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Sustainability assessment of recycled aggregate concrete structures: A critical view on the current state-of-knowledge and practice

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Abstract

The environmental impacts of activities such as raw material extraction, construction of infrastructure, and demolition, place construction as one of the sectors that exert the highest pressures on the environment, society, and economy. Some of the major environmental impacts for which the construction industry is responsible are mineral resource depletion, greenhouse gas emissions, and waste generation. Among the different strategies that exist to decrease such impacts, recycling demolition waste into recycled concrete aggregates has been considered a promising alternative. As such, at present, the literature dealing with the impact assessment of recycled aggregate concrete structures is very extensive. Therefore, the objective of this article is to present a critical view of the state-of-the-art in terms of sustainability assessment of recycled aggregate concrete structures, taking a holistic perspective by considering environmental, social, and economic impacts.

KEYWORDS

construction, economic impact, environmental impact, recycled aggregate concrete structures, societal impact, state-of-the-art, sustainability

1 | INTRODUCTION

The construction industry—including a range of activities from the extraction of raw materials; manufacturing/distribution of construction products; construction, use, and management; maintenance, renovation, and demolition; and recycling of construction and demolition waste (CDW)—is alone responsible for a large portion of Europe's environmental footprint: 50% of natural raw materials use, 40% of total energy consumption[1] (as the single largest consumer), 46% of total waste generated,[2] and 36% of all greenhouse gas (GHG) emissions.[3] Within the construction industry, mineral materials, concrete, and other cement-based materials are responsible for a large share of the industry's environmental impacts. Extremely large production of concrete, with 25 billion tonnes produced annually at the global level,[4] causes the following major environmental impacts:

- large consumption of natural mineral resources and energy (mostly for cement and reinforcement steel

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production; in addition, for maintenance of buildings, and other structures; finally, for transportation, construction, demolition, and recycling at smaller extent and except in the case of re-use);
- large emissions of greenhouse gasses, primarily CO_2, which are responsible for climate change and originate mostly from cement production (due to clinker content) and energy consumption; to a smaller extent, emissions of SO_2 are responsible for acidification and mostly originate from the transportation phase;
- large amount of produced CDW (mostly inert and non-dangerous wastes).

For these reasons, it is of crucial importance to find way(s) of greening the concrete industry, that is, to decrease its impacts on the environment. Recycling CDW into recycled concrete aggregates (RCA) is a way to reduce simultaneously the amount of waste and consumption of natural mineral resources. Besides the fact that sources of quality sand and stone for aggregate production are not endless, especially at a regional level, their unlimited extraction also has a strong impact on the environment and leads to direct local devastation of the natural environment, whether it is a crushed stone or river aggregate.[5] Landfill capacity is becoming a very important and scarce resource nowadays in many countries. On the other hand, the utilization of RCA in new concrete structures helps close the concrete loop within the circular economy context.

However, replacing the virgin aggregates with RCA does not necessarily and directly lead to better environmental performance in the course of a concrete structure's life cycle. Besides, the holistic sustainability assessment should include social and economic assessments as well. Therefore, scientifically based methods for the assessment of all three sustainability aspects are needed. For the environmental assessment, the well-recognized and standardized methodology of Life cycle assessment (LCA) is usually applied. It allows for evaluating the environmental impacts of processes and products during their life cycle. The LCA is used according to the ISO 14040 standards,[6] which provide a framework, terminology, and methodological phases of the assessment: (1) goal and scope definition (including the system boundaries and functional unit [FU] definition), (2) creating the life cycle inventory (LCI), (3) assessing the environmental impact (LCIA) and (4) interpreting the results. Besides these four mandatory steps, normalization, grouping, weighting, and additional LCIA data quality analysis are optional steps within the LCIA phase. Social and economic assessments should be performed within the same framework and for that purposes social LCA and economic LCA (Life Cycle Costing) were developed. Currently, LCA standards related to the built environment are EN 15804 (product),[7] EN 15978 (buildings)[8] and EN 17472 (civil engineering works),[9] whereas standards related to the Built Social and Economic assessment are, respectively, EN 16309-A1[10] and EN 16627[11] concerning buildings. EN 17472[9] for civil engineering works includes social, economic, and environmental aspects. Compared with environmental LCA, economic and especially social LCA were rarely applied in the sustainability assessment of recycled aggregate concrete (RAC) structures in the previous research—these methodologies are less systematized currently. Finally, there is a big step between the assessment and standardized design of any kind of concrete structure.

The objective of this work is to provide the critical review of the existing LCA methodologies and those applied in standards; to explain the limitations and their consequences; to recommend certain methodological choices depending on the goal of the LCA study; and to point out the directions of the future research in the area of sustainability assessment and design of RAC and structures made of RAC.

2 | ENVIRONMENTAL ASSESSMENT

To correctly interpret the results of a LCA applied to RAC structures, it is important to understand the calculation assumptions and the resulting uncertainties for each phase of the method. Thus, first, the most important methodological issues regarding the application of LCA in the assessment of RAC structures are discussed. Then, basic environmental impacts suitable for RAC concrete/structures are presented.

2.1 | System boundaries and LCI data modeling

A basic methodological issue in applying LCA for RAC structures is modeling the concrete waste recycling. The choice of system boundaries and LCI data modeling depend on the way concrete recycling is modeled, which is directly related to the type of LCA that is performed.[12–17] Generally, two types of LCA are distinguished: attributional (ALCA) and consequential (CLCA). In assessments using an attributional approach, the purpose is to estimate the impacts of a product system within cradle-to-grave boundaries, at a given point of time assuming a *status-quo situation*. A consequential approach on the other hand is intended to estimate the environmental burdens that occur, directly or indirectly, as a consequence of a certain decision. In that case results represent the net environmental impacts of the change caused by this decision—change in demand. Two major differences between these two approaches are the following:

- Different system boundaries—an ALCA model does not include processes other than those of the life cycle investigated, so allocation based on either physical flows or economic value is usually applied when dealing with multifunctional processes. The CLCA model includes processes that are significantly affected irrespective of whether they are within or outside the cradle-to-grave boundaries, so system expansion is the only way for resolving multifunctional processes. In practice, instead of system expansion, a substitution or so-called avoided burdens approach method is usually applied as being mathematically equivalent at the process level.[12]
- Different data—an ALCA model should include average data on each unit process within the system's life cycle, while the CLCA model should include marginal data on all the interlinked processes/technologies that will be affected by the decision.[16] Besides, changes in the production and alternative use of co-products in CLCA should be based on the analysis of how the relevant markets and price elasticity of supply and demand of each product/co-product are affected.[16]

It is the goal of the LCA study (assessment of the product system impact or assessment of the impact of a decision to change) that should determine whether ALCA or CLCA is used. CLCA is much more complex—marginal data on many processes and many assumptions on relevant markets, alternative products, price elasticities, and so forth are needed.

CDW concrete recycling is considered a case of open-loop recycling since the inherent material's properties are changed over the product system and the possibility of endlessly repeated recycling is questionable (not investigated yet). However, it is a multifunctional and multiproduct process: it has two functions (waste management service for the upstream product that is natural aggregate concrete—NAC structure), and material production for the downstream product (RAC structure) and two co-products (RCA and additional iron scrap recovered during the recycling process). The question of how to allocate inputs and outputs between these functions and co-products arises.

ISO 14040[6] describes a three-step procedure with regard to allocation—partitioning of inputs and outputs between the co-products or functions. As a first step, allocation should be avoided where possible by dividing the process into subprocesses or by expanding the system boundaries to include all the products/functions involved. As a second step, when allocation cannot be avoided, allocation must be done in a way that reflects an underlying, causal, physical relationship, usually mass allocation. The third step is about "other relationships" such as market value—economic allocation.

LCA can be performed at three levels:

- Cradle-to-gate—includes the product production phase ("factory gate," commonly used when materials are assessed);
- Cradle-to-grave—includes all life cycle phases from raw materials extraction to disposal;
- Cradle-to-cradle—goes beyond the boundaries of the studied system including the secondary life of recovered materials.

At cradle-to-gate and cradle-to-grave levels both ALCA and CLCA are applicable and consequently, two approaches are used when modeling concrete recycling.

In the attributional approach, inputs, and outputs of the recycling process at both functional and co-product levels are allocated between NAC and RAC systems. The most traditional allocation procedure for reuse and recycling is not to apply allocation at all. This is a simple cut-off rule, where a product made out of primary materials carries the burdens of those primary materials, and a product made out of secondary materials carries the burdens of the recycling activities of those secondary materials.[13] In the concrete case, this means that environmental burdens of all stages from raw material production to the disposal of non-recyclable wastes are included in the NAC system under study. The environmental burdens of recycling are excluded from the system, as they are considered burdens to the next product system-RAC,[18] Figure 1. This method however has variations. For example, in France the EPD of RCA excluded impacts due to the reduction of block size which is allocated to the parent concrete with a specific EPD "treatment of deconstruction waste to produce aggregates."[19] Beside the cut-off rule, the 50:50 partition rule and economic allocation are sometimes used. If an economic allocation is used, recycling activities are allocated proportionally to the shares of waste management service and recycled material in total economic proceeds.[20] For example, in the case of blast furnace slag economic allocation (between steel and blast furnace slag) is often used to determine its environmental burdens; it's also the case of fly ash or silica fume. The economic proceeds are calculated based on the quantity and market price of the service or product. The main

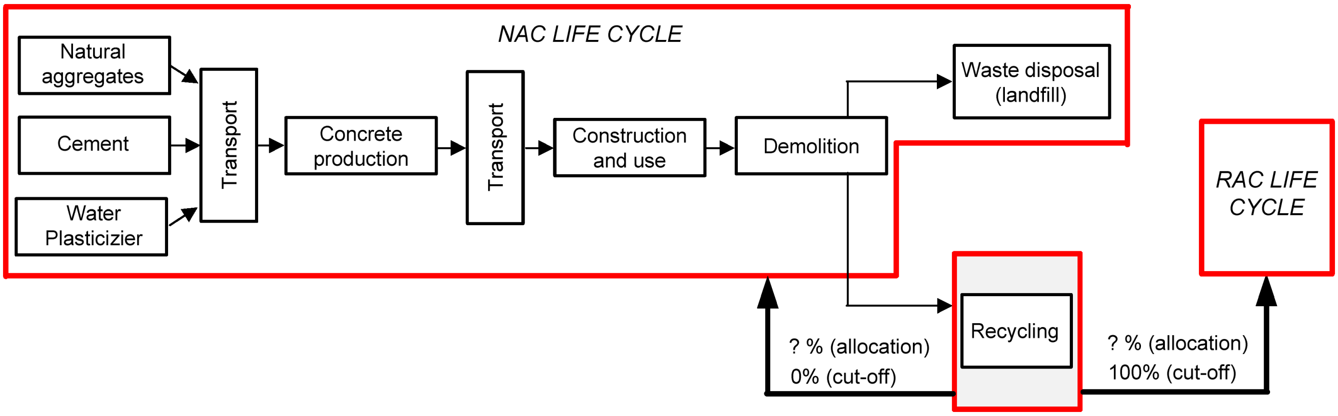

FIGURE 1 Allocation of recycling between natural aggregate concrete (NAC) and recycled concrete aggregates (RAC) life cycle.

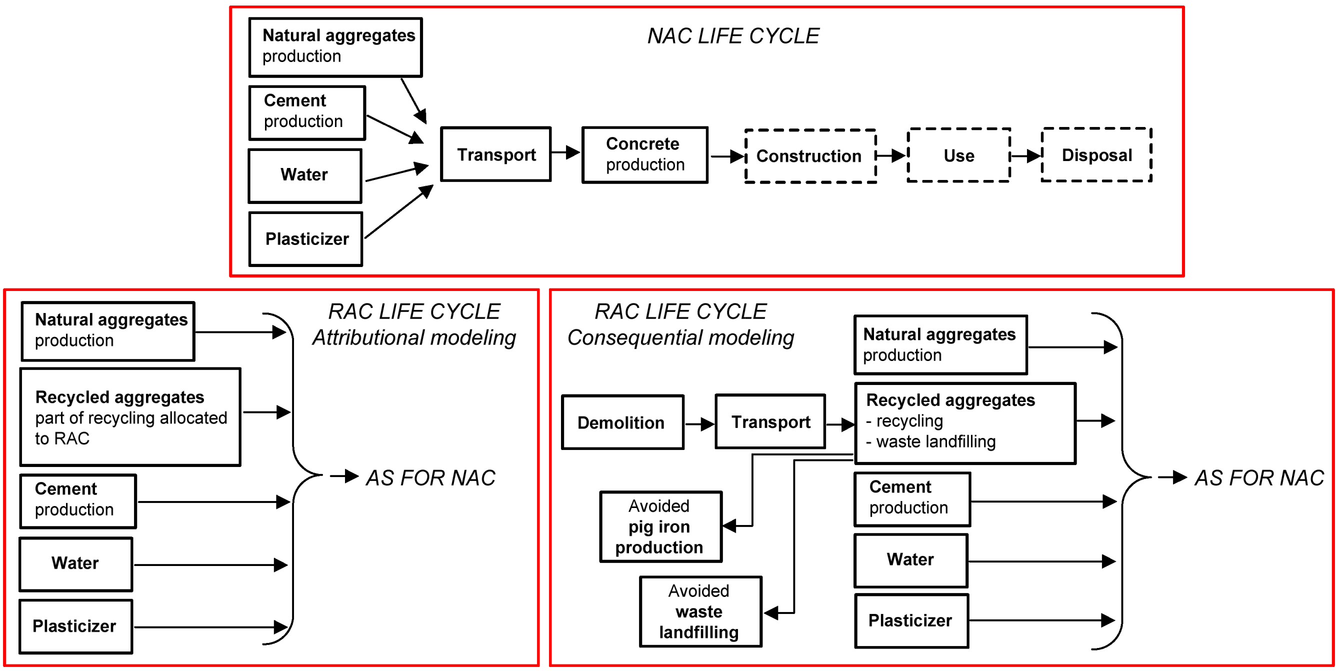

FIGURE 2 Comparison between attributional and consequential modeling of recycling at cradle-to-gate level (- - -: cradle-to-grave).

disadvantage of economic allocation is that it is based on market prices, which can be unstable and fluctuate in the case of recycled materials and markets. The main disadvantage of mass allocation is claiming that mass is the best (and only) representative of physical casualties.

Previous research showed that the type of allocation makes no significant differences at the scale of concrete (or RAC) when airborne emissions and energy use are considered. Marinković et al.[21] reported the energy use and emissions increase by up to 2% when economic allocation is compared with the cut-off rule. Visintin et al.[22] reported a maximum of 1.5% change in $CO_{2\text{-equ}}$ emissions when varying the allocation from 0% (a cut-off rule) to 100% (all recycling burdens allocated to NAC). Gervasio et al.[18] tested the cut-off rule, 50:50 partition rule, and economic allocation with similar conclusions. Therefore, the simplest cut-off rule can be recommended when calculating energy use and impact categories based on airborne emissions. This is because the cement production (especially if it has a high clinker content) is by far the largest contributor to these impacts, while the contribution of aggregate production is rather small, and allocation cannot affect the total result for more than a few percent.

In the consequential approach, the single function system is obtained by subtraction of the alternative substituted process. At the cradle-to-gate and cradle-to-grave levels of assessment, the recycling process is commonly ascribed fully to the RAC life cycle (at the stage of RCA burdens), and RCA production is credited for the processes that are displaced by recycling: waste landfilling and pig iron production.[23,24] This means that burdens from these processes are considered avoided burdens for the RAC system (see Figure 2). Credits from avoided natural aggregate (NA) production are not included since "the material-production" function of the recycling process serves the function of the whole system—the production of concrete/concrete structure.

At the cradle-to-cradle level, only CLCA is applicable because ALCA includes only processes within the boundaries of the studied system. Recycling is considered a waste management option, and credits through the avoided NA production and transportation are subtracted from the natural aggregate concrete (NAC) life cycle.[25–27] For instance, Blegnini[26] compared two different end-of-life scenarios for an actual residential building in Turin (Italy): complete recycling of waste (except for a small amount of common waste which was landfilled) and complete landfilling. He performed a comprehensive LCA of the whole life cycle of this building with special attention paid to the demolition and recycling phases. For these phases, analysis was based on actual, field-measured data. Since the building was reinforced concrete framed, waste consisted mostly of concrete (83%) and steel rebars (4%). The net impacts from steel and concrete recycling were calculated as the difference between avoided impacts (avoided primary NA production and transportation) and induced impacts (recycling process and transportation). Obtained results showed that the reductions in the recycling option are quite small in comparison with the whole life cycle impacts (0.2%–2.1%). When the comparison was restricted to the pre-use phase (material production and construction phases), 19% savings of energy use and 10% savings of GWP were found for the concrete waste recycling scenario.

The cradle-to-cradle approach is supported (partly) by EN15804[7] and EN16757[28] standards related to the sustainability of construction products and concrete and concrete elements, respectively. These standards present a mix of attributional and consequential modeling. They use the modular structure for the environmental impact report: module A—material production and construction phase, module B—use phase, module C—end-of-life (EoL) phase, and module D—benefits and loads beyond the system boundaries for reuse, recovery and recycling expressed as net loads and benefits. It should be noted that module D is not included in the calculation of the total life cycle, it is considered an optional module. Then, it is in fact an ALCA approach, but it can become simplified CLCA if module D is included. The EoL system boundary is set where outputs of the system have reached the "end-of-waste" state.

Following the "polluter pays" principle CEN standards prescribe practically the 100:0 cut-off rule for allocation of EoL processes[7]: all loads of secondary materials before they reach the "end-of-waste" are attributed to the system that produces waste, while processes after having reached the "end-of-waste," required to reach the functional equivalence to replace the primary material, are beyond the system boundary and declared in module D. Applied to waste concrete recycling, this means that crushing and stockpiling of crushed concrete is attributed to the system under study, while direct substitution of primary material (in road construction for instance) or sizing into fractions, stockpiling and use as aggregate in new concrete are reported in module D. This way of allocating burdens and reporting benefits in module D leads to promote RCA, and to penalize the stockpiling without crushing. In comparison, NA issued from careers and crushed have burdens due to this crushing step as opposed to RCA (burdens due to crushing steps are highly reduced).

Hence, when the "end-of-waste" is reached and material is recycled, the producer of recycled material can declare credits for avoided primary material production in module D. For example, if reinforcement is made of steel with 54% recycled content and after demolition 96% of the reinforcement is recycled, the net benefit due to avoided virgin steel production reported in module D is $96-54=42\%$.[29]

System expansion is preferred by ISO 14040[6] as the method of resolving multifunctional processes. However, when avoided burdens approach is applied within an ALCA framework with an assumption that the substitution of virgin with recycled material is 1:1 and without analysis of how the relevant markets and price elasticity are affected, and how the EoL processes technologies will develop in the future, the results of such analysis are questionable.[27] Simple subtracting the avoided burdens from the system's life cycle does not make a CLCA—in fact, avoided burdens approach should be applied only within a framework of a proper CLCA.[17] For instance, in previous analyses, the assumption is made that iron scrap is fully utilized in steel-making processes and actually displaces pig iron there, and that there are no differences between them regarding quality and cost.

The same assumption is made for the replacement of NA with RCA in cradle-to-cradle assessments, and EoL practices of today are assumed to be valid in the distant future. Nevertheless, it is accepted in the research community and, as already mentioned, is supported by EN15804[7] and EN16757[28] standards if module D is implemented. The application of the avoided burdens approach within the ALCA framework is partly justified when using these standards since several criteria must be fulfilled for reaching the "end-of-waste" state. These criteria make the substitution of primary materials and fuels with secondary ones more realistic (common application, existence of a market or demand, fulfillment of existing legislation and standards for secondary materials, etc.). The attempt was made also to deal with a functional non-equivalence of secondary and substituted primary material, although the provision is rather vague. It says that when the output flow does not reach the functional equivalence of the substituting process, "a justified value-correction factor" should be applied to reflect the difference in functional equivalence.

At the same time, ALCA fails to encompass the benefits of natural mineral resource preservation and waste reduction brought by waste concrete recycling. This can be taken into account through appropriate impact categories with corresponding indicators (for instance, landfill capacity depletion and mineral resources depletion). Unfortunately, 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. Therefore, it seems reasonable to aim the research towards the development of special indicators for natural bulk resources depletion[5] and landfill space depletion.

The previous version of EN 15804, from 2013, fails to properly include those indicators. Particularly the indicator of resource depletion is not representative of the territorial context. It is a global indicator even though the resources of natural aggregates are not the same all around the globe. Moreover, depletion of aggregate should be evaluated by silicon inventory (element instead of mineral), well this resource is considered with very little depletion. As a result, no significant difference is observed between inventories of RCA and NA when using the indicators recommended in the previous standard EN 15804.[30] The land consumption due to waste storage was also not recognized as impact indicator. The most recent version (2020) of this standard[7] includes a new indicator "land use/soil quality" but there is no improvement regarding the mineral depletion—standard "abiotic depletion—minerals and metals" impact category is calculated from elements inventory and expressed in antimony (Sb) equivalents.

2.2 | Functional unit

If made with the same water-to-cement ratio without admixture, RAC has lower compressive strength and modulus of elasticity, larger creep and shrinkage, and lower carbonation resistance compared with NAC.[31–34] To perform the comparative assessment in such a situation, two solutions are possible. First one is to keep FU equal to unit volume but to adapt mix designs to obtain the same mechanical and durability-related properties if possible, for example by the use of admixture. Second solution is to correct the unit-volume FU to account for different properties. Both approaches were used in previous research but in both cases, it is the material properties that determine the FU. Such FU based only on the material properties can be used for generic comparisons in order to evaluate the sustainability potential for material substitution. Besides, such comparative LCAs should clearly state that the performance of a specific application of assessed materials is not taken into account.[35]

Since the concrete structure is a specific product with a known area of application, the same functional requirements regarding safety, serviceability, and durability (expressed through service life) should be satisfied for all compared alternatives in comparative LCA. Service life is not the property of material but a function of the concrete structure and therefore it is hardly possible to obtain functional equivalence with FU based only on the concrete properties. Therefore, in comparative assessments of concrete structures, FU based on a specific concrete structure performance should be applied.

Some standards however rely on the material properties when taking into account the service life of a structure. For example, in the case of a French regulatory calculation, if a material has a 50 years reference service life, its impacts will be counted twice for the LCA of works and once for the LCA of a building. Similarly, if a material has a 100 years reference lifetime, its impacts will not be divided by two for a building; if a material has a 40-year lifetime, its impacts will be counted twice for a building[36]; this way of calculating can be criticized.

When comparing structures made of concrete with different properties two approaches for obtaining the functional equivalence are possible: (1) either to correct the FU volume to obtain same performance or (2) to normalize the calculated environmental impacts with compressive strength and duration of service life if FU has the same volume. What however can lead to very different assessment results is the fact that for the service life prediction in the case of deterioration mechanisms which cause the reinforcement corrosion, only the depth of the concrete cover in reinforced concrete (RC) member matters. In previous research, the second approach was mostly applied. For instance, in References 37 and 38, an RC column with chosen size dimensions was applied as FU. Silva et al.[39] chose a linear member with a specified length and cross-section size as FU and performed the service life prediction for different concrete mixes. These assessments were based on the FU which represents the same volume of compared RC alternatives, that is, the same depth of concrete cover, resulting in different service lives of compared alternatives. With this approach, the functional equivalence regarding durability is not achieved and for the proper comparison environmental impacts must be calculated per year of service life or a certain number of repairs within service life must be included. This leads to a situation that concretes with poor resistance to chosen deterioration mechanisms have significantly larger environmental impacts per year of service life (since they have much shorter service lives).

At the same time, the same service lives of compared alternatives can be obtained with different cover depths keeping the same structural member's strength. Since the depth of concrete cover is usually equal to several centimeters, this causes small changes of volume in FU and can cause totally different (for the order of magnitude) environmental impacts during service life.[35] The serviceability aspect should also be taken into account, especially when analyzing structural members where long-term deflections are detrimental in design. In Reference 40, RC slabs representing a typical floor structure of a residential building and made either of NAC or RAC with 100% replacement of natural coarse aggregate were compared. Both slabs were designed to fulfill the same functional requirements: strength, long-term deflections, service life of 50 years (for XC3 exposure class according to Eurocode 2), and fire resistance. In the NAC case, required slab's height was 160 mm, while in the RAC case it was 170 and 180 mm for RCA with water absorption equal to 4.45% and 5.73%, respectively. If 1 m^2 of slab area is adopted as a basis, the FU volume for RAC was 6% and 12% higher compared with the FU of NAC, respectively. If beams were analyzed, the increase in FU volume would be smaller. Such a small increase in the FU volume has a much lower influence on the environmental impacts compared with the influence that different service lives would have. Hence, care should be taken when choosing FU in comparative LCA of concrete structures no matter which type of concrete is used—functional equivalence of compared structures can be obtained in more than one way. Thus, considering the FU on the material level is not sufficient, and the FU on the element or structural level should be considered with a multifunctional approach.

2.3 | CO_2 uptake

It is a well-known fact that cement production is responsible for large carbon dioxide emissions, on average of 850 kg of CO_2 per ton of clinker.[41] About 60% of this amount is emitted from the calcination process of limestone, and the rest comes from the burning of fossil fuels in the clinker kiln. The calcination process is a chemical reaction in which limestone (which mainly contains calcium carbonate) is converted to calcium oxide and carbon dioxide at high temperatures, named decarbonation ($CaCO_3 + \text{heat} \rightarrow CaO + CO_2$). On the contrary, when exposed to air, concrete structures will over time reabsorb CO_2 from the atmosphere through carbonation. It is a chemical process reversed to calcination in which atmospheric CO_2 diffuses into concrete to react with hydration products (calcium hydroxide and other calcium-rich hydrated oxides) and form calcium carbonate again ($Ca(OH)_2 + CO_2 \rightarrow CaCO_3 + H_2O$ in the case of portlandite carbonation); theoretically, the resulting hydration products can react with the same amount of atmospheric CO_2. Therefore, over the life cycle, chemical CO_2 released from the calcination process will be reabsorbed or uptaken by the concrete structure.

CO_2 absorption by cement-based materials through carbonation can actually be considered a natural form of carbon capture and storage. It happens during the entire life cycle of concrete structure from the moment when concrete is produced, through service (primary) life, end-of-life, and secondary life phase. The major problem is that carbonation is a very slow process—complete reabsorption of chemical CO_2, which is theoretically possible, would take decades, even centuries. Indeed, the carbonation rate is limited by the progression of the carbonation front which progressively modifies the pore network. If, after service life, which is normally taken as 50 years for buildings, demolished concrete waste is crushed into RCA and then again used in new construction, the carbonation process will continue during the "secondary life" of concrete.[42] The presence of adhered mortar on the surface of RCA enables further carbonation. The amount of captured CO_2 in the secondary life phase depends on the RCA application: whether it is used in unbound applications (for sub-base and base of road structures, embankments, and fillings, where it is commonly used) or as aggregate in new concrete construction or asphalt. In the former case, there is a much larger potential for natural CO_2 absorption during its second life because the exposed surface area relative to the volume of RCA is greatly increased compared with the concrete structure itself. Between primary and secondary life there is an intermediate phase where RCA has to be stockpiled for a certain period of time until it is used again. In this phase, the carbonation is very fast due to the many times larger surface area exposed to atmospheric CO_2, as it is in the case of secondary life unbound applications. But unlike the secondary life, this phase has the potential of maximizing the CO_2 absorption by proper choice of stockpiling manner (for instance, stockpiling of different particle sizes separately), exposure conditions (sheltered from rain), and enhancing the exposure time to the maximum which is acceptable in industrial practice.

The research in the area of CO_2 uptake started in the last decade in Nordic countries initiated by Nordic Innovation Centre. This research was devoted to CO_2 uptake modeling and calculation of reabsorbed CO_2 by mineralization in the course of the service life of building stock in Nordic countries. Andersson et al.[43] proposed a model for CO_2 uptake calculation during the service life of existing concrete structures. The model has been applied to data from Sweden and results showed that a CO_2 uptake in 2011 was about 17% of the total emissions (calcination and fuel) from the production of new cement for use in Sweden in the same year. Other researchers have also advocated that CO_2 uptake should be considered to offset the production emissions within the concrete structure life cycle.[44–46]

EN 16757:2017 in its Annex BB[28] and the new CEN technical report on carbonation and CO_2 uptake in concrete[47] provide some practical recommendations on the calculation methods for CEM I concrete and indicative correction factors for a selected range of other binders (cement with commonly used additions: fly ash in the range 10%–20% and 30%–40%, limestone in the range 10%–20%, and slag in the range 10%–80%). The report also provides some information on the CO_2 uptake in the EoL phase (concrete crushed into RCA), based on the published research, but no recommendation is given regarding the CO_2 uptake of RAC structures. According to these documents, the CO_2 uptake, in kg per square meter of concrete surface, during t years can be calculated as follows:

$$CO_2uptake = \left(k\sqrt{t}/1000\right)\cdot\left(CO_2uptake_{tcc}\right)\cdot C\cdot M \qquad (1)$$

where $x_c = k\sqrt{t}$ is the carbonation depth (mm) and k is a carbonation rate coefficient (mm/year$^{0.5}$) $CO_2uptake_{tcc} = (w\%reactiveCaO/100)\cdot\frac{m_{CO_2}}{m_{CaO}}$ is the maximum theoretical uptake of CO_2 in fully carbonated concrete (kg of CO_2/kg of binder); w% of reactive CaO, part of reactive CaO (100·kg CaO/kg binder); m_{CO2}, molar weight of CO_2 = 44 g/mol; m_{CaO}, molar weight of CaO = 56 g/mol; C, binder content; M, degree of carbonation.

For instance, if ordinary Portland cement is the binder (CEM I), which includes at least 95% clinker and 65% of reactive CaO typically, the maximum theoretical CO_2 uptake is $(65/100) \times 0.95 \times (44/56) = 0.49$ kg CO_2/kg cement. From Equation (1) it follows that the basic parameters are the carbonation depth, the binding capacity of binder and its content, and degree of carbonation. Unlike the carbonation and CO_2 uptake of NAC, the RAC CO_2 uptake consists of the uptake of new hardened cement paste in RAC and the uptake of old hardened cement paste in RCA. Florea and Brouwers[48] have tested the RCA produced in several different ways. For RCA obtained by one-stage crushing of laboratory concrete with a water-to-cement ratio equal to 0.5, authors reported that hydrated cement paste content ranged from 10% to 25%, depending on the RCA fraction size (from 32 to 4 mm). This means that RAC contains reactive CaO in addition to CaO available in cement, as if the mass of cement is increased for a certain percentage of the RCA mass. How much of the reactive CaO remains in the old cement paste depends on the duration of the exposure period and conditions while RCA was stockpiled before being used as aggregate in RAC. Many researchers have proven, on the basis of accelerated carbonation tests, that carbonation depth is larger for RAC than for corresponding NAC and that increase depends on the RCA amount—the more RCA in the mix, the larger the carbonation depth. A comprehensive study is given in Reference 49 and several prediction models for the RAC carbonation were proposed.[34,50,51] Therefore, the RAC structure should have greater CO_2 uptake within production and service phase compared with the corresponding NAC structure. However, results reported in previous research vary significantly depending on the assumed prediction model.

Modeling of RAC and especially RCA uptake is a complex task since many parameters are involved; however, several models were developed recently. Fang et al.[52] developed an empirical CO_2 uptake model for the carbonation of stockpiled RCA while Xiao et al.[53] and Huang et al.[54] developed models which include the carbonation of RAC structures during service life as well as the carbonation of the stockpiled RCA. For the same FU (1 m^3 of concrete with an exposed surface area of 5.68 m^2), exposure conditions (relative humidity of 76% and volume CO_2 concentration of 0.034%) and scenario (50 years of service life and 30 days of RCA stockpiling period prior to manufacturing RAC) and for similar concrete's strength, they obtained rather different results regarding the CO_2 uptake of RAC, Table 1.

Huang's model seems to be more elaborate and complete. However, this model is valid only for RCA with a diameter larger than 5 mm and the fine RCA which has the fastest carbonation rate in the stockpiling phase is not included. Nevertheless, the major part of total RAC CO_2 uptake comes from the RCA carbonation in the stockpiling phase—about 65% of total CO_2 uptake. Compared with NAC, RAC100 CO_2 uptake is 3.6 times larger. With stockpiling phase prolonged to 90 days total CO_2 uptake increases to 31 kg/m^3 with about 75% belonging to RCA carbonation (five times larger compared to NAC) according to authors of Reference 54.

TABLE 1 Results of the CO_2 uptake model by Huang et al.[55] and Xiao et al.[54]

		CO_2 emissions per m^3 of concrete (kg)	**CO_2 uptake per m^3 of concrete (kg)**	$\frac{\lvert(2)\rvert}{(1)} \cdot 100(\%)$
	Concrete type	(1) Cement production	(2) Service life + stockpiling	
Huang et al.[54]	NAC	269	−6.2 (no stockpiling)	2.3
	RAC100[a]	290	−(8 + 14.4)	7.7
Xiao et al.[53]	NAC	242	−5 (no stockpiling)	2.1
	RAC100	261	−(23.4 + 52.2)	29.1

[a]Concrete with 100% replacement of coarse NA with coarse RCA.

A different approach was applied in Wijayasundara et al. work.[55] The CO_2 uptake in the production phase was calculated on the basis of experimental results of Dayaram[56] which were extrapolated to industrial conditions (assumed RAC production period from 9 to 16 days, assumed composition of concrete waste, and participation of different RCA fractions, exposure conditions, cement type, etc.). The authors obtained 4.9 and 16.4 kg of uptaken CO_2 per m^3 of RAC for 30% and 100% replacement rates, respectively. The CO_2 uptake over the service life of 50 years was calculated for two case study concrete house buildings and reported as incremental values compared to CO_2 uptake obtained for those buildings if they were made of NAC. The authors used the model recommended by CEN/TR 17310:2019 with increased cement content in RAC based on the residual cement mortar in RCA, which was assumed to be 25%. The incremental CO_2 uptake in those two case studies were 6.8 and 22.8 kg/m^3 of RAC for 30% and 100% replacement rates, respectively. Prediction models for the RCA uptake are based (among other simplifications) on the assumption that all RCA in the stockpiling phase is equally exposed to CO_2, which in reality does not have to be the case. Since aggregates are usually stored in piles it is possible, especially if different fractions are mixed, that airflow through the pile is prevented and that only outer layers carbonate. While prediction models for NAC and RAC CO_2 uptake can be tested in the laboratory (accelerated carbonation in combination with thermogravimetric analysis and mass spectrometry analysis for instance), prediction models for RCA CO_2 uptake should be tested in large-scale field conditions which are lacking. Very limited field measurements do not support high values of RCA uptake in the stockpiling period obtained with the described prediction models. Kikuchi and Kuroda[46] performed measurements of CO_2 uptake of RCA that was obtained from several recycling plants in Japan. Usually different fractions are mixed together within piles and fully exposed to the atmosphere for a period between 1 and 3 months. They reported a CO_2 uptake of 11 kg/tonne of RCA, corresponding to about 26 kg CO_2/m^3 of demolished concrete. Having in mind that RAC100 usually contains about 1000 kg of RCA, the closest to this measured value are the Huang et al. predictions.[54] Andersson et al.[43] recommended that, if an improved procedure with air access in the fractions and at least 4 months of storage in at least three fractions is applied, CO_2 uptake of RCA could be set to 20 kg CO_2/m^3 of demolished concrete.

To go further in CO_2 capture and its environmental assessment, accelerated carbonation of RCA was studied. It is a process based on the mineralization of industrial gas, especially CO_2 within portlandite and calcium silicate hydrates of RCA cement paste. In laboratory or industrial conditions, gas is in contact with RCA and the carbonation reaction, by the capture of CO_2, leads to obtaining carbonated recycled concrete aggregates (CRCA) with improved qualities (clogging the porosity).[57–59] A French National Project (FastCarb), aimed to transpose the accelerated carbonation at an industrial scale at a suitable economic and environmental cost.[60] In addition to capturing CO_2, by improving the mechanical properties of recycled aggregates, which are weaker than those of natural aggregates and potentially require the use of more cement in concrete, the environmental impact of concrete could be reduced.

An environmental assessment applied in two industrial demonstrators was performed to validate the relevance of the project and to identify the critical points in order to minimize the impacts. Due to the quantity of cement paste of RCA, the amount of CO_2 captured is much more significant for sand (31–39 kgCO_2/t for 0–4 mm) than for gravel (16 kgCO_2/t for 4–16 mm).[61] The impact of accelerated carbonation on the environmental assessment of RAC was carried out by comparing the environmental impacts of concretes formulated with the same cement content (320 kg/m^3) and water-to-cement ratio (0.55 in terms of efficient water) and containing varying amounts of natural, recycled and recycled carbonated aggregates. Considering global warming impact, the lowest emissions are observed for the NAC due to its small quantity of admixture. However, if the CO_2 captured by RCA is deducted, the lowest emission is obtained for CRAC (Carbonated RAC) composed of 40% CRCA sand and 100% CRCA gravel. On other

TABLE 2 Impact categories according to Eco-indicator 99 and CML.

Eco-indicator 99 (damage-oriented)		**CML (baseline) (problem-oriented)**
Impact category	Sub-categories	Impact category
Damage to human health	Caused by carcinogenic substances	Depletion of abiotic resources
		Impacts of land use
	Caused by respiratory effects	Climate change
	Caused by climate change	Stratospheric ozone depletion
	Caused by ionizing radiation	Human toxicity
	Caused by ozone layer depletion	Ecotoxicity
Damage to ecosystem quality	Caused by ecotoxic substances	Photo-oxidant formation
	Caused by acidification and eutrophication by airborne emissions	Acidification
		Eutrophication
	Caused by land use	
Damage to resources	Caused by the depletion of minerals and fossil fuels	

environmental aspects, RAC (carbonated or not) are more damaging than NAC due to the use of additives in higher quantities, the recycling and accelerated carbonation installations and the longer transport distances in the studied case. To further improve the environmental impact CRCA is to increase the CRCA sand rate, which is only 40% in this case. In order to reduce transport distances, the case of prefabrication should be studied.

2.4 | Impact assessment

Based on the inventory of input and output flows conducted under the defined conditions, the emissions, and consumptions are aggregated and assembled to assess the environmental impact indicators. Characterization factors are defined for each emission or consumption and thus weigh the flows within each impact indicator.

Generally, there are two different types of impact assessment methods. The first one is called the damage-oriented approach where the category indicator is chosen at the endpoint of the environmental mechanism (also the top-down approach or "endpoints") and the second one is called the problem-oriented approach where the category indicator is chosen at an intermediate level somewhere along the mechanism—at midpoint (also the bottom-up approach or "midpoints"). The most representative examples of damage-oriented and problem-oriented are Eco-indicator 99 methodology, developed at Pré Consultants B.V., the Netherlands,[62] and CML methodology, developed at The Institute of Environmental Sciences (CML) of the Faculty of Science, Leiden University in the Netherlands,[63] respectively, Table 2.

Impact categories defined at the endpoint of the environmental mechanism which express, for instance, the damage to human health, ecosystem quality and resources, are much easier to comprehend than the rather abstract definitions of midpoints such as infrared radiation, UV-B radiation or proton release. However, the problem with this approach is that it is not easy to establish a clear relationship between the LCI results and damage categories. Besides in the "endpoints" approach, it is not possible to avoid normalization, grouping and weighting. Weighting is not a scientifically based operation, but it relies upon the opinion and attitude of experts towards different environmental effects. On the other hand, the relationship between midpoint category indicators and LCI results is easily established through appropriate, scientifically based, characterization models. That is why the "midpoints" approach is often used to quantify the results in the early stage of the cause-effect chain to limit the uncertainties.[64]

Other methodologies for impact assessment are available like TRACI (US Environmental Protection Agency's National Risk Management Research Laboratory), ReCiPe (Dutch National Institute for Public Health and Environment, Radboud University Nijmegen, Norwegian University of Science and Technology, Pré Sustainability), EDIP (Institute for Product Development at the Technical University of Denmark), IMPACT 2002+ (EPFL Laussane), and so forth. Some of them use the midpoint approach (TRACI, EDIP), while some combine both approaches (ReCiPe, IMPACT 2002+). Dreyer et al.[65] compared EDIP97 and CML2001 methodologies and found differences of up to two orders of magnitude for the impact categories describing toxicity to humans and ecosystems due to different characterization models. For the other impact categories, the two methods showed only minor differences.

If the use phase is excluded, the most significant emissions in the course of a concrete structure's life cycle originate from cement production and transport of constituent materials and concrete: GHG (carbon dioxide, methane, and nitrogen dioxide), sulfur dioxide, nitrogen oxide and non-methane volatile organic compounds (NMVOC). Relevant impact categories related to these emissions are climate change (GWP), acidification (AP), eutrophication (EP), and photochemical-oxidant creation (POCP).[66] Besides them, energy consumption and especially fossil fuel consumption are of interest when assessing concrete structures. Similarly, EN 15804 (problem-oriented approach) prescribes that environmental product declarations (EPD) for construction products shall contain information on abiotic depletion for nonfossil and fossil resources, total climate change declined in fossil fuel, biogenic, land use, and land use change (global warming), acidification, eutrophication declined in aquatic freshwater, aquatic marine, land, photochemical-oxidant creation, ozone depletion, and water need. Besides, EN 15804 assesses additional Environmental Impact Indicators (fine particle emissions, ionizing radiation, ecotoxicity, and human toxicity) and requires additional information on resource use, types and amounts of waste as well as amounts of reused and recycled components and materials.

2.5 | Uncertainties in LCA and how to deal with them

LCA is an inherently uncertain methodology due to many reasons. According to Huijbregts,[67] three types of uncertainty can be distinguished within LCA. The first one is parameter uncertainty caused by the imprecise, incomplete, outdated, or missing values of LCI data. The non-representative territorial scope between countries of European data enhances this uncertainty, especially concerning cement, aggregate, and concrete recycling processes. The second is model uncertainty, for instance, the adoption of linear models instead of nonlinear ones for environmental phenomena modeling. The third is scenario uncertainty due to choices such as, for instance, the choice of system boundaries (which upstream and downstream flows are included), methodological choices (attributional or consequential modeling), choice of FU, allocation approaches, assessment method (damage-oriented or problem-oriented) how to assess future technology trends, etc.

Uncertainty analysis is usually performed using Monte Carlo simulation which requires particular probability distributions of parameter values. It is a probabilistic model parameter uncertainty analysis where a predefined number of combinations (typically 10,000) of random parameters is used to calculate the results. However, the problem is how to estimate the probability distribution of involved parameters. The LCI data uncertainty, for instance, can be dealt with using a simplified approach which includes a qualitative assessment of data quality indicators based on a pedigree matrix.[68] Then, uncertainty factors are attributed to each of the quality indicators depending on their scores (from 1 to 5) and the square of geometric standard deviation can be calculated for the assumed probability function for each elementary flow. Such a procedure is applied for instance in the Ecoinvent database where uncertainty factors based on expert judgments are used.[69] The uncertainty estimations on the level of a unit process are then obtained using Monte Carlo simulation. However, it is stated in this Ecoinvent report[69] that "the deterministic results are regarded as more reliable than the probabilistic mean values because they are often based on roughly estimated distribution parameters," which have to be used for probability distribution definition and Monte Carlo simulations. Besides, there can be situations when the probabilistic approach does not help to rank, although it provides a "measure" of uncertainty (or degree of reliability). If the mean values are similar, a clear ranking between the alternatives can be identified if we consider them only. If the standard deviation in the form of 95% confidence intervals is taken into consideration as well, the ranking may become ambiguous due to the overlapping of these intervals.[18]

There are two uncertainty aspects specific to concrete structures assessment. The first one is connected to the ignorance of future operation and maintenance of the construction within its service life, future waste treatment technologies in the EoL phase, future utilization of recovered materials etc. Due to the very long life of concrete structures, we have to make certain assumptions which cannot be taken for granted since the distant future cannot be known (especially concerning second-generation RAC, concrete cannot be recycled ad infinitum). In a cradle-to-cradle type of analysis we "borrow" an environmental loan from future generations—the risk of accepting an environmental credit from future generations is taken deliberately.[70] Furthermore, the quantification of this credit in the case of long-lived constructions can be the source of large uncertainties. Sandin et al.[27] showed that assumptions in the EoL phase can significantly influence impact results which may hamper sound decisions regarding the sustainable future; particularly assumptions regarding ways of disposal, the expected technology development of disposal processes and any substituted technology and the choice between attributional and consequential approach. This uncertainty due to ignorance cannot be measured, it needs to be supposed.

The second one is introduced by the variability of constituents' content in the mix design of concrete. Mixture proportions, that is, amounts of component materials are essential for the impact assessment. This is especially important for the cement amount (clinker content) because it practically determines the main environmental impacts of concrete. The cement amount in the mix design is influenced not only by the strength but also by the workability requirement. For different target workability, the same compressive strength can be obtained with (very) different cement content, depending also on the amount of water-reducing admixture. The analysis performed in Reference 35 showed that compressive strength dependence on the concrete mix design introduced similar, maybe even larger uncertainty than that brought by the LCI environmental data uncertainty if Monte Carlo simulation is performed (for instance see Reference 71). This uncertainty due to variability could be sometime estimated by analysis of RCA or historical investigations.

Uncertainty analysis was rarely performed in concrete LCA studies but an increasing trend is observed in recent years. For instance, Mostert et al.[72] chose to perform the uncertainty analysis regarding the variability of the concrete constituents' content, energy consumption in the EoL phase, and transport distances. They found in their specific case study that CoV of the aggregated LCA results for raw material consumption, water use, cumulative energy demand, and GWP ranged between 1.8% and 8.8%, respectively, with higher uncertainty belonging to RAC compared to NAC structure. Hafez et al.[73] established several important sources of uncertainty in concrete LCA: upstream data regarding raw materials depending on the source used, the selected characterization factors in the midpoint approach, service life prediction, the energy use in forecasted technologies for maintenance, demolition, and EoL, CO_2 uptake model. They recommended quantifying the uncertainties by performing scenario analysis on each of the mentioned sources and then running a Monte Carlo simulation repeatedly to aggregate the uncertainty in the final LCA outcome (result).

Unlike uncertainty, which is the consequence of the lack of knowledge about the true value of a quantity, sensitivity is the influence that one parameter has on the value of another parameter. In previously published research sensitivity analysis on the main influencing parameters/choices were often performed. Impact assessment is performed for chosen parameters varied within "realistic" limits and the interval of impact result is obtained instead of one value. It should be kept in mind that the function of the structure must not be affected by these variations, otherwise a sensitivity analysis has no sense. The results of any sensitivity analysis, as well as the results of any LCA, depend strongly on the specific assumed scenarios and usually cannot be generalized.

The most influential parameters in RAC assessment are the cement (clinker) content which in RAC specifically is reflected through the amount and quality of RCA; transportation distances and types; and modeling approach—ALCA or CLCA.

Transport distances considered in the study context must be determined to be truly representative of the territory. If a long delivery distance for NA contrary to RCA is considered, many environmental impacts (energy use, global warming, eutrophication, acidification, photochemical oxidant) are dependent on the transport conditions (type and distances).[21] On the contrary, in some territorial cases where the transport distances for the RCA and NA remain low (under 40 km), even when RCA transport distances are longer than the ones for NA, the influence of transport distance and transport route are negligible or remains moderate.[30] In those cases, transport distance is not a major factor for environmental effects. Such territorial cases can occur when a ready-mix concrete plant is located in the proximity of natural alluvial aggregate quarry along river. Thus, territorial representativeness is a major factor to be determined.

Sensitivity analysis was most commonly used for testing the effect of transportation distances and types, specifically for the determination of the limit RCA transport distance above which the RAC scenario has no benefit over the NAC scenario. In this case, the results depend mainly on the replacement percentage of NA with RCA, modeling approach, and assumed NA transportation distance. For instance, Marinković et al.[74] obtained the RCA limit distance of 20 km (100% replacement of coarse NA with coarse RCA, ALCA, NA transport distance up to 150 km), whereas Turk et al.[24] calculated this limit distance to be 100 km (about 30% replacement percentage, CLCA with avoided burdens approach, NA transport distance 1 km).

As for the RCA amount, its impact on the environmental results depends on the cement content increase required to maintain the concrete performance similar to referent NAC. Braymand et al.[30] showed that the use of RCA in RAC increased environmental impacts at different levels when the increasing of RCA (0%, 30%, 100%) replacement was accompanied by higher cement content; when the RAC was formulated at same component content, the positive effects of RCA amount remain small (effect of 10% max for 100% replacement).

Colangelo et al.[75] concluded that the effect of the RCA amount (25%, 50%, and 100%) is small, but only RAC with 25% of RCA had lower impacts compared with a reference NAC. Mostert et al.[72] compared RAC with 43% and 100% replacement and obtained the lowest GWP for RAC with 100% RCA due to very small cement content increase compared to referent NAC (below 3%). The concretes compared in this work were designed for low aggressive environment leading to low durability requirements and therefore the effect of the RCA amount on the

TABLE 3 Impact categories per kilogram of aggregate.[19,76–78]

	NA (kg)[76]	RCA (kg)[77]	Crushed NA (kg)[19]	Rolled NA (kg)[19]	RCA (kg)[19]	NA (kg)[78]	Dry RCA (kg)[78]	Wet RCA (kg)[78]
Total non-renewable energy (MJ)	5.18E−02	6.49E−02	7.14E−02	7.73E−02	3.19E−02			
Global warming potential GWP (kg $CO_{2-eq.}$)	4.44E−03	4.73E−03	2.60E−03	2.75E−03	1.50E−03	1.43E−02	2.94E−02	3.81E−02
Acidification potential AP (kg SO_{2-eq}.)	2.93E−05	3.09E−05	1.47E−05	1.58E−05	9.60E−06	1.98−05	2.93E−05	4.13E−05
Eutrophication potential EP (kg PO_4^{3-}–eq.)	4.50E−06	5.65E−06	3.08E−06	5.71E−06	2.08E−06	3.67E−06	5.44E−06	6.33E−06
Photochemical ozone creation potential POCP (kg ethene–eq.)	2.06E−06	3.65E−06	7.05E−07	7.87E−07	5.04E−07	1.41E−05	1.25E−05	1.53E−05

cement content was small. Even in this scenario favorable for RAC, the GWP of RAC with 100% of coarse RCA was only 4% lower compared with referent NAC. It can be seen from last two examples (both cradle-to-gate ALCA with 100% allocation to RAC system) how conclusions depend on the assumed scenario. The quality of RCA also affects the environmental impacts since the RCA of lower quality requires a higher cement content increase; however, no such sensitivity analysis was found in the literature. The choice of modeling approach is perhaps the most important parameter. Within the ALCA framework, RCA with 100% replacement of coarse NA has a chance to show slightly lower impacts compared with a reference NAC if the cement content increase is below 5% and RCA transport distances are kept low. On the other hand, within avoided burdens approach framework chances are much higher. Knoeri et al.[23] showed that RAC environmental impacts were reduced to about 70% of the NAC impacts using the Ecoindicator 99 method. Turk et al.[24] obtained that AP, EP, and ADP of fossil fuels were reduced to 88% of the corresponding NAC impacts, while GWP was reduced to only 96% although avoided burdens approach was applied.

2.6 | Comparative assessment of RCA and NA

The results of the environmental assessment of RCA depend on the assumptions concerning system boundaries, that is, on the approach to LCI data modeling. As these hypotheses may differ from one study to another and depend on the territorial context, as discussed in the previous sections, it is difficult to present universally applicable conclusions on the LCA results of RCA and consequently on the comparisons between NA and RCA.

Moreover, not many studies focused on the LCA of aggregates alone, as they are rarely used alone. Literature mainly focuses on the use of aggregates (natural or recycled) in their application (concrete, road structures, ...). Therefore, assessment results are application-dependent, usually assuming transport and manufacturing scenarios depending on the type of usage of the aggregates. It is usual to take directly EPDs of aggregates (especially natural) using reference databases or manufacturers' EPDs for LCA studies on NAC and RAC.

Table 3 shows impact category indicators per kg of aggregate taken from Ecoinvent V2.0 database,[76] French Manufacturer database,[19] Josa et al. data used in a concrete LCA study,[77] and Won-Jun et al. data assessed in a specific Korean study.[78] In the case of the French EPD of RCA,[19] impacts due to the reduction of block size is excluded and allocated to the parent concrete, which leads to decreased impacts. Won-Jun et al.[78] assessed the influence of process production (dry/wet) of RCA on environmental impacts in comparison to production of NA using the ISO 14044 standard in the Korean context. The environmental impact of the wet process (RCA) was found to be up to 16% ∼ 40% higher compared with the dry process due to the energy used by impact crushers while producing wet RCA. The environmental impact of RCA was found to be up to twice as high as that of NA, largely due to assumed lower energy process used for the production of Korean natural aggregate issued from rivers. Dias et al. collected data (per kg of aggregate) from Portuguese companies indicate value ranges from 8.72E−04 to 2.45E−03 for GWP, and 9.98E−03 to 3.86E−02 for non-renewable energy concerning RCA, and 2.86E−02 for GWP and 4.01E−01 for non-renewable energy concerning NA.[79]

These results in terms of variability confirm the difficulty of comparing the assessment of NA and RCA independently of the territorial context, production processes, and LCA calculation assumptions.

2.7 | Results of the research on the environmental assessment of RAC and RAC structures and recommendations

The published research is relatively large in the last decade and it is mostly dedicated to the comparative environmental assessment of NAC and RAC. Several review articles on LCA of recycled aggregate concrete were published recently.[73,80–82] The authors dealt with basic methodological issues and the way these issues were resolved in the published research in this area. Based on the reviews performed by mentioned authors some conclusions can be drawn:

- cradle-to-gate analysis was often applied: 75%–87% of all studies, while only 13%–23% and 2–4% was cradle-to-grave and cradle-to-cradle analysis, respectively[73,81,82];
- FU based on unit volume was most frequently used: in 65%–74% of all studies; FU normalized with strength in 14%–25% and FU normalized with strength and service life in 10%–12% of analyzed studies[73,81];
- most of the studies applied an ALCA approach with some type of allocation considering recycling; if avoided burdens approach is used, it is applied within an ALCA framework;
- most of the published research relies on the midpoints method. Hafez et al.[73] for instance in their review study found that out of 107 papers only 6 used the endpoints approach. Hossein et al.[82] found that 84.4% of studies used midpoints, 6.2% endpoints and 9.4% both methods;
- CO_2 uptake during the use phase was rarely taken into account and in very different amounts: only seven out of the 107 references according to Reference 73 included the sequestered carbon in the LCA study;
- uncertainty analysis was also rarely applied: according to Hossein et al.[82] in only 2 out of 32 analyzed studies analysis regarding the inventory data uncertainty was performed; sensitivity analysis on the other hand is more common, especially regarding transportation distances (in 8 out of 32 studies in Reference 82).

On the methodological level, the results of the assessment mostly depend on FU and on the system boundaries—whether an attributional or consequential approach to inventory data modeling is chosen and whether the secondary life of concrete is included or not. On the material level, results significantly depend on mix proportions including replacement percentages of NA with RCA. Finally, LCI data and transportation data are geographically dependent. Review papers tried to recommend the "proper" way of performing the LCA in order to obtain more reliable results and decrease discrepancies. Generally, FU based on volume (for concrete), strength, and durability is recommended as well as the cradle-to-cradle type of analysis (read avoided burdens approach within ALCA framework including secondary life of recovered aggregate) and CO_2 uptake inclusion. Uncertainty analysis or sensitivity analysis in combination with uncertainty analysis is also advised.

At the structural level, these recommendations can however be discussed due to the reasons explained above. FU based only on material properties, even if it includes strength and service life, is not recommended in the assessment of structures. When comparing different structures, the best choice for FU is the whole structure. Cradle-to-grave and cradle-to-cradle analysis bring large uncertainties regarding the distant future operation, maintenance and waste treatment technologies, recovered materials utilization, as well as energy carriers and electricity mixes. Several fundamentally different scenarios are needed when modeling future EoL processes, especially if a consequential approach with avoided burdens is applied (having in mind that this approach within the ALCA framework is not justified methodologically and that results are questionable). CO_2 uptake during service life and eventually during stockpiling and secondary life should be taken into account only based on a proper prediction model. Finally, uncertainty analysis should include uncertainties due to different concrete mix design methodologies since they can have a larger impact on the LCA results than for instance, LCI data or scenario uncertainties.

Despite the discrepancies found in LCA studies dealing with the environmental assessment of concrete and concrete structures, the following conclusions are generally valid:

- the contribution of various phases to total life cycle impacts depends on the type of construction. For buildings, life cycle impacts are often dominated by energy consumption during the use phase: it is estimated that the use phase in conventional buildings represents ~80% to 94% (for a common house, not a passive house) of the life cycle energy use, while 6% to 20% is consumed in materials extraction, transportation and production, and less than 1% is consumed in EoL phase.[83] Blegnini[26] found that, when taking into account other environmental impacts (GWP, AP, EP, POCP), the contribution of the use phase was 93%. With the growing interest toward the development of energy-efficient buildings, the other life cycle phases are becoming more significant. Here, the material production and EoL phases are of special interest, since they are energy, resources, and waste intensive. For other types of construction works such as bridges or dams, the contribution of material production,

TABLE 4 Energy use and impact categories per m^3 of middle-strength concrete (assumed mix proportions given in table).

	CEM I	CEM II	Coarse aggregate (crushed) for equivalent volume		Fine aggregate (river)	Plasticizer	Concrete	Reinforcement
			NA (1100 kg)	RCA (925 kg)				
	350 kg	350 kg			800 kg	3.50 kg[a]	m^3	75 kg[b]
Total non-renewable energy (MJ)	1295.0	1064.0	56.98	60,03	21.52	109.9	38.4	1005.0
Global warming potential GWP (kg CO_2−eq.)	314.3	258.3	4.884	4,38	1.752	6.58	3.15	92.25
Acidification potential AP (kg SO_2−eq.)	0.518	0.424	0.0322	0.0286	0.0139	0.0102	0.0242	0.6322
Eutrophication potential EP (kg PO_4^{3-}−eq.)	0.0739	0.0609	0.0050	0.0052	0.0023	0.0036	0.0040	0.0314
Photochemical ozone creation potential POCP (kg ethene-eq.)	0.0497	0.0410	0.0023	0.0034	0.0013	0.0011	0.0022	0.0430

[a]Plasticizer mass—1% of cement mass (wt/wt).
[b]Reinforcement mass—3% of concrete mass (wt/wt).

construction, and EoL phases is more significant, and usually, the impacts from these phases exceed the impacts from the use phase.

- cement (clinker) is by far the largest contributor to all impact categories if the use phase (operation energy) is excluded. Table 4 shows non-renewable energy use and impact categories per m^3 of typical medium-range strength NAC and RAC including reinforcement (for RAC it was assumed that coarse NA is replaced with coarse RCA at equivalent volume). Impact category indicators per kg of concrete constituent materials were taken from European manufacturers' EPDs for cement, plasticizer, and reinforcement[84–87] or calculated using the Ecoinvent V2.0 database for aggregate and concrete production.[76] RCA data was taken from Josa et al.[77] It is evident that the contribution of other phases in the concrete production process is for an order of magnitude (and even more) smaller compared to the cement production contribution. Only comparable to it is the reinforcement production but with exception of GWP which is smaller in the reinforcement case. Transportation of constituent materials to the concrete plant was not taken into account since it largely depends on assumed distances (according to the territorial network) and types. Its contribution however should not be neglected in LCA studies because it is usually larger than aggregate and concrete production contribution.
- CLCA (understood as avoided burdens approach) applied within the ALCA framework is usually beneficial for RAC. The main reasons for improvements in the environmental behavior are avoided burdens: avoided waste landfilling and avoided iron production if iron scrap as a co-product of recycling is recovered. In attributional LCA studies where allocation is used instead avoided burdens approach, results are not so beneficial for RAC. With this approach, at best, for low cement increase in RAC, impacts of RAC and the corresponding NAC are similar, depending on the replacement ratio, mix design, and transport distances. Visintin et al. in their large-scale study[22] concluded that the use of RAC should be limited to concrete strength of 45 MPa or less. According to the authors, in that case, it is possible to produce RAC with the same strength and durability as NAC without any additional emissions via the optimization of mix design. Specially, according to Jiménez et al.[88] with the help of a method called "equivalent mortar volume method," when a mix proportioning of concrete with a lower cement content is obtained (thanks to admixture) used in RAC design, the RAC impacts are lower compared with those of the corresponding NAC, even with an attributional approach.

- If the replacement percentage of coarse NA with coarse RCA is up to 25%, this will not affect the basic properties of concrete (strength, workability) and functional performance of concrete structure but it will bring benefits through NA preservation and waste reduction. According to the Annual Review of the European Aggregates Association,[89] about 630 million tons of aggregates for the ready-mixed concrete industry were produced in 2016 in 28 European countries (EU-28). In the same year, about 350 million tons of mineral waste from CDW were generated in EU-28 countries.[90] Assuming that concrete waste makes up at least 40% of that amount,[1,91] ∼140 million tons of demolished concrete were generated in that year. Therefore, available sources for RCA are enough to replace about 20% of NA in ready-mixed concrete and to reduce the amount of landfilled concrete waste by 40%. This means that all of the demolished concrete waste can be used as a source of quality RCA for structural concrete application, without jeopardizing concrete structures' performance or the aggregate industry. The future availability of RCA depends on the ratio of CDW generation and concrete production increase rate.

- When admixtures are used to reach the same mechanical and durability-related properties without increasing cement content, several impacts (GWP, AP, ADP, POCP, EP, ODP) became smaller than those of referent NAC, but the eutrophication potential (EP) increased slightly.[92]
- Using RCA as a carbon sink instead of aggregate in new concrete can significantly increase the CO_2 uptake during secondary life. Visintin et al.[22] have calculated that if concrete waste is crushed to the size of aggregate and buried for another 100 years (for instance if it is used for road base or geotechnical fill), the maximum theoretical limit of CO_2 absorption can be reached: 40%–55% of the emissions associated with manufacture. Using crushed concrete waste as aggregate in new concrete reduced this offset to only 31%.

3 | SOCIAL AND ECONOMIC ASSESSMENT

While the assessment of RAC structures in environmental terms has long and extensively been examined in the literature, the analysis of the socioeconomic impacts related to their different lifecycle stages is still in its infancy.

LCA can also be used for the sake of social and economic impacts, even though the most well-known and extended tool can be considered to be environmental LCA. On the one hand, social LCA (S-LCA) focuses on the social aspects of products and services, including both actual and potential positive and negative impacts during their life cycle. On the other hand, the economic LCA (referred to as Life-Cycle Costing, LCC) focuses on economic impacts throughout the life cycle of products and services. More details on these tools as well as the impact categories and corresponding indicators that can be used for social and economic assessment will be described in the following subsections.

3.1 | Choice of social impact categories and indicators

There are several tools that can be used to make a choice on social impact categories and indicators, including rating tools, social LCA, or individual indicators.

TABLE 5 Summary of social categories and sub-categories in sustainability rating tools for buildings.

Category	Issues
Accessibility	Accessible public services and amenities
	Accessible public transport
	Accessible pedestrian network
	Accessible bicycling network
	Alternative transport modes
Communication	Building management
	Building design
Occupant wellbeing	Building user comfort
	Health and safety
	Spatial access
Security	Designing out crime
Social and cultural value	Social and ethical responsibility
	Sensitivity to the local community
	Building aesthetics and context

First, the social impact categories and indicators may be obtained from rating tools aimed at assessing buildings' impacts. A summary of social aspects that are considered in these tools can be found in Table 5. The table includes categories and issues, even though it needs to be noted that there is a third level, sub-issues, which is more specific.

As it can be seen, even though there are multiple categories and indicators regarding social impacts, their application in the context of recycled aggregate concrete structures is not straightforward. Some sub-issues that could have a direct relationship with the use of recycled aggregates are the toxicity of finishing, thermal comfort, or acoustic comfort, which belong to the category of occupant wellbeing, and which might be affected by the material used in the construction.

Second, as it was mentioned above, the framework of S-LCA may also be used in order to determine what social impacts should be assessed. S-LCA differs from environmental LCA in the fact that it not only considers impact categories but also stakeholder groups. The five main stakeholder groups that are proposed by UNEP's guidelines[93] are workers, consumers, local community, society and value chain actors.

To the best of the authors' knowledge, there are no studies yet in the literature dealing with the development of an S-LCA for the case of RAC structures. Studies[94,95] did use life-cycle approaches to analyze the sustainability of recycled aggregates, but none of them carried out a complete S-LCA.

Sou et al.[94] assessed the sustainability of using recycled bottom ash as construction material, which included analyzing its social and legal implications. They did not use the LCA methodology, but fuzzy set theory, and evaluated two indicators, namely social acceptability and legal feasibility. Shi et al.[95] mentioned the assessment of social impacts in their study on the LCA of concrete pavement containing RCA. They used the TRACI category indicators,[96,97] which include social impacts through indicators of human health particulate air, human health cancer, and human health non-cancer.

Aruakala et al.[98] analyzed the sustainability of alternatives of coarse aggregates, including recycled options. They chose several indicators, including several social ones: effects on human health and safety and human satisfaction.

Finally, other individual indicators that have been developed until the moment are the Occupational Risk Index (ORI index), which is used to measure the health and safety of workers and occupants, or the third-party effects, measured through building site space. An example of the application of these indicators in the context of recycled aggregate can be found in Reference 77.

3.2 | Assessment of social impact categories and indicators

A complexity inherent to the nature of social impact indicators is the data collection methodologies. Social indicators tend to be more subjective, and fewer data are usually available at large scales. The assessment of social indicators may be done quantitatively or qualitatively.

In the case of using quantitative indicators, data can be obtained from open databases from governments or results of surveys distributed to representative samples.

In the case of qualitative indicators, these may be collected through interviews, small-scale surveys, or expert seminars. Usually, they are represented in the form of scales, such as the Likert scale. For instance, Sou et al.[94] evaluated the indicators using questionnaires distributed among engineering consultants, government officials, and academic institutions. The questionnaires were based on five-point scales. The results of these questionnaires were then analyzed using the fuzzy set theory. A questionnaire was also used by Arukala et al.[98] to assign values to the Preference Selection Index (PSI) of different alternatives to ultimately choose a sustainable material.

Very commonly, the choice of the type of indicator is not to be made by the researcher but is a constraint of the data available.

TABLE 6 Summary of economic categories and sub-categories in sustainability rating tools for buildings.

Category	Issues
Financing and management	Value management—function analysis
	Value management—risk and value management
Whole life value	Whole life costs
	Asset value
	Maintenance
Externalities	Local and regional impacts
	Image value

3.3 | Choice of economic impact categories and indicators

Regarding the choice of economic impact categories and indicators, one can also resort to rating tools aimed at assessing buildings' impacts. A summary of such categories can be found in Table 6. A differential factor of economic impacts with respect to social and environmental ones is that there is less diversity in indicators, and they tend to be more homogeneous among different products and services.

In addition to rating tools, life cycle costing (LCC) also represents a means of measuring the economic sustainability of products and services. LCC considers both costs that are related to the life-cycle of a product, as well as those economic impacts related to externalities (see Table 7). Such externalities are usually regarded as negative costs because they are considered as impacts from factors such as emissions. Some recommendations on how to calculate externalities can be found in Reference 99. LCC is commonly carried out in compliance with international standard ISO 14040:2006.[6] As it happened with S-LCA, LCC is not yet as systematized as the environmental LCA.

Related to the issue of externalities, Santero et al.[101] discussed the role of economics in the context of GHG emissions and concrete. They also used a method called cost-effectiveness analysis to calculate the cost to reduce GHG emissions in the sector.

There are a few examples of LCCs in the context of concrete in the literature. For instance,[100] performed an LCC of lightweight artificial aggregates from industrial waste. Mah et al.[102] carried out an LCC of concrete waste management alternatives.

Finally, other approaches have been used by researchers to measure economic sustainability.[95] used the economic input–output life cycle assessment (EIO-LCA) to analyze the impacts of using concrete

TABLE 7 Cost items that may be included in LCC.

Cost item	Examples
Direct production costs	Raw materials
	Direct labour
Indirect costs	Overheads
	Indirect labour
	Waste treatment
	Pollutant abatement costs
Externalities	Costs related to CO_2 emissions

Source: Adapted from Reference 100.

pavement with recycled concrete aggregate. EIO-LCA uses economic inputs to estimate the materials and energy resources that are required for activities in the economy, as well as the environmental emissions resulting from them.

The cost–benefit analysis is also a well-known method for economic impact assessment. It was used by Reference 94 to analyze the sustainability of bottom ash generated with construction waste. More simplified cost analysis can be found in Reference 103 to analyze the cost of recycled asphalt concrete mixtures or in Reference 104, where the costs of 3D printed buildings with recycled concrete were assessed.

3.4 | Assessment of economic impact categories and indicators

Different to social indicators, economic indicators are seldom assessed qualitatively. Data on costs are commonly available either in construction databases or through direct inquiry with specific companies.

For instance,[103] used data from the Department of Statistics of Malaysia, and[77] used data from a Spanish costs database.

4 | SUSTAINABILITY ASSESSMENT

Having seen the existing methods for assessing environmental, social, and economic impacts, this last section describes how all these impacts can be grouped together in order to make a more holistic assessment. Given that this is a timely and relevant topic, there are several reviews in the literature on the different issues involved in sustainability assessment. Therefore, only a brief comment on this topic will be made in this article.

Methods to do such holistic analyses could be grouped into three main types. First, one could perform a life cycle sustainability analysis (LCSA), which would involve the evaluation of all environmental, social, and economic impacts throughout the life cycle of a product or service. Namely, the environmental, social, and economic LCAs would be carried out separately and then conclusions would be drawn from the results.

Second, multi-criteria decision-making methods (MCDM) can be used as a means of simplifying the number of indicators used and aggregating them into a single index.

Third, rating tools recognize and reward organizations that build and manage sustainable services or products (usually buildings). Some of the categories and indicators that these tools contemplate may also be used as a basis to assess the sustainability of recycled concrete aggregate.

These methods are described in more detail next.

4.1 | Life cycle sustainability analysis

Costa et al.[105] reviewed the literature on life cycle sustainability assessment. They considered LCSA as the sum of LCA, S-LCA, and LCC. They found that there is a lack of harmonization of the methodology, which has become a central challenge to its operationalization.[106,107] While their review encompassed studies in different fields, Backes and Traverso[108] did a review on LCSA studies conducted in the construction sector.

While LCSA provides a very thorough method to assess sustainability, it may also be resource-consuming. Therefore, studies that incorporate the three LCAs, environmental, social, and economic, are very scarce.

4.2 | Multi-criteria decision-making

MCDM methods are also tools that have been used to assess sustainability in different contexts, including concrete structures with recycled aggregate. The best well-known methods include TOPSIS, PROMETHEE, ELECTRE, VIKOR, and MIVES.

The examples of the application of MCDM to recycled aggregate concrete that have been found are those by References 77,109. On the one hand, Josa et al.[77] assessed the sustainability of continuous flight auger piles and, among the alternatives that they defined, they included piles both with natural crushed and recycled aggregates. In their study, Josa et al.[77] used MIVES. On the other hand, Revilla-Cuesta et al.[109] assessed the sustainability of various mixes of self-compacting concrete including different amounts of recycled concrete aggregate. For the assessment, methods TOPSIS, AHP, and PROMETHEE were utilized.

Other studies that fall into similar areas are the following ones:

- Petrillo et al.,[100] who used a multi-criteria model to select lightweight artificial aggregates from industrial waste.
- Hafez et al.,[110] who reviewed the literature to analyze the influence of using fine recycled aggregated on the technical, environmental and economic performance of concrete. One of their main conclusions was that the sustainability of using recycled fine aggregates largely depends on the transportation distance.

4.3 | Rating tools

Finally, in addition to the previous two methods, there are rating tools, which are used to assess and recognize those products that meet certain requirements or standards. While there are multiple tools that have been developed until the present, in the context of this chapter, the most relevant ones are those that assess the sustainability of buildings. Some of the most well-known tools are BREEAM, CASBEE, SB Tool or LEED. These environmental quality certifications consider the use of RCA in the concrete as a criterion of allocation of points often flooded in more global criteria. As a consequence, the relative weight of recycling becomes relatively weak. The introduction of more explicit subcriteria like that implemented in Minergie-Eco (Switzerland) is a way of improvement. This certification comprises an exclusion criterion for applicants who do not comply with the concrete recycling provision except if the distance between the concrete plant and the site exceeds 25 km.[111]

5 | CONCLUSIONS

The sustainability assessment of recycled aggregate structures should consist of environmental, social, and economic assessments. While environmental assessment has a two-decade-long tradition, social assessment was much less investigated probably due to the inherent complexity of the nature of different possible social impacts. On the other hand, some type of economic assessment was always applied in the evaluation of construction projects and introducing the Life Cycle Costing should present the advancement of existing practices.

The basis for the assessment in all three cases should be LCA. However, common referring to LCA methodology is not precise enough since ISO standards provide a general framework and leave plenty of room for interpretation. In the sustainability assessment of RAC structures special attention should be paid to:

- choice of the assessment approach: If CLCA is applied, marginal data on many processes and many assumptions on relevant markets, alternative products, price elasticities, distant future operations, and so forth are needed. It should not be applied within the ALCA framework. If ALCA with some type of allocation is applied, which is recommended at the RAC structures' assessment level as simpler and less uncertain than CLCA, indicators describing sand and stone depletion and land use should be included;
- choice of FU: it should be based on the function of the structure (strength, serviceability and durability) and in that sense the best option for FU is a structure as a whole; FU based only on the material properties can be used for generic comparisons in order to evaluate the sustainability potential for material substitution;
- CO_2 uptake during the RAC life cycle should be included in the LCI; the prediction model by Huang et al.[54] for the production and service phase can be recommended although probably unconservative for the production phase part; large-scale field measurements are needed to improve the reliability of predictions;
- both "endpoints" and 'midpoints' approaches in the impact assessment are applicable;
- uncertainty analysis is recommended: preferably, it should include LCI data, concrete mix design, the CO_2 uptake prediction model (if used), and distant future operations uncertainties; at least, sensitivity analysis on the most influential parameters is advised;
- the application of S-LCA in the context of RAC structures is extremely scarce at present. Some methodological challenges (e.g., the selection of the impact categories, the definition of the FU) hinder the development of such applications;
- similar to the case of S-LCA, thorough applications of LCC in the context of RAC structures are not abundant. While there are a few economic analyses where costs are evaluated, a Life Cycle Thinking approach is not always adopted (i.e., other approaches, such as Cost Benefit Analysis, are more common in this context).

The majority of environmental impacts for both RAC and NAC originates from the cement (clinker) production and this is especially valid for GWP; the replacement of NA with RCA cannot significantly affect the CO_2 emissions amount. At the same time, using RCA as a carbon sink can significantly increase the CO_2 uptake during the secondary life. The wise utilization of RCA is therefore recommended. The best environmental results are probably obtained by combining two applications: first, use RCA as carbon sink for as long as possible, then apply it as aggregate in new concrete. The unbound applications are also recommendable since they provide the prolonged CO_2 uptake. Regardless of the future applications, it is important to crush the demolished concrete waste and separate it into various RCA fractions—the waste management practices should be adapted to maximize the CO_2 absorption.

CEN has issued several standards regarding environmental assessment (EN 15804, EN 16757 and EN 15978 for buildings, and EN 17472 for civil engineering works) and several standards regarding social and economic assessment (EN 16309-A1 and EN 16627 concerning buildings, and EN 17472 for civil engineering works). These standards may serve as guidelines for the sustainability assessment of concrete structures while working on their further development and resolving currently existing flaws and limitations.

Despite all the effort put into the development of sustainability assessment methods and procedures, we are still far away from implementing the "design for sustainability" in the concrete structural codes. The *fib* Model Code for Concrete Structures[112] for instance introduced sustainability into the design of concrete structures as a performance criterion along with structural safety and serviceability. The Code however offers only the general procedure for the verification of the environmental and social aspects of sustainability. For the practical application in the design of concrete structures, specific evaluation and verification methods must be defined. Within the performance-based and limit states framework, the verification method includes the target reliability level definition, the safety format choice, and performance requirements and criteria definition. Future research in the area of the sustainability of all types of concrete structures should be focused on the standardization of the evaluation and verification methods for all three aspects of sustainability.

DATA AVAILABILITY STATEMENT

Data sharing not applicable to this article as no datasets were generated or analysed during the current study.

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