg per tyre, this results in a use of 1,018,500 t steel cord just for tyres in 2006.

The functional unit is 1 kg of micro steel fibres with a diameter of about 0.15 mm and a tensile strength of about 3,000 N/mm^2. These steel fibres are finally made by cutting to length a steel cord of the same diameter. The high tensile strength of the steel cord is mainly achieved by cold deformation of steel wire during the production process. In Europe, steel cord and fibres made from it are produced by, for example, NV Bekaert SA Belgium, Cord Romania SRL and Chircu SRL, Romania. Steel cord is mainly produced for use as reinforcement in car tyres. General information on the production of steel cord (e.g., types of machines used in a steel cord plant) was taken from the manufacturers and from Golis *et al.* (1999). No plant-specific data were used for this study. The production of micro steel fibres was divided into seven subprocesses: electric steel production, hot rolling, descaling, dry wire drawing, wet wire drawing, tempering, steel cord strand production and cutting to length. Sometimes three drawing steps with upstream patenting and surface treatment are used (Golis *et al.*, 1999). Steel cords used for tyre reinforcement or micro steel fibres are usually made of carbon steel (Golis *et al.*, 1999). The steel grade and its quality are important to the production of very thin wires.

Electric steel production

The electric arc furnace (EAF) steel production represents about 34% of the overall European steel production (see World Steel Association, 2009). Concrete reinforcing steel and steel wire for the production of steel fibres are mainly derived from EAF steel. LCI information on the EAF steel production was taken directly from Althaus *et al.* (2004) and the Ecoinvent database. The EAF steel production process is a discontinuous process which involves seven subprocesses. First of all the raw material (mainly scrap) has to be handled and stored. Thus an appropriate infrastructure, e.g. cranes and storage facilities, is necessary. Charging the furnace with the scrap is the next step in the production process, followed by the EAF scrap melting. Afterwards the steel and the slag have to be tapped. In a ladle furnace, the melted steel is treated for quality adjustments. The slag has to be removed so that continuous casting to strands can finally be performed. In view of the lack of detailed information on the infrastructure, rough assumptions have been made in Althaus *et al.* (2004) based on aluminium production. The yearly production capacity was assumed to be 500,000 t.

The EAF steel production plant is modelled for a production period of 50 years. The disposal of the machines (complete recycling) and the buildings (all mineral-bound construction materials to inert landfill, all metals to recycling) is included. The operation of the buildings is not included. For 1 kg EAF steel, the electric arc furnace was assumed to require 0.425 kWh electricity (medium voltage, European mix, UCTE) and 0.975 MJ natural gas. All other inputs as well as the emissions to air are given in Althaus *et al*. (2004). Beyond this, the landfill disposal of slag from the furnace and the cradle, dust and refractory material as well as 0.928 tkm transport by lorry and 0.929 tkm transport by rail are taken into consideration.

Hot rolling

The hot rolling of strands to wire 5.5 mm in diameter comprises five main processes: conditioning of the input (scarfing, grinding), heating to rolling temperature, descaling, rolling down to the final diameter and finishing (trimming, slitting, cutting). LCI information on the hot rolling process was taken directly from Althaus *et al*. (2004). The basic data originate from the Joint Research Centre (JRC) of the European Union and a Swiss steelwork. The auxiliary materials in Althaus *et al*. (2004) and the energy sources per kilogram of hot rolled steel were taken into account for the input. It should be remembered that all values depend strongly on the amount by which the cross section of the rolled wire is reduced. Moreover, a rolling mill is included as infrastructure in the production process. The data for the rolling mill were determined using available information on a Swiss steelwork. The production hall for the rolling mill was assumed to cover an area of 73.5 m^2 and the volume of the administrative building to be 40 m^3. The service life of the buildings was 50 years. The rolling mill is made of 57.7 t non-alloyed and low-alloyed steel and stands on a concrete foundation having the same mass. The operating life of the rolling mill is 25 years.

Descaling

As wire produced by hot rolling cools down, an oxide layer known as scale forms on its surface. The scale is hard and brittle. Descaling, i.e. the mechanical or chemical (pickling) removal of the scale, is essential before dry wire drawing because otherwise the hard scale would lead to rapid wear of the wire drawing dies. The usual descaling method is to bend the wire in different directions in order to loosen the scale which is then removed by sand blasting followed by brushing. Since descaling can be performed as a continuous feed process, the descaling machine can be coupled directly to the wire drawing machine which follows. In this study a weight of 1.5 t is taken for the mechanical descaling machine (see Ruge and Wohlfahrt, 2001;

Table 22.3 Ecoinvent processes used for modelling the descaling process

Ecoinvent process	Quantity
Industrial machine, heavy, unspecified, at plant	1.42×10^{-5} kg
Electricity, low voltage, production UCTE, at grid	0.0357 kWh

Schimpke *et al.*, 1977). The output of the descaling machine must be adjusted to match the feed rate of the dry wire drawing machine, which is between 1.0 m/s and 1.5 m/s. For this reason a feed rate of 1.25 m/s is chosen. A motor power of 30 kW is necessary to bend the wire in different directions (see Ruge and Wohlfahrt, 2001; Schimpke *et al.*, 1977). It has been assumed that the loosened scale is composed completely of iron oxide. According to statements made by the suppliers of rolled wire, 1 g scale forms per metre of hot rolled wire. The operating life of the descaling machine is set at 15 years. A total operating time of 23 hours per day is used in the calculation, as the idle time was estimated at 1 hour per day. This results in a total operating time of 125,925 h and a total output of 105.68×10^6 kg steel wire. The raw data determined are presented in Table 22.3.

Dry wire drawing to wire Ø 1.6 mm

After having reduced the diameter as much as possible by hot rolling (smallest diameter approximately 5.5 mm), smaller diameters must be produced by cold drawing (see Golis *et al.*, 1999; Schruff, 2004). Tension is applied to draw the hot rolled wire in several pulls down to a final required diameter between 1 and 2 mm. Calcium and sodium stearate are usually applied as a drawing lubricant. These soaps are particularly suitable at high drawing speeds because the lubricant film remains intact. Due to friction, temperatures above 150°C and as high as 400°C develop, which cannot be reduced by the cooling effect of the lubricant and air alone. Thus the dies are cooled with water and the wire with compressed air. Compared with earlier machines, modern dry wire drawing machines are more efficient with regard to the consumption of cooling water and lubricant as well as the filtration of stearate dust emissions. Dry wire drawing machines are machines that pull in the forward direction (Ruge and Wohlfahrt, 2001; Schimpke *et al.*, 1977). They pull the wire as many as 14 times, i.e. a series of 14 drawing dies which are usually powered separately using rotational speed regulators. Typical technical data may be found in Table 22.4.

To estimate the electrical power, it is assumed that the electric motors operate continuously at their rated power (= mechanical power) with an

Table 22.4 Technical data for a dry wire drawing machine for drawing wire from Ø 5.5 mm to Ø 1.8 mm

Value	Unit	Average
Mechanical power	kW	148
Mechanical power per pull	kW/pull	27
Maximum drawing speed	m/s	20
Input diameter	mm	5.9
Output diameter	mm	1.8

Table 22.5 Compilation of data for the dry wire drawing process

Input	
Rolled wire, descaled Ø 5.5 mm	1.0 kg
Forward drawing machine – 25 t	1.38×10^{-4} kg
Electrical energy (164 kW)	0.114 kWh
Dry drawing lubricant (Ca and Na stearate)	0.012 kg
Cooling water	Not specified
Output	
Wire Ø 1.8 mm	1.0 kg
Consumption of dry drawing lubricant	0.012 kg
Dry drawing lubricant – solid dust	Not specified
Consumption of cooling water	Not specified
Wear of drawing dies	Not specified
Wear of drawing disks	Not specified

efficiency of 90%. This is a large simplification because the energy consumption of electric motors depends decisively on their speed of rotation and the torque applied to the shaft when the motor is running. As well as energy consumption, a total mass of 25 t non-alloyed and low-alloyed steel is assumed for the dry wire drawing machine. The operating life is taken to be 15 years. For a daily operating time of 23 hours, the total output is 181.11×10^6 kg steel wire with a diameter of 1.6 mm. The resulting composition of the total process may be taken from Table 22.5. For simplicity, the dry wire drawing process was modelled using the ecoinvent processes in Table 22.6.

Wet wire drawing to wire Ø 0.15 mm

Two consecutive drawing processes are always necessary for the production of high strength wire with a diameter of 0.15 mm. After dry drawing, the wet drawing process is carried out to reduce the wire diameter to its final

Table 22.6 Ecoinvent processes used for modelling the dry wire drawing process

Ecoinvent process	Quantity
Industrial machine, heavy, unspecified, at plant	1.38×10^{-4} kg
Electricity, low voltage, production UCTE, at grid	0.114 kWh
Soap, at plant	0.012 kg

value between roughly 0.2 and 0.1 mm. A lubricant is continuously supplied to the die and the wire to cool them and prevent the wire running dry. Such systems usually operate in a closed loop where the reprocessed lubricant is used again. In some wet drawing machines, the complete drawing process is cooled by immersion in a lubricant bath. In this case, cooling coils in the drawing liquid prevent a gradual increase in temperature. Modern machines are fitted with lubricant reprocessing systems to reduce the consumption of lubricant and water. The drawing dies and deformation cones (stepped taper) in wet wire drawing machines are usually placed beside each other in a 'parallel' arrangement (Ruge and Wohlfahrt, 2001; Schimpke *et al.*, 1977). The drawing liquid is an emulsion based on vegetable oils and fats with dispersed liquid additives. In addition, copper or stannous sulphate in dilute sulphuric acid may be added to the drawing baths. A very thin coating of copper or tin forms on the wire which significantly increases the ease of drawing (Schimpke *et al.*, 1977). Parallel wet drawing machines are much more compact than dry drawing machines (forward direction drawing). Including reeling and unreeling systems, they have a maximum length of 10 m. The drawing cones in this type of wire drawing machine are powered by a common electric motor. If the drawing cones operate in a drawing liquid bath, 25 drawing stages are possible. In addition, wet drawing machines are fitted with pumps, heat exchangers and reprocessing systems. The data used for a wet drawing machine in the present analysis are presented in Table 22.7.

The electrical power for operating the machine was estimated in the same manner as for the dry drawing machine. The total mass of the wet drawing machine was set at a total of 2 t non-alloyed and low-alloyed steel. Based on daily operating time and operating life in analogy to a dry drawing machine, the total output of this wet drawing machine is 2.05×10^6 kg high strength micro steel wire. The dies and cones used to reduce wire diameter operate in two baths containing the drawing liquid (lubricant and water). It is assumed that each bath has a capacity of 350 l on average. A period of 3 months' use may be taken for each bath. During this time, fresh drawing liquid is added as necessary. This results in a total of 42,000 l lubricant in the operating life of the machine. The consumption of lubricant is assumed

Table 22.7 Technical data for a wet wire drawing machine for drawing wire from Ø 5.5 mm to Ø 1.8 mm

Value	Unit	Average
Mechanical power	kW	41
Mechanical power per pull	kW/pull	2.1
Maximum drawing speed	m/s	24
Input diameter	mm	1.7
Output diameter	mm	0.37

Table 22.8 Compilation of data for the wet wire drawing process

Input	
Wire Ø 1.8 mm	1.0 kg
Wet drawing machine 2 t	9.74×10^{-4} kg
Electrical energy (41 kW)	2.452 kWh
Wet drawing lubricant (emulsion + additive)	2×10^{-3} kg
Wet drawing lubricant (mainly H_2O)	0.02 kg
Cooling water	Not specified
Output	
Wire Ø 0.15 mm	1.0 kg
Consumption of drawing liquid	0.022 kg
Drawing liquid vapour	Not specified
Consumption of cooling water	Not specified
Wear of dies	Not specified
Wear of cones	Not specified

to be 2 kg per tonne of wire. No data are available on material wear and cooling water consumption. The resulting composition of the complete process is given in Table 22.8. For simplicity, the wet drawing process was modelled using the Ecoinvent processes in Table 22.9.

Tempering

Cold deformation is the most economical method to increase tensile strength of steel. For steel, strengths above 3,000 N/mm^2 (tyre cord up to 4,000 N/mm^2) are achieved merely by drawing. However, the increase in tensile strength is accompanied by a reduction in ultimate strain and impact toughness (Golis *et al.*, 1999). Tempering after cold working can be used to lower internal stress and improve toughness. Tempering involves heating up to 400–500°C followed by cooling in a water or oil bath (Schimpke

Table 22.9 Ecoinvent processes used for modelling the wet wire drawing process

Ecoinvent process	Quantity
Industrial machine, heavy, unspecified, at plant	9.74×10^{-4} kg
Electricity, low voltage, production UCTE, at grid	2.452 kWh
Lubricating oil, at plant	0.002 kg
Tap water, at user	0.02 kg

et al., 1977). In the case of small diameter wire, this is performed as a continuous process in a tempering machine which can be placed immediately after the drawing machine. Normally, tempering is performed using electricity or gas as an energy source. In modern plants, continuous feed tempering systems are installed which operate with induction heating. Up to 50 wires can be treated at the same time. The steel wire is heated up under a blanket of inert gas to avoid unwanted oxidation reactions. Lubricant residuals adhering to the surface of the wire are automatically removed during the heat treatment. The technical specifications for the present study were taken from the technical data sheet of a 16-wire continuous feed tempering machine (see Bongard, 2006). This machine permits simultaneous treatment of 16 wires with diameters of 0.16 mm. The machine weighs 5.0 t and has a total electric heating power of 44 kW. The mean feed speed may be taken to be 16 m/s. It is not known how often the coolant is changed. Like the above machines, continuous operation may be assumed for tempering which yields a total output of 21.91×10^6 kg. The corresponding composition of the total process is given in Table 22.10. For sake of ease, the Ecoinvent processes in Table 22.11 were used to represent the total process.

Steel cord wire strand fabrication

Laying of single wires in cord wire strands is not an essential process in the production of micro steel fibres. When the fibres are cut to length, the advantageous properties of cord wire strand are lost because the individual wires separate. However, the handling of strand is easier on account of its high total cross section compared with a single wire. This enables a higher output for the last stage of fibre production. In this study, the laying process is considered in the production of micro steel fibres because currently only steel cord strand is used as basic material for the production of micro steel fibres. During the laying process, the single wires are coupled by twisting them in a helix around a common axis. Double-twist bunching machines operating in the OUT-IN configuration are usual in cord production. The pay-off bobbins are placed outside the machine and the cord wire strand is

Table 22.10 Compilation of data for the tempering process

Input	
Wire Ø 0.15 mm untreated	1.0 kg
16-wire continuous feed tempering machine 5.0 t	2.28×10^{-4} kg
Electrical energy (44 kW)	0.126 kWh
Inert gas	Not specified
Cooling water	Not specified
Output	
Wire Ø 0.15 mm tempered	1.0 kg
Consumption of inert gas	Not specified
Consumption of cooling water	Not specified

Table 22.11 Ecoinvent processes used for modelling the tempering process

Ecoinvent process	Quantity
Industrial machine, heavy, unspecified, at plant	2.28×10^{-4} kg
Electricity, low voltage, production UCTE, at grid	0.126 kWh

produced and spooled inside the cording system. The size of the machine is limited by the size of the take-up bobbin. Sections for wire pay-off, twisting with a rotor unit and a pay-out strand section result in total lengths up to 15 m. The steel cord wire strand machines weigh about 15 t and have a mechanical power of 43 kW. For a twist length of 2 cm and a twisting frequency of 7,500 Hz, the cord production speed is 2.5 m/s. Based on an operating life equivalent to the above machines, the total production is 5.78×10^6 kg. No data are available on material wear or consumption of other materials. This, together with the technical data in SKET (2006), results in the composition of the total process shown in Table 22.12. For sake of ease, the Ecoinvent processes in Table 22.13 were used to model cord production.

Cutting to length

In this part of micro steel fibre production, the high strength steel cord on metal bobbins is cut to the required length of the fibres. This is performed by a wire cutting machine. The wire is pulled into the machine by two oppositely rotating rollers and cut to length by blades rotating at high speed. The fibre length can be adjusted by varying feed speed and the rotational

Table 22.12 Compilation of data for steel cord wire strand production

Input	
27 wires Ø 0.15 mm each	1.0 kg
Cord production machine 15 t	2.60×10^{-3} kg
Electrical energy (43 kW)	0.937 kWh
Output	
Strand from 27 wires	1.0 kg
Wearable parts	Not specified

Table 22.13 Ecoinvent processes used for modelling steel cord wire strand production

Ecoinvent process	Quantity
Industrial machine, heavy, unspecified, at plant	2.60×10^{-3} kg
Electricity, low voltage, production UCTE, at grid	0.937 kWh

speed of the blades. An electric motor with continuously variable rotational speed powers the feed rollers and the cutting unit. Owing to the high rotational speed of the cutter, the fibres are thrown to the rear of the machine. A variation in fibre length of ±5% is usual. Moreover, the end of the fibres are crushed and the fibres slightly twisted along their length owing to the helical form of the cord wire strand. Cutting machines with a power of, for example, 5 kW and weighing, on average, 700 kg are used. The actual fibre production process is not continuous so a daily operating time of 8 h for 200 days per year is assumed. This results in a total operating time of 30,000 h. According to information supplied by the manufacturer, a good estimate of mean output is 150 kg/h which results in a total output of 4.50×10^6 kg. No information is available on material wear of the wire cutting machine. The composition of the total process is shown in Table 22.14. For simplicity, the Ecoinvent processes in Table 22.15 were used to model the cutting process.

Environmental impact of the production of 1 kg micro steel fibres Ø 0.15 mm

The potential environmental impact by the production of 1 kg high strength micro steel fibres is GWP100: 2.67 kg CO_2-eq., ODP: 1.39×10^{-7} kg CFC-11-eq., POCP: 6.80×10^{-4} kg C_2H_4-eq., AP: 1.40×10^{-2} kg SO_2-eq. and NP:

Table 22.14 Compilation of data for the cutting to length process

Input	
Strand made from 27 filaments	1.0 kg
Wire cutting machine 700 kg	1.56×10^{-4} kg
Electrical energy (5 kW)	0.033 kWh
Output	
Steel fibres, cut to length Ø 0.15 mm	1.0 kg
Wear of cutter	Not specified

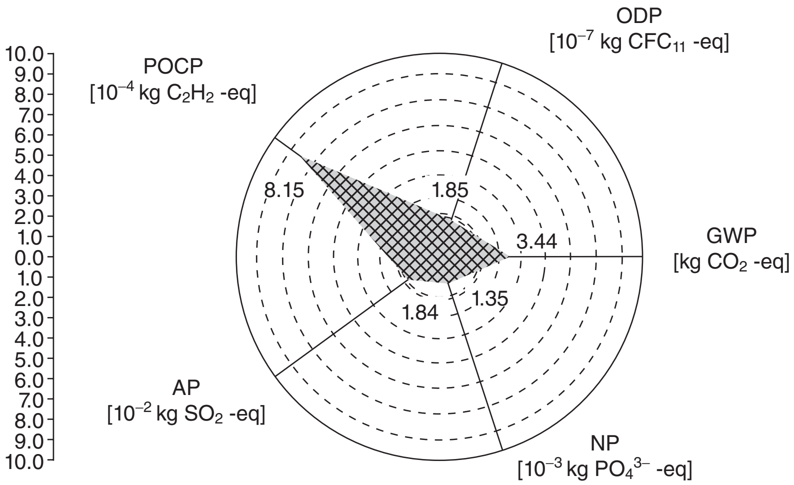
Table 22.15 Ecoinvent processes used for modelling the cutting to length process

Ecoinvent process	Quantity
Industrial machine, heavy, unspecified, at plant	1.56×10^{-4} kg
Electricity, low voltage, production UCTE, at grid	0.033 kWh

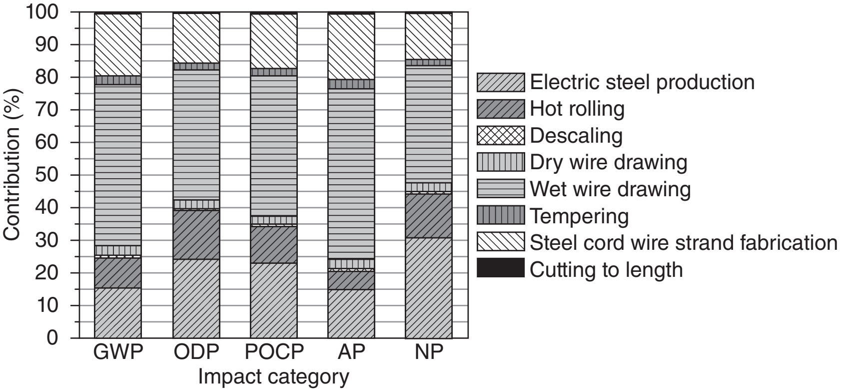
1.00×10^{-3} kg PO_4^{3-}-eq. (see Fig. 22.4). The results of the dominance analysis are shown in Fig. 22.5. The largest contributions come from the subprocesses 'production of electric steel' and 'wet wire drawing' which are 15–31% and 36%–52%, respectively, depending on the particular impact category. Both subprocesses 'steel cord wire strand fabrication' and 'hot rolling' contribute, depending on the impact category, approximately 14%–20% and 6%–5%, respectively, to the potential environmental impact. The subprocesses 'descaling', 'dry wire drawing', 'tempering' and 'cutting to length' may be neglected because they each contribute less than 3% to the total result. In the following, the subprocesses 'electric steel production' and 'wet wire drawing' are considered in more depth on account of their importance regarding the potential environmental impact of the total process.

22.3.6 Reinforcing and prestressing steel

For reinforcing steel the Ecoinvent data sets 'Steel, electric, unalloyed and low alloyed, at plant' and 'Steel, converter, unalloyed, at plant' as well as 'Hot rolling, steel' were taken into consideration. Thirty-four per cent was assumed to be electric steel (EAF steel) and 66% oxygen steel (converter steel), corresponding to the worldwide crude steel production in 2007 (see World Steel Association, 2009). To model prestressing strands, the aforementioned data sets were used, too. In addition to this, the data set



22.4 Results of impact assessment for production of 1 kg micro steel fibres.



22.5 Results of dominance analysis for the production of 1 kg micro steel fibres.

'Electricity, medium voltage, production UCTE, at grid' (0.1 kWh/kg strand) was used to account for the strand production process (i.e., energy for dry wire drawing and twisting). Results for reinforcing and prestressing steel are given in Table 22.16.

22.3.7 Water

Data for water was taken from the Ecoinvent database ('Tap water, at user') without any changes. Results for tap water are given in Table 22.17.

Table 22.16 Environmental impact of production of 1 kg reinforcing and prestressing steel

Material	GWP100	ODP	POCP	AP	NP
	kg CO_2-eq.	kg CFC-11-eq.	kg C_2H_4-eq.	kg SO_2-eq.	kg PO_4^{3-}-eq.
Reinforcing steel	1.106	5.106×10^{-8}	7.911×10^{-4}	4.608×10^{-3}	1.187×10^{-3}
Prestressing steel	1.156	5.315×10^{-8}	8.030×10^{-4}	4.880×10^{-3}	1.202×10^{-3}

Table 22.17 Environmental impact of 1 kg tap water at user

Material	GWP100	ODP	POCP	AP	NP
	kg CO_2-eq.	kg CFC-11-eq.	kg C_2H_4-eq.	kg SO_2-eq.	kg PO_4^{3-}-eq.
Mean UHPC	1341.3	5.627×10^{-5}	8.097×10^{-1}	4.498	4.391×10^{-1}

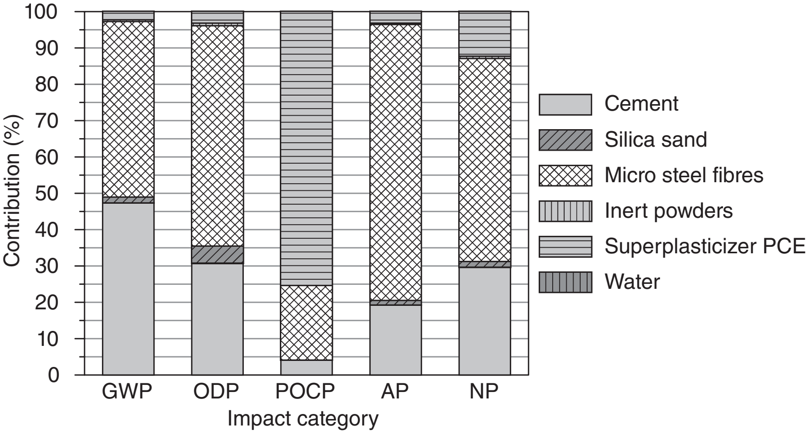
22.4 Impact assessment of UHPC at material level

Based on the process data derived in this study and the Ecoinvent process data referred to in Section 22.3, it is now possible to estimate the environmental impact of UHPC taking all its components into account. The mean UHPC composition in Section 22.1 is used in this calculation. The results for the estimation of the environmental impact of 1 m^3 UHPC are presented in Table 22.18. The corresponding contributions of the subprocesses are shown in Fig. 22.6.

The manufacture of micro steel fibres dominates with 48–76% the categories GWP, ODP, AP and NP. The contribution from cement production, approximately 20–47%, is also considerable for these categories. The impact POCP is essentially due to the production of PCE, 75%, and the production of micro steel fibres, 21%. The main effect of UHPC production on the environment is caused by the manufacture of micro steel fibres, cement and PCE. Lowering the amount of these materials in UHPC is the easiest way of producing UHPC which is more environmentally friendly. The production of PCE and its constituent acrylic acid cause the majority of POCP in UHPC production. Heat treatment of UHPC was not considered so far. If Portland cement and microsteel fibres are, for example, reduced by a factor of 0.5 and superplasticizer by a factor of 0.4, the environmental impact of UHPC can be lowered by 30%–45%.

Table 22.18 Environmental impact of 1 m^3 mean UHPC (see Section 22.1 for composition)

Beam type	Name	Height (mm)	Width (mm)	Moment of inertia (cm^4)	Mass (kg/m)
IPBv 320/305	Wide flange I-beam, heavy pattern	320	305	40,950	177.0
IPB 360	Wide flange I-beam	360	300	43,190	142.0
IPBl 400	Wide flange I-beam, light pattern	390	300	45,070	125.0
I 450	Narrow flange I-beam	450	170	45,850	115.0
IPE 500	Medium flange I-beam	500	200	48,200	90.7
559 / 6.3	Welded tube	559		41,780	85.9

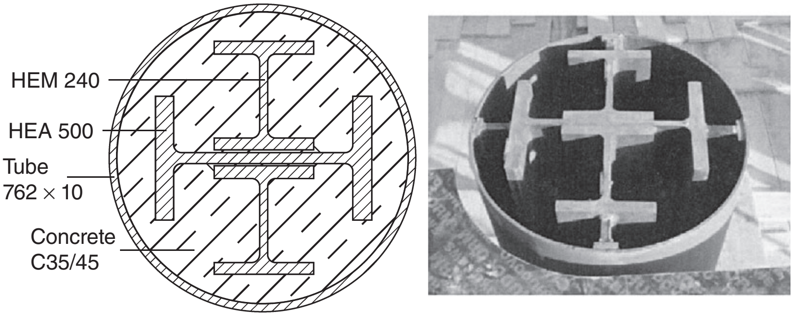


22.6 Results of dominance analysis for the production of UHPC (see Section 22.1 for composition).

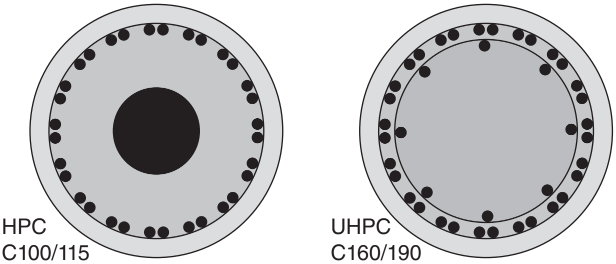
22.5 Impact assessment of structures made with UHPC

22.5.1 High rise building columns

Columns of high-bearing capacity and high slenderness are preferably used within high rise buildings when a translucent outer shell is realized (Sobek *et al.*, 2001). Steel and composite steel columns are normally used for such

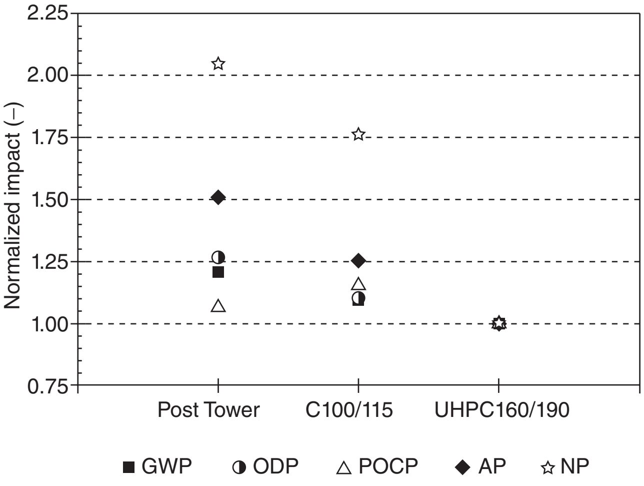


22.7 Cross section of Post Tower columns (left: Stahl-Informations-Zentrum, 2003; right: Sobek *et al.*, 2005).



22.8 Cross section of composite steel column with HPC and reinforced UHPC column (after Empelmann *et al.*, 2008).

purposes (Sobek *et al.*, 2001). The Post Tower (Bonn, Germany) is a high rise building with translucent outer shell and composite steel columns in the ground floor foyer. The columns have a height of up to 15.6 m, a diameter of 762 mm and a design load of 30,000 kN (Sobek *et al.*, 2001; Stahl-Informations-Zentrum, 2003; Empelmann, 2008) (see Fig. 22.7). The composite steel columns are made of a welded tube, a HEA 500 hot rolled girder and two HEM 240 hot rolled girders (Stahl-Informations-Zentrum, 2003). This yields a total steel cross-sectional area of 835 cm^2 and a corresponding mass of 655 kg/m steel. It is assumed that steel quality S355 is used for all hot rolled girders and the tube. Normal concrete (C35/45) is necessary as fire protection for the hot rolled steel girders (Sobek *et al.*, 2005). Within this study it is assumed, that normal concrete consists of 340 kg/m^3 cement (CEM I 42,5), 145 kg/m^3 fly ash, 175 kg/m^3 water and 1,750 kg/m^3 gravel. Besides the original Post Tower columns, another composite steel column made of high performance concrete (C100/115) and a reinforced UHPC (C160/190) column is analysed (see Fig. 22.8). The cross-sectional area of steel and reinforcing steel was taken from Empelmann *et al.* (2008).



22.9 Normalized environmental impact of columns analysed in this study.

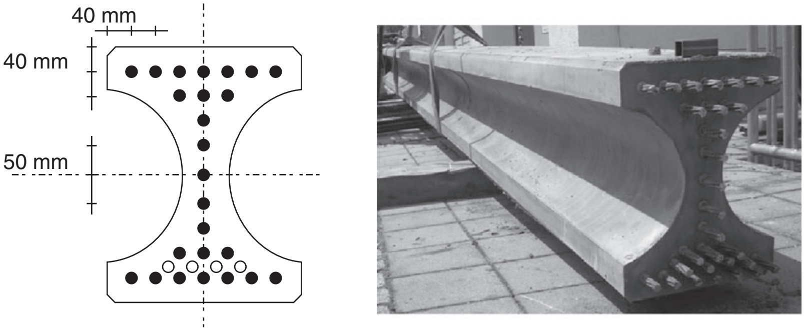
The composite steel column is made of 32 reinforcing rebars 32 mm in diameter (BSt 500 S) and a steel core 250 mm in diameter (S355). For high performance concrete, it is assumed that 470 kg/m^3 cement CEM I 52.5 R, 60 kg/m^3 silica fume, 80 kg/m^3 fly ash, 720 kg/m^3 sand, 1,150 kg/m^3 crushed gravel, 125 kg/m^3 water and 15 kg/m^3 superplasticizer (PCE) is used. The UHPC column is reinforced by 40 rebars 32 mm in diameter (BSt 500 S). A mean composition is taken into consideration as given in Section 22.1. To increase fire resistance of UHPC, 0.7 vol.% PP-fibres are added. Formwork was not taken into consideration. All columns were designed to have the same bearing capacity.

Results of the analysis are given in a normalized way in Fig. 22.9. Despite a high amount of Portland cement, micro steel fibres and superplasticizer, the UHPC column shows the lowest environmental impact. GWP and ODP of composite steel columns lies around 10–27% above the UHPC column. AP is 25–50%, NP 75–more than 100% higher than the UHPC column. POCP is roughly 7% higher than the UHPC column. All impact categories of Post Tower columns are dominated by hot rolled steel girders (90% to more than 95%). Roughly 60–70% of the environmental impact of composite steel columns (C100/115) is due to the steel core, 25–30% comes from reinforcing steel, and 5–20% from the high performance concrete. The impact categories GWP, ODP, POCP and AP of the UHPC column are dominated (63–67%) by the raw materials of UHPC. Only NP (62%) is mainly due to reinforcing steel. Therefore the overall environmental impact of the UHPC column could be lowered significantly using more eco-efficient concrete compositions, as given in Section 22.4.

22.5.2 Single span bending beam

Five different hot rolled I-beams and a welded tube are compared to a prestressed UHPC girder. The dimensions of the hot rolled I-beams and the welded tube were chosen to have a similar midspan deflection as the UHPC girder at a bending moment of 470 kNm. To take into account a realistic load-bearing behaviour of the prestressed UHPC girder, a four-point bending test was conducted on a 6 m single span beam. The moment of inertia necessary for the hot rolled I-beams and the welded tube was calculated employing the principle of virtual work. Deformations due to shear forces were neglected. In the LCA, only the production of raw materials necessary for the different girders and transportation processes are taken into account.

The UHPC beam tested had an I-shaped cross section with a height of 420 mm and a width of 320 mm (see Fig. 22.10). The cross-sectional area of the girder was 0.082 m^2 and the weight 236 kg/m. A total of 25 prestressed strands grade 1570/1770 (cross section each: 140 mm^2) were used. The vertical and horizontal centre-to-centre separation of the strands was 40 mm for all prestressed strands except the three centre strands (see Fig. 22.10). The strands were prestressed at a tensile stress of 1,410 MPa against the bed. Four additional strands without prestress were placed in between the two bottom strand lines. A UHPC as mentioned in Section 22.1 was used for the girder. The girder was cast in the prestressing bed using a concrete pump. The prestress was applied at an age of 5 days when the UHPC had a compressive strength of about 100 MPa. Afterwards the girder was heat treated at 90°C for 48 h in a water basin. The final compressive strength was about 160 MPa. The load controlled four-point bending test was performed with a bearing clearance of 6 m (girder length: 7 m) and a load distance of 2 m (third point). A midspan deflection of 20 mm was observed at a bending moment of about 470 kNm. The prestress at the lower flange was



22.10 Cross section of the prestressed UHPC girder.

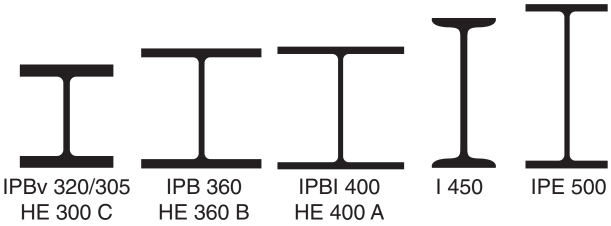
compensated at about 500 kNm. The four-point bending test was stopped when a bending moment of 700 kNm was reached. At this stage, multi-cracking with crack width below 0.1 mm (crack separation ≈10 mm) was observed over the whole tensile zone (bottom flange). Similar cracks, but parallel to the girder's centre-axis were found in the compressive zone (top flange). The web was free of cracks. Accounting for 29 strands, a volume of 0.543 m^3 of UHPC was necessary for the girder. A total of 223 kg of strands were used. For the heat treatment, a basic heat requirement of 53.7 kWh/m^3 UHPC with an additional 112 kWh for losses, etc., were taken into account. The haulage distance from the precast concrete plant to the construction site was assumed to be 30 km.

Employing the principle of virtual work, a moment of inertia of approximately 43,000 cm^4 was found to be necessary for the hot rolled I-beams to keep a deflection of 20 mm at a bending moment of 470 kNm. A Young's modulus of 210,000 MPa was assumed for the hot rolled steel. Hence five different standardized hot rolled I-beams and a welded tube with a length of 7 m were chosen with moment of inertia as near as possible to 43,000 cm^4 (see Fig. 22.11). Dimensions, masses and sectional properties of the hot rolled I-beams and the welded tube can be found in Table 22.18.

The hot rolled I-beams were modelled using the Ecoinvent data sets 'Steel, electric, unalloyed and low-alloyed, at plant', 'Steel, converter, unalloyed, at plant' and 'Hot rolling, steel' (66% of the steel needed for the girders was assumed to be oxygen steel, 34% electric steel). For the welded tube, the data set 'Welding, arc, steel' was added. A haulage distance to construction site (30 km) was taken into consideration for all girders using the data set 'Transport, lorry 28 t'.

Table 22.19 gives the LCA results of the different girders. The normalized ecological impact is shown in Fig. 22.12. For normalization, the results of the UHPC girder were set to 1.

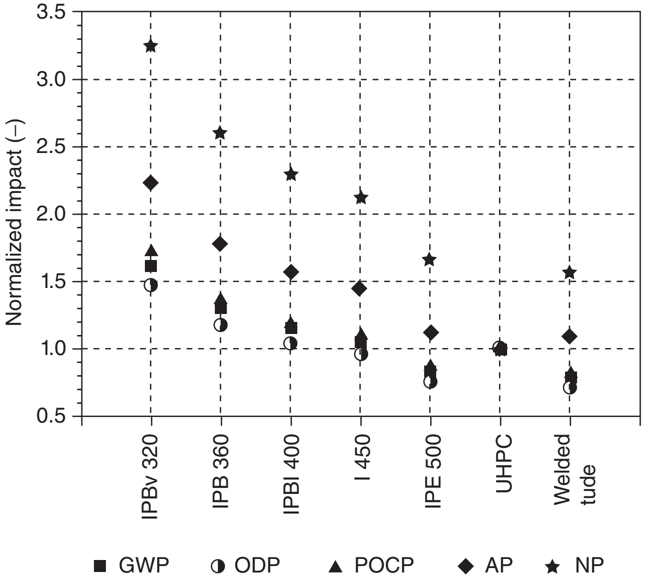
The ecological impact of the hot rolled I-beams decreases almost linearly with increasing height, i.e. decreasing mass. The impact of the hot rolled I-beam IPBv 320 is roughly twice as high as the impact of the IPE 500 I-beam. The ecological impact of the UHPC girder is significantly lower



22.11 Hot rolled I-beams analysed in this study.

Table 22.19 LCA results of the different girders investigated

Impact category	IPBv 320	IPB 360	IPBl 400	I 450	IPE 500	UHPC	Welded tube
GWP [kg CO_2-eq.]	1751	1405	1237	1138	897	1080	851
ODP [10^{-5}kg CFC11-eq.]	8.30	6.66	5.86	5.39	4.25	5.65	4.03
POCP [kg C_2H_4-eq.]	1.081	0.867	0.763	0.702	0.554	0.634	0.525
AP [kg SO_2-eq.]	8.641	6.932	6.103	5.614	4.428	3.895	4.198
NP [kg PO_4^{3-}-eq.]	1.853	1.486	1.309	1.204	0.949	0.571	0.900



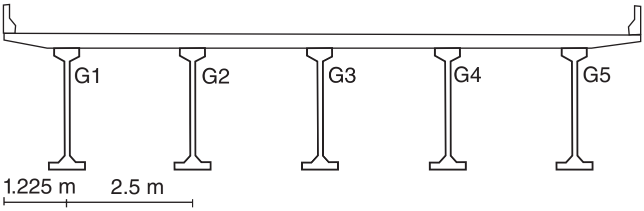
22.12 Normalized ecological impact of the different girders.

than the impact of the first four hot-rolled I-beams except ODP of I 450. In the case of the IPE 500 and the welded tube, only the GWP, ODP and POCP are lower when compared to the UHPC girder. When comparing girders with similar height, UHPC shows a lower or similar impact as hot-rolled I-beams (see IPBl 400 and I 450). The dominance analysis of the UHPC girder indicates that, depending on the impact category, 42–69% of the impact is due to the UHPC's raw materials. Between 21% and 47% of the impact is caused by the production of the strands. The heat treatment contributes appreciably to the GWP and ODP at 4% and 11%, respectively. Haulage accounts for 1–14% of the girders' impact. In view of the different possibilities available to reduce the ecological impact of UHPC, more sustainable load carrying structures can be made compared to hot rolled I-beams. Besides this, the UHPC girder investigated in this study could be optimized by, for example, using internal post-tensioning strands after the heat treatment is finished and finally a higher strength is reached. Furthermore, the compressive strength could be increased using an optimizing concrete composition and heat treatment. This would result in reduced cross sections and requirement on raw materials.

22.5.3 Precast single span bridge girder (bridge design model)

Two traffic bridge design models having one 45 m single span were analysed (see Almansour and Lounis, 2008). They consist of precast/prestressed girders and a cast-in-place deck slab. The deck slab is made of normal concrete in both cases. The bridge design was performed according to the Canadian Highway Bridge Design Code (Almansour and Lounis, 2008). The requirements include no cracking (i.e., fully prestressed) at serviceability limit state (SLS). One model was designed for the use of normal concrete with a compressive strength of 40 MPa for the girders, whereas the second model was designed for the use of UHPC with a compressive strength of 175 MPa for the girders. In both cases, low-relaxation strands grade 1,860 with a nominal diameter of 12.7 mm and a nominal cross-sectional area of 98.7 mm^2 were used (Almansour and Lounis, 2008). About 55% of the strands were arranged in straight tendons, whereas the remaining 45% were conventional deflected strand pattern groups (Almansour and Lounis, 2008). The deck slab thickness was 175 mm for both bridges. A normal concrete with a compressive strength of 30 MPa was used for the deck slab (Almansour and Lounis, 2008).

It was found that in the case of normal concrete, five so-called CPCI-1600 girders with a height of 1.6 m and a cross-sectional area of 0.499 m^2 are needed (see Almansour and Lounis, 2008). The corresponding girder spacing is 2.5 m and the length of the cantilever slab is 1.225 m on each side (Fig. 22.13). The prestressing steel ratio necessary was between 11‰ and 12‰ (Almansour and Lounis, 2008). A mean ratio of 11.5‰ was chosen for this study. No information on normal reinforcement was provided in Almansour and Lounis (2008). A mean composition of the 40 MPa normal concrete as listed in Table 22.20 was taken into account. A total amount of 110.98 m^3 concrete is necessary for the five girders. The Ecoinvent data sets used to model the materials as well as the raw materials chosen and the haulage



22.13 Cross section of the normal concrete traffic bridge design model.

Table 22.20 Composition of 40 MPa normal concrete and corresponding Ecoinvent data sets

Type of material	Amount (kg/m^3)	Haulage distance (km)	Total amount (kg)	Ecoinvent data set name
Portland cement 42.5	335	150	37,179.6	Portland cement, strength class 42.5, at plant
Fly ash	146	150	16,203.6[a]	–
Water	173	–	19,200.2	Tap water, at user
Superplasticizer	5	300	554.9	See Section 22.3.4
Round gravel	987	20	109,541.0	Gravel, round, at mine
River sand	759	20	84,236.7	Sand, at mine

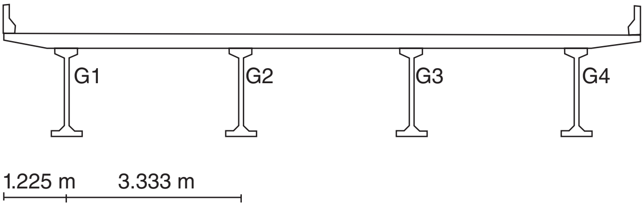
[a] Considered as industrial by-product without any environmental burdens, only transport is considered.

distance to the prefabrication plant can be found in Table 22.20. A 28 t lorry was considered to perform all the haulage.

To model the strands, the two data sets ‘Steel, electric, unalloyed and low alloyed, at plant’ and ‘Steel, converter, unalloyed, at plant’ were used. A total amount of 10,135.6 kg strands was used for the five girders. A haulage distance of 30 km to the construction site was taken into consideration for the precast girders.

For the 30 MPa normal concrete deck slab (98.0 m^3), the Ecoinvent data set ‘Concrete, normal, at plant’ which accounts for ready-mix concrete up to strength class C30/37 was chosen. A distance of 30 km to the construction site was taken into consideration for the ready-mix concrete.

For the bridge made of UHPC, only four CPCI-1200 girders are needed (see Almansour and Lounis, 2008, and Fig. 22.14). The height of a girder is 1.2 m, the cross-sectional area is 0.320 m^2 (see Almansour and Lounis, 2008). Accordingly, the girder spacing is 3.3 m and the side cantilever slabs of the



22.14 Cross section of the UHPC traffic bridge design model.

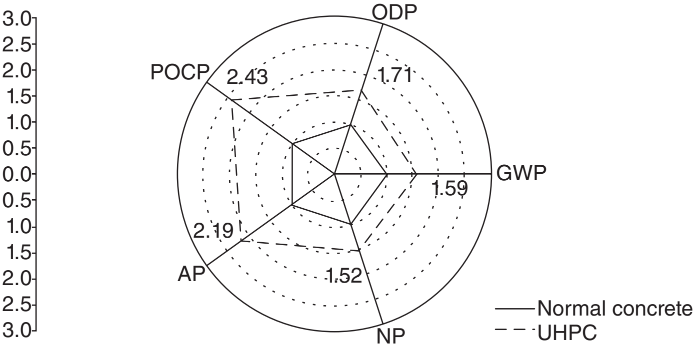
deck are 1.225 m each. The prestressing steel ratio necessary was between 23% and 26‰ (see Almansour and Lounis, 2008). A mean ratio of 24.5‰ was chosen for this study. No information on normal reinforcement was provided in Almansour and Lounis (2008). A mean composition of UHPC as mentioned above was taken into account. A total amount of 56.19 m^3 concrete is necessary for the four girders. The distance to the prefabrication plant was chosen to be 20 km for silica sand and quartz flower, 150 km for Portland cement and silica fume and 300 km for superplasticizer and micro steel fibres. A 28 t lorry was considered to perform all the haulage. A total amount of 11,077.9 kg of strands was used for the four girders. The strands were modelled as given above.

The UHPC girders were assumed to be heat treated for 48 h at 90°C. This heat treatment was modelled based on energy demand calculations for concrete heat treatments according to Altner and Reichel (1981). Therefore 53.7 kWh/m^3 UHPC basic heat requirement was taken into account (total basic heat requirement: 3,017.4 kWh). An additional 1,235.0 kWh heat demand was considered to cover all heat losses as well as the energy necessary for heating the body of the heat treatment chamber. Fifth-six per cent of the total heat requirement was assumed to be supplied by light fuel oil, 40% by natural gas and 4% by heavy fuel oil, which is a typical energy mix for European sand lime brick production (see Kellenberger *et al.*, 2000). The Ecoinvent data sets 'Heat, light fuel, at boiler 10 kW, non-modulating', 'Heat, natural gas, at boiler condensing modulating' and 'Heat, heavy fuel oil, at industrial furnace 1 MW' were chosen to model the three energy sources. The normal concrete deck slab was modelled as described before.

The results obtained by the LCA for the two bridge design models are given in Table 22.21. The normalized ecological fingerprint of the two bridge design models is shown in Fig. 22.15. To normalize the diagram, the results of the normal concrete bridge design model were set to 1. Despite the fact that in the case of the UHPC bridge design model only half the amount of concrete was needed for the girders, it can be seen that using a mean UHPC results in a significantly higher ecological impact compared to normal concrete. The impact categories for the UHPC used in this study are between 1.5-fold and 2.4-fold higher with respect to the normal concrete design

Table 22.21 Results of the LCA for the two bridge design models

Impact category	Normal concrete bridge design model	UHPC bridge design model
GWP [kg CO_2-eq.]	76,277	121,278
ODP [kg CFC11-eq.]	0.00334	0.00572
POCP [kg C_2H_4-eq.]	23.5	57.2
AP [kg SO_2-eq.]	175.6	385.3
NP [kg PO_4^{3-}-eq.]	34.4	52.1

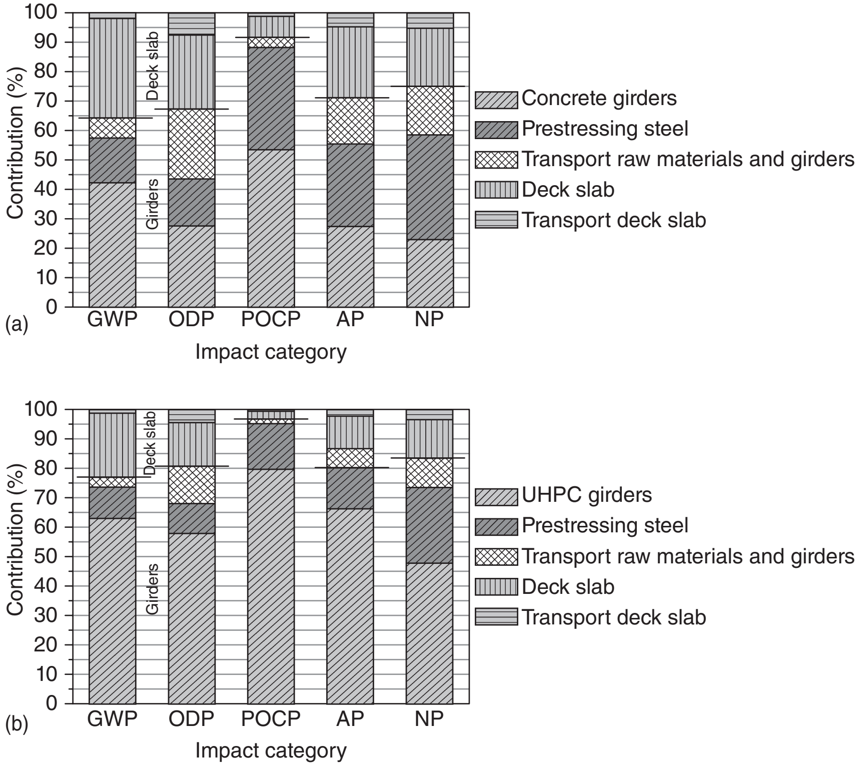


22.15 Normalized ecological fingerprint for the two bridge design models.

model. To determine the reasons for this, a dominance analysis was carried out which gives the contributions of the different raw materials to the overall ecological impact (Fig. 22.16).

The ecological impact of the normal concrete bridge design model is mainly (between 65% and 92%) due to the girders. In the case of GWP, POCP, AP and NP, the normal concrete together with the prestressing strands account for more than 50% of the impact of the girders. Haulage of raw materials and the girders is only important for ODP. The haulage of ready-mix concrete for the deck slab is less than 10% of the overall impact. In the case of the UHPC bridge design model, 75–97% of the overall ecological impact is caused by the girders. Within the girders, the highest contribution comes from the UHPC. For the impact categories GWP, ODP, AP and NP, this is mainly due to the high amount of Portland cement (about 20–48%) and micro steel fibres (about 50–75%). The POCP is mainly caused by the production of PCE based high-performance superplasticizer.

The use of UHPC with a high cement and a high micro steel fibre content as in this study did not yield a more environmentally friendly bridge



22.16 Results of dominance analysis for (a) the normal concrete bridge design model and (b) the UHPC bridge design model.

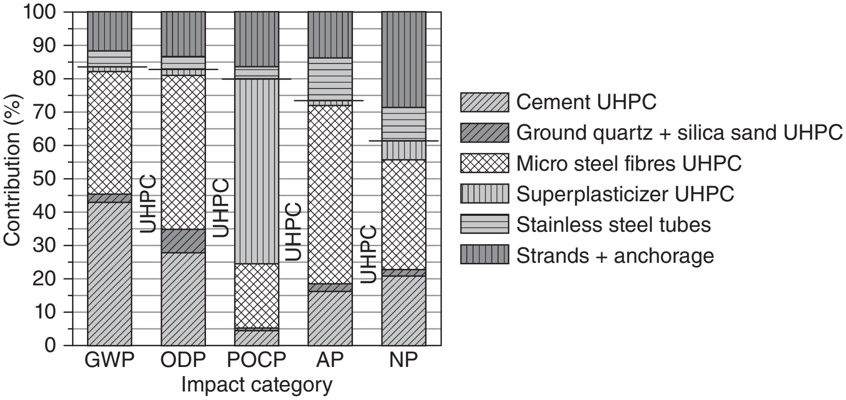
construction. The ecological impact of the UHPC could be lowered significantly by reducing the amount of Portland cement as in Gerlicher *et al.* (2008) and Stengel (2008) and by reducing or exchanging the micro steel fibres as far as possible. A reduction of 50% of the UHPC's ecological impact would lead to an overall ecological impact of the UHPC bridge design model in the range of the normal concrete design model. However, it should be kept in mind that according to the current literature, UHPC shows a better durability when compared to normal concrete. Denarié *et al.* (2009) assumed a service life of a bridge rehabilitation using UHPC at least twice that of normal concrete. Therefore a longer service life of the structure can be expected when using UHPC. This may compensate the higher ecological impact of the raw materials used. Besides this, the girders are designed according to current design codes and methods. Using a design method appropriate for the material involved may also result in a more efficient structure.

22.5.4 Bridges made of UHPC

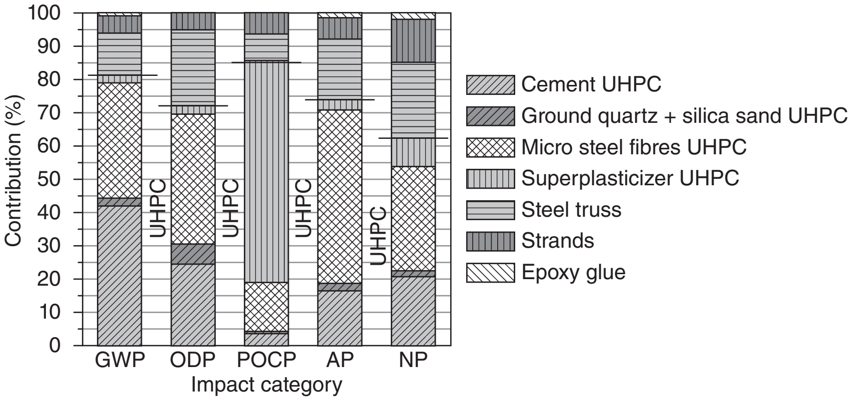
In this section, the results of life cycle assessments (LCA) performed for three bridges in which UHPC was an essential part of the structure are summarized. The bridges investigated are the Sherbrooke footbridge in Canada (spanning 60 m, precast, prestressed pedestrian bridge, post-tensioned open-web space reactive powder concrete truss, truss made of stainless steel tubes filled with reactive powder concrete), the Kassel Gärtnerplatz footbridge in Germany (steel and UHPC composite space frame, consists of precast prestressed upper chords and precast prestressed bridge deck elements both made of UHPC, lower chords and diagonals made of tubular steel), and the Wapello road bridge in the US (simple span bridge with a three-beam cross section, prestressed prefabricated UHPC beams). Details on the structures and the materials used can be found in Stengel and Schießl (2008) and (2009).

The contributions of the different materials to the potential environmental effect of the Sherbrooke footbridge truss may be taken from the results of the dominance analysis (see Fig. 22.17). Depending on the impact category, the UHPC used contributes between 60% and 84% to the environmental effect of the Sherbrooke footbridge truss. The stainless steel tubes contribute between about 3% and 10% and the strands and anchors together between roughly 12% and 28%. The effect of the mixing water for the UHPC is negligible and has thus not been included in the results.

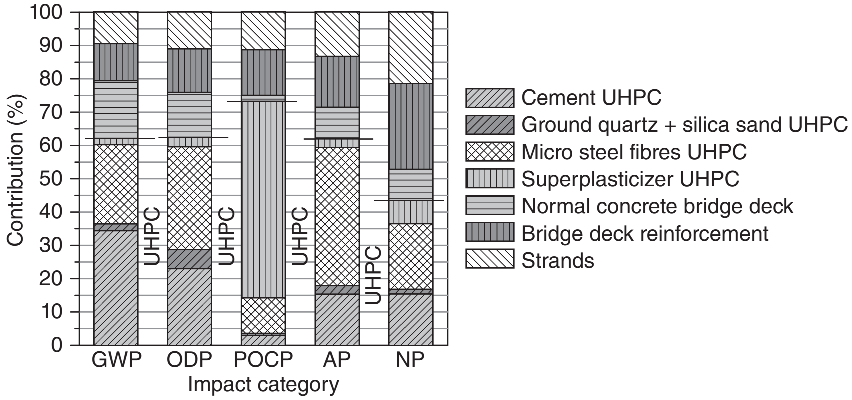
The contributions of the different materials to the potential environmental effect of the Gärtnerplatz footbridge may be taken from the results of the dominance analysis (Fig. 22.18). Depending on the impact category, the UHPC used contributes between 72% and 85% to the environmental effect of the Gärtnerplatz footbridge. The stainless steel tubes of the web



22.17 Contribution of materials to the environmental impact of the Sherbrooke footbridge.



22.18 Contribution of materials to the environmental impact of the Gärtnerplatz footbridge.



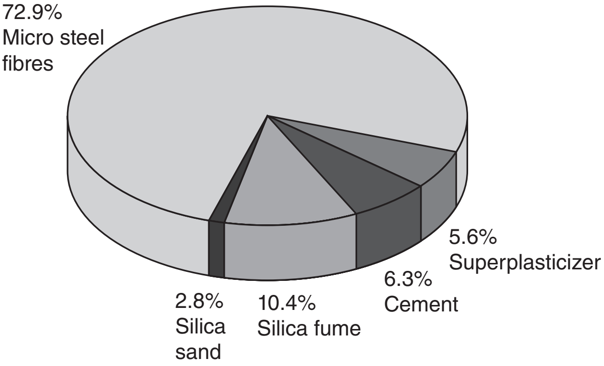
22.19 Contribution of materials to the environmental impact of the Wapello road bridge.

diagonals contribute between about 8% and 23% and the strands between roughly 5% and 13%. The effect of the epoxy resin mortar with respect to the impact indicators GWP, ODP and POCP can be neglected. It contributes 1.4–1.9% to the categories AP and NP, respectively. The effect of the mixing water for the UHPC need not be considered.

The contributions of the different materials to the potential environmental effect of the Wapello road bridge may be taken from the results of the dominance analysis (Fig. 22.19). Depending on the impact category, the UHPC used contributes between 44% and 74% to the environmental effect of the Wapello road bridge. The bridge deck made with normal concrete and its reinforcing steel contribute about 15–34%, the strands roughly 10–22%.

Table 22.22 Market price of raw materials used for UHPC

Raw material	Market price (€/1000 kg)
Well cement	110
Portland cement	70
Silica fume	500
Quartz powder fine	55
Quartz powder very fine	75
Limestone powder fine	30
Limestone powder very fine	40
Silica sand	26
Superplasticizer (PCE)	1,500
Micro steel fibres (0.15 mm)	2,500
Steel fibres (0.40 mm)	1,800



22.20 Contribution of raw materials to the overall cost of UHPC.

22.6 Cost of UHPC

Table 22.22 gives an overview of the cost of raw materials used for UHPC. All figures are market prices evaluated in the year 2009. Taking into consideration a mean composition as given in Section 22.1, the cost for 1 m^3 of UHPC is around 830 €/m^3. The contribution of raw materials to the overall cost can be seen in Fig. 22.20. More than 70% of the overall cost is due to micro steel fibres.

22.7 Conclusions and future trends

The analysis of the bridge design models showed that the structure made of UHPC with a high content of Portland cement, micro steel fibres and superplasticizer (PCE) has an up to 2.4-fold higher ecological impact than the normal concrete structure. Using UHPC with a high content of Portland cement and micro steel fibres therefore does not automatically result in a more sustainable structure when compared to normal concrete. A future goal in the development of UHPC and UHPC constructions should therefore be the reduction of the content of high strength micro steel fibres and the use of fibres produced by less elaborate methods. This can be achieved, for example, by optimizing fibre geometry, distribution and orientation as well as the bonding properties of the fibre in the UHPC matrix. Moreover, an attempt should be made to use fibres with a larger diameter which would reduce the energy needed for the energy-intensive wet drawing process. Since energy requirement has a large effect on the environmental impact of fibre production, an attempt should be made to determine the real energy consumption by monitoring the production process in cooperation with the steel wire manufacturing industry. Besides this, the content of Portland cement should be reduced using, for example, finely ground blast furnace slag, inert lime powder or inert quartz powder. An amount of 450–600 kg/m^3 should be sufficient to achieve compressive strength of about 200 MPa. A reduction in, for example, 50% or more of the Portland cement and the superplasticizer would already lead to an ecological impact closer to the normal concrete structure. If, then, the amount of micro steel fibres can be reduced, a structure with similar ecological impact could be obtained. The aforementioned conclusions are only valid when the durability of the materials is not taken into consideration. Taking into consideration a higher durability and therefore a longer service life, UHPC ultimately offers a great chance for building in a more sustainable way with concrete.

The aim of this study was to determine the extent to which the environmental effects of construction can be reduced by using the new UHPC material. Therefore with the help of life cycle assessment methods (LCA) the ecological impact of both a UHPC design model and a normal concrete design model for a 45 m single span traffic bridge with precast girders was investigated. The two design models possessed the same load-bearing capacity, dimensions of the deck slab and requirements on the serviceability limit state (SLS). Therefore the two materials used can be compared directly with each other. Besides this, a 7 m single span prestressed UHPC girder was investigated and compared with different hot rolled I-beams and a welded tube. To account for a realistic load-bearing behaviour of the prestressed UHPC girder, a four-point bending test was conducted on a 7 m single span beam. The dimensions of the hot rolled I-beams and the welded tube were chosen to have a similar deflection as the UHPC girder under a bending moment of 470 kNm. Therefore the UHPC and steel girders can be compared with each other.

The analysis of the bridge design models showed that the structure made of UHPC with a high content of Portland cement, micro steel fibres and superplasticizer (PCE) has an up to 2.4-fold higher ecological impact than the normal concrete structure. Using UHPC with a high content of Portland cement and micro steel fibres therefore does not result in a more sustainable structure. The most efficient way to reduce the impact of the UHPC is to lower the amount of Portland cement and micro steel fibres. A reduction of, for example, 50% or more of the Portland cement and the superplasticizer as done, for example, in Gerlicher *et al*. (2008) and Stengel and Schießl (2012) would already lead to an ecological impact closer to the normal concrete structure. If, then, the amount of micro steel fibres can be reduced as proposed by Stengel (2008) and Stengel and Schießl (2012), a structure with similar ecological impact could be obtained. Taking into consideration a higher durability and therefore a longer service life, UHPC ultimately offers a great chance for building more sustainability with concrete.

The ecological impact of hot rolled I-beams was found to decrease almost linearly with increasing height, i.e. decreasing mass of the beam. When comparing the single span prestressed UHPC girder to the hot rolled I-beams, a lower or at least similar ecological impact was observed for the UHPC. In view of the numerous ways of reducing the ecological impact of UHPC already mentioned, more sustainable load-bearing systems could be made with UHPC compared to hot rolled steel.

The investigations show that UHPC can enable more sustainable buildings compared to normal concrete or steel structures. This is only valid for the structures and load-bearing systems investigated here. The ecological impact of state-of-the-art UHPC can be lowered to a great extent by reducing the amount of Portland cement and micro steel fibres. According to the literature, there is no need to keep the amount of Portland cement and micro steel fibres as high as currently practised. Together with a higher durability and therefore a longer life span, this would lead to significantly more sustainable structures (see, e.g., a study on the use of high performance concrete by Habert *et al*., 2012).

The study on LCA of UHPC was performed at the Centre for Building Materials (cbm) at the Technical University of Munich (TUM) within the six-year priority programme No. 1182 launched by the German Research Foundation (DFG) in 2005.

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23

Life cycle assessment (LCA) of fibre reinforced polymer (FRP) composites in civil applications

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Abstract: Fibre reinforced polymer (FRP) composites have been increasingly used in civil engineering applications. These materials provide cost-effective alternatives to conventional materials and their use often has associated financial benefits that are immediate and/or can be foreseen over the service life of the structure. As concerns grow over climate change and depletion of natural resources, it is important that the environmental implications of their use should be properly considered. One of the techniques developed to address this issue is life cycle assessment (LCA). In this chapter, a brief overview is first given of FRP composites and their application in construction, after which the general framework and key features of LCA are discussed in detail. Three LCA case studies of FRP composites, from both the composites and civil engineering industry, are then presented and discussed within the general LCA framework and with reference to current industry practices.

Key words: fibre reinforced polymer, life cycle assessment, environmental impact, composites, construction, civil engineering, bridge.

23.1 Introduction

23.1.1 Fibre reinforced polymer (FRP) composites

Fibre reinforced polymer (FRP) composites, also referred to as advanced polymer composites (APC), consist of high-strength and stiffness fibres embedded in a high-performance thermosetting polymer matrix. The fibres provide the primary structural performance of the material, e.g. strength and stiffness, while the polymer matrix locates, transfers load between and provides environmental and damage protection to the fibres. FRP composites may be manufactured from a variety of fibres and polymers and thus offer great flexibility in material choices to meet any specific application requirements. For structural applications in construction, common fibres used are carbon, glass and aramid (better known under the trade name Kevlar) fibres. In the case of the polymer matrix, vinyl-ester, epoxy and polyester are the most commonly used materials.

There are often other compounds within the matrix such as fillers, catalysts and accelerators, which, in the case of fillers, are used to replace part of the polymer within the matrix and in the case of catalysts and accelerators, are added to adjust the cure time and temperature. The addition of fillers does not compromise the structural properties of the composite but will bring significant environmental benefits, which will be further discussed later in the chapter.

FRP composites can be manufactured using a variety of processes, amongst which are wet lay-up, vacuum pre-preg, resin infusion and pultrusion. Pultrusion is an automated process for making constant cross-section profiles, but cannot reliably produce components with a curvature or varying cross section. Wet lay-up, vacuum pre-preg and resin infusion methods provide great freedom for changing geometry but are considerably more labour intensive. In the construction industry, standard components (e.g., FRP bars, plates, bridge deck sections) made from the pultrusion process are most commonly utilized. For more complex or more aesthetically pleasing profiles, the structure would typically need to be bespoke, using methods such as vacuum pre-preg.

For the basic mechanical properties of the component parts, their interaction and the manufacturing techniques of FRP composites, readers may refer to publications by, among others, Hollaway and Head (2001), Hollaway (2009a) and Kim (1995). The important physical and in-service characteristics of thermosetting polymers and their effects on the properties of the FRP materials and FRP structural components have been comprehensively discussed in Hollaway (2009b, 2010).

23.1.2 Uses of FRP composites in construction

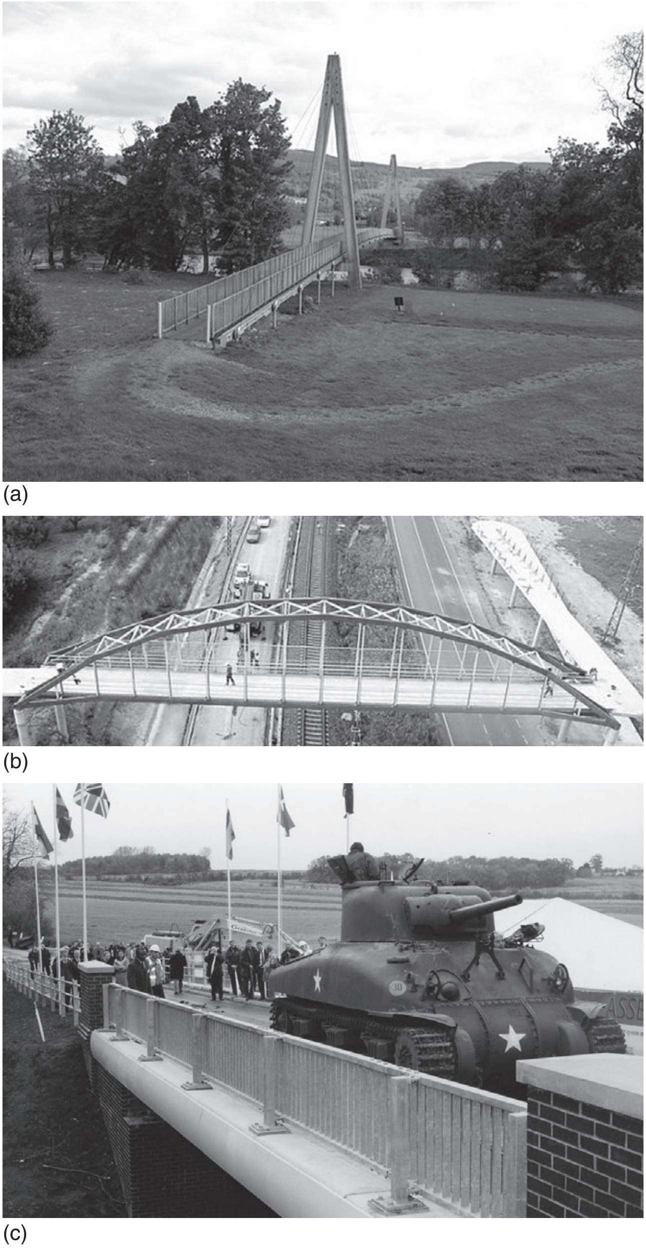
The use of FRP composites in construction may be traced back to the 1970s when semi-load-bearing and infill panels made of glass FRP (GFRP or GRP) composites using the wet lay-up technique, were used in building constructions. Architectural embellishment rather than structural performance and durability motivated these early applications of FRP composites (Hollaway, 2010). In the late 1980s, automated unit FRP building blocks (Advanced Composites Construction System (ACCS) plank units, now marketed by Strongwell as Composolite), manufactured by the pultrusion method, were introduced into the construction industry. So far, the use of FRP composites in building applications has been rather limited, mainly due to problems associated with their lack of fire resistance as polymers begin to soften at temperatures near its glass transition temperature (T_g) and decompose when heated to higher temperatures. A review is given in Kendall (2007) of FRP buildings for the future.

The use of FRP composite as a structural material started in the 1980s, largely driven by the need for durable and high performance materials to replace the more conventional materials in aggressive environments sometimes encountered in civil engineering applications. The use of advanced composites in civil infrastructure has been comprehensively introduced in Hollaway (2009b, 2010) and Canning *et al*. (2007). Structural applications in buildings have largely been limited to strengthening of structural components (e.g., beams, slabs and columns) using FRP plates or sheets. In these applications, probable effects of accidental loss of FRP strengthening, e.g. in case of fire, should be considered and mitigation measures taken to provide the necessary fire endurance required in relevant design codes.

Structural applications of FRP composites for civil infrastructure have been mainly in the area of bridge engineering: (a) all-composite or hybrid bridges, the latter typically in the form of FRP bridge decks in conjunction with conventional steel or concrete girders; and (b) rehabilitation and strengthening of existing bridge structures.

The first all-FRP pedestrian bridge was built in Tel Aviv, Israel in 1975 (Hastak *et al*., 2004). Since then, many others have been constructed in Asia, Europe, North America and other parts of the world. These bridges are mainly constructed of pultruded FRP shapes (e.g., ACCS planks) or using resin infusion techniques. The developments in all-composite FRP footbridges have been followed by FRP decking systems, often made of proprietary experimental sections and details, for vehicular bridge constructions. The first FRP bridge on a public highway in the UK, West Mill Bridge over the River Cole in Oxfordshire, was completed in 2002 using the Asset decking system. The Asset decking system, wholly composite or in conjunction with more conventional materials such as steel and concrete, has also been used in the UK for bridges over railway lines (e.g., Standen Hey bridge near Clitheroe, Lancashire) and motorways (e.g., Mount Pleasant Bridge over the M6 in Lancashire). A state-of-the-art review is given of composites for highway bridge applications in Mosallam (2007). Figure 23.1 shows examples of some FRP bridges. The main advantages of FRP bridge decks are their light weight (typically 10–20% of the structurally equivalent of a reinforced concrete (RC) deck) that enables swift installation and leads to much reduced traffic disruption, and high durability during their service life that requires minimal maintenance of the structure.

Another major area of structural application for the FRP composites is their use for strengthening (e.g., flexural, shear, seismic) of existing concrete or steel members (e.g., beams, columns, slabs and walls). FRP plates/sheets are typically adhesively bonded to the soffit of existing concrete/metallic members for flexural strengthening (see Fig. 23.2); FRP sheets can be wrapped around existing columns to improve its compressive and/or seismic



(a)

(b)

(c)

23.1 Examples of FRP bridges: (a) Aberfeldy Footbridge, Scotland (picture adapted from www.geograph.org.uk); (b) Lleida Footbridge, Spain (courtesy of Fiberline Composites, Denmark); (c) West Mill Bridge, England (courtesy of Mouchel, UK); (d) Mount Pleasant Bridge, England (courtesy of Mouchel, UK).

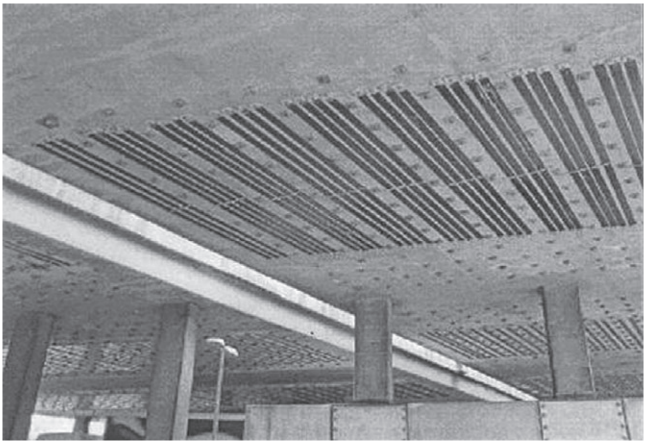
(d)

23.1 Continued

resistance. FRP bars have also been used to replace steel bars as reinforcement in concrete structures. This latter use could be particularly beneficial in coastal areas, or in areas where the seasonal use of de-icing salt causes traditional steel reinforcement to corrode, or in concrete structures which may be required to be devoid of metal. There is a vast body of research and publications on these topics (e.g., Meier *et al*., 1993; Hollaway and Cadei, 2002; Teng *et al*., 2002; Oehlers and Seracino, 2004; Rizkalla *et al*., 2006; Zhang, 2010) with numerous field applications around the world, and this will not be discussed in detail here.

FRP composite is most efficiently utilized when in conjunction with more conventional materials – concrete in particular – in such a way that the two materials are used to their strengths (i.e., FRP composites in tension and concrete in compression). Therefore, hybrid structural systems innovatively combining FRP composites with conventional materials are currently and continue to be the main focus for use of FRP composites in new construction. Indeed, novel ideas have already materialized with the construction of a novel advanced composite motorway bridge using ‘duplex’ beams on the Highway Cantabrico in Spain. Detailed information on the duplex beam elements may be obtained from Hulatt *et al*. (2004).

The superior mechanical properties and high durability of FRP composites together with their light weight may provide more cost-effective solutions over the life cycle of the structure due to reduced installation and maintenance cost. This is often the case for structures such as bridges, for which maintenance cost (and associated cost for traffic disruption)



(a)

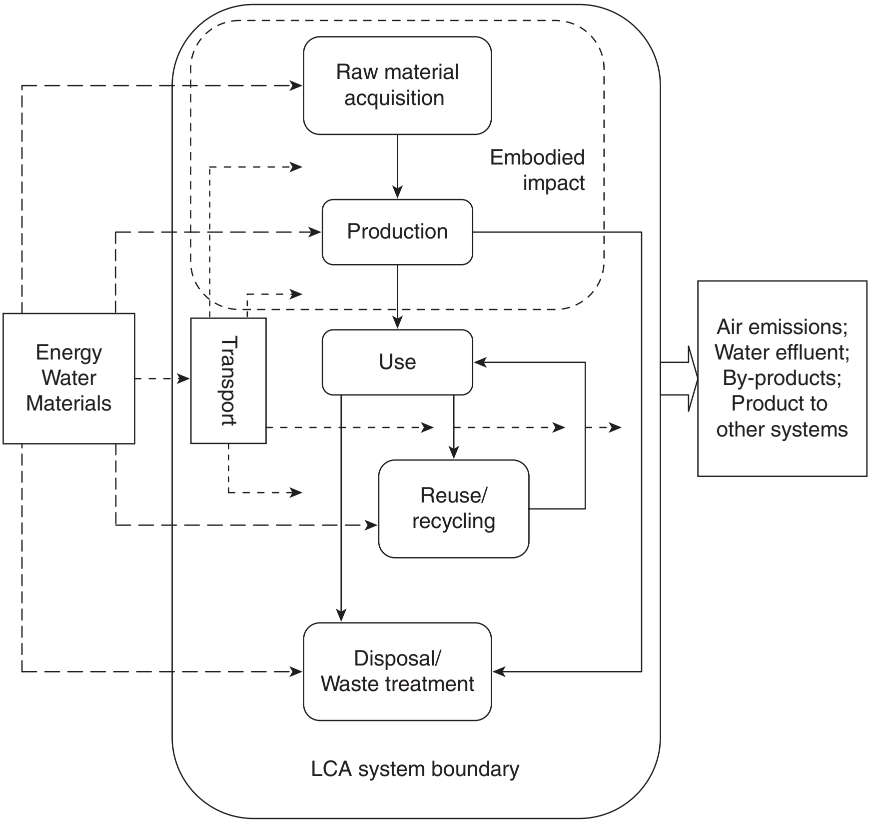
(b)

23.2 Examples of FRP strengthening of existing structures: (a) concrete bridge in Greater Manchester; (b) cast iron girder on East London line (pictures courtesy of Mouchel, UK).

represents a high proportion of the total life cycle cost. Thus, their use often has associated financial benefits that are immediate and/or can be foreseen over the service life of the structure. However, as concerns grow over climate change and resource depletion, it is important that the environmental – in addition to cost – implications of their use should be properly considered (Zhang and Canning, 2011). One of the techniques developed for this purpose is life cycle assessment (LCA), which will be introduced in the following section.

23.2 Life cycle assessment (LCA) method

There has been growing awareness of the importance of the environmental impacts associated with products or services. In the case of FRP composite, their in-service environmental benefits can be more easily appreciated, as (a) its light weight enables swift installation, and (b) less maintenance, and thus traffic disruption, is required due to its high durability. However, to gain a thorough understanding of the environmental impacts of FRP composite and indeed, any materials, products or services, it is important that the environmental impacts throughout its life cycle (i.e., cradle to grave) should be considered from raw material acquisition through production, use and end-of-life treatment. The technique to address this issue is known as life cycle assessment (LCA) (ISO, 2006). Figure 23.3 shows the flow process of LCA.



23.3 Flow process of LCA (based on ISO, 2006).

23.2.1 Introduction

LCA studies typically include four phases: (a) goal and scope definition; (b) inventory analysis; (c) impact assessment; and (d) interpretation of results.

The scope, including the system boundary and level of details, depends on the subject and the intended use of the study.

The life cycle inventory analysis (LCI) involves the compilation and quantification of input–output data for the product system throughout its life cycle. There are two fundamentally different approaches to inventory analysis, namely process-based analysis and input–output-based analysis, each with their own strengths and limitations. This will be discussed in detail later in the section.

The life cycle inventory assessment (LCIA) provides additional information to help understand and evaluate the magnitude and significance of the environmental impacts for the product system throughout its life cycle. Examples of common impact categories include extraction of natural resources, energy consumption, climate change, hazardous releases to water, air and soils.

In the life cycle interpretation phase, the results of the LCI and/or LCIA are summarized to form the basis for conclusions and recommendations in accordance with the defined goal of the study. The life cycle interpretation phase also makes provisions for links between LCA and other environmental management techniques (e.g., environmental impact assessment and environmental performance evaluation) when the information developed in an LCA study is used in more comprehensive decision-making processes.

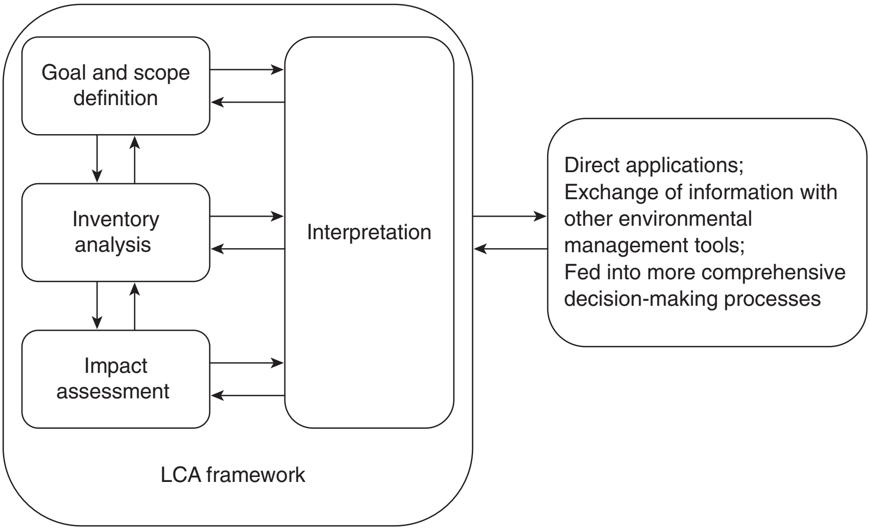
LCA is an iterative process (as shown in Fig. 23.4), when the results from later stages are constantly fed back to the previous stages and adjusted accordingly in order for the goal of the study to be fulfilled. Where the goal of an LCA can be satisfied by performing only an inventory analysis and an interpretation, an LCI study may be adequate. LCA may have many derivatives when only certain aspect(s) such as energy consumption and carbon dioxide (CO_2) emissions of the environmental impacts are considered. For more detailed information on the technique and its applications, readers may refer to the ISO 14040 series of standards.

23.2.2 Key features of LCA

There are several key features of LCA, as will be discussed below.

Life cycle perspective

LCA takes a life cycle approach by considering the entire life cycle of a product/service, from raw materials extraction, product manufacturing

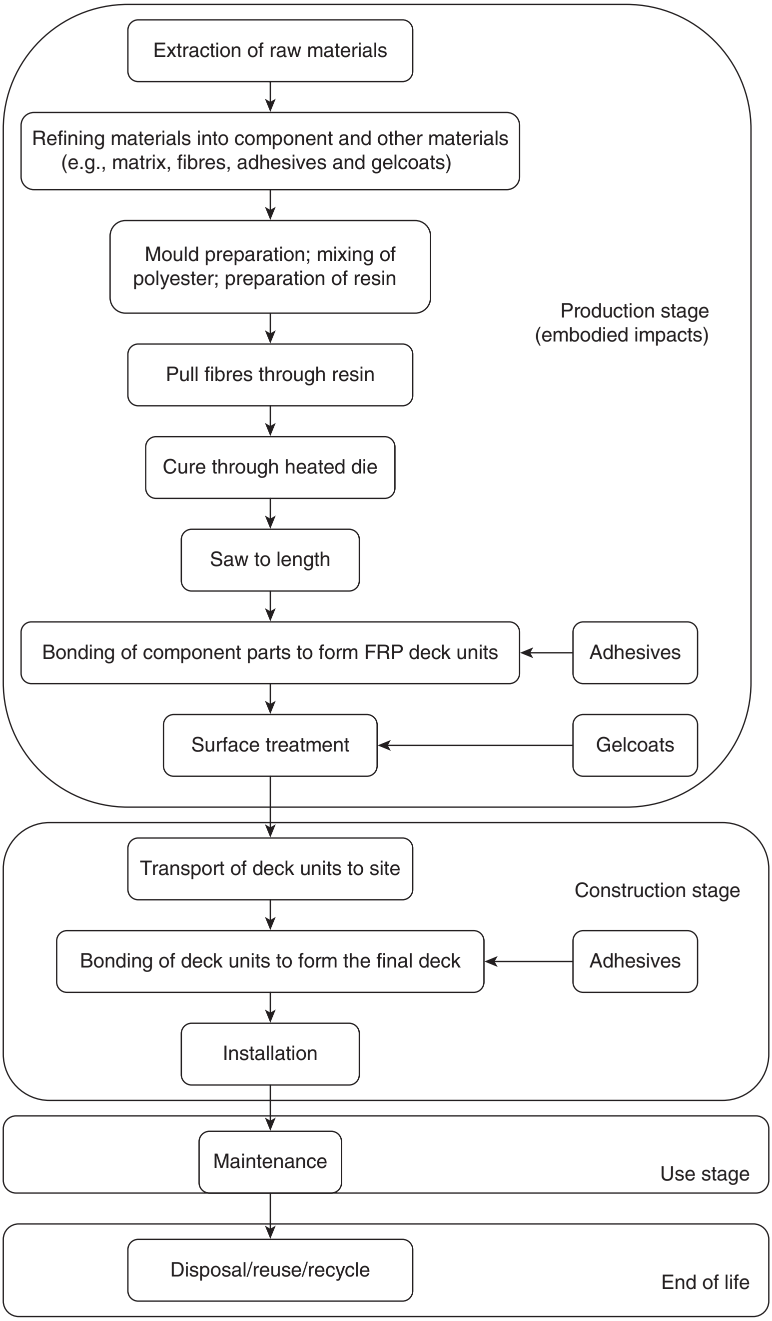


23.4 LCA methodology (based on ISO, 2006).

through use (and maintenance), to end-of-service-life treatment (e.g., reuse, recycle or disposal). Through a systematic overview, the occurrence and/or shifting of environmental burdens between different stages or processes can be identified (as can be seen in Fig. 23.3).

The inventory should include all – direct and indirect – processes and activities that occur in the course of making the product system. Take the manufacturing of FRP composite decking units using the pultrusion technique (see Fig. 23.5) as an example; it should include all the main processes occurring at the production phase, as well as indirect processes which may include transport of equipment and materials between sites (e.g., transport of equipment to the site of extraction, transport of raw materials to the production plant), operation of machines (at stages prior to manufacturing), manufacturing of the mould and generation of electricity and fuels.

Whether a process should or not be included in the inventory depends, in the present author's opinion, upon whether the process is induced by making the particular product system. In the above example, processes such as making of the pultrusion machine and construction of the manufacturing plant, although essential, do not occur as the result of making the particular product system and should not be included in the inventory. Erection of temporary tent for site fabrications, on the other hand, should be part of the inventory as this will otherwise not be necessary. Environmental impacts associated with a machine can be evaluated, for example, by considering the proportion of the embodied environmental impacts based on the ratio of the total working hours (in making the production system) to the service



23.5 Pultruded FRP composite bridge deck through its life cycle.

life of the machine (Itoh and Kitagawa, 2003). In the present author's opinion, however, these impacts may be more appropriately accounted for in LCA studies of different product systems, e.g. construction machinery.

The concept of LCA is complete in nature as it aims to cover all stages of the life cycle. In practice, the life cycle approach is sometimes taken to assess the environmental impacts over certain stages of a product or service, e.g. cradle to gate. The impacts that occurred from cradle to gate are often called 'embodied impacts' (see Figs 23.3 and 23.5). When an LCA covers stages that have not already occurred in time, the environmental impacts associated with these stages can only be predicted values and therefore are approximate in nature.

Function-based product system

LCA models the life cycle of a product as a product system, which performs one or more functions. The function unit is essential as it, within the goal of the study, inherently defines the product system and its component processes, and therefore provides the basis for tracing the product and elementary flows (e.g., use of resources, emissions to air, water and land). Without considering the function unit, assessment of the environmental impact of a product will be largely limited to its embodied environmental impact, which in itself is less meaningful as, for example, there is little significance in knowing that manufacturing a steel plate consumes more energy than a piece of a wood.

As an LCA is structured around a function unit, the assessment of the environmental impacts using this technique is relative in nature, and any results are typically case- or application-specific. This is important in cases where LCA studies are carried out for environmental comparisons of different materials/systems. Particular care needs to be exercised when any findings are to be generalized.

23.2.3 Process-based versus input–output-based approach for inventory analysis

Process-based analysis is a bottom-up technique, in which one starts from the final product and traces upwards the main product and elementary flows up to extraction of raw materials and, if required, downwards through its use, reuse and recycle until its final disposal. In practice, it is virtually impossible to include all the processes due to a combination of many factors including labour and cost intensities, data availability and lack of knowledge about the system. A system boundary is usually drawn around the inputs where data are available (Menzies *et al.*, 2007), or defined with the intention that, for the particular goal of the study, the environmental impacts associated with processes outside the boundary will have insignificant effects on the results (Lenzen and Dey, 2000; Lenzen, 2001). The setting of system boundary and therefore omission of processes outside of the boundary mean that it is not possible to achieve 100% system completeness. The discrepancy thus incurred is referred to as 'truncation error', which is systematic in nature.

Input–output analysis is originally an economic technique developed by Leontief (Leontief and Ford, 1970) that uses sectoral monetary transactions to account for the complex interdependencies of industries in a country. By using national energy data, monetary flows between sectors can be transformed to energy flows (and further to emission flows by taking account of the prevailing energy structure of the country), and can therefore be used in inventory analysis. Input–output-based analysis starts with the national energy/emissions data, so all upstream processes and activities in the economy are inherently covered and 100% system completeness can be achieved. However, since data on which input–output analysis is based are only available for a limited number of industrial sectors for a country, particular processes of interest may only be contained in one or more aggregated classifications. Therefore, input–output analysis generally suffers from lack of accuracy in determining direct contributions of particular processes. Also, input–output analysis cannot be applied in cases where future environmental impacts (e.g., impacts associated with its maintenance, recycling and disposal) need to be considered.

Studies have shown that the results from process analysis are consistently lower than those from input–output analysis. Nassen *et al*. (2007) evaluated the energy use and carbon emissions for Swedish building productions using input–output analysis. By breaking down and regrouping the results into different activities and materials, the results from the input–output analysis become comparable to those from previous process-based analysis. It is found that the results from the input–output analysis are considerably (almost 90%) higher in terms of specific energy use (GJ/m^2). The differences are only around 20% for the share coupled to production and processing of building materials, and significantly higher for results associated with other sectors such as transport, construction activities, production of machines and service sectors. Some of the differences can be explained by the truncation errors in the process-based analysis. Lenzen and Dey (2000) studied the energy requirements for the manufacture of basic iron and steel products by the Australian steel industry using input–output analysis and, by separating the results into different order requirements, compared their results to a previous process-based study by Michaelis *et al*. (1998). The truncation error was found to be on the order of 50%.

Truncation errors decrease with the increasing order of production stages considered in the process analysis. The magnitude of truncation errors also varies greatly among commodities – low for energy-intensive commodities (e.g., electricity, basic metals, refinery products) and high for less energy-intensive sectors (e.g., finance, services). According to Nassen *et al.* (2007), the number of stages included in process-based LCA studies typically corresponds to level 2 (e.g., for the Swedish economy, level 0 corresponds to direct energy use in a representative industry, level 1 constitutes 134 paths to the other sectors of the economy, level 2 is 134^2 paths) in the input–output system. By comparing the results from input–output analysis and those from process analysis with different truncation orders (e.g., 0th order process analysis only considers energy consumed in the representative industry, 1st order process analysis also includes the first order upstream energy inputs), Lenzen (2001) estimated the truncation errors for 132 Australian industries. The truncation errors for second order (considering direct and first order contributions) process analysis of main construction materials/products are between 24% and 30% for bricks and concrete, and around 65% for plastic and structural metal products.

Truncation errors will lead to consistent underestimation of the embodied impacts in products, buildings and other infrastructure, and should therefore be properly accounted for in analyses to compare different stages of a life cycle. In contrast, truncation errors can be expected to cancel out in comparative analyses of different materials and processes (Nassen *et al.*, 2007; Lenzen, 2001), which, as will be seen in the later sections of the chapter, has been the main area of applications of LCAs for environmental analysis of composites.

Hybrid approaches have been developed to combine the two techniques so their relative advantages can be fully explored, or rather, their shortcomings can be overcome. In one such approach, process analysis is used to calculate environmental impacts of direct and some higher order (usually limited to first order) upstream processes, while the remaining processes are covered by input–output analysis (Bullard *et al.*, 1978). Another such approach uses input–output analysis to extract the most significant energy paths to which process analysis is applied (Treloar, 1996). The remaining paths are covered by input–output analysis or may simply be ignored.

23.3 LCA of fibre reinforced polymer (FRP) composites: case studies

In the previous section, a general introduction is given to the methodology of LCA. This section presents three case studies of the environmental impacts of FRP composites, one from the composites industry and two from the bridge engineering industry. Note that the study by Daniel (2003) was not intended to be an LCA study, but is presented here with reference to the general framework of LCA.

23.3.1 Embodied environmental impacts of composites

The study

A Green Guide to composites (BR 475) has been developed by Anderson *et al.* (2004), through a two-year research project 'A simplified guide to assessing environmental, social and economic performance for the composites industry (COMPASS)', partly funded by the UK's then Department of Trade and Industry (DTI), to allow the composites industry to understand the environmental (and social) impacts of different materials and manufacturing processes.

The life cycle impacts of each material and process choice from the cradle to factory gate are studied using the BRE (Building Research Establishment) Environmental Profiles Methodology. A total of 12 environmental impact categories (and two social impact categories mainly related to health and safety at work), including minerals extraction, fossil fuel depletion and climate change, are considered for three generic product types: (a) a double curvature monolithic panel, (b) a flat sandwich panel with core, and (c) a complex moulded component.

The results of different impact categories are combined using a weight factor determined from a previous BRE research programme to give an overall environmental summary rating, the Green Guide Ratings. The results are presented in a simple comparative ranking of A (good) to E (poor), allowing users to make quick comparisons between a wide range of material and process options to determine which are the most environmentally optimum for their particular products.

Results and discussion

The Green Guide provides quick and easy guidance to composite manufacturers to better understand the environmental impact of their products. A parallel web-based guide allows more specific and detailed comparisons to be made across a broader range of materials and processes and is available at www.netcomposites.com/green-guide/launch-tool.

Several general observations can be made from the guide. A strong message emerging from the work is that, in general, polymers in the matrix are environmentally adverse (as they are based on fossil fuels) and anything that reduces their use, from increasing fibre volume fraction to using fillers (to replace some polymers within the matrix) to techniques that would lead to less waste of the material during mixing, will be environmentally desirable. This includes gelcoat, a resin-based material often used in composite manufacture to provide a good surface finish and also act as a moisture barrier. Epoxy resin has lower environmental impact than polyester, but using calcium carbonate (chalk) as filler halves the impact of polyester and makes it more environmentally advantageous than epoxy resin. Fibres are less environmentally adverse than polymers. Carbon has inherently high environmental impact due to high energy demand during its manufacture, though its superior mechanical properties mean that for the same structural performance, less carbon fibre is needed when compared with other fibres such as glass. Finally, there is usually good potential for reduced environmental impact of composites without compromising their mechanical and other in-service properties, when making the right choice of materials and process methods.

However, attention is drawn to a few areas of the study. First, the guide takes a cradle-to-gate approach by focusing on composites production with the environmental impacts beyond the factory gate not being included. As such, information on the environmental impact over the life cycle of the products is not available and the environmental credentials of composites in comparison with more conventional materials are not known. Second, the results on the embodied environmental impacts of composites are presented in comparative rankings. Quantitative information (in terms of material unit embodied energy/carbon) is not available for use in studies that account for the environmental impacts of composites over their whole service life. Finally, the studies are carried out for three generic product types that are judged by the authors to be representative of the composites industry. Materials that are more specific to a single sector (e.g., carbon or glass fibre composites commonly used in civil engineering) are not considered.

23.3.2 Materials selection for a pedestrian bridge in the Netherlands

The study

Using a method called ecological material analysis, Daniel (2003) compares composite with four other materials (i.e., structural steel, stainless steel, aluminium and concrete) for construction of a new pedestrian bridge to replace an existing steel bridge in the Noordland inner harbour, an area with severe weather conditions and high chloride exposure. Concrete was not included in the original analysis as its heavy weight would jeopardize the stability of the existing bridge support, but was later added for comparison purposes. The bridge consists of two simply supported spans of 13.5 m each with a deck width of 1.6 m. The service life is taken as 50 years for all material options.

The LCA method is considered by the author not to be suitable for complex infrastructure such as a bridge, especially for which data required for the inventory analysis are not readily known at the early stage of design when the material selection is made. Instead, the study uses a two-way evaluation approach:

1. The 'Exergy' method is used to evaluate the energy consumption and environmental effects associated with fossil fuel consumption such as climate change. Exergy represents the potential of the energy 'stored' in materials to deliver work, and typically differs from energy requirement of a similar process by the order of 5–10% (Boustead and Hancock, 1979).
2. Pollution analysis to evaluate water and air pollution as a result of the material winning, processing and fabrication of the final product. The legal threshold of pollutants is used to account for a different spectrum of qualitatively different pollutions resulting from each material option, so specific pollution and total emissions to water or air can be represented by a single value for direct comparison between materials.

Embodied energy in materials and products for both construction and maintenance are included in the analysis, while other activities such as transportation and operation of construction equipment (during and after the construction phase) and the phase of dismantling or demolition are not included. The evaluation of embodied energy is limited to basic materials, with influence of wooden bridge decks in the steel bridges or metal bolts in the aluminium and composite bridges being ignored. Inspection and maintenance requirements (e.g., blast-cleaning and painting for steel) are determined for each material option. In the absence of appropriate data, estimations are made based on engineering judgement and practical experience.

Preliminary designs are conducted to the relevant NEN (Netherlands Standardization Institute) standards to determine the structural form, cross sections and material grade for each material option. The mass of each material used is obtained from geometric dimensions obtained in the preliminary design for each option. Although the analysis is brief with little effort put into optimizing cross sections and joints, the design is considered to be representative of different materials and comparable in terms of strength and durability.

Embodied energy data (in terms of MJ/kg material) and emission data (in terms of kg pollutant/m^3 product) available in the literature are used directly, used after modification or, as in the case of embodied energy data for reinforced concrete and emission data for GRP composites, computed based on data on component materials. Assumptions are made on the relative proportion of the primary and secondary source of materials. The unit embodied energy value used for GRP composite is 33 MJ/kg.

Results

The analysis of consumed energy makes the composite bridge a clear winner with an equivalent energy consumption of 2,778 MJ/m^2 deck area. Each other option results in a more than twice as high energy use. The composite option also comes out the best in the comparison of water and air pollution. The emission of cobalt from polyester resin processing is less significant when compared to the zinc emission in structural steel processing or the mercury and cadmium emissions in the processing of aluminium.

In that particular project, the analysis resulted in advice pointing to a bridge of pultruded GRP sections as the environmentally best option. The customer (Regional Directorate for Public Works and Water Management in the province of Zeeland) followed the advice and the composite bridge has been in service since November 2001.

23.3.3 FRP bridge decks in UK highway bridge deck replacement applications

Background information

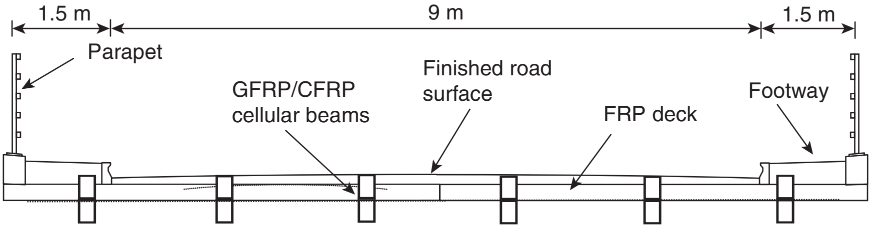
In this study, the environmental credentials of FRP decks are examined by comparing its life cycle environmental performance – in carbon terms – with the conventional concrete option. The study is based on a project to replace a nineteenth-century highway bridge in North London. The new bridge has a single span of 12 m with a total deck width of 12 m, and is designed to carry the full highway traffic on both lanes.

In the preliminary design stage, two options are considered: (a) the FRP option, where the superstructure comprises a pultruded FRP deck adhesively bonded onto six hybrid CFRP/GFRP cellular beams; and (b) prestressed concrete option, where reinforced concrete deck is supported by eleven internal precast prestressed concrete beams and two external reinforced concrete beams. The prestressed concrete option is a common method of construction for a highway bridge of this size. A schematic representation of typical cross sections of the two options is shown in Fig. 23.6.

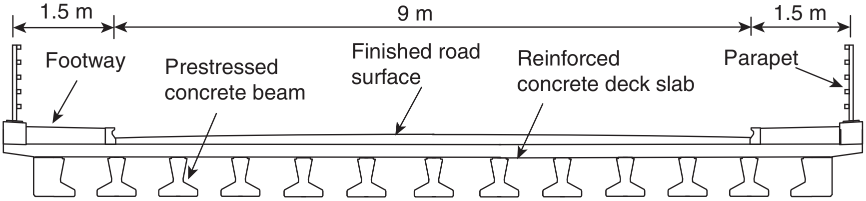
Across the UK highway network, there are a considerable number of bridges of similar size and in need of deck rehabilitation. In order for the findings to be more general and informative, the bridge is taken as a representative bridge carrying a minor road (as defined in DfT, 2010), the part of the highway network where they are most likely to be present.

System boundaries

Figure 23.7 depicts the main stages of the bridge, from conception of the project (to ensure the continuous service of a bridge) to end of design life



(a) FRP option



(b) Prestressed concrete option

23.6 Proposed cross sections: (a) FRP option and (b) prestressed concrete option.

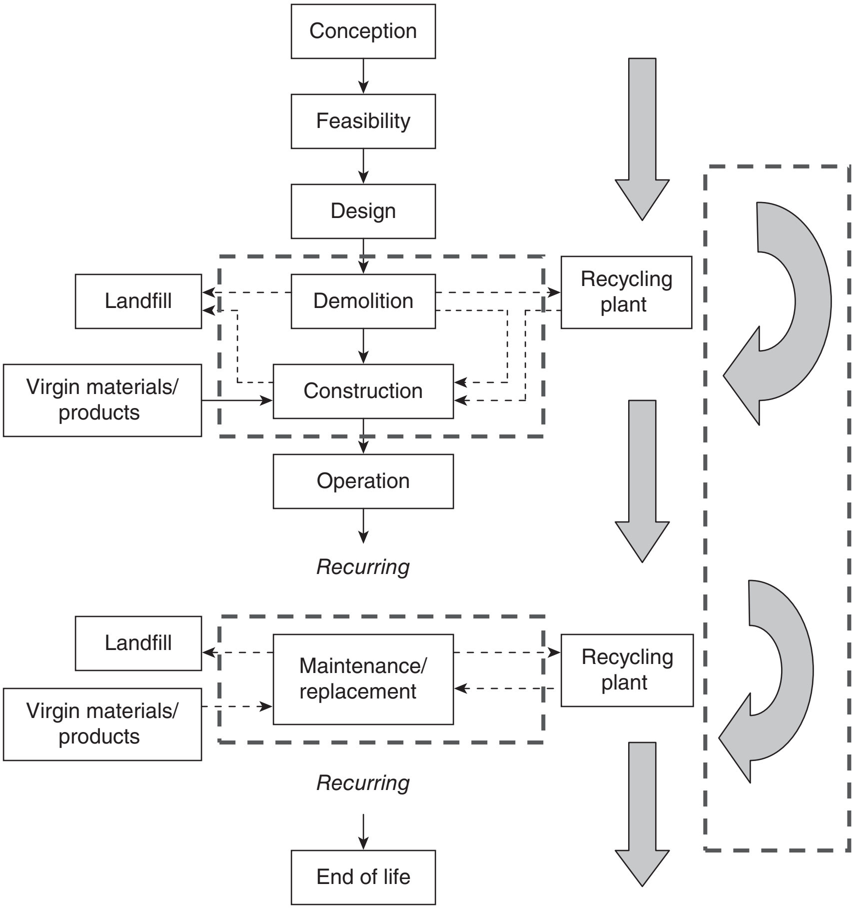
for the new bridge deck structures. The study focuses on areas where decisions/actions by bridge designers/engineers will have a direct influence on the final results. Demolition, although not typically a variable to engineers' designs, is also included for a more complete picture. The system boundary is illustrated by the thick dashed boxes.

Three stages are considered in the study: (a) demolition of the existing bridge superstructure; (b) construction; and (c) maintenance of the new superstructure. Within each stage, three sources of carbon emissions are considered: (a) embodied carbon of any new materials/products; (b) transportation, including transportation of materials/products to site and waste disposal/recycling; and (c) traffic diversions.

Input data

1. Material embodied carbon emissions

Preliminary design work is carried out to determine the initial sections of the main structural elements. For non-structural and auxiliary elements, typical sections are used where specified in the preliminary design, for example, parapets. Wherever it is not specified, the materials and dimensions are taken based on the authors' many years of practical experience.



Note: Arrow – sequence of activities; dashed arrow – direction of material flows; block arrow – indication of traffic flow; curved block arrow – indication of traffic diversions.

23.7 Schematic representation of major activities/components considered in the study.

Therefore, although detailed design may not have been undertaken, the material quantities taken for the study are believed to be able to reasonably represent the final bridge section. Kerbs are reused for the new construction.

Material unit embodied carbon values for different materials are largely based on the Inventory of Carbon and Energy (ICE) Version 1.6a (Hammond and Jones, 2008). Since GRP is a relatively new material to civil engineering, information on its embodied energy/carbon values is very limited. The unit embodied energy value quoted in the ICE database for GRP is 100 MJ/kg, a value taken from a single source published in 1998 (Eaton and Amato, 1998). The last 10 years or so has seen rapid expansion of the use of FRP materials in the civil engineering industry, and the GRP manufacturing process has undergone continuous development. The value of embodied energy for the material can reasonably be expected to be lower than this value. Indeed, a value of 33 MJ/kg (the same as the value used in Daniel, 2003) is believed to be more representative of the current industry practice when the present author contacted a major GRP manufacturer in Europe. In order to be more objective about the competitiveness of the FRP decking system, an average of the two (a principle adopted in preparing the ICE database) is taken for GRP in the study.

Polymer concrete is usually used as the surfacing material for the FRP decking system due to performance considerations (Canning *et al.*, 2007). After consulting with specialist providers, its unit energy and carbon values are calculated based on an epoxy resin to aggregate weight ratio of 1:4 from the values for epoxy resin and aggregate given in the ICE database. The unit energy and carbon values for steel reinforced concrete are obtained from the values for steel bars/rods and general concrete using the method suggested in the database.

2. Material/products and waste transportation

Carbon emissions from transportation of materials/products to site and for disposal of site wastes to landfill or recycling plant were estimated using the carbon calculator prepared by the Environment Agency (2007). Given the good level of local market knowledge that can reasonably be expected of the specialist demolition contractors, it is assumed that a local landfill site or recycling plant is used. A transportation distance of 10 miles is used for disposal of all site waste.

Things can be quite different for sourcing of materials/products, on which cost and market availability will have more significant impact. There are a limited number of manufacturers of FRP decks, and currently, many FRP sections in the UK are transported from Denmark or the US. Road transport for construction materials to site in the UK will typically vary from 5 to 20 miles for ready-mix concrete, to 100 miles for precast concrete. To generate a range of possible transport scenarios, the study considers 10–400 miles transport by road. A transportation distance of 10 miles is initially taken, assuming local sourcing. All materials/products are transported a constant distance except the FRP sections, which are taken as being sourced from Denmark. Road transportation is assumed, again except the FRP sections, for which water transport is necessary.

3. Traffic diversion

Carbon emissions from traffic diversions depend on: (a) the daily traffic volume on the bridge, (b) the diversion distance, that is, the extra distance a vehicle needs to travel using an alternative route, and (c) the period of disruption.

The period of disruption is based on the proposed works programme, which is representative for the specific construction method (14 days for FRP and 85 days for prestressed concrete). A bridge of the size considered in the present study is most likely to be carrying a minor road as defined in Road Statistics (DfT, 2010), either in an urban or rural area. The average daily traffic volume on a minor road in England is used to provide more general information to bridge designers/engineers. The diversion distance in real projects is not necessarily related to the volume of traffic using the bridge but heavily depends on the availability and capacity of any alternative routes in the surrounding areas. In the present study, a diversion distance of 5 miles is assumed. The carriageway is assumed to operate with one lane working during site works.

The unit value of carbon emissions for the vehicles is taken as 350 g/km, a value used in Collings (2006). This value is somewhat higher than typical values for a car. For example, the values used in the EA carbon calculator are in the range of 104–291 g/km for cars. However, cars and taxies account for only around 80% of the road traffic (DfT, 2010); other vehicles such as trucks and lorries have much higher unit values of carbon emissions. Also, typical unit values are given for normal or smooth driving conditions, but where traffic diversion is necessary, congestion can often be expected. Carbon emissions of vehicles in congestion, characterized by slow speed and more frequent start-stop, can be significantly higher, as has been demonstrated in Thomas *et al.* (2009).

4. Maintenance

The main structural elements are designed for 120 years. The design life for non-structural elements including surfacing and waterproofing, and also for bearings is 30 years. For maintenance, it is assumed that good practice is taken so resurfacing, re-waterproofing and bearing replacement take place at one operation. Based on the author's practical experience, a disruption period of 2 weeks is used for both options in estimating carbon emissions from traffic diversions.

Results and discussion

Similar life cycle CO_2 emissions are predicted for the two options, indicating that the FRP decking system can be environmentally competitive with – and even advantageous over – conventional materials for the bridgework considered in the study. The relative merits of the two options are largely affected by the total traffic diversion distance, defined as the product of the average daily traffic volume and the diversion distance.

Carbon emissions at the initial construction stage are comparable to those arising from the maintenance activities. The latter is dominated by embodied carbon in non-structural component replacement and additional emissions from traffic diversions, both of which could be changed during the life of the bridge as technology evolves. In particular for the FRP decking system, the embodied carbon value of the surfacing materials will need to be reduced.

The most controllable emissions are at the construction stage. The supply of materials for the FRP option contributes to 83.6% of construction carbon emissions, whereas the traffic diversion for the prestressed concrete option contributes to 83.5% of construction carbon emissions. The central case therefore highlights that the aim for improvements in the FRP option is to reduce embodied emissions in materials, and for the prestressed concrete option, to focus on schedule and traffic management to reduce traffic associated carbon emissions.

For heavy materials such as precast concrete, road transport over long distances can add more than 10% to the embodied impact of the element. Road transport of heavy materials should be reviewed and alternative transport modes or local suppliers used if possible.

Based on the results, good practices are identified to enable engineers to reduce their carbon footprint. Discussions have also been made on the uncertainties and limitations of the results. More details of the study can be found in two previous publications (Zhang *et al*., 2011a,b).

23.4 Summary and conclusions

Some general observations can be made from the previous case studies and are here discussed with reference to the general LCA methodological framework and industry practices.

23.4.1 General discussions

Process analysis is invariably used for the three case studies. This may be partly because of its conceptual simplicity being more appealing to practitioners in both the composites and engineering industries, and partly because of a general lack of knowledge within the industries about the input–output technique. Bridge engineers largely rely on available material unit embodied energy/carbon data to account for the environmental impacts in manufacturing the materials/products. Preliminary designs are usually carried out to obtain the initial dimensions of the structure for calculating the mass of the materials. To the best of the present author's knowledge, there has so far been no study conducted to trace and record the *actual* environmental impacts (e.g., energy consumption) in a real bridge project, in contrast to buildings, for which the operational energy use is dominated by more easily estimated direct energy use (e.g., for lighting, air-conditioning, heating and cooling).

LCA studies have been carried out for different purposes with their own target users. They are performed to different levels of detail with different system boundaries, and each has their unique way of presentation of results. Estimation is inevitable with regard to, for example, distance of traffic diversion, and the service life and maintenance requirements for a structure. Transportation distance of materials may be directly measured from a map. In addition, there is significant variation in the basic input data, e.g. material embodied energy/carbon data, even for common construction materials such as steel (Menzies *et al.*, 2007), and the transport-related energy/carbon data. In the case of GRP composites, the unit embodied energy data used in the second and third case studies differ by a factor of 2. As has been boldly stated by Trusty (2004), 'quality of LCA results = quality of life-cycle inventory (LCI) data, no matter what tool is used or how results are presented'. However, sensitivity analysis is not commonly conducted to estimate the effects of the choices made regarding methods and data on the outcome of the study, although a parametric study is carried out in the third case study to investigate the effects of traffic diversions.

These issues make the results of the LCA studies rather case-specific and indeed, even questionable. The findings are not directly comparable between LCA studies and can not be transferred to a different case. Great caution needs to be taken in generalizing any findings.

23.4.2 Recycling of FRP composites

Recycling of FRP composites is not considered in the previous case studies. In practice, FRP structural components have been in service for no more than 40 years and have yet to reach the end of their service life. FRP wastes related to the construction industry are mainly those from FRP production and removal of materials on site. However, with the passage of time and the introduction of increasingly stringent legislation on waste management, these would increasingly become an issue facing both the composites and construction industries.

According to Conroy *et al.* (2004), landfill is currently the main route for disposal of FRP composites and, together with incineration, accounts for 98% of FRP waste. This is mainly due to difficulty in recycling

FRP composites, as thermosetting polymers (e.g., vinyl-ester, epoxy and polyester, which are most commonly utilized in construction), once fully cured, are non-deformable under reheating. These properties make FRP composites more durable under many aggressive environmental conditions and therefore the preferred option for these applications. However, these same properties make FRP composites difficult to recycle when they reach the end of their service life.

As alternatives to landfill, FRP composite components may be reused for non-structural or downgrade structural applications. However, there are currently no recognized certification schemes for recycled FRP products. Considering the bespoke nature of some FRP composite constructions, again a property that makes the materials more desirable in many cases, they may be difficult to reuse for different applications in a different environment. The best opportunity for minimizing FRP wastes and re-use of FRP products at the end of their service lives may well lie in the early stages of manufacturing and design.

Recently, the US Army has commissioned two bridges constructed entirely from recycled consumer and industrial plastics at Fort Eustis, VA, the home of the US Army Transport Corps (Hollaway, 2010). This must be seen as an excellent example of reusing waste FRP materials, although the overall environmental impact of such applications is not known.

23.4.3 Closing remarks

LCA studies of FRP composites result in improved understanding of the environmental impacts of this relatively new material to the civil engineering industry. It is particularly effective in identifying environmental 'hotspots' for targeted measures to be taken, as has been shown in the first and third case studies in the previous section. However, due to issues related to the inherent nature of LCA (e.g., function unit-based, approximation), quality of data (e.g., availability, reliability, transparency) and lack of knowledge about the method (e.g., truncation errors in process-based inventory analysis, lack of attention on sensitivity analysis), thorough understanding of the environmental impacts of FRP composites is not available at this time. The situation will improve with time as better understanding of both the method and its applications is gained and more reliable data become available.

Currently, the efforts in applying LCA to FRP composites are rather isolated, in which the best opportunities are usually lost to make these excellent efforts to be most effective. For example, detailed analyses of embodied environmental impacts in composites in the first case study did not yield quantitative information (in terms of material unit embodied energy data) for direct use in LCA studies by, among others, bridge engineers. For an issue as complex as this, coordinated efforts involving key stakeholders at various stages over the life cycle of the FRP composites are desirable. Inputs from material designers, manufacturers and experienced engineers with good knowledge of both bridge design and maintenance would be particularly beneficial.

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