Cement & Concrete Composites 34 (2012) 431–442

Contents lists available at SciVerse ScienceDirect

Cement & Concrete Composites

journal homepage: www.elsevier.com/locate/cemconcomp

Environmental impact and life cycle assessment (LCA) of traditional and 'green' concretes: Literature review and theoretical calculations

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ARTICLE INFO

Article history:
Received 18 August 2011
Received in revised form 3 January 2012
Accepted 3 January 2012
Available online 12 January 2012

Keywords:
Sustainability
Functional unit
Allocation
Life cycle inventory
Impact assessment
Cement
Blast-furnace slag
Fly ash

ABSTRACT

With the current focus on sustainability, it is necessary to evaluate concrete's environmental impact properly, especially when developing new 'green' concrete types. Therefore, we investigated the available literature on every step in the LCA of concrete. The adopted functional unit for which the environmental impact is calculated, influences the outcome significantly. When comparing different concrete compositions, this unit should incorporate differences in strength, durability and service life. Hence, a cradle-to-grave or modified cradle-to-gate approach is advised as system boundary. When using industrial by-products as cement replacing material in 'green' concrete, an economical allocation of impacts is recommended. Inventory data on energy use, CO_2, PM_{10}, SO_x and NO_x emissions were collected and assigned to the impact categories of the problem oriented CML 2002 and the damage oriented Eco-indicator 99 impact method. Compared to Portland cement, the impact of blast-furnace slag and fly ash is about an order of a magnitude lower.

1. Introduction

Concrete is one of the most widely used building materials in roads, buildings, bridges and other infrastructures. On average, approximately 1 ton of concrete is produced each year for every human being in the world [1]. Because of this global extensive use, it is imperative to evaluate the environmental impact of this material correctly.

Nowadays, a material's environmental impact is often equated with its effect on greenhouse gas emissions (GHGs) and climate change. From this point of view, a great variance of so-called 'green' concrete concepts have been developed over the years. Very often these concepts focus on partially replacing the cement, the concrete constituent responsible for the highest CO_2 emissions, by other materials [2]. Worldwide, the cement industry alone was estimated to be responsible for 5–7% of all anthropogenic CO_2 generated [3–5]. Since this branch of industry emits almost no other GHGs, it is held accountable for only about 3% of the total GHG emissions generated by human activities [6].

However, the implementation of these 'green' concepts implies that certain parameters in the mix design need to be changed to obtain a sufficiently workable, strong and durable concrete. Moreover, the specific application and the environment in which the concrete will be used, needs to be considered. A more correct environmental evaluation takes into account all differences between the 'green' and traditional concrete. Therefore, it is necessary to adopt a LCA approach. This methodology compares the environmental impact of a strength and durability/service life related functional unit (FU) of traditional and 'green' concrete over its entire life cycle (production, use and end-of-life phase) for predefined system boundaries. The LCA framework is generally accepted as a reliable assessment tool. It is a multi-stage process that traditionally consists of four steps: the definition of goal and scope, the inventory analysis, the impact analysis and the interpretation.

This review paper addresses the concrete properties that influence the environmental impact significantly. Life cycle inventory (LCI) data on concrete were collected from literature and compared with each other. Finally, it was investigated how these inventory data are assigned to the impact categories of different impact assessment methods and how the environmental score obtained can be affected by important input parameters, such as the FU choice, the system boundaries assumed and the applied allocation rules.

2. Environmental impact of concrete

2.1. Composition

2.1.1. Cement

Environmental impacts of cement manufacturing can be global, regional or local in scale [7].

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doi:10.1016/j.cemconcomp.2012.01.004

2.1.1.1. Global scale. Over the last decades, the emphasis has clearly shifted towards a global focus on climate change. Table 1 presents a summary of values found in literature [4,6,8–16] for the cement related CO_2 emissions. They are usually the sum of the CO_2 emitted during the calcination process (raw material CO_2:RM-CO_2 [16]) and the CO_2 associated with energy use. With respect to the latter, a distinction can be made between indirect and direct energy bound CO_2 (IEB- and DEB-CO_2). IEB emissions comprise the CO_2 emissions associated with the generation of electrical power to operate the cement plant, while the direct energy bound emissions are associated with the fuel combustion in the cement kiln.

Regarding DEB-CO_2, the efficiency of the cement kiln plays an important role. A clinkering energy efficiency of about 3 GJ/ton is not far above the real thermodynamic limit [16]. 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 DEB-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 [6].

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 practise, replacement levels of about 10% are commonly reported. For a limestone replacement of 10%, the total CO_2 reductions can in theory be as high as 25 % [6].

To reduce the emissions any further, alternative clinker chemistries need to be considered. Low energy belite cements seem a valuable alternative. If all the alite in cement could be substituted with belite, RM-CO_2 emissions can be reduced with about 8%. Because of the lower burning temperature, DEB-CO_2 emissions can also be lowered with about 8%. However, due to its slow setting and hardening, the successful implementation of this cement is doubtful. A more promising material is sulphoaluminate cement. Although quite expensive, this cement type has a much lower embodied RM-CO_2 content than Portland cement due to its significantly lower CaO contents (30% lower than a modern OPC clinker). Moreover, the raw materials can be burned at lower temperatures [17].

The greatest potential to reduce CO_2 emissions lies in the replacement of conventional carbon based fuels by alternative low fossil carbon based fuels, e.g. carbon neutral biomass [6], non-carbon neutral scrap tyres [18], etc.

Table 1
Summary of CO_2 (inclusive its distribution over RM-CO_2, IEB-CO_2 and DEB-CO_2), SO_x, NO_x and CKD emissions for Portland cement expressed in g/kg cement* or g/kg clinker** [4,6,8–16,19,20].

Ref.	RM-CO_2	IEB-CO_2	DEB-CO_2	Total
[4]	–	–	–	870 g/kg*
[6]	530 g/kg**	310–600 g/kg**		840–1150 g/kg**
[8]	50%	50%		810 g/kg*
[9]	59%	6%	35%	800 g/kg*
[10]	57%	6%	37%	–
[11]	50%	0–10%	40–50%	–
[12]	–	80 g/kg*	–	–
[13]	–	–	–	820 g/kg*
[14]	–	–	–	690 g/kg*
[15]	–	–	–	810 g/kg*
[16]	425 g/kg*	80 g/kg*	390 g/kg	815 + 80 g/kg*
			Mean value	842 g/kg*
	SO_2			
[9]	0.40–0.60 g/kg*			
[14]	0.82 g/kg*			
[15]	0.58 g/kg*			
[19]	0.27 g/kg**			
[20]	0.54 g/kg**			
Mean value	0.53 g/kg*			
	NO_x			
[9]	2.40 g/kg*			
[11]	10.00 g/kg**			
[14]	1.20 g/kg*			
[15]	1.50 g/kg*			
Mean value	3.65 g/kg*			
	CKD/PM			
[14]	0.49 g/kg*			
[15]	0.04 g/kg*			
[19]	150–200 g/kg**			
Mean value	83.3 g/kg*			

2.1.1.2. Regional scale. Regional environmental impacts include SO_2 and NO_x emissions which contribute to acid rain. Table 1 includes an overview of the estimated SO_2 and NO_x emissions for Portland cement according to literature [9,11,14,15,19,20].

The majority of SO_2 emitted is derived from the fuel combustion and the processing of raw materials in the kilns. Around 70–95% of the fraction not attributable to energy production is absorbed due to the high alkalinity of clinker [21]. Thus, the majority of the SO_2 leaves the kiln with the clinker [22].

The NO_x values refer to both NO_2 and NO emitted to the air. These are mainly an output from fuel usage during clinker production and energy consumption throughout the entire process chain.

2.1.1.3. 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 [22]. Since the diameter is smaller than 10 μm, CKD is classified as PM_{10}. According to the EPA [23,24], 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 1, the amount of CKD generated per kg of clinker produced equals about 15–20% (by mass) [19]. Nowadays, both the environmental and health risks associated with CKD can be reduced significantly by means of mineral carbonation. As observed in the carbonation of other industrial wastes, sequestering carbon may yield additional benefits by stabilizing the waste (reducing the pH) which reduces health risks and the generation of harmful leachate [7]. In addition, the utilisation of CKD for carbon sequestration by means of mineral carbonation appears to have its advantages on the global scale, since about 7% of the carbon emissions can be captured this way [7].

On the local scale, attention should also be paid to the emission of metals and polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/Fs). After emission into the atmosphere, these chemicals can be transmitted to humans through direct (air) and indirect (groundwater, soil, vegetation) pathways. Health risks turned out to be quite low [22]. When located in an urban area, non carcinogenic and cancer risks derived from exposure to metal and PCDD/Fs coming from a cement plant were within the ranges acceptable according to national and international regulations, with the exception of only a few elements (e.g. As and Cr) [25]. In addition, no significant increases in the environmental levels of metals and PCDD/Fs were detected when comparing a conventional fossil kiln fuel with an alternative fuel (15% on average partial substitution of fossil fuel by refuse-derived fuel from municipal solid waste). Also, no changes in airborne particulate matter were noted, while significant reductions were found for a number of pollutants (PCDD/Fs, Co, Cr, Mn and Ni) in vegetation, as well as in soil (Ni) and air (Sn) [26].

Quite some attention is also being 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 [27]. Hexavalent chromium or Cr(VI) is not stable. When dissolved, Cr(VI) can penetrate the unprotected skin and be transformed into to Cr(III) which combines with epidermal proteins to form the allergen that causes sensitivity to 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 [28].

2.1.1.4. Variability and quality of the data. An important issue regarding all cement related emission data as found in literature is the aspect of variability and quality of the reported data. The varying CO_2, NO_x, SO_x and CKD emissions as shown in Table 1, are a clear proof of this. Obviously, assuming constant emission values will not lead to an accurate environmental evaluation. Until now, only few sources have reported standard deviations on their average values, e.g. Chen et al. [14] and ATILH [15]. Von Bahr et al. investigated the data variability and quality problem by analysing the monthly dust, NO_x and SO_2 emissions of six Nordic cement plants for a period of 7 years (1993–1997). Significant differences were observed between cement plants and with time. These differences may depend on raw materials, production processes (including stability in the process) and varying performance of/investments in air pollution control systems. Von Bahr et al. aimed for the highest possible data quality through a close cooperation with the cement plants and through internal and external review. The resulting comparable data quality indicates that the variability of emissions remains [29]. This conclusion should also hold true for other constituents used in concrete.

2.1.2. Cementitious materials

Partial replacement of Portland cement clinker with FA or BFS is considered as beneficial and has become common practise. From an environmental perspective, it is obvious that a high content of cementitious by-products in blended cements is desirable. Since the production process will involve less greenhouse gas emissions compared to an ordinary Portland cement (OPC) consisting for 95–100% out of clinker. FA or BFS, can also be added separately to the concrete mix. The latter approach is more commonly accepted in the United States [6].

According to Feuerborn, the application of FA (total utilisation:20.0 million tonnes) as concrete addition, cement raw material and constituent for blended cements in Europe in 2007 amounted to 29.5%, 26.9% and 14.5%, respectively. Due to a recent retrofitting of existing coal fired power plants with flue gas desulphurisation, the construction of new coal fired power plants in some countries and the increased use of imported coal with higher ash content, FA production and use is even expected to increase in the next years [30]. However, as many cement kilns are operating at or above their effective capacity, the use of blended cement could only allow for an increase in concrete production without any real increase in clinker or cement production [31]. Due to the increasing cement demand, the overall CO_2 emissions attributable to the cement industry are even expected to increase dramatically in the coming decades [4].

From the legislative viewpoint, it is worth mentioning that the *k*-value concept of NBN EN 206-1 [32] strongly limits the maximum FA content (FA/cement $\leqslant$0.33). When calculating the total binder (cement + FA) content for a concrete composition, the FA amount needs to multiplied with a *k* factor of 0.4. As a consequence, the total binder content is lower than the sum of the cement and FA present and the water-to-binder (W/B) ratio increases. According to NBN 206-1 [32], the exposure class of a concrete composition is determined by a minimum binder content and a maximum W/B ratio. Logically, the *k*-value concept limits the use of concrete mixes with a higher FA content to the less demanding exposure classes mentioned in the standard. Moreover, the minimum cement content required for the relevant exposure class may only be reduced by a maximum amount of $k \times$ (minimum cement content – 200) kg/m^3 and additionally the amount of (cement + FA) shall not be less than the minimum cement content required in accordance with NBN EN 206-1 [32]. According to NBN B15-001 [33], it is the amount of $k \times$ (cement + FA) that shall not be less than the minimum cement content required. Thus, the Belgian design approach is even more strict. When following the *k*-value concept, the use of high-volume fly ash (HVFA) concrete – with a FA content of at least 50% [34] – is not allowed in more demanding environments. However, this concrete type can be used in Belgium, but only if its equivalent performance compared to the proper reference concrete is proven according to NBN B15-100 [35]. A *k*-value concept for BFS also exists, but it is only specified in NBN B15-001 [33]. There, the maximum BFS/cement ratio amounts to 0.20. The *k*-factor equals 0.9. Note that the *k*-values used for both FA and BFS in the different EU member states can vary considerably [36].

Table 2 gives an overview of the CO_2, particulate matter, SO_x and NO_x emissions associated with steel production and coal fired electrical power generation [37,38]. Depending on the allocation principle adopted (Section 3.2.2), part of these emissions should be attributed to the production of the forthcoming by-products BFS and FA.

The presence of toxic metals in slags and ashes is not seen as a problem. Mehta suggests that the concrete industry offers ideal conditions for the use of these by-products, since the metals can be immobilized and safely incorporated in the hydration products of cement [39]. Leaching tests conforming to NEN 7345 [40] showed that paving concretes made with CEM III/A 42.5 LA, only leach heavy metals at very low concentrations, significantly lower than the parametric values given in European Directive 98/83/EC [41], which defines the quality of water intended for human

Table 2
CO_2 emissions, total energy use, particulate matter, SO_x and NO_x emissions associated with steel production (in kg/kg) and coal fired power generation (in kg/kW h) [37,38], and the amounts attributable to 1 kg of the corresponding by-products BFS and FA using the mass and economic allocation precentages of Chen et al. [90] cfr. Table 4.

[37]	Emissions (kg/kg steel)	19.4% mass allocation 0.24 kg BFS/kg steel (kg/kg BFS)	2.3% economic allocation 0.24 kg BFS/kg steel (kg/kg BFS)
CO_2	1.356	1.0961	0.1300
Total energy	16.575*	13.3981**	1.5884**
Particulate matter (unspecified)	0.003	0.0024	0.0003
SO_x as SO_2	0.012	0.0097	0.0012
NO_x as NO_2	0.005	0.0040	0.0005
[38]	**Emissions (kg/kW h Electricity)**	**12.4% mass allocation 0.052 kg FA/kW h electricity (kg/kg FA)**	**1.0% economic allocation 0.052 kg FA/kW h electricity (kg/kg FA)**
CO_2	1.022	2.4371	0.1965
Total energy	12.552***	29.9317****	2.4138****
Particulate matter (unspecified)	0.010	0.0238	0.0019
SO_x as SO_2	0.007	0.0167	0.0013
NO_x as NO_2	0.003	0.0072	0.0006

* MJ/kg steel.
** MJ/kg BFS.
*** MJ/kW h electricity.
**** MJ/kg FA.

consumption. Thus, the partial replacement of clinker with BFS, within the limits defined in NBN EN 197-1 [42] for CEM III/A type cements (36–65% of slag), has no effect on the leaching behaviour of the concrete [43].

Another problem might be the potential Radon exhalation. Most building materials contain naturally occurring radioactive elements. Building inhabitants may be externally exposed to gamma rays originating from these radioisotopes. As the presence of Radon is responsible for the largest fraction of the natural radiation dose to the population, the tracking of this Radon concentration is of great importance. The Radon in FA originates from the coal burned in the electrical power plants. Kovler et al. [44] found that despite the higher ^{226}Ra content of FA (more than three times, compared to OPC), Radon emanation from cement-FA pastes is significantly lower (7.65% for cement versus 0.52% only for FA). However, note that few information is currently available on the emanation behaviour of FA in concrete during the use phase of the structure and after demolition.

2.1.3. Water

A low water-to-cement (W/C) ratio is advised especially in more demanding applications. To make sure that the water does not contain organic substances, chlorides or alkalies, drinkable water is usually applied in practise. Apart from in agriculture, the demands for freshwater in industry are causing groundwater resources to be depleted and surface waters to be abstracted in ways which compromise the freshwater ecosystem health [45]. This impact can be assessed by calculating the water footprint of a product. This is typically the sum of all water consumed in the various stages of production and therefore the same as its virtual water content [46]. Lafarge uses approximately 343 L of water to manufacture 1 ton of cement. In addition, about 284 L of water are needed on average to produce 1 m^3 of concrete [47]. Evidently, within concrete manufacturing a reduced use of ground water and surface water, the so-called blue water according to Ridoutt and Pfister [48], would be beneficial. Thus, concrete preferably has a low water content. Nevertheless, the increased use of superplasticizers to achieve this low water content without losing workability cannot be neglected.

2.1.4. Admixtures

The European Federation of Concrete Admixture Associations (EFCA), has published an eco-profile on superplasticizers [49]. It is valid for the four main groups of superplasticizers: sulphonated naphthalene formaldehyde, sulphonated melamine formaldehyde, vinyl copolymers and polycarboxylic ethers. All of them are dissolved in water and typically contain 30–45% active matter.

It is worth noticing that the amount of CO_2 (720 g/kg) emitted for the production of 1 kg of superplasticizer is only a little bit lower than the CO_2 emissions associated with the production of cement (842 g/kg). The same is true for the NO_x emissions (1.8 g/kg versus 3.65 g/kg). On the other hand, the amount of SO_x emitted to manufacture the superplasticizer is significantly higher (3.6 g/kg versus 0.53 g/kg). However, as the amount of superplasticizer used in concrete is almost negligible when compared with its cement content, these emissions should not contribute significantly to the overall environmental impact.

2.1.5. Fine and coarse aggregates

A distinction must be made between naturally rounded and crushed aggregates. Rounded aggregates are the result of weathering and erosion and do not require any processing once collected from the sea or river bed. Crushed aggregates are exploited from quarries and require mechanical crushing.

Over the years, a lot of research has been done on the recycling of construction and demolition waste [50]. However, the use of recycled aggregates in structural concrete is still lacking confidence [51].

To date, few reliable data exist on the environmental impact of sand and aggregates consumption. Habert et al. [52] stated that the indicators currently available to assess resource consumption in Life Cycle Impact Assessment (LCIA) are not fully adapted to the concrete industry. Since concrete and its constituents are not transported over long distances, the regional scale is a more relevant scale upon which resource extraction policies should be based. Since the price per ton aggregates transported doubles every 30 km, the reason for this is clear. The assessment approach of Habert et al. [52] confirmed that depending on the size of the studied territory the depletion of bulk resources is different. At a world scale, depletion of bulk resources is negligible [53], at a country scale, depletion is low and at a regional scale, depletion is clear.

Besides the regional availability and accessibility, the aggregate type is of importance. According to Steen [54] the extra cost for crushing rock to a size as small gravel is about 2 euro/ton. Emissions and energy consumption per metric ton of glacier rock produced by a Nordberg HP400 SX rock crusher [55] are shown in Table 3. Flower and Sanjayan found that for granite/hornfels and basalt aggregates, GHG emissions amount to 45.9 kg CO_2 equivalents/tonne and 35.7 kg CO_2 equivalents/tonne, respectively. These figures include the average contribution of transport from the quarry to the concrete batching plants. The amount of CO_2 released during the production and subsequent transport of concrete–sand was found to be 13.9 kg CO_2 equivalents/tonne. The production process included the strip-mining of raw sand by excavators, the subsequent loading into haulers, the washing into a pumpable slurry, the piping to the grading plant and the filtering into standard grades. The lack of a crushing step explains the difference between the fine and coarse aggregate related emissions [13].

2.2. Workability

The required workability of the fresh concrete highly depends on its specific application field. In road construction, where fast setting after casting is desirable, concrete with rather low consistency is needed, e.g. roller compacted concrete [56]. In case of complex formworks or a high density of steel reinforcements, a much higher consistency is necessary. The same goes when the implementation of the concrete on site involves pumping. When adequate mechanical compaction is not possible, it is advised to use self-compacting concrete (SCC). De Schutter et al. [57] estimated that a pipe factory can save annually about 1 GW h of energy when the shift is made from traditional concrete to SCC. This substantial reduction in energy use is about 60% of the actual energy consumption for the production of concrete pipes.

2.3. Strength and mechanical loading

The dimensioning of reinforced concrete structures subjected to a given mechanical load is traditionally based upon the concrete's 28 day characteristic compressive strength and the characteristic tensile strength of the steel reinforcements [58]. When using a

Table 3
Emissions and energy consumption to produce 1 metric ton of crushed glacier rock with the Nordberg HP400 SX rock crusher [55].

Emissions	CO_2, fossil	0.6465 kg/ton
	NO_x as NO_2	0.0021 kg/ton
	SO_x as SO_2	0.0036 kg/ton
	Particulates (unspecified)	0.0038 kg/ton
Energy	Total primal energy	9.8192 MJ/ton

concrete with a high mechanical strength, this could decrease the amount of concrete needed to build a given structural element [59]. Of course, the dimensions highly depend on the type of structural element. Habert and Roussel evaluated the impact of a higher strength for a horizontal element carrying only itself, a horizontal element carrying an external load and a vertical element carrying an external load. In all three cases, the total CO_2 production was found to be less when a stronger concrete was used. The vertical element turned out to be the most environment friendly structural element.

Van den Heede et al. obtained similar findings when evaluating the environmental impact of a column supported beam made of HVFA concrete: structure dimensions of the columns could be reduced more than the cross-section of the beam [60].

According to Habert and Roussel, a doubling of the strength of common use concretes, would result in 30% less CO_2 emissions. When a certain high strength concrete is also characterised by a high cement replacement level, an additional CO_2 reduction of about 15% could be obtained [59]. Van den Heede et al. took the benefits of both a higher strength and a higher cement substitution into account for the column supported beam made of HVFA concrete located in a dry carbonation exposed environment. Compared to the proper reference concrete, the amount of CO_2 equivalents had reduced with 25.8% [60].

Increasing the compressive strength is seen as one of the key options to increase the efficiency of cement use [6]. Therefore, Damineli et al. proposed the binder intensity index bi_{cs} which measures the total amount of binder per m^3 of concrete necessary to deliver 1 MPa of strength. This approach is very useful within the LCA framework, as it defines a unit of functional performance (cf. the functional unit FU, see also Section 3.2.) instead of a unit of concrete volume or weight [61].

Note that a general shift towards high-strength concrete for every application will require a revision of NBN EN 206-1 [32] where an indicative minimum strength class is given per concrete exposure class.

Revisions may also be necessary with respect to the age at which this minimum strength is specified. For instance, the relatively slow strength development of FA concrete is a disadvantage in applications where high early strength is required. However, in many situations, especially those involving mass concrete structures such as dams and heavy foundations, which are not loaded to their design values until months if not years after their placement, it is quite common to specify 90-day strengths instead of the conventional 28-day strength [2]. In contrast with NBN EN 206-1 [32], the Canadian standard CSA A23.1-09/A23.2-09 [62] already specifies a 56-day strength for some exposure classes.

2.4. Durability, environment and service life prediction

2.4.1. Environment specific durability based design approaches

2.4.1.1. Prescription based design, European school. When looking at the NBN EN 206-1 [32], six main exposure classes can be identified (X0: No risk of corrosion or attack, XC: Corrosion induced by carbonation, XD: Corrosion induced by chlorides other than from seawater, XS: Corrosion induced by chlorides from seawater, XF: Freeze–thaw attack with or without de-icing agents, XA: Chemical attack).

2.4.1.2. Prescription based design, American school. Only four main exposure categories are defined in the American standard ACI 318-08 [63] (F: Freezing and thawing, S: Sulphate, P: Requiring low permeability, C: Corrosion protection of reinforcement). In contrast with the European approach, the risk of carbonation-induced corrosion is not specifically covered in a separate exposure class.

2.4.1.3. Prescription based design, considerations. Since concrete structures are easily catalogued in a different way depending on the applicable standard, it is of importance to look at the minimum concrete requirements per exposure class for the different standards. In Europe, limiting values are imposed on the minimum cement content, the W/C ratio and the compressive strength class. In North America, there are no requirements regarding the minimum cement content. The mix design of a concrete is only governed by a maximum W/B ratio and a minimum compressive strength.

Another important difference between the European and the North-American approach involves the performance attributed to mineral by-products. Within the European framework this performance is not considered as equivalent to Portland cement. The *k*-factor concept strictly limits the amount of exposure classes in which concretes with a significant by-product content can be used. In the North-American standards, much less limitations are imposed on the FA and slag content.

Evidently, fixing the minimum cement content has its consequences when choosing an appropriate reference for evaluation of new concrete types. For less demanding applications this cement content may already be quite low. Applying a high cement replacement level with mineral by-products can result in a poor performance, especially at early age. To avoid this, an increase in total binder content (cement + by-product) may be necessary. As a consequence, the environmental benefit will not simply equal the percentage of by-products used [60]. When only the maximum W/B ratio and the minimum strength are being limited, the choice of reference will be somewhat different. For a given maximum W/C ratio, the minimum strength required highly influences the cement content and thus the composition of the reference.

2.4.1.4. Performance based design. For a given deterioration process, this design approach describes the involved mechanisms using mathematical models and numerically predicts the induced degradation extent (as described in Fib Bulletin 34 [64], DuraCrete [65], etc.). This degradation should remain acceptable within the predefined service life of the structure (see Section 2.4.2). Although seemingly more universal than the prescriptive design approach, some problems remain, e.g. the definition of the critical chloride concentration threshold level when evaluating concrete's resistance to chloride-induced corrosion. A wide range of values can be found in literature [66], but a reliable value is still not available.

2.4.1.5. Recommendations. Li et al. [67] point out that both design methods are complementary and not opposite to one another. Therefore, a combined methodology may be advised when assessing the environmental impact of concrete in which the choice of reference is prescription based and the calculation of service life is model-based.

2.4.2. Service life prediction

For service life prediction, the DuraCrete [65] methodology requires the definition of the desired structure performance, usually by setting a required target service life and by specifying the event which corresponds with the end of this service life. With respect to corrosion, four different events can be seen as the end of service life: depassivation of the reinforcement, cracking, spalling or collapse of the concrete structure.

The first event limits service life to the initiation period of the corrosion process. When corrosion is chloride induced, the initiation phase ends when the chloride concentration at the reinforcement reaches a critical threshold value. For carbonation-induced corrosion, the initiation period comes to an end when the carbonation front reaches the reinforcement. In both cases, the depassivation event does not represent structural failure. However, since from then on the actual corrosion process begins, the end of the initiation period is often equated with the end of service life. The propagation period for chloride-initiated corrosion is usually very short compared to the initiation period. As a consequence, it is sufficiently accurate to assume that the sum of the initiation period and the propagation period is simply equal to the initiation period [65].

The other three events are all situated in the propagation period of the corrosion process. Val and Stewart see spalling as the most influencial failure mode for the estimation of life cycle cost, because corrective actions such as repair or replacement usually are made to the structure almost immediately after spalling [68]. When the aim of the study is a quantification of the environmental impact instead of the monetary cost, the same reasoning should be valid. Obviously, it is the additional concrete manufacturing for repair or replacement after concrete spalling that will result in a substantial extra environmental load.

Once the failure event is set, the applicable environmental actions and degradation mechanisms need to be identified. DuraCrete [65] defines a limit state function for chloride and carbonation-induced corrosion.

A similar but updated design approach can be found in Fib Bulletin 34 [64]. The required safety levels associated for the different limit states are usually expressed in terms of a reliability index β. Depending on the type of limit state – Service Limit State (SLS) or Ultimate Limit State (ULS) – and the consequences of failure, values for β are specified in Eurocode 0. For instance, depassivation will be classified as a SLS as there is no immediate consequence on structural safety. β-values in the range of 1.0–1.5 may be appropriate for depassivation.

It should be noted that the currently available models to predict carbonation- and chloride-induced corrosion assume that the concrete is uncracked during the initiation period. In practise, this is often not the case. It is known that the presence of cracks accelerates the penetration of CO_2 and chlorides inside concrete [69,70] and therefore, shortens the initiation period. This observation suggests that an update of the existing models is needed to include the cracking aspect. Also, both the cracking resistance and the healing capacity of potentially ‘green’ concrete mix designs should be verified and improved if necessary.

Besides the design methods for carbonation- and chloride-induced corrosion, Fib Bulletin 34 [64] also provides a probabilistic design approach for frost induced internal damage and for salt-frost induced surface scaling. However, the models are less straightforward as the ones used for chloride- and carbonation-induced corrosion. For environments where chemical attack is at risk, the necessary models for adequate service life prediction are not yet available.

With a notion of the concrete’s service life based on durability tests representative for its environment, it would be possible to expand the definition of the binder intensity proposed by Damineli et al. [61] (see Section 2.3.) as follows: the total amount of binder per m^3 concrete necessary to deliver one 1 MPa of strength and 1 year of service life. This way, the unit of functional performance is obtained on two levels – strength and durability/service life – which is ideal for LCA of concrete.

3. Life cycle assessment of concrete

LCA is defined as “the compilation and evaluation of the inputs, outputs and potential environmental impacts of a product system throughout its life cycle [71]”. In other words, LCA is a tool for the analysis of the environmental burden of products at all stages in their life cycle. According to this definition, the impact of a product is studied from “the cradle to the grave”.

Within the construction industry, LCA studies can be performed at many different levels. First of all, there is the cradle-to-gate approach, that only considers the impact of raw material extraction, the production of materials and product parts until the end product leaves the gate of the factory. However, a simple cradle-to-gate analysis cannot be used when evaluating the environmental benefit of potential ‘green’ concrete types. On this level, only the influencing parameters workability (Section 2.2) and strength (Section 2.3) can be considered, durability (Section 2.4) not. The demolition and waste phase are also not taken into account. Therefore, it may neglect the impacts of heavy metal leaching from industrial by-products contributing to human toxicity and ecotoxicity. The same is true for the processing steps taken to make the material recyclable or even reusable.

A cradle-to-grave approach on the other hand, can take the latter aspect into account, since the LCA looks at the material’s impact over its entire life cycle. However, in contrast with the well known concrete production process, representative data related to the use phase and end-of-life phase for a specific concrete structure, are not always known. To include all influencing parameters – workability, strength and durability – into the LCA without knowing all the details, a modified cradle-to-gate approach can be adopted. This methodology intends to quantify the environmental impact of the total amount of concrete that the manufacturer will need to produce to construct and maintain a specific structure during its predefined service life [72].

As the aim for sustainability should stimulate profound recycling and reuse, the cradle-to-cradle concept [73] is gaining importance today. Products should be designed in such a way that at the end of the life cycle the materials can be recycled as raw material for production. Completely recyclable concrete is an example of this concept. Its mix design is adjusted in such a way that after demolition the concrete debris can simply be reused as raw materials for cement production [74].

Either way, ISO 14040-14044 specify a four step LCA methodology for each approach: definition of goal and scope, inventory analysis, impact analysis and interpretation [71,75].

3.1. Definition of goal and scope

Obviously, the goal of LCA here, is to accurately compare the overall environmental impact of traditional and ‘green’ concrete.

Regarding the scope of the LCA, system boundaries must be described clearly using a flow diagram or process tree. Of great importance is the definition of the functional unit (FU). This unit is seen as the reference unit of the product system for which the environmental impact will be calculated [75,76]. Literature review shows that the scale of this FU for LCA of concrete can vary significantly from the material level (i) onto the structure level (ii).

(i) When comparing the environmental impact of different concrete mix designs, a small scale FU on the material level can be appropriate. An efficiency indicator similar to the one proposed by Damineli et al. [61] (see Section 2.3.) is seen as a first example, although it should not only be expressed in terms of a compressive strength unit, but also relate to a unit of service life (see Section 2.4.2.). Another possible functional unit choice is the amount of concrete needed in a simple structural element (column, beam, slab,...) with a given mechanical load and a predefined service life in a given environment. This way, additional concrete manufacturing due to replacement or repair over time, is included in the LCA [72]. The same goes for differences in strength. The use of a high strength concrete implicates that structure dimensions and thus the overall concrete amount needed can be reduced considerably. The resulting environmental benefit will be visible in the LCA output [60].

(ii) When LCA is used to evaluate the environmental impact of a specific structure, the FU usually corresponds with the structure itself. For a LCA study on pavements, Sayagh et al. used a 1 km pavement with a lane width of 3.5 m as FU. The service life was set at

30 years with a traffic load of 9.4 million trucks per lane within this period [77]. This definition is in correspondence with the FU choice recommended in (i) as both durability and the mechanical load aspects are included in the study. Park et al. calculated environmental impacts for 1 km of a four lane highway with a predefined service life of 20 years and repair once every 7 years [78]. However, construction activities and maintenance are not always included. For instance, Chowdhury et al. chose a road section with a thickness, width and length of 600 mm, 2.5 m and 1000 m, respectively, as FU [79]. The focus of the study was simply on the material level with no specification of a service life.

Besides road structures, entire buildings can be the object of an environmental evaluation. Xing et al. [80] studied the difference in impact between steel and concrete-construction office buildings with a use life of 50 years. Both the construction and use phase were taken into account. One m^2 of building area was adopted as FU. For the assessment of wood and steel reinforced housing construction, Gerilla et al. [81] expressed the FU in kilogram emission per year and per m^2. It was assumed that the detached houses had a 150 m^2 floor space and a design life of 35 years. The whole life cycle of the houses was considered. Specifying a 1 m^2 area is not always correct [82]. When comparing the construction related impact of in situ cast floors with precast floors, the smallest precast hollow core slabs on the market performed much better than a normally dimensioned in situ cast floor regarding structural strength. As a consequence, the spans achievable with precast concrete slabs are higher and this reduces the number of columns and spread footings. Therefore, the FU comprised the whole building, although the main goal of the study was only an evaluation of the floor type used.

In conclusion, regardless the scale of the FU for a LCA study comparing the environmental impact of traditional and 'green' concrete, the unit should be able to deal with strength and durability/service life differences between the two concrete types.

Note that even for a strength and durability related FU, it often remains difficult to decide what to and what not to include in the system. For instance, transport of materials can be incorporated in different ways. First of all, one can decide to attribute the impact of transport between systems to the system under investigation or not. In case of inclusion, the partial transport impact associated with the transported material or good has to be specified [83]. Since railway carriages, ships and aircraft normally return with other goods after delivery, the impacts of the return travel do not have to be considered. Trucks transport other goods only on a part of the return travel to limit empty tours. This observation suggests to include the impact of half of the return travel in the LCA. Another issue is the status of capital goods. Certainly, production processes require the appropriate infrastructure of which the construction has a certain environmental impact. Peuportier [83] states that the effects of making the infrastructure available for use is in general negligible. ISO 14044 [75] provides several cut-off criteria to decide which inputs are to be included in the assessment, such as mass, energy and environmental significance.

Finally, the necessary criteria regarding the quality of the data used in the LCA need to be set. The following data requirements should be addressed: time-related coverage, geographical coverage, technology coverage, precision, completeness, representativeness, consistency, reproducibility and (un)certainty of the information [75].

A distinction can be made between primary and secondary data [84]. Primary data are required for the key constituents of a product (e.g. cement), meaning that they should be as accurate and representative as possible. Therefore, they should be obtained from the manufacturer directly. Note that the IPCC provides three methods to calculate the cement related CO_2 emissions depending on the information available: (i) the estimated clinker production data through use of cement production data, (ii) the available clinker production data or (iii) the original carbonate input data of the cement manufacturer [85]. Obviously, (iii) gives the most accurate results and is recommended when primary data are required.

On the other hand, inventory data regarding transport can be treated as secondary data. It means that the more general data available in LCA databases, such as Ecoinvent [86] can be used for this.

3.2. Inventory analysis

3.2.1. Data collection

The necessary information can be obtained directly from the industries involved using detailed questionnaires or from publicly available annual environmental reports (ERs) and environmental product declarations (EPDs). Obviously, data from questionnaires will result in a more reliable LCI because ERs and EPDs will always hold a certain risk of misinterpretation and double counting. However, first hand data are not always provided by the companies because of confidentiality issues. As a consequence, the larger part of the LCIs are based on data from ERs, EPDs and LCA related journals. Therefore, it is understandable that ISO 14044 [75] requires detailed documentation referencing for all public sources used. A sensitivity analysis is also very useful. For instance, Josa et al. [9] made an extensive comparative analysis of the available life cycle inventories of cement in the EU to obtain a general trend in CO_2, NO_x, SO_x and dust emissions. LCA databases (e.g. Ecoinvent [86]) are seen as another important data source. As data availability and quality are identified as critical problems affecting all four LCA phases, there is still an existing need for more peer-reviewed, standardised LCA inventory databases [87].

3.2.2. Allocation

While collecting the inventory data, attention needs to be paid to allocation. Problems occur whenever a system produces more than one product. Somehow, the environmental impacts have to be divided over the different end products. Although this allocation of impacts is preferably avoided, it is often impossible. When this is the case, the inputs and outputs of the system should be partitioned between its different products or functions in a way that reflects the underlying relationships between them, e.g. allocation by mass or by economic value [75].

Allocation is of particular importance when FA or BFS are in play. Whenever these materials are used as a cement replacing material, attention needs to be paid to their allocated environmental impact. In Van den Heede and De Belie [72] no environmental impact was attributed to the FA incorporated in HVFA concrete. The FA was considered as an avoided waste from the electrical power plant. The environmental load is at expense of its producer, in this case the electricity company. However, FA and BFS are no longer considered as merely waste, but as useful by-products. Both of them meet the necessary requirements imposed by the recent European Union directive 2008/98/EC [88] to qualify for the by-product status: (i) further use of the substance is certain, (ii) the substance or object is produced as an integral part of a production process, (iii) the substance or object can be used directly without any further processing other than normal industrial practice, and (iv) further use is lawful. Moreover, sometimes the use of these by-products is even highly recommended to obtain a sufficient performance. For instance, the use of slag in concrete has been proven beneficial to increase the concrete's resistance to acid attack [89]. As a consequence, the question rises whether it would not be more appropriate to allocate a part of the environmental load to the concrete producer. Sayagh et al. [77] studied two extreme allocation procedures for BFS. Firstly, the BFS is considered as a steel waste with no steel plant environmental loads allocated to it. Secondly, the BFS is seen as a steel plant by-product with 20%

of the steel plant environmental flows allocated to the BFS. This percentage is in correspondence with the BFS/steel mass ratio. The environmental indicator results were found to be highly sensitive to the adopted allocation hypothesis. A 20% allocation by mass resulted in a contribution increase to the greenhouse effect of roughly 60 %. Acidification potentials and eutrophication indexes were found to be around 25% higher. Ecotoxicity potentials on the other hand almost doubled.

Chen et al. [90] evaluated the influence of three allocation procedures on the environmental impacts of BFS and FA: no allocation, allocation by mass and allocation by economic value. The latter two approaches resulted in the calculation of mass and economic allocation coefficient (Table 4). These coefficients were used to calculate the amount of energy use, CO_2, SO_x, NO_x and particulate matter emissions attributable to the by-products BFS and FA (Table 2).

Each of these two allocation principles has its advantages and disadvantages. Mass allocation imposes enormous environmental impacts to the industrial by-products which may discourage the concrete industry to continue applying them as cement replacement [90]. When comparing the mass allocated emission values for BFS and FA (Table 2) with the corresponding cement related emissions (Table 1), it is clear that considerably more CO_2 (BFS: 1.1 kg/kg; FA: 2.4 kg/kg; cement: ±0.8 kg/kg) and SO_x (BFS: 9.7 g/kg; FA: 16.7 g/kg; cement: ±0.5 g/kg) are emitted for 1 kg of BFS and FA. Similar conclusions can be drawn with respect to the total energy use (BFS: 13.4 MJ/kg; FA: 29.9 MJ/kg; cement: ±2.9 MJ/kg [16]). Especially, the environmental impacts attributed to FA are high. This is mainly caused by the fact that very little FA (0.052 kg) is produced per kW h of electricity. As a consequence, the mass allocation coefficient of 12.4% needs to be applied to the impacts of about 19.2 kW h of electricity to obtain the impact of 1 kg of FA. Since a lot more BFS is produced per kg of steel (0.24 kg), the mass allocated impacts of BFS are much lower when compared with FA.

When adopting the economic allocation principle, the impacts imposed onto 1 kg of BFS and FA are much lower. However, there is the disadvantage of price instability which can make the LCA outcome subject to significant fluctuations. When attributing an environmental load to FA or BFS through mass allocation, the corresponding value will remain constant over a long period of time [90].

3.3. Impact analysis

The main aim of the impact analysis is to connect each LCI result to the corresponding environmental impacts. Usually, this approach results into a classification of impact categories, each with a category indicator. Two main schools of methods can be distinguished [99].

(i) The first school comprises classical impact assessment. Its category indicator is located right in between the LCI results and the category end points (where the environmental effect or damage occurs). The corresponding methods restrict quantitative modelling to relatively early stages in the cause-effect chain to limit uncertainties and group LCI results related to a certain environmental problem, into midpoint categories. Therefore, these methods (e.g. CML 2002) are considered to be problem oriented. For example, a material's impact on climate change can be expressed in kilograms CO_2 equivalents. Obviously, this is merely a quantification of an emission that contributes to the problem of climate change and not a quantification of the actual environmental damage.

(ii) The second school focuses much more on the actual effect. So-called damage oriented impact methods (e.g. Eco-indicator 99) try to model the cause effect chain up to the

Table 4
Allocation percentages by mass and economic value for FA and granulated BFS as calculated by Chen et al. [90] using data from [91–98].

Product	Mass produced	Market price	Mass allocation (%)	Economic allocation (%)
Steel	1 kg	400 euro/t	80.6	97.7
BFS	0.24 kg	40 euro/t	19.4	2.3
Electricity	1 kW h*	0.1 euro/kW h	87.6	99.0
FA	0.052 kg	20 euro/t	12.4	1.0

* Equivalent to 0.367 kg of hard coal used to produce electricity. The remaining produced FA and bottom ash.

Table 5
Link between the LCI data and the damage categories of the Eco-indicator 99 methodology (cf. Goedkoop et al. [103]).

Life cycle inventory (LCI)	Effect	Damage category
Extraction of minerals and fossil fuels	Surplus energy for future extraction	Damage to mineral and fossil resources (MJ surplus energy)
Land use: occupation and transformation	Occurrence of vascular plant species (POO)	Damage to ecosystem quality (% vascular plant species km^2 yr)
NO_x, SO_x, NH_3	Acidification/ eutrophication (PDF)	
Pesticides, heavy metals	Ecotoxicity: toxic stress (PAF)	
CO_2, Hydrochlorofluorocarbons (HCFC)	Climate change	Damage to human health (DALY)
Hydrochlorofluorocarbons (HCFC)	Ozone layer depletion	
Nuclides	Ionising radiation	
Suspended particulate matter (SPM), volatile organic compounds (VOCs), NO_x, SO_x	Respiratory effects	
Polycyclic aromatic hydrocarbons (PAHs)	Carcinogenics	

Table 6
Characterisation factors and indicator units of the CML 2002 impact method (cf. Guinée et al. [53]).

Impact category	Characterisation factor	Indicator unit
Abiotic depletion	Abiotic depletion potential (ADP)	kg (antimony eq)
Land competition	1 for all types of land use (dimensionless)	m^2 yr (land use)
Climate change	Global warming potential (GWP)	kg (carbon dioxide eq)
Stratospheric ozone depletion	Ozone depletion potential (ODP)	kg (CFC-11 eq)
Human toxicity	Human toxicity potential (HTP)	kg (1,4-dichlorobenzene eq)
Freshwater aquatic ecotoxicity	Freshwater aquatic ecotoxicity potential (FAETP)	kg (1,4-dichlorobenzene eq)
Marine aquatic ecotoxicity	Marine aquatic ecotoxicity potential (MAETP)	kg (1,4-dichlorobenzene eq)
Terrestrial ecotoxicity	Terrestrial ecotoxicity potential (TETP)	kg (1,4-dichlorobenzene eq)
Photo-oxidant formation	Photochemical ozone creation potential (POCP)	kg (ethylene eq)
Acidification	Acidification potential (AP)	kg (SO_2 eq)
Eutrophication	Eutrophication (EP)	kg (PO_4 eq)

endpoint, or the actual environmental damage, sometimes with high uncertainties. With respect to climate change, the damage on human health is quantified in terms of disability adjusted life years (DALYs). This unit counts as a measure for the Years Lived Disabled (YLD) and the Years of Life Lost (YLL) due to this damage.

According to Benetto et al. [100] the problem related approach provides reliable results, although it is sometimes difficult to compare them with each other. On the other hand, a damage oriented impact analysis, allows a much easier interpretation of the LCA output, but is considered to be not so reliable.

3.3.1. The IPCC approach

When the aim of the LCA study is a quantification of the concrete related GHG emissions only, it is justified to use the IPCC 2007 GWP impact method. According to this method, the corresponding Global Warming Potential (GWP) index is calculated for every emitted GHG. The index is based on the time-integrated global mean radiative forcing of a pulse emission of 1 kg of some compound relative to that of 1 kg of the reference gas CO_2 [101]. Logically, the GWP value for CO_2, the main GHG associated with cement production, equals 1 for the 3 commonly used time horizons, i.e. 20, 100 and 500 years [102]. The GWP index proposed by the IPCC is a problem oriented indicator because it merely quantifies

Table 7
Link between the most relevant LCI results for OPC cement, BFS and FA, and the LCIA categories (CML 2002 [53] and Eco-indicator 99 [103]) to which they are assigned.

LCI	Problem oriented LCIA (CML 2002)	Damage oriented LCIA (Eco-indicator 99)
1 kg CO_2	Climate change 1 kg CO_2 eq	Damage to human health 2.10×10^{-7} DALY
1 MJ fossil energy	Abiotic depletion 4.81×10^{-4} kg antimony eq	Damage to mineral and fossil resources 0.00859 MJ surplus energy (coal) 0.15000 MJ surplus energy (gas) 0.14400 MJ surplus energy (oil)
1 kg PM_{10}	Human toxicity 0.82 kg dichlorobenzene eq	Damage to human health 3.75×10^{-4} DALY
1 kg SO_x (as SO_2)	Human toxicity 0.096 kg dichlorobenzene eq Acidification 1 kg SO_2 eq	Damage to human health 5.46×10^{-5} DALY Damage to ecosystem quality 1.041 PDF m^2 yr
1 kg NO_x (as NO_2)	Photo oxidant formation 0.028 kg ethylene eq Acidification 0.70 kg SO_2 eq Eutrophication 0.13 kg PO_4 eq	Damage to human health 8.87×10^{-5} DALY Damage to ecosystem quality 5.713 PDF m^2 yr
1 kg OPC cement*	Climate change 0.84 kg CO_2 eq Abiotic depletion 1.37×10^{-3} kg antimony eq Human toxicity 6.83×10^{-2} kg 1,4-dichlorobenzene eq Photo oxydant formation 1.02×10^{-4} kg ethylene eq Acidification 3.08×10^{-3} kg SO_2 eq Eutrophication 4.75×10^{-4} kg PO_4 eq	Damage to human health 3.18×10^{-5} DALY Damage to ecosystem quality 2.14×10^{-2} PDF m^2 yr Damage to mineral and fossil resources 0.0245 MJ surplus energy (coal) 0.4275 MJ surplus energy (gas) 0.4104 MJ surplus energy (oil)
1 kg BFS* (economic-mass)	Climate change 0.13–1.10 kg CO_2 eq Abiotic depletion $0.76–6.44 \times 10^{-3}$ kg antimony eq Human toxicity $0.36–2.92 \times 10^{-3}$ kg 1,4-dichlorobenzene eq Photo oxydant formation $0.13–1.13 \times 10^{-4}$ kg ethylene eq Acidification $0.15–1.25 \times 10^{-2}$ kg SO_2 eq Eutrophication $0.62–5.25 \times 10^{-4}$ kg PO_4 eq	Damage to human health $0.24–2.03 \times 10^{-6}$ DALY Damage to ecosystem quality $0.39–3.32 \times 10^{-2}$ PDF m^2 yr Damage to mineral and fossil resources 0.0136–0.1151 MJ surplus energy (coal) 0.2383–2.0097 MJ surplus energy (gas) 0.2287–1.9293 MJ surplus energy (oil)
1 kg FA* (economic-mass)	Climate change 0.20–2.44 kg CO_2 eq Abiotic depletion $0.12–1.44 \times 10^{-2}$ kg antimony eq Human toxicity $0,17–2.12 \times 10^{-2}$ kg 1,4-dichlorobenzene eq Photo oxydant formation $0.16–2.00 \times 10^{-4}$ kg ethylene eq Acidification $0.18–2.17 \times 10^{-2}$ kg SO_2 eq Eutrophication $0.75–9.30 \times 10^{-4}$ kg PO_4 eq	Damage to human health $0.09–1.10 \times 10^{-5}$ DALY Damage to ecosystem quality $0.47–5.82 \times 10^{-2}$ PDF m^2 yr Damage to mineral and fossil resources 0.0207–0.2571 MJ surplus energy (coal) 0.3621–4.4898 MJ surplus energy (gas) 0.3476–4.3102 MJ surplus energy (oil)

* Values for 1 kg OPC cement, 1 kg BFS and 1 kg FA were calculated from the data available in Table 1 and 2.

GHG emissions (in kilograms CO_2 equivalents) and not the resulting climate change related damage (in DALYs) induced by them (see Section 3.3). The same indicator is used within the CML impact method for the impact category climate change (see Section 3.3.3).

3.3.2. The Eco-indicator 99 approach

Eco-indicator 99 combines a series of scores representative for different environmental impacts into one single score through weighing by a panel of specialists [103]. Naturally, the weighing part to obtain an aggregated indicator is the most critical and controversial step in the impact method. A panel was asked to weigh three types of environmental damages, namely damages to human health, ecosystem quality and resources extraction (Table 5).

The first damage type combines respiratory and carcinogenic effects, the effects on climate change, ozone layer depletion and ionising radiation into one value expressed in DALY.

The damage to Ecosystem quality is expressed in terms of the percentage of species that have disappeared in a certain area due to the environmental load (% vascular plant species km yr). With respect to ecotoxicity, this definition covers the percentage of all species present in the environment living under toxic stress (PAF: Potentially Affected Fraction). Regarding acidification/eutrophication, the damage to a specific target species (vascular plants) in natural areas is modelled (PDF: Potentially Disappeared Fraction). Land use and land transformation on the other hand are based on empirical data regarding the occurrence of vascular plants as a function of the land use type and the area size (POO: Probability Of Occurence).

The damage category dealing with resource extraction gives a value expressed in MJ surplus energy to indicate the quality of the remaining mineral and fossil resources. Logically, further extraction of mineral resources will result in a decreased ore grade and increased energy requirements for future mining. For fossil resources, a shift towards exploitation of unconventional resources can also induce an increase in extraction energy. The mining of resources such as sand or gravel are adequately covered by the effects on land use, not by the damage category on extraction of resources.

3.3.3. The CML 2002 approach

The list of best available practise impact categories drawn up by the Society of Environmental Toxicology and Chemistry (SETAC) Working Group on LCIA served as a basic list for the problem oriented impact method CML [53]. For LCA on concrete, we will mainly look at the baseline impact categories. For each category, a category indicator can be calculated based on the applicable characterisation model and the characterisation factors derived from the underlying model. The applicable characterisation factors and indicator units per impact category are summarised in Table 6.

3.4. Interpretation

One of the key parts of the interpretation phase is the identification of the significant issues based on the results of the LCI and LCIA phases. The literature review conducted in this paper allows not only for their identification starting from each phase, but also for revealing the link between the phases. Table 7 shows this link between the most important OPC cement, BFS and FA related LCI results and the appropriate problem (cfr. CML 2002 [53]) or damage oriented (cfr. Eco-indicator 99 [103]) LCIA categories. The values for 1 kg OPC cement, 1 kg BFS and 1 kg FA were calculated from the data available in Tables 1 and 2. Table 7 allows for an easier identification and check-up of the possible origins of a certain environmental score per impact category. Compared to the impact of 1 kg OPC cement (consisting of 95% Portland clinker), substantially lower values were recorded for BFS and FA when applying economic allocation, and this for every problem and damage oriented impact category that was studied. When applying a mass allocation of impacts for BFS and FA, practically all category indicator values exceed their corresponding values for OPC cement. Impact indicators covering human toxicity and damage to human health are the only exception. Thus, the environmental benefit of using BFS or FA is only pronounced when the economical allocation principle is adopted. The impacts of FA are clearly much higher than for BFS. As already stated in Section 3.2.2., this is mainly attributed to the fact that very little FA (0.052 kg) is produced per kW h of electricity, while a lot more BFS is produced per kg of steel (0.24 kg).

Similar environmental impact calculations can also be done for the aggregates and admixtures used in concrete. Yet, they were not included in Table 6 for the following reasons. As can be seen from Table 3, the air emissions and energy use associated with crushed aggregates (which are more labour-intensive than natural aggregates) are quite low compared to the values recorded for cementitious materials (Tables 1 and 2). Since superplasticizers are usually used in small amounts, their contribution to the impact of a concrete mix is very small. Note that the impact of the mixing water can not be calculated with either of the impact methods used. The term water footprint has evolved independently from the discipline of LCA and accordingly there is no clear relationship between a water footprint and potential environmental harm [48].

Finally, it should be noted that this paper mainly focused on the environmental impact associated with the material concrete. It is known that the embodied energy of building materials is normally much less than the energy required to operate building facilities (energy for heating, air conditioning, ...) during their service life [104]. In this sense, it would be interesting to include the thermal efficiency of the material within comparative LCA studies of traditional and 'green' concrete.

4. Conclusions

Within LCA, small variances in the definition of goal and scope (i), the inventory analysis (ii) and the impact analysis (iii) may induce important differences in the environmental score eventually obtained in the interpretation phase.

Within step (i), the functional unit (FU) choice is seen as one of the most influencing factors. Preferably, this unit includes all relevant concrete aspects, being its strength, its durability and to some extent its workability. To take into account both strength and durability, it should comprise the concrete amount needed to manufacture a structural element or even a whole building with a given mechanical load and a predefined service life. The concrete's life span should be evaluated in relation to the latter, using probabilistic service life prediction models based on experimental durability tests representative for the application field. As LCA usually involves a comparison of impacts, the choice of the reference concrete, characterised by a minimum cement content and strength according to the applicable standards, is also extremely important. This is because the environmental benefit of a potential 'green' concrete is mainly due to its reduced cement content and strength governed structure dimensions in comparison with the reference. Besides material aspects, LCA system boundaries (cradle-to-gate, cradle-to-grave, ...) play an important role. Since concrete durability is an important issue, at least the amount of concrete needed (including repairs and replacements) to maintain the structure during its predefined service life (modified cradle-to-gate) should be considered as system boundary.

Within step (ii), the inventory data used can obviously influence the LCA outcome. The preference goes to first hand data to ensure correctness and technological, geographical and time related representativeness of the LCA. Of importance is also the allocation approach used (none, by mass or by economic value) on the inventory results associated with by-products. Economic allocation is advised to guarantee an enduring use of BFS and FA as cement replacing materials. Mass allocation results in environmental impacts of BFS and FA that are about an order of a magnitude higher. As a consequence, their environmental burdens, especially when considering FA, become higher than the burden of traditional cement.

Within step (iii), the choice of the impact assessment method, which aggregates the relevant concrete related LCI results, should be carefully considered. The method used preferably covers more than only the impact on climate change and should be problem oriented.

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