Defra Environment Bill Targets
Rapid Evidence Assessments

Environmental Life Cycle Impacts of Inert / Less Reactive Waste Materials

A report prepared for Defra

By Brunel University London

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EXECUTIVE SUMMARY

This report presents the results of a rapid evidence assessment of the environmental life cycle impacts of inert/less reactive materials that might typically be expected to have minimal impact on the environment, to generate insights on where environmental improvements might be needed in England. From a material standpoint, according to the European Waste Catalogue codes (Commission Decision 2001/118/EC, 2020), inert/less reactive waste materials arise primarily from activities under Category 17 (Construction and demolition), Category 19 (Waste management), Category 10 (Thermal processes), and Category 01 (Exploration, mining, quarrying, physical and chemical treatment of minerals); justifying a construction sector focus.

Identifying ways to improve resource efficiency and productivity in the construction sector is critical, as the UK Government is set to achieve net-zero greenhouse gases (GHG) emissions by 2050. Therefore, the report focuses on understanding which of the materials used in the construction sector have the greatest environmental impact across their full life cycle, such that it can allow Defra to understand which interventions may lead to the greatest environmental improvements. A systems approach was employed to look at materials from their extraction and processing to their end-of-life (EoL) management. The study adopted a bottom-up and top-down method of data collection and analysis, and proposed the use of a novel scoring system to present evidence on materials environmental life cycle impacts. To ensure the reliability and replicability key findings the study used the environmental impact indicators used in Life Cycle Assessment analysis (LCA), and thereby, collected evidence only from LCA studies.

The main findings of the study are summarised below:

- The terms recovery and recycling are used in an elusive, ambiguous way, making unclear the way in which inert/less reactive waste materials are being managed. In some studies recovery is believed to refer to backfilling, while in other studies recovery can refer to anything from preparation to reuse, recycling and backfilling. This ambiguity is further aggravated by the legal framework, in which ‘recovery’ means anything from preparation for reuse, recycling, backfilling to energy recovery, whereas ‘material recovery’ includes all of the above except energy recovery. Lack of clarity with respect to the waste management options obscures our ability to see where waste materials end up, which in turn, may result to inefficiencies in the way inert/less reactive waste materials are being managed. This can ultimately prevent efforts to promote resource efficiency and productivity in the construction system.
- The environmental impacts of quarrying / dredging activities are hugely underexplored, which disguises the externalities caused at this stage. Some suggest that the embodied carbon emission measures include carbon emissions at this stage; however, it is unclear what is actually included

in these measures. Additionally, quarry waste and fines waste generation and management at this stage are overlooked. Limited insights suggest that the in-situ management of quarry and fines wastes may present considerable environmental threats over the long-term.

- Generally, there is an increased focus on concrete and clay bricks production, due to the prevalence of these materials in the UK construction sector, which places little attention on other materials. While, the production of cement (calcination) and the firing of clay contribute the most to the environmental impacts of concrete and clay bricks full life cycle, this is based on skewed data, as the environmental impacts arising from limestone and clay quarrying remain unclear. Moreover, efforts to reduce the environmental impacts of concrete and bricks focus on the use of substitutes and/or alternative materials. The environmental savings of these practices could lead to knock-on effects on the technical performance of materials, which, in turn, could result to net negative environmental impacts over the materials' lifecycle. Most importantly, it misplaces efforts to reducing construction materials input in the system, and improving their management at the end-of-life stage.
- Information on the fate of excavated soil, composed of topsoil, subsoil and spoil is scant. The prevalence of soil waste management options remains unclear, creating a blind-spot in the system. Soil is not always inert, and its mismanagement (e.g., improper disposal to landfill sites, or land) could ultimately lead to important environmental impacts if left unexplored.
- Construction and demolition waste (CDW) are a highly heterogenous stream composed of a large amount of inert/less reactive waste materials, often mixed with reactive and potentially hazardous materials. This makes the EoL management of inert/less reactive materials a challenging task, which might be the reason information at this stage lacks transparency. There is a multitude of parameters that affect the environmental performance of inert / less reactive materials at their EoL stage, which must be considered when assessing the environmental performance of inert/less reactive waste materials. Evidently recycling and backfilling can offer environmental savings compared to landfilling, but studies on structural components reuse suggest that this option contributes the least to environmental impacts.

Granularity over the material type and properties, and environmental assessment of their full life cycle can help to uncover opportunities and inefficiencies in the system. To be able to see the big picture, environmental analyses should be complemented by the economic, social, and technical performance of materials, to highlight inefficiencies and blind-spots in the system. Such an integrated, holistic analysis of the construction full value chain, can ultimately facilitate an improved and sound decision-making process that supports the development of sustainable, zero-carbon management strategies.

CONTENTS

LIST OF ABBREVIATIONS

ADP	Abiotic Depletion Potential
AP	Acidification Potential
BAT	Best Available Techniques
Cd	Cadmium
CD&E	Construction, Demolition and Excavation
CDW	Construction and demolition waste
CFC (also R11)	Chlorofluorocarbon
CH_4	Methane
CO	Carbon monoxide
CO_2^-eq.	Carbon dioxide equivalent
DCB	Dichlorobenzene
DQI	Data Quality Indicator
EoL	End of Life
EP	Eutrophication Potential
ETP	Ecotoxicity Potential
EWC	European Waste Catalogue
GGBS	Ground granulated blast-furnace slag
GHG	Greenhouse Gases
GWP	Global Warming Potential
HCFCs	Hydrochlorofluorocarbons
HTP	Human Toxicity Potential
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LO	Land Occupation
LoW	List of Waste
MgO	Reactive magnesia
MJ	Megajoule
MMP	Materials Management Plan
Mt	Million tonnes
NO_X	Nitrogen oxides
ODP	Ozone Depletion Potential
PC	Portland Cement

PFA	Pulverised Fly Ash
Pb	Lead
PM	Particulate Matter
POCP	Photochemical Ozone Creation Potential
RSD	Relative Standard Deviation
SCC	Self-consolidating concrete
SO_2	Sulphur dioxide
w/w	weight for weight

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1. INTRODUCTION

Excavation and construction activities, and the demolition of existing and/or derelict structures generate an enormous amount of soil and rock, and construction and demolition waste. In 2016, *construction, demolition and excavation* (**CD&E**) waste accounted for ca. 62% by weight (**w/w**) of the total solid waste generated in the UK. The largest waste material category in CD&E waste is *mineral waste* (63.4 out of 136.2 million tonnes of CD&E waste in the UK), which typically consists of aggregates, bricks, stone, and asphalt. In 2016, mineral waste comprised 81.1 million tonnes across all waste streams (ca. 36.7% w/w of total UK waste) and *soils* comprised 58.7 million tonnes (ca. 26.5% w/w of total UK waste) (DEFRA, 2020). CD&E waste is not just a voluminous waste stream; it poses several environmental, social, and economic impacts, largely due to the lack of a coherent framework for the utilization of these wastes.

A large amount of the soil and mineral fraction of the CD&E waste can be treated via backfilling, i.e., a recovery operation where wastes are used for engineering purposes in landscape (e.g. slope reclamation) or in excavated areas (e.g. underground mines and gravel pits) replacing natural aggregates and soil. In 2016 in the UK17 million tonnes of waste were treated via backfilling operations, of which ca. 89% consisted of *soils*, and 4% of *mineral wastes*. Out of the 52 million tonnes of waste sent to landfill in the UK in 2016, ca. 55% of this tonnage consisted of *soils*, and 5% of *mineral wastes*. Of the 121 million tonnes of waste sent to recycling and/or other recovery operations (excluding backfilling and energy from waste incineration), ca. 12% of this tonnage was made up of soils, and 55% by mineral wastes (DEFRA, 2020). Whilst it is important to account for the environmental impacts of CD&E waste at the end-of-life (**EoL**) stage, it is commonly acknowledged that "*at least 70% of the environmental impact of an average construction material is attributed to the energy required for its production (Kay and Essex, 2009) (a notable exception being concrete, where 60% of emissions are associated with decarbonation of limestone)*" (Iacovidou and Purnell, 2016).

With pressures from the UK Government to achieve net-zero greenhouse gases (**GHG**) emissions by 2050 (i.e. at least a 100% reduction in emissions from 1990), urgent action is needed across all sectors to accelerate reductions in their carbon footprint. The construction sector is at the forefront, being the most material and energy intensive sector, and therefore decisive action is needed to accelerate the pace of innovation and find ways to enable the sector become resource efficient and productive. This can be a challenging task, not only because the construction sector operates based on a conservative model hampered by the heterogeneous landscape of stakeholders across the value chain, but also because it requires an in-depth understanding of where interventions are needed to help the sector move towards the right direction.

A systematic and systemic assessment of the sector's current practices could illuminate where inefficiencies occur in the system and help develop policies and management strategies able to deliver circular economy solutions to the construction sector. An area most important in helping the construction sector become more resource efficient, and achieve zero-carbon emissions, is introducing interventions in materials use across the entire construction value chain. This involves the way materials are produced and used - upstream of the construction value chain - and the generation of CD&E waste - downstream of the construction sector. Gaining a good insight in the life cycle environmental impact analysis of materials produced, used and managed across the sector, can highlight where changes are needed for environmental improvement, contributing to a reduction in the GHG emissions, and maximising reuse and recycling by both improving the quality of secondary materials and optimising the efficiency of waste treatment methods (Gálvez-Martos et al., 2018).

The aim of this report is to carry out a rapid evidence assessment of environmental life cycle impacts of materials used in the construction sector, and gain an insight on which of these materials have the greatest environmental impact across their full life cycle. Emphasis is needed particularly on the inert/less reactive materials - from extraction and processing, to manufacture, use and EoL treatment – in order to identify strengths and limitations associated with their production, use and management, and illuminate gaps in the research literature. Inert / less reactive materials are those that are neither chemically nor biologically reactive and will not decompose, e.g., sand. The analysis, therefore includes the following aspects: (1) synthesis of information on the environmental impacts of selected materials (e.g. concrete, bricks, tiles) across the various stages of their life cycle collating different datasets; (2) depiction of the processes and factors that come into play during the extraction, processing, manufacture, use (incl. service life) and end-of-life treatment; (3) collection and illustration of key environmental impacts at each stage of the materials life cycle; (4) identification of key blind spots and making recommendations for resource efficiency in the construction sector.

In **Section 2**, we provide the list of inert / less reactive materials as found in the European Waste Catalogue (**EWC**) and the UK Waste Classification Technical Guide. **Section 3** outlines our methodology on collecting and analysis the evidence from the global literature on the environmental life cycle impacts of inert / less reactive materials. **Section 4** describes our analytical approach and provides clarifications to definitions used in the government documents as well as on the global literature. **Section 5** presents our result on the environmental life cycle impacts of inert / less reactive materials, and **Section 6** our proposed approach to comparing different materials across their life cycle and against key environmental impacts. Finally, in **Section 7** we present our key conclusions and recommendations for further research.

2. LIST OF INERT/LESS REACTIVE WASTE MATERIALS

The List of Waste (**LoW**) as specified by the amended Commission Decision 2001/118/EC (Commission Decision 2001/118/EC, 2020) is a catalogue of all wastes divided into 20 chapters and can be used to indicate if a waste is hazardous waste. It is a legal waste classification system and provides a considerable and detailed list of wastes classified by their type (e.g. 15 *Waste packaging, absorbents, wiping cloths, filter materials and protective clothing not otherwise specified*), or the industrial process or business activity at which they are produced (e.g. 10 *Wastes from thermal processes*).

To accurately identify all inert / less reactive materials (and thereby wastes) produced in the UK economy we considered the entire LoW, rather than focussing on a single process chapter (e.g. 17 *Construction and Demolition Wastes (Including Excavated Soil from Contaminated Sites)*). This is because inert and less reactive waste materials can be generated by a range of business activities and sectors. In **Table 2-1**, a list of the most prevalent waste component categories of inert and less reactive waste materials is presented according to the European Waste Catalogue codes (Commission Decision 2001/118/EC, 2020), used in the UK Waste Classification Technical Guide.

Table 2-1 Types of inert and less reactive materials ending up as waste according to the EWC by the amended Commission Decision 2001/118/EC (Commission Decision 2001/118/EC, 2020)

Category	*Source*	*Waste category*	*EWC code*
Bricks	Thermal processes	wastes from manufacture of ceramic goods, bricks, tiles and construction products	10 12 08
	Construction and demolition	concrete, bricks, tiles and ceramics	17 02 02 17 01 07
Ceramics	Thermal processes	wastes from manufacture of ceramic goods, bricks, tiles and construction products	10 12 08
	Construction and demolition	concrete, bricks, tiles and ceramics	17 01 03 17 01 07
Concrete	Thermal processes	wastes from manufacture of cement, lime and plaster and articles and products	10 13 14
	Construction and demolition	concrete, bricks, tiles and ceramics	17 01 01 17 01 06
Fines	Waste management operations	soil, dust, plastic fragments, glass, ferrous, organic, lithoid, ferrous,	Not specified

	Household, commercial and industrial waste production	and other fragments	
Glass	Thermal processes	wastes from manufacture of glass and glass products	10 11 03 10 11 12
	Packaging		15 01 07
	Construction and demolition	wood, glass and plastic	17 02 02 17 02 04
	Waste management facilities	mechanical treatment of waste (sorting, crushing, compacting, pelletising)	19 12 05
	Household, commercial and industrial waste production	separately collected fractions	20 01 02
Gravel	Exploration, mining, quarrying, physical and chemical treatment of minerals	wastes from physical and chemical processing of non-metalliferous minerals	01 04 08
Gypsum	Construction and demolition	gypsum-based construction material	17 08 02
Minerals	Exploration, mining, quarrying, physical and chemical treatment of minerals	wastes from mineral excavation	01 01 0102
	Waste management facilities	mechanical treatment of waste (sorting, crushing, compacting, pelletising)	19 12 09
Soil	Agriculture	wastes from sugar processing	02 04 01
	Construction and demolition	soil (including excavated soil from contaminated sites), stones and dredging spoil	17 05 04 17 05 06 17 05 08
	Waste management facilities	wastes from soil and groundwater remediation	19 13 02
	Household, commercial and industrial waste production	garden and park waste	20 02 02
Steel	Construction and demolition	metals (including their alloys)	17 04 05
	Waste management facilities	wastes from shredding of metal-containing wastes	19 10 01
Stones/ rocks	Exploration, mining, quarrying, physical and chemical treatment of minerals	wastes from physical and chemical processing of non-metalliferous minerals	01 04 08 01 04 13
	Construction and demolition	soil (including excavated soil from contaminated sites), stones and dredging spoil	17 05 04

	Waste management facilities	mechanical treatment of waste (sorting, crushing, compacting, pelletising)	19 12 09
	Household, commercial and industrial waste production	garden and park waste	20 02 02
Tiles	Thermal processes	wastes from manufacture of ceramic goods, bricks, tiles and construction products	10 12 08
	Construction and demolition	concrete, bricks, tiles and ceramics	17 01 03 17 01 07
Sand	Exploration, mining, quarrying, physical and chemical treatment of minerals	wastes from physical and chemical processing of non-metalliferous minerals	01 04 09
	Thermal processes	wastes from power stations and other combustion plants (except 19)	10 01 24
	Waste management facilities	wastes from incineration or pyrolysis of waste	19 01 19
	Waste management facilities	mechanical treatment of waste (sorting, crushing, compacting, pelletising)	19 12 09

Table 2-1 gives very quickly an insight on which economic activity, and hence sector, each of the resulting wastes are generated and/or assigned to. This implicitly provides also an insight into the sources and potential management (also referred to herein as ‘disposal’) pathways. In numerical order the codes provided in **Table 2-1** denote that the inert / less reactive wastes identified are classified the following categories:

– 01 Wastes Resulting from Exploration, Mining, Quarrying, and Physical and Chemical Treatment of Minerals
– 02 Wastes from Agriculture, Horticulture, Aquaculture, Forestry, Hunting and Fishing, Food Preparation and Processing
– 10 Wastes from Thermal Processes
– 15 Waste Packaging, Absorbents, Wiping Cloths, Filter Materials and Protective Clothing not Otherwise Specified
– 17 Construction and Demolition Wastes (Including Excavated Soil from Contaminated Sites)
– 19 Wastes from Waste Management Facilities, Off-Site Waste Water Treatment Plants and the Preparation of Water Intended for Human Consumption and Water for Industrial Use

- 20 Municipal Wastes (Household Waste and Similar Commercial, Industrial and Institutional Wastes) Including Separately Collected Fractions

As shown in **Table 2-1,** the majority of inert waste materials arise during construction, demolition and excavation activities, which justifies our focus on the construction sector, i.e. Category 17, and their management, i.e. Category 19. Mineral waste and waste gravel and sand are considered to be indirectly associated with construction activities, and as a result, wastes produced by these activities are also considered, i.e. Category 01. Thermal processes are an integral part of the processing of materials used in the construction sector, and therefore inert waste listed in **Table 2-1** under Category 10 are also considered in our analysis. Wastes that fall under Categories 02, 15 and 20 are excluded from our analysis, as well as the steel and glass due to their nature, or variety of end-uses.

3. METHODOLOGY

We carried out a literature review to identify Life Cycle Assessment (**LCA**) studies of the inert materials presented in **Table 2-1.** Considering the prevalence of CD&E waste in total waste arisings in the UK, the inert materials under consideration in this report meant a construction sector focus. The scope of our research was to gather all evidence on the environmental impacts of inert materials across the various stages of their life cycle – from extraction and processing, to manufacture, transportation, use, and EoL management. The literature search focused on recent research, particularly between 2015 and 2021 (as agreed with Defra), although earlier studies were also included to expand our analysis and support our arguments when the availability of information in the identified studies within the selected review period was scarce. Due to the high variability of materials used in the construction sector, at the stage of raw material extraction we looked at quarry and dredging activities; at the stage of production we focused on physical and thermal processes of aggregate materials; at the on-site construction / use stage we looked at structures built in the UK, and at the management stage we focused on the management of CD&E waste.

Under this strategy, we developed two main research objectives: 1) quantify the environmental impacts of prevalent inert construction and excavation/quarrying materials across their entire life cycle; 2) quantify the environmental impacts of CD&E waste under different EoL management pathways as well as their contribution to full life cycle of materials evaluated upstream. Several combinations of keywords were searched in the most popular scientific databases such as Scopus, Google Scholar and Web of Science, grouped into: i) material context, such as “minerals”, “excavated soil”, “topsoil”, “gravel”, “stone”, “rocks”, “fines”, “clay”, “sand”, “bricks”, “concrete”, “aggregates”, and “CD&E waste” using several nomenclatures; ii) life cycle assessment context, such as “LCA”, “life cycle”, “circular”, “extraction”, “excavation”, “mining”, “quarrying”, “processing”, “production”, “manufacturing”, “construction”, “service life”, “use”, “demolition”, “EoL”, “end of life”, “landfill”, “recycling”, “recovery”, and “backfilling”; iii) environmental impacts context, such as “global warming potential” using several nomenclatures, “acidification”, “eutrophication”, “human health”, “human toxicity”, “ecosystem toxicity”, “ozone depletion”, “abiotic depletion”, “energy consumption”; “air emissions”, “soil contamination”, “natural resources”, “photochemical ozone creation”, “smog formation”, “embodies carbon”, “land occupation”, “land use” and “particulate matter”. We were not able to pose any regional eligibility criteria (e.g. UK-based or European-based studies) due to limited information on related studies and therefore we included LCAs around the world. However, the representation of quantified data aims to provide information that could be considered in the UK.

Literature searching showed a wide variety of environmental impact categories that were quantified using different methodologies (e.g. CML, ReCIPE, TRACI, Ecoindicator99, and EN 15804:2012+A1:2013 based on CML in Europe) and reference units – that is the units (e.g. kg, m3, m2 use, etc.) of the analysed system for which all the environmental impacts are calculated - making the comparison amongst them difficult or even impossible in some cases. Predominantly, the LCA studies identified in our search were carried out following the principles of ISO 14040 and ISO 14044 (ISO 14044, 2006, ISO 14040, 2006). As a result, quantified data were retrieved from studies under related ISOs. From that perspective, we selected a list of the most common impacts as shown in **Table 3-1**.

Table 3-1 Environmental metrics along with corresponding units selected to investigate the environmental impacts of inert materials.

Environmental metric	*Description*	*Unit*	*Ref.*
Global warming potential (GWP)	Recent and ongoing rise in global average temperature near Earth's surface caused by increasing GHG emissions leading climate patterns to change, such as sea level rise, increased temperature, and extreme weather events. GWP is primarily associated with combustion of fuels and energy intensive production processes.	kg CO-eq.	(Harvey et al., 2016)
Acidification potential (AP)	The increasing concentration of hydrogen ions (H+) leading to acid producing gases (e.g. sulphur dioxide (SO2), nitrous oxides (NOx), hydrogen chloride (HCl), hydrogen sulphide (H2S), and hydrogen fluoride (HF)) deposited on the earth's surface (water, soil, plants, buildings, etc.) most commonly through rain. AP is associated with fuel combustion (e.g. SO2 from coal combustion and calcination process, NOx emissions from the combined heat and power engines used to treat biogas, and NH3 from agricultural soil).	kg SO2 eq.	(Harvey et al., 2016, Iacovidou et al., 2017)
Eutrophication potential (EP)	The increasing concentration of nutrients such as ammonia, nitrates, nitrogen oxides, and phosphorous in aquatic systems leading to excessive growth of biomass (e.g. algae) causing a rapid consumption of oxygen dissolved in the water and therefore damage to fish populations. EP is associated with the production and/or management of nitrogen or phosphorus containing resources and wastes, fertilizers used in agriculture and runoff from livestock operations.	kg Phosphate eq.	(Iacovidou et al., 2017, Harvey et al., 2016)
Ozone depletion formation (ODP)	The relative contribution of ozone depleting substances to depletion of stratospheric ozone by the release of reactive chlorine and bromine atoms in the stratosphere leading to increased pass-through of UV radiation that can cause effects to human health (e.g. skin cancer and	kg R11 eq. (R11 refers to CFC (chlorofluorocar	(Iacovidou et al., 2017, Harvey et al., 2016)

	cataracts) and ecosystem. ODP is associated with the consumption of chlorofluorocarbons as coolants in refrigeration and air conditioners, solvents in cleaners, blowing agents in the production of foam, and as propellants in sprays and other aerosols.	bon) - 11)	
Abiotic depletion potential (ADP)	Reduction in availability of fossil or non-renewable resources and raw materials. ADP is subdivided into $ADP_{elements}$ related to the availability of natural elements and ADP_{fossil} related to fossil energy carriers (e.g. crude oil, natural gas, and coal).	MJ (for fossil) AND kg Sb eq. (for element s)	(Harvey et al., 2016, Christoforou et al., 2016)
Ecotoxicity potential (ETP)	The effect of toxic substances released into ecosystem. ETP is often calculated for terrestrial, marine and freshwater ecotoxicity separately, as the cause-effect chain depends on the environmental compartment that pollutants are released.	kg DCB eq.	(Harvey et al., 2016)
Human toxicity potential (HTP)	The increase in human morbidity caused by exposure to a pollutant (whether cancer or non-cancer related).	kg DCB eq.	(Harvey et al., 2016)
Photochemical ozone creation potential (POCP)	The relative contribution of volatile organic compounds and other substances (e.g. NO_x) to the photochemical oxidation effect, or 'summer smog', through complex reactions under the presence of sunlight. POCP is associated with fuel combustion, vehicle emissions, gasoline vapours, and use of paint solvents.	kg ethyl eq.	(Iacovidou et al., 2017)
Particulate matter (PM)	Particulates such as PM_{10} and $PM_{2.5}$ can cause respiratory effects on humans and lead to illnesses such as asthma.	kg $PM_{2.5}$ eq.	(Harvey et al., 2016)
Land occupation (LO)	The reduction in availability of usable land that can constitute an indicator of biodiversity loss.	m^2	(Harvey et al., 2016)

4. ANALYTICAL APPROACH AND CLARIFICATIONS ON WASTE DEFINITIONS

With an expected population increase of 3.6 million people by 2026[1], the construction sector activity is set to rise sharply. A niche area when it comes to the future of the construction sector is related to the sustainable management of the existing (and future) stocks and flows of materials, components and products produced, used, and disposed of in the construction value chain, including interdependences with the value chains associated with them. A life cycle perspective that considers all phases in the materials (incl. components and products) flow across the construction value chain can promote informed decision-making and implementation of new sustainable and circular trends in the construction sector.

4.1. Analytical approach

The environmental aspects involved in, and associated with, the production, use and management of construction materials, components and products are far more complex than most other resource flows. This is due to the high variability of materials used, and the complexity of processes involved in their life-cycle stages, from raw materials extraction, transport, processing, through to use (e.g. the way by which they are bound/ fixed in different types of engineering structures) and final disposal/ management. A material flow framework can facilitate the improved understanding of the flows of raw and processed / manufactured materials used in the construction sector, and associated waste. This can track their environmental impacts across their life cycle, thereby providing a useful means to assess the entire construction value chain and underpin informed decision-making. With increasing political focus placed on sustainable resource management and productivity, the combined use of a mass flow analysis with LCA studies, can thus generate useful insights.

In LCA studies boundary conditions are used to define the life cycle stages included in the assessment of environmental impacts. The most common boundary is the *cradle to gate* that considers activities related to the extraction of materials from the earth (the cradle), transportation, refining, processing and manufacture until the material or product is ready to leave the factory gate. Some studies take this one step further to adopting a *cradle to site* boundary, which includes the *cradle to gate*, plus the transportation of the material or product(s) to the site of use. Another boundary is the *cradle to grave* that includes the *cradle to site, plus* the use and EoL (disposal, reuse, recycling) of the waste materials, components and products generated when the asset reaches the end of its service life as shown in **Figure 4-1** (Circularecology, 2021).

[1] Office for National Statistics, 2020.

Following the stages illustrated in **Figure 4-1**, we began our analysis looking at the extraction stage of raw materials, moving across all stages that lead up to their final disposal as part of construction and demolition and excavation waste.

In **Figure 4-1**, we aimed to include all the evidence gathered from the LCA studies identified in our review, which proved to be a rather challenging task. This is due to the lack of proper measurement of the LCA impacts of the extraction, processing, use and final disposal, alongside the difficulty of defining accurate material use and disposal pathways (Marcelino-Sadaba et al., 2017). It is worth mentioning that for construction and demolition waste (and everything that falls within that definition), the negation of materials (including components and products), and therefore the contribution to key environmental impacts, begins from the on-site construction stage (e.g. excess material disposal, inappropriate storage that leads to damaged materials, and off-cuttings).

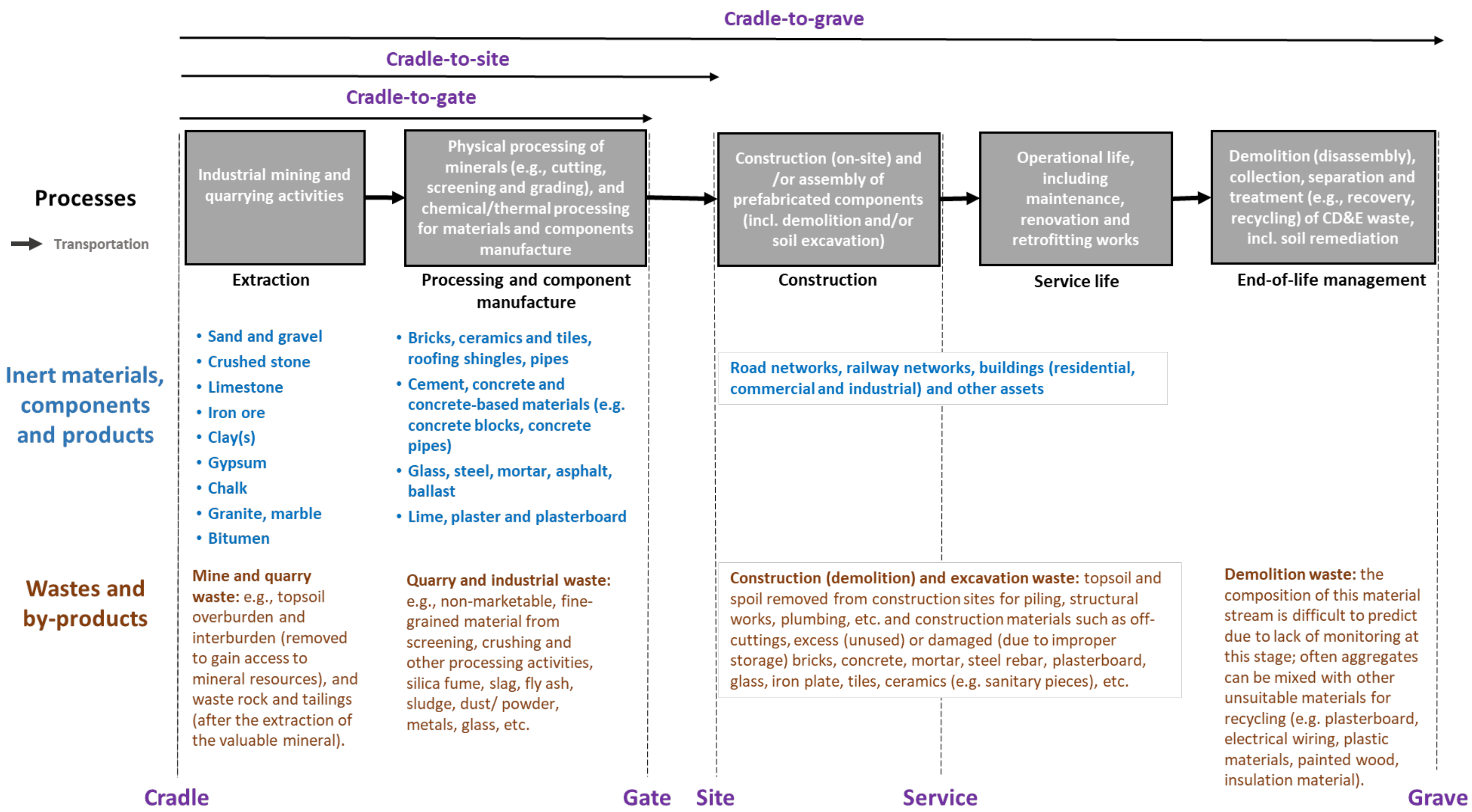

Figure 4-1 Depiction of the major stages involved in the life cycle of inert /less reactive materials from production through to EoL management and LCA boundaries

4.2. Definitions on recovery

It is worth mentioning that at present the upstream parts of the construction industry, such as extraction of raw materials, known as quarrying and dredging activities, and raw materials processing and manufacturing stages, are not well connected with the downstream parts of the construction industry that involve the on-site construction, demolition and excavation activities, use and EoL management. It is therefore expected that the CD&E definition includes only waste materials generated downstream of the construction value chain. While this is justifiable based on the presentation of statistics according to economic activities, it creates discrepancies in the life cycle evaluation of materials, components, and products used in the construction sector.

Furthermore, there is ambiguity in the way waste materials in the CD&E stream are being accounted for, which adds to the complexity of gaining an insight into their disposal/ management pathways. The following c*larifications were possible to make:*

Dredging spoils from a construction sector point of view, are a waste by-product of the construction and maintenance of water projects, and are included in the excavation soil and spoil. From a material standpoint, dredging spoils are not the same as soils (i.e., can include rocks and other aggregates) and they tend to have a different fate from soils—i.e., generally speaking, they tend to end up released back into water bodies as opposed to in landfill, and seem to be much less commonly used for backfilling.

Excavated soil and spoil in England, that the holder discards, intends to discard, or is required to discard is considered to be a 'waste', unless if it is to be reused for the purposes of construction on the site of origin and is not considered to be contaminated soil / spoil. If uncontaminated excavated soils up to 1000 tonnes are to be reused as a fill material on another site, this can only be carried out under a U1 exemption from the Environment Agency, or under a Materials Management Plan (**MMP**) prepared in accordance with the CL:AIRE code of practice, known as the DoWCoP[2].

Furthermore, there is also a multitude of meanings attached to the definition of 'recovery'. This term, and particularly its use, its contested, ambiguous and elusive. Figure 4-2 clarifies the terminology used in legal documents.

[2] DoWCoP: stands for Definition of waste code of practice and enables the direct transfer and reuse of clean naturally occurring uncontaminated soil materials between sites.

KEY DEFINITIONS

'TEATMENT': means recovery or disposal operations, including preparation prior to recovery or disposal.

'RECOVERY': means any operation the principal result of which is waste serving a useful purpose by replacing other materials which would otherwise have been used to fulfil a particular function, or waste being prepared to fulfil that function, in the plant or in the wider economy.

'MATERIAL RECOVERY': means any recovery operation, other than energy recovery and the reprocessing into materials that are to be used as fuels or other means to generate energy.

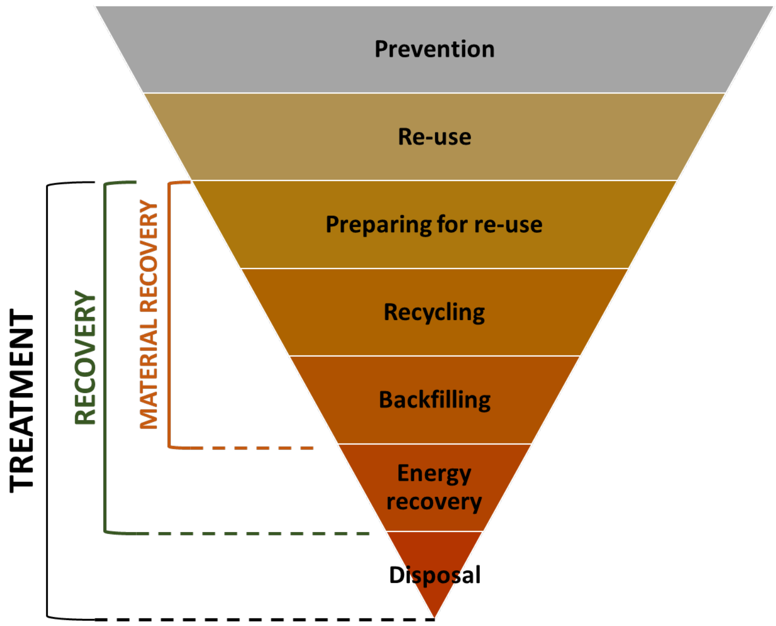

Figure 4-2 Clarification on the definition of 'recovery' as described in legislation

Historically, the Defra published statistics on total waste generation and final treatment have been produced in line with EU reporting requirements under the Waste Statistics Regulation. They are calculated at opposite ends of what can be a complex and multi-staged treatment process and different methodologies are used to estimate generation and final treatment figures. By definition, generation figures include waste that is exported for treatment outside of the UK (but exclude imports) and treatment figures include waste that is imported for final treatment from outside of the UK (but exclude exports). Furthermore, final treatment excludes some treatment processes identified as predominantly intermediate, which nevertheless may effectively be the final treatment for some waste. As a result, there is no direct reconciliation between generation and final treatment of total waste. Additionally, in most cases it is not possible to estimate the final treatment of waste generated by specific economic activities.

The treatment processes for CD&E waste management reported in the literature appear to be contested and elusive. A notable example, for which clarification were not possible to make is soil.

Soil recycling when it's not backfilling (as defined in the Eurostat guidance here), could refer to various 'recycling' options as follows:

- reuse on site or other projects (AUTHORS NOTE: *that could in same cases also be referred to as backfilling);*
- storage in landfills or other sites where it is inactively stocked (AUTHORS NOTE: *that could in fact refer to disposal*);

- used as cover material in closed landfills and quarries, or for rehabilitation purposes.

The latter is often considered recycling of low quality; yet is closer to recycling that any of the other options (Magnusson et al., 2015).

The waste materials generated during the quarrying and dredging activities as well as the waste materials generated during the physical, mechanical and thermal processing of raw materials for the manufactured of the finished materials, components or products are included under the mining and quarrying and industrial waste respectively (as shown in **Figure 4-1**). Herein, we consider these wastes to play a notable part in our analysis, as they represent a key environmental impact associated with inert materials used in construction sector across their life cycle.

5. LIFE CYCLE ENVIRONMENTAL IMPACTS

In 2018, the production of aggregates – that is sand and gravel (incl. land and marine dredged), and crushed rock (e.g. limestone, including dolomite, igneous rock and sandstone) - amounted to 198 Mt, and of other construction materials (i.e., clay, shale, gypsum) was at 11.7 Mt in the UK (BGS, 2020b), of which clay and shale accounted for ca. 5Mt. According to BGS (2020c), aggregates make up around 85% of the non–energy minerals extracted in the UK, and are largely used in the construction sector. The following Sankey diagram illustrates the mass flow from *cradle-to-gate* for aggregates produced in the UK in year 2018, and potential end-uses (using data from Great Britain) (**Figure 5-1**).

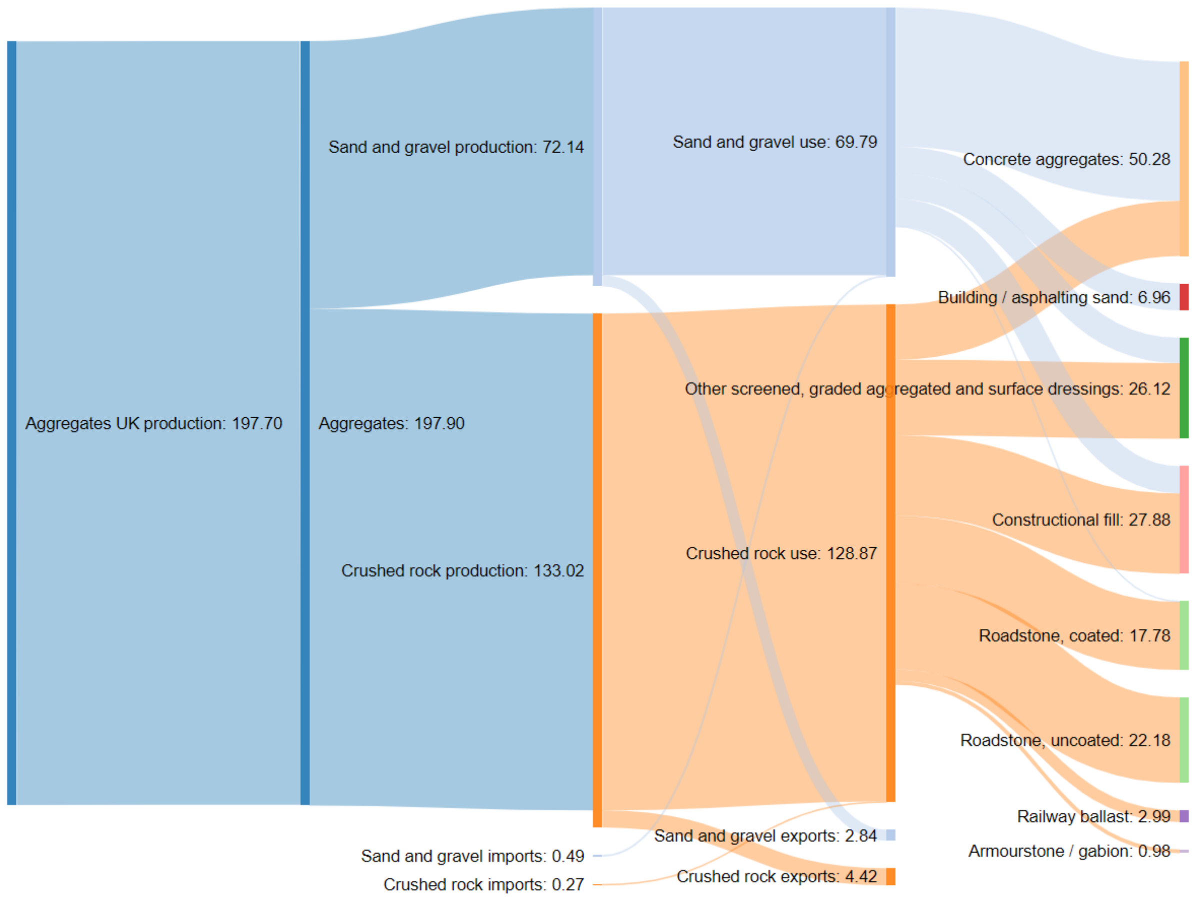

Figure 5-1 Production of aggregates (in Mt) in the UK, and potential end-use pathways. The gaps in the end use pathways may be due to: 1) use of data from Great Britain only; 2) data sets used are from different years; 3) discrepancies in the data sets used.

Aggregates can be distinguished into primary, secondary, or recycled:

- Primary aggregates are rock, sand and gravel that are extracted from naturally occurring mineral deposits for use as aggregates, and their production and flow in the UK economy is depicted in **Figure 5-1**;
- Secondary aggregates are a by-product of other quarrying and mining operations, such as china clay waste, slate waste and colliery spoil, or material arising as unavoidable consequence of construction works, as well as manufactured aggregates obtained as a by-product of other industrial processes.
- Recycled aggregates are those produced from the processing of construction and demolition waste (**CDW**).

Clay and shale is the second largest category of inert materials extracted and used in the construction sector. The mass flow of clay and shale materials from production to end-use is depicted in **Figure 5-2**.

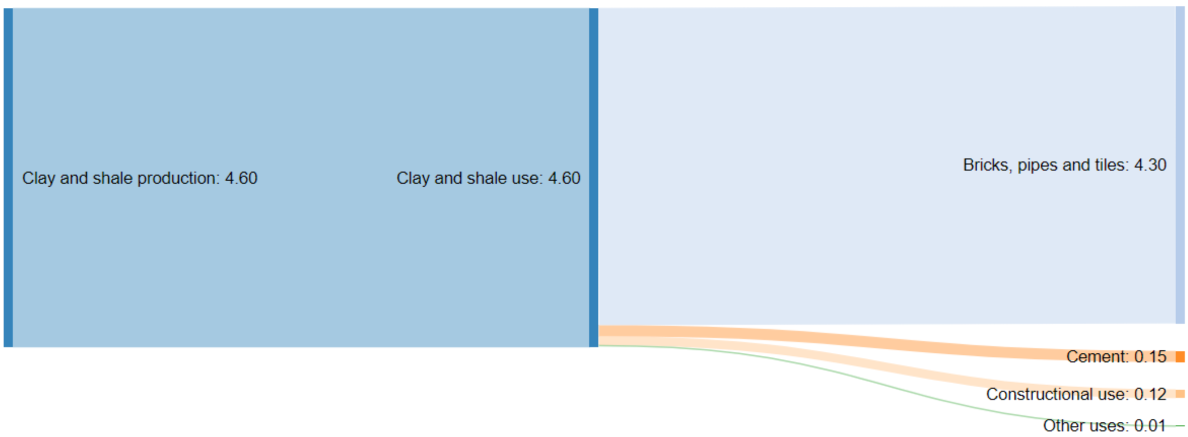

Figure 5-2 A cradle-to-gate mass flow depiction of clay and shale production and potential application pathways (in Mt) in the UK in year 2018.

Even though the aggregates, clay and shale are generally inert and non-hazardous, their extraction and processing may give rise to a number of environmental impacts. Quarry waste and fines of which disposal / management pathways are hugely underexplored, can give rise to significant environmental concerns. There are currently limited studies on the environmental impacts of quarrying activities; yet, there is convergence on the fact that the nature and extent of the environmental impacts will vary from site to site, according to their characteristics and specific local context.

The high variability in raw materials end-uses, their transformation and final (EoL) fate, alongside the lack of base data (emission, energy etc.) for each stage of the life cycle signifies that gaining a clear and accurate life cycle impact evaluation of these materials can be grim (Marcelino-Sadaba et al., 2017).

The majority of studies makes no quantification or reference on the waste fractions or materials, which means that data granularity in this area is limited.

5.1. Quarry and dredging of raw materials

The extraction and processing of aggregates creates a substantial amount of unavoidable waste minerals (e.g. clay, mudstone and siltstone) and fines (produced during quarrying and aggregates sizing process). These (quarry waste and fines) are usually temporarily or permanently disposed within the boundaries of the quarrying operations in-situ, e.g. stored in bunds or tips[3] and settlement lagoons (BGS, 2020a, GoodQuarry, 2020). These waste streams are often distinguished into the: i) overburden (i.e. topsoil and subsoil removed, and used in restoration); ii) interburden (low-value material that has no use); and iii) processing wastes (non-marketable, mostly fine-grained material from screening, crushing and other physical-mechanical processing activities) (GoodQuarry, 2020).

According to BGS, the UK quarrying industry produces more than 50 Mt of 'quarry fines' and over 20 million tonnes of 'quarry waste' per year (BGS, 2020c), as shown in **Figure 5-3**. This is congruent with the latest UK Statistics, in which it is reported that under the 'other mineral waste' category, the Mining and Quarrying industry (NACE B) generated around 17 Mt of waste. However, this figure is an estimate and there isn't robust evidence on the actual amount of quarrying and dredging waste produced for the extraction of aggregates and other minerals used in construction activities.

The removal of both overburden and interburden soil can lead to geomorphological changes and land use change, which in turn can modify the natural drainage and increase soil erosion and siltation, as well as destabilisation of slopes (Langer, 2001). It can also damage the surrounding ecosystem, due to the loss and/or displacement of species, and the loss of land productivity (Saviour, 2012). Large amounts of silt and other effluents from quarries (e.g. other waste, fuel, oil) may pollute rivers as well as underground water bodies within and far beyond the vicinity of the quarries. These impacts can vary depending on the size of the quarry, the location and local landform, and there is always a visual impact associated with the quarry waste and fines.

[3] Tips are defined as accumulated quarry wastes, including waste and soil heaps, stockpiled materials, backfill, screening embankments, and lagoons and settling ponds.

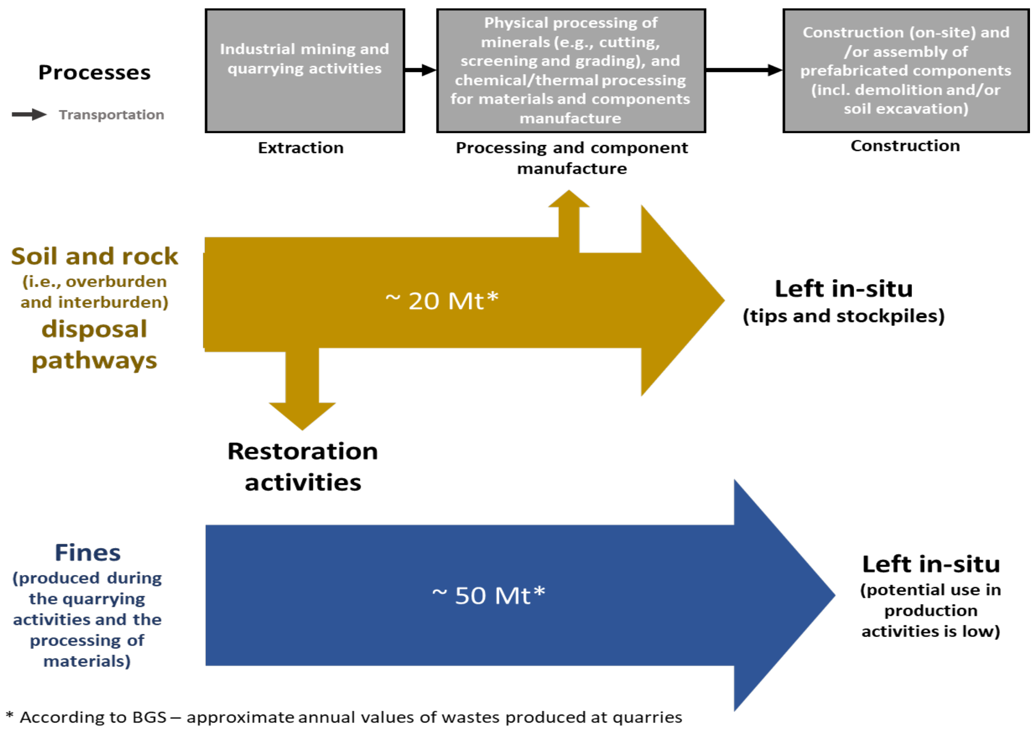

Figure 5-3 Generation and disposal/ management pathways of quarry waste and fines in the UK

Commonly, the quarry waste, i.e. the overburden (topsoil and subsoil) and interburden, removed during site clearance for the extraction activities to be stockpiling *in-situ*. Even though both quarry waste and fines are considered to be inert, non-hazardous wastes, their *in-situ* management presents considerable environmental threats in the long-term. The environmental, and by extent social, impacts can vary widely depending on the nature of materials quarried, geology and proximity of housing, amenity areas and local businesses. Some of the key potential environmental impacts associated with the disposal/ management of quarry waste and fines are summarised in **Table 5-1**.

Table 5-1 Environmental impacts associated with the generation of quarry waste and fines during the extraction and physical / mechanical processing of raw inert, non-hazardous materials used in construction activities (excludes, thermal processing of raw materials, e.g. limestone, dolomite and clay)

Environmental impact	*Description*	*References*
Air quality	Dust produced during quarrying and from air filtration units/ stacks, haulage trucks, conveyors and transfer points can have a substantial impact on air quality, and is most acute in enclosed spaces (for example in the processing plants) or in close proximity to major sources.	(Zhang et al., 2020, GoodQuarry, 2020, BGS, 2020a)

	Indirect impacts: *health implications (typically linked to occupational health impacts), visual intrusion (also considered as eye sore), nuisance for surrounding communities and businesses.*	
Water consumption	Water is used to prevent fines from being dispersed to the surrounding environment.	(Zhang et al., 2020, GoodQuarry, 2020, BGS, 2020a)
Land use change	Waste disposed of to flood plains may exacerbate flooding. Settled silt and clay can also be washed out and displaced from settling ponds and lagoons during storm events. Temporary or permanent land sterilisation may result from the use of space within or outside the working area, some of which could otherwise be put to beneficial use. Temporary or permanent loss of the associated fauna and flora are also likely, although this can be mitigated by appropriate restoration of the disposal areas.	(Zhang et al., 2020, GoodQuarry, 2020, BGS, 2020a, Al-Dadi et al., 2014)
Ecotoxicity potential	**In water:** water run-off from quarry waste tips or quarry fines stockpiles can carry solids that may cause erosion and contaminate local watercourses. Suspended solids (and acid drainage), can impair quality and use of water (e.g. drinking water, industrial uses, irrigation, and fisheries), and impact on the fauna and flora that it supports. **In soil:** the presence of chemically active mineral phases in the dust (e.g. sulphides occasionally present at hard rock quarries) may alter soil's chemistry and suitability for the fauna and flora that the soil supports	(Zhang et al., 2020, GoodQuarry, 2020, BGS, 2020a)
Noise pollution	Noise from the extraction of aggregate and dimension stone from earth-moving equipment, processing equipment, and blasting can negatively impact workers and biota (Langer, 2001), as well as nearby communities (depending on how close they are to the quarry).	

A study that looked into the particulate matter (**PM**) and gas emissions in the mining and quarrying sector, using data from 2012, concluded that they lead to considerable influences on human health and ecosystem (Fugiel et al., 2017). For instance, they can cause respiratory diseases both in humans and biota (Losacco and Perillo, 2018), especially when the quarry is at close proximity to nearby communities (Langer, 2001). The concentrations of PM generated can depend on the size of the operation, rock properties, moisture, ambient air quality, while its deposition may depend on air currents and prevailing winds (Langer, 2001). Another study has shown that mining and industrial processing can also be a source of heavy metal (such as cadmium, Cd, and lead, Pb) contamination in the environment, which may accumulate to toxic concentration levels and lead to ecological pollution. It must be emphasised that while the life cycle environmental impacts in the mining and quarrying sector have been poorly investigated, this is a key issue that requires attention due to the non-renewability of the resource and the considerable environmental impacts arising from related processes (Fugiel et al., 2017).

Finally, in post-mining/ post-quarrying activities there is provision for soil rehabilitation. This involves topsoil mining separately at the beginning of the operations before any surface disturbance such as drilling, mining or blasting, and stockpiling when it is impractical to promptly redistribute it (Ghose, 2001). The best practice of soil rehabilitation is the stripping and immediate replacement of dry and fresh topsoil, but if topsoil is stockpiled, the biological activity should be maintained, avoiding compaction, nutrient leaching, loss of organic matter, and dilution of seed bank (Goosen, 2014).

5.2. Processing / Production of materials, components and products

The processing and production stages of raw materials, components used in pre-fabrication facilities and on-site construction, include the quarrying and/or mining, and physical-mechanical treatment processes, e.g. cutting, grading and finishing techniques. The main challenge in evaluating the environmental impacts at these stages is the acquisition of Life Cycle Inventory (**LCI**) datasets in terms of quality and availability (Bianco and Blengini, 2019). For example, according to a study carried out in Thailand, the environmental impacts arising from limestone quarrying (extraction, transportation and processing) contributed to energy use and GWP nearly 79.6 MJ and 2.8-3.1 kg CO_2-eq. per tonne of limestone product respectively, mainly caused by diesel fuel and electricity consumption (Kittipongvises, 2017). Also in Thailand, the total amount of GHGs emitted from limestone mining operations (including transportation) also accounted for 18.8-22.4 kg CO_2-eq. per tonne of limestone product mostly related to transport emissions, while GWP was nearly two times higher for basalt mining compared to limestone rock mining (31.9-35.7 kg CO_2-eq per tonne of basalt product) (Dubsok and Kittipongvieses, 2016). Furthermore, a cradle-to-gate study in Australia estimated the GHG emissions arising from quarrying (including transportation) of graphite/hornfels and basalt in Australia at nearly 45.9 and 35.7 kg CO_2^-eq. per tonne product, respectively, while the production and transportation of concrete-sand, that is strip mined and processed in a quarry, accounted for 13.9 kg CO_2-eq. per tonne (Nazari and Sanjayan, 2016).

A UK-based LCA that compares the environmental impacts of the extraction and processing of natural aggregates (crushed rock, land won gravel and sand, and marine gravel and sand), takes also into account overburden stripping, drilling and blasting, and restoration, with those of processing the equivalent recycled aggregates (Korre and Durucan, 2009). Impact assessment results per tonne of aggregate produced in terms of GWP are illustrated in **Figure 5-4**. Authors reported that impacts quoted as averages for the aggregate systems (e.g. crushed rock) are not representative of the impacts associated with the individual products (e.g. crushed rock category A, vs crushed rock category B in **Figure 5-4**), while it is important to consider product category specific results since significant differences between them can be found even for one aggregate production site (Korre and Durucan, 2009). For crushed rocks, crushing unit processes contribute mostly to impacts due to high energy demand indicating that processing phase has higher impacts than extraction, while for land won gravel and sand, the impacts are lower since the extracted material is similar to the product specifications required, minimising the need for processing (Korre and Durucan, 2009). The processes for land won gravel and sand with higher contribution to GWP are loading and conveying, followed by washing and scrubbing, product storage and crushing sub-phases (Korre and Durucan, 2009). The production of marine aggregates exceeded by far the production of all other aggregates with respect to all impacts due to high gas oil consumption for the dredger and the corresponding upstream indirect emissions, while no significant difference can be observed between marine gravel and marine sand (**Figure 5-4**) as their proportions in the primary resource is equal (Korre and Durucan, 2009). Finally, in the case of recycled aggregate production, the stage of washing was found to be the most intensive phase in all impact categories, while the relative range is considerably higher compared to natural aggregate production depending on the product quality and related applications (e.g. bound vs unbound aggregate) (Korre and Durucan, 2009).

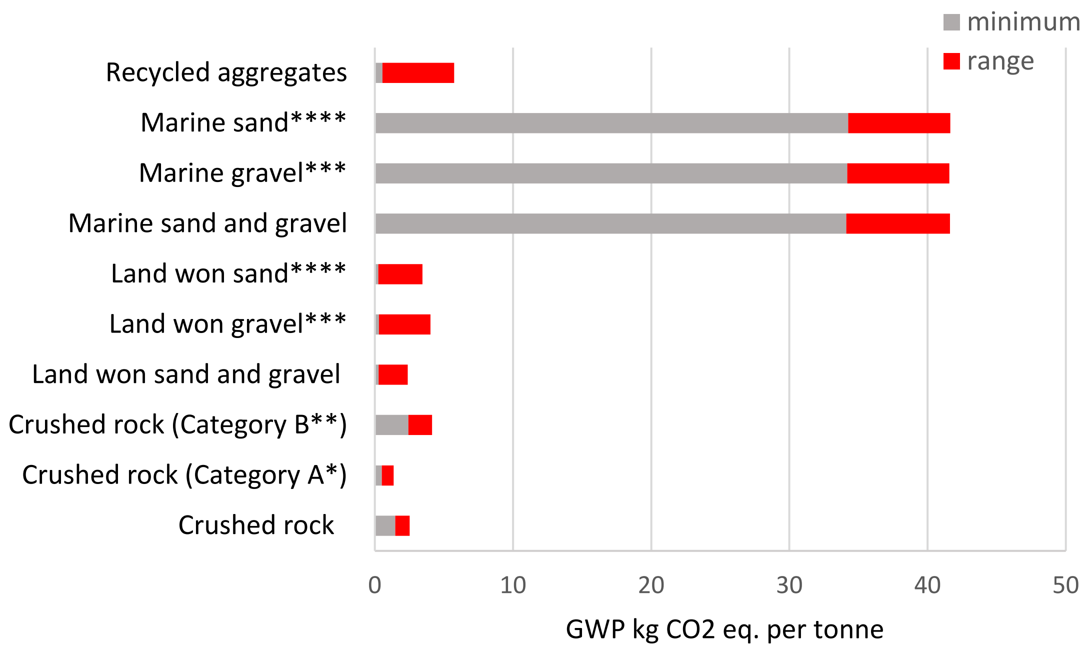

Figure 5-4 Range (max-min) of global warming potential (GWP) per tonne of aggregate produced arising from the production of aggregates in the UK including extraction and processing for natural aggregates and recycling for recycled aggregates. *subbase, capping layers, crusher runs, agricultural lime, scalping, 80-40 mm, 150 mm, 125 mm, 40 mm, dust 6mm, dust 3mm; **28 mm, 20 mm, 14 mm, 10 mm; ***40-20 mm, 20-10 mm, 5-10 mm, 3-5 mm, oversize; ****coarse sand, building sand, fine sand. Adopted by Korre and Durucan (2009).

At present, there is limited attention at the environmental impacts at this stage. Nonetheless, the embodied carbon data on materials, component and products can provide a *cradle-to-gate* insight into one of the key LCA impacts. Embodied carbon is the carbon released in the manufacture of construction materials, components, and products and includes raw material(s) extraction, processing, transportation, site operations, etc. From an environmental perspective, the carbon emissions throughout a material's life cycle are important milestone to be considered, and there many studies that have focused on this metric. In the UK, there is database dedicated to the calculations of embodied carbon emissions of materials used in the construction sector (Circularecology, 2021). In **Table 5-2**, the average embodied carbon for most prevalent construction materials in the UK obtained from this database is presented indicating that aluminium has the highest average value followed by steel, glass, and cement. The highest relative dispersion can be seen for aggregates arising from the high variability of aggregate products and therefore extraction and processing phases leading to significantly different footprint profiles. A detailed list on the embodied carbon of materials used in construction projects is given in the **Appendix** (**Table A.1**).

Table 5-2 Descriptive statistics of averaged cradle-to-gate embodied carbon of the most common materials used in the construction sector within the UK as obtained from ICE database V3.0 (Circularecology, 2021).

Material category	***Embodied carbon (kg CO_2-eq. per kg of produced material)***						***DQI* (%)***
	Average	**Median**	**Min**	**Max**	**RSD (%)[1]**	**N[2]**	
Aggregates (land won, marine, recycled and secondary)	0.017	0.004	0.000	0.397	336.78	164	72.8
Aluminium (virgin and recycled)	8.719	8.120	0.330	22.078	71.26	84	73.3
Asphalt (blend of bitumen and aggregates)	0.043	0.031	0.021	0.098	53.94	21	65.4
Cement (UK average cement)	0.795	0.785	0.088	2.670	60.56	70	68.2
Concrete (cement, aggregates, sand, and admixtures)	0.166	0.134	0.034	0.558	68.94	16	69.5
Steel (virgin and recycled)	2.364	2.384	0.282	16.155	76.84	169	76.9
Glass	1.594	1.530	0.920	5.062	27.20	189	77.7

Clay bricks	0.255	0.243	0.179	0.354	17.46	74	64.5
Timber	-1.040	-1.061	-1.546	0.580	-30.96	211	75.9

*[1]RSD: relative standard deviation; [2]N: sample population; *DQI: Data quality indicators that show the quality attribute of data*

Mitigation strategies to reduce the embodied carbon of construction materials have focused on the utilisation of industrial by-products, such as ground granulated blast-furnace slag (**GGBS**), from primary steel production, pulverised fly ash (**PFA**) from coal combustion, and silica fume from silicon or ferrosilicon metal manufacture, at the production stage as an aggregate replacement. This practice has gained precedence over the last decades as a resource efficiency and a waste minimisation strategy. For example, PFA or GGBS, can replace up to 55% wt. or 80% wt., respectively, of the required cement, which is by far the most carbon intensive ingredient of concrete production (Millward-Hopkins et al., 2018). Substituting natural aggregates with by-products can offer many benefits; which in the environmental domain in particular includes reduction in the wastage of valuable resources, resource efficiency and carbon emissions reduction (e.g. 15-73%) depending on the type and amount of materials substituted (Meek et al., 2021).

5.2.1. Cement and concrete production

The assessment of the environmental impacts of concrete materials has been largely researched mostly in relation to the origin of the aggregates used in cement manufacture (Serres et al., 2016, Estanqueiro et al., 2018, Napolano et al., 2016, Knoeri et al., 2013, Colangelo et al., 2018, Ding et al., 2016, Guo et al., 2018). Concrete is a composite material, made of Portland cement (**PC**), or cement substitutes (10-15%), large amounts of aggregates (gravel, sand or rock) (60-75%) and water (15-20%) (PCA, 2020) .

Cement is the ingredient responsible for a large inflow of fuels (e.g. coal) and resources (i.e. limestone and clay), whereas aggregates are abundant and also low-carbon (see **Table 5-3**). Mining the limestone and clay from rock quarries, and breaking them into smaller pieces for transportation can have a considerable impact on the environment via the production of PM, dust and heavy metals contamination (Al-Dadi et al., 2014), but a thorough evaluation of the environmental impacts at this stage is largely underexplored. The quarried limestones and clay are further processed by crushing or pounding to chunks approximately 1½ inches in size, are then grinded and pulverised, and finally they are proportionally mixed according to the particular type of cement being produced. The homogenised raw material mixture (also known as, *raw meal*) is transformed into clinker in a process called calcination. Calcination is a process that requires a temperature as high as 1450 °C for the chemical and physical transformations to occur, which turn raw meal into small, dark grey nodules 3-4 cms in diameter (i.e. clinker). These nodules are further grounded with additives and other mineral components such as gypsum, slag, and fly ash that build up the required properties of the final fine-powdered material – the cement. The process of cement production is responsible for the highest carbon emissions across all materials used in the construction sector.

The production of Portland cement is an energy intensive process, accounting for ca. 5%-7% of global anthropogenic CO_2 emissions (Ruan and Unluer, 2016). The calcination process is responsible for 60% of the carbon emissions associated with cement production (Ruan and Unluer, 2016). Nonetheless, it is worth noting that in one study it is reported that the main environmental impacts of cement production comes from quarrying, waste disposal, or storage of materials on site, and atmospheric deposition (Al-Dadi et al., 2014). According to CEMEX, the GWP per tonne of cement produced is ca. 850 kg CO_2-eq. for clinker, and can be reduced with the addition of cement substitutes such as PFA and GGBS (achieving reductions of approximately 35% in mixes with PFA, and 60% in mixes with GGBS; without allocation (Salas et al., 2016)). Besides CO_2 emissions, cement production accounts for significant emissions of air pollutants, including sulphur dioxide (**SO_2**), nitrogen oxides (**NO_X**), carbon monoxide (**CO**), PM, and cement kiln dust, which may lead to significant regional and global environmental impacts (Lei et al., 2011, Salas et al., 2016). In Europe, the production of 1 tonne of the ordinary Portland cement at 100% purity (i.e. CEM I, or Type I) results in the average emissions of 2.4 kg of NO_x, 0.6 kg of SO_2 and 2.7 kg of dust (Josa et al., 2007). For instance, NO_x and SO_2 emissions can cause acidification, whilst NO_x emissions are also responsible for eutrophication potential. The amount of air pollutants released depends largely on the type of fuels, mostly from energy production processes (electricity and fuel refining (Salas et al., 2016)), impurities in the raw materials used during the calcination process, as well as the manufacturing process(es)/ technologies employed (e.g. cement production technologies, Best Available Techniques (BAT), or the use of alternative fuels).

Blends of cement with substitute materials (e.g. PFA, GGBS, limestone powder, etc.) (Celik et al., 2015)) can also affect the environmental performance of concrete. As shown in **Figure 5-5**, partial or whole replacement of Portland cement with fly ash and limestone powder can provide significant environmental benefits (Marcelino-Sadaba et al., 2017). For example, high-volume replacement of ordinary Portland cement with fly ash (up to 55% w/w), or a mixture of fly ash and limestone powder can provide workable concrete with lower GWP (ca. up to 50%) in concrete production (Celik et al., 2015). However, CO_2 emissions induced by mixtures with higher fly ash were found to be higher compared to limestone powder (e.g. 5-10 times) due to the higher fuel consumption per unit mass of fly ash in the drying process employed before mixing in the concrete (Celik et al., 2015).

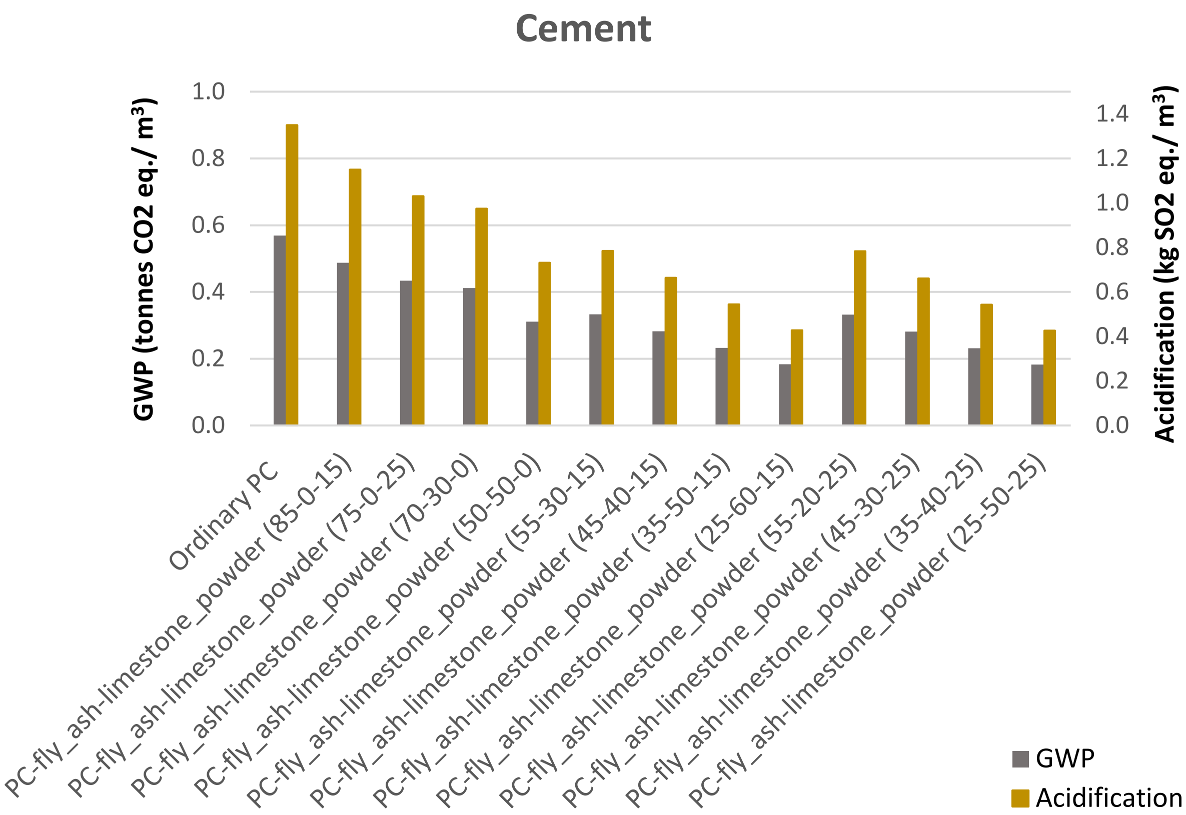

Figure 5-5 Environmental impacts of self-consolidating concrete (SCC) cement mixtures made with blended Portland cements containing fly ash and limestone powder, expressed in 1 m3, with respect to GWP and Acidification potential from a cradle-to-gate LCA considering raw material production (energy used); transportation; production. Abbreviations: PC: Portland cement; Ordinary PC: concrete manufactured from gravel and basalt (coarse aggregate), sand (fine aggregate), and PC. Adapted by Celik et al. (2015).

Figure 5-6 below, illustrates the results from a recent cradle-to-gate LCA study of concrete blocks (**Figure 5-6**: bars in pattern fill driven from a cradle-to-cradle LCA) that examined the environmental impacts of seven concrete blocks from the use of different concrete curing technologies, mixes of cement with substitute materials, and origin of aggregates used. The CO_2 mineral carbonation curing manufacturing process includes ordinary Portland cement block, Wollastonite-Portland cement block, reactive magnesia (**MgO**)-Portland cement block, limestone-Portland cement block, slag-Portland cement block, calcium silicate cement block, and waste concrete aggregate block (Huang et al., 2019). Reactive magnesia (MgO) cements, produced via the calcination of magnesite, are considered to offer net environmental benefits compared to PC due to their lower production temperatures (ca. 800 vs. 1450 °C) and ability to fully carbonate and gain strength during setting (Ruan and Unluer, 2016). Results from the comparative study indicate that GWP of 1 m^3 CO_2-cured non-hollow concrete block ranged from 292 to 454 kg CO_2-eq, while for the conventional steam-cured (not mineral-cured) ordinary-

Portland cement block, GWP was 419 kg CO_2-eq indicating that the replacement of steam curing by mineral carbonation curing and adjustment of binder types may lead to up 30% CO_2-eq emission savings (Huang et al., 2019).

Reducing the use of Portland cement, while increasing the blending ratio in binary binders and lightweight redesign (non-hollow, lower volume density) are important solutions to mitigating the environmental impacts of CO_2-cured concrete blocks. Without that action, the environmental advantage of CO_2-cured concrete block would not be considerable compared to steam-cured concrete block (Huang et al., 2019). Therefore, both recycled aggregates under specified replacement ratio (e.g. 30-75%) (Guo et al., 2018, Knoeri et al., 2013) and Wollastonite-Portland cement block (Huang et al., 2019) and/or GGBS-Portland cement (Huang et al., 2019, Colangelo et al., 2018) can be considered attractive environmental solutions (Huang et al., 2019).

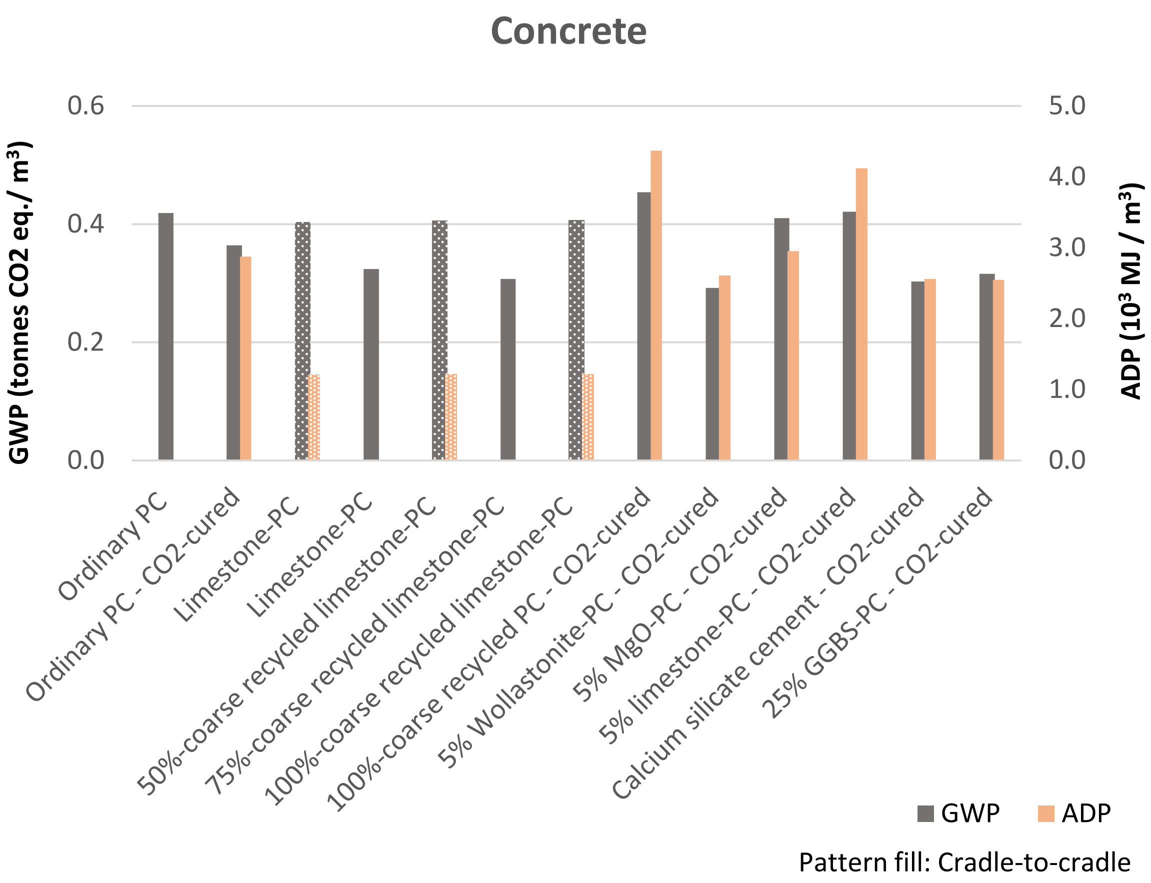

Figure 5-6 Environmental impacts of different types of concrete with similar workability focusing on concrete curing technologies and mixes of cement used, as well as the origin of aggregates (primary vs recycled), expressed in 1 m^3, with respect to GWP and ADP$_{fossil}$ arising from cradle-to-gate LCAs considering production of raw materials (e.g. aggregate extraction and PC production), transportation, and concrete production. Abbreviations: PC: Portland cement; GGBS: ground granulated blast-furnace slag; Ordinary PC: concrete manufactured from gravel

(coarse aggregate), sand (fine aggregate), and PC; Limestone-PC: concrete similar to Ordinary PC but is manufactured from limestone instead of gravel; %-coarse recycled: the proportion of coarse aggregates substituted by recycled concrete waste; %: the proportion of total aggregates (coarse and fines) substituted by an alternative raw material; CO_2-cured: mineral carbonation instead of steam-curing applied to other concrete samples. Noted: Bars with pattern fill obtained from a cradle-to-cradle study considering aggregate production (extraction for natural and recycling for recycled: emissions and energy used); transportation at each stage; concrete production; service phase; demolition; EoL (landfill and recycling); and secondary life (Ding et al., 2016). Adapted by (Ding et al., 2016, Huang et al., 2019, Guo et al., 2018).

Besides using cement substitutes to lower the embodied carbon of concrete (and other environmental impacts), the utilisation of recycled aggregates has also gained traction. The use of recycled aggregates from CDW can lower the environmental impacts of concrete production via the substitution of natural aggregates (Ding et al., 2016), and it can also offer a potential management pathway for CDW and eliminate the need of waste management (Serres et al., 2016). Evidence on the environmental impacts of several types of concrete based on the composition of aggregates in the concrete mixture in terms of GWP and ADP fossil was collected and illustrated in **Figure 5-6**.

In **Figure 5-6**, 100% replacement of natural aggregates (e.g. gravel or limestone) by concrete waste can lead to higher CO_2 emissions and higher energy consumption than the utilisation of natural aggregates. This is attributed to the fact that the incorporation of recycled concrete aggregates tends to degrade the properties of concrete blocks (e.g. compressive strength and durability) (Ding et al., 2016), resulting in the use of higher amount of cement (up to 60 kg per m^3) and additional water content (ca. 3.5 times higher than natural aggregates) (Ding et al., 2016), The higher the amount of cement used, the higher the net GWP arising from the production of concrete with 100% recycled aggregates (Knoeri et al., 2013). Improving the quality of recycled aggregates, so that the use of additional amount of cement does not exceed 10% (corresponds to 22-40 kg per m^3) it can outperform the benefits of conventional concrete (from natural aggregates), especially when the transport distance of recycled aggregates is no more than 15 km (Knoeri et al., 2013). The employment of viable and advanced technologies to produce better quality recycled aggregates (e.g. shaping, pre-soaking, and carbonization modification), could improve the environmental benefits of using recycled aggregates in concrete production (Ding et al., 2016).

Transport distance is a critical factor in assessing the environmental impacts of construction materials, and concrete production specifically (Colangelo et al., 2018). When long transport distances are involved in the use of natural aggregates, it can make the use of recycled concrete aggregates particularly favourable from an environmental perspective even with a slightly higher amount of cement used, as we explained above when using recycled aggregates (Ding et al., 2016). Specifically, Knoeri et al. (2013) reported that the use of recycled aggregates (28-45% w/w of total aggregates in concrete production) can induce significant environmental benefits (ca. 30%) with respect to all impacts at endpoint level (effect on human health, biodiversity, and resource scarcity) due to avoided burdens arising from reinforcing steel recycling and reduced disposal of CDW.

With a focus only on raw materials, ignoring the environmental impacts associated with the use of demolition waste, 100% replacement by recycled aggregates in concrete production can be beneficial only with respect to land occupation and respiratory inorganics mainly due to avoiding quarrying, which highlights the importance of optimising the recycling process aggregates (e.g. by means of selective demolition through careful dismantling of buildings) (Estanqueiro et al., 2018). It's been shown that coarse recycled aggregates can become more favourable than natural aggregates (23% lower GWP in concrete production), when fine recycled aggregates (e.g. recycled/secondary sand) are also used in concrete production via means of selective demolition instead of being landfilled depending on transportation distances (Estanqueiro et al., 2018).

A mixed proportion of 75% recycled and 25% natural aggregates in concrete production was found to provide environmental benefits with respect to all impacts, while this proportion is able to produce concrete blocks with favourable mechanical and durability performances (Guo et al., 2018). This is in line with another statement reporting that replacement of 30% w/w of total aggregates used in concrete mixture by recycled aggregates may lead to considerably better environmental performance (Kua and Kamath, 2014). In **Figure 5-7**, a qualitative comparison between concrete produced by natural and recycled coarse aggregates based on the above-mentioned evidence is provided.

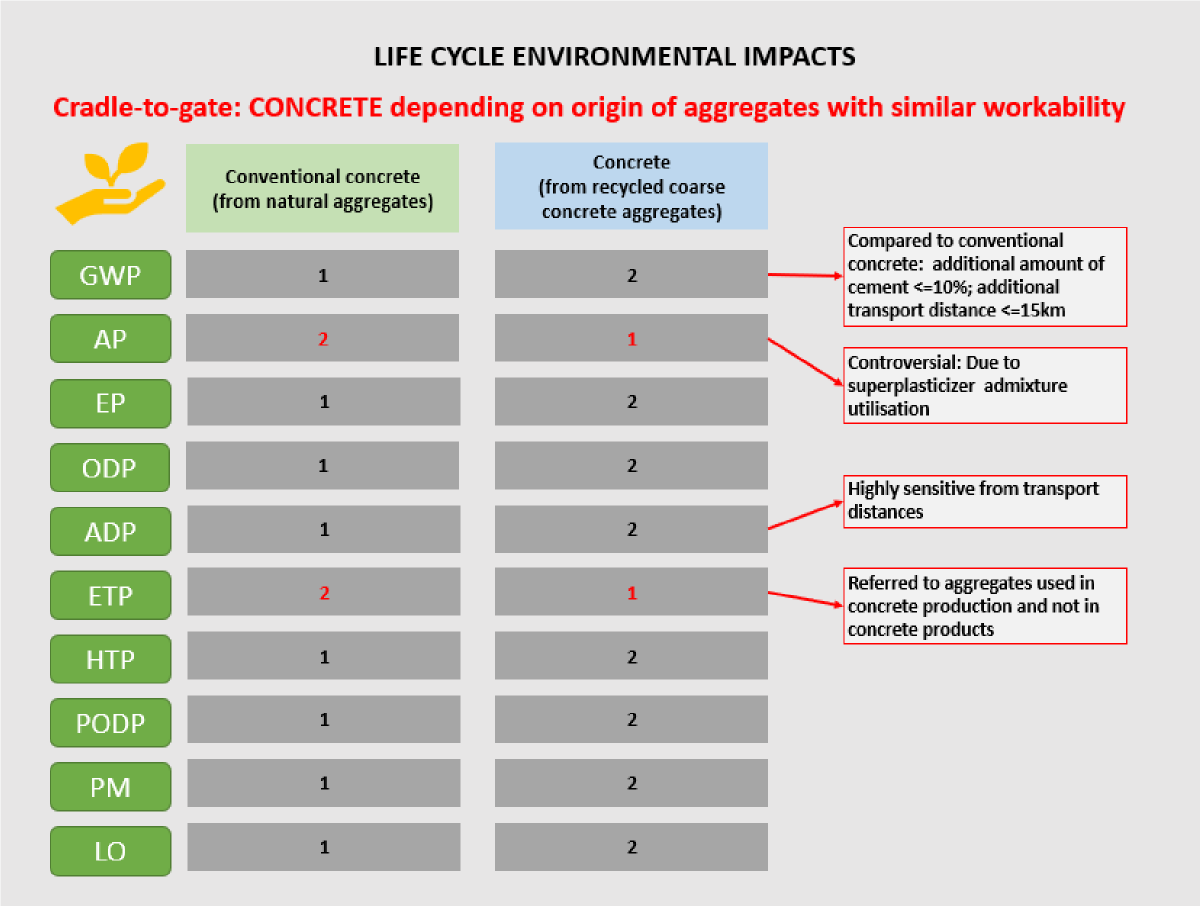

Figure 5-7 Scoring that indicates the contribution of concrete produced by natural (e.g. gravel) and recycled coarse aggregates against all LCA impacts (value 1 indicates the highest contribution). Abbreviations: Global warming potential (GWP); Acidification potential (AP); Eutrophication potential (EP); Ozone layer depletion potential (ODP); Abiotic depletion potential (ADP); Ecotoxicity potential (ETP); Human toxicity potential (HTP)l; Photochemical ozone creation potential (POCP); Particulate matter (PM); Land occupation (LO). Adapted by (Estanqueiro et al., 2018, Knoeri et al., 2013).

The functional unit to quantify the environmental impacts of concrete products (e.g. 1 m^3 of concrete block, as in **Figure 5-3**, or strength) is critical, as different functional units may offer a different understanding of the impacts. We found one study (Serres et al., 2016) that investigated the environmental impacts of concrete samples using functional units from two perspectives: i) concrete samples made with natural and recycled (gravel, sand and terracotta brick) aggregates with the same volume composition of the granular skeleton and a grain size 8 mm (**Figure: 5-8-A**); ii) concrete samples made with varying mixture of natural, and recycled aggregates having identical strength with a grain size 20 mm (**Figure: 5-8-B**). It is worth noting that the recycled aggregates used were secondary aggregates, and not recycled aggregates from CDW. **Figure 5-8** shows that the recycled concrete samples performed better than traditional and mixed concrete samples even if the utilization of recycled materials involved more operations, such as crushing (Serres et al., 2016). The only exclusion was the category of AP, in which recycled materials had greater impact than traditional samples (consisting of natural aggregates) due to the use of superplasticizer admixture applied to recycled samples in order to increase the fluidity of the fresh concrete reaching the desired workability (Serres et al., 2016). The concrete sample produced by recycled brick aggregate provided the lowest indicator of environmental impacts among concrete samples due to exhibition of low aggregate density and good mechanical strength leading to decrease of transport-related impacts and improvement of its lifetime, respectively (Serres et al., 2016).

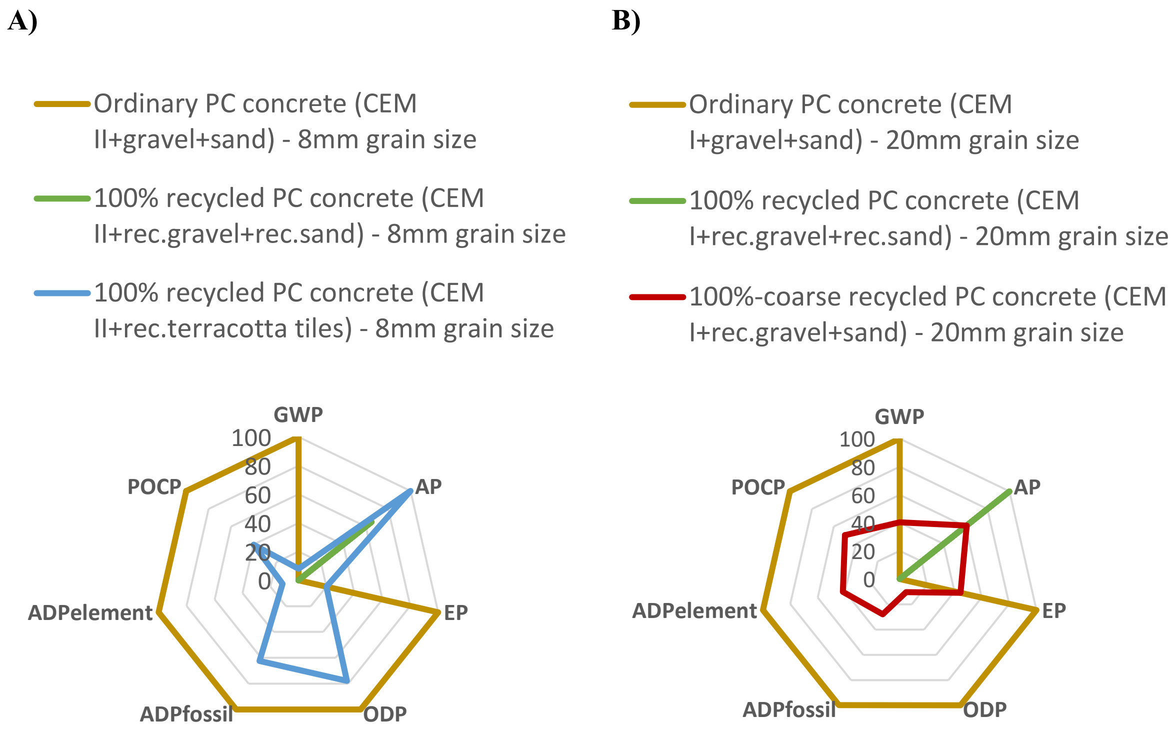

Figure 5-8 Normalised environmental impacts of concrete samples with: same volume composition (A); or same strength (B); depending on the source of aggregates (natural vs recycled gravel; natural vs recycled sand; and natural aggregates vs recycled terracotta tiles) – with respect to GWP, AP, EP, ODP, ADP, and ETP from a cradle-to-gate LCA considering production of raw materials (gravel, cement, aggregates and admixtures); transportation; concrete production. Abbreviations: PC: Portland cement. Note: Min-max normalisation applied to concrete samples for each impact to visualise the sample with the greatest environmental burdens. Adapted by Serres et al. (2016).

The contribution to environmental impacts of each stage involved in concrete's life cycle is presented in **Figure 5-9**, using a scoring system with values ranging from 1 to 5 (score value 1 indicates the highest environmental impact, and score value 5 indicates the lowest contribution to environmental impacts). Cement production has a considerable contribution to most LCA impacts (e.g. 30-40% for GWP, AP, EP, and ADP_{fossil}) due to use of fossil fuels in electricity production and use, largely for the calcination of limestone, followed by aggregate extraction (e.g. 20-35% for GWP, AP, EP, and ADP_{fossil}). A study in China reported that cement production followed by transportation were the top two contributors to GWP (81-87% and 9-12%, respectively) and to energy consumption (81-87% and 11-14%, respectively). This evidence is in line with a cradle-to-gate LCA study in France demonstrating that the contribution of cement production to total impacts had a range of 71.2-95.9%, followed by the production of aggregates (0.9-9.6%) and concrete production (0.3-5%), while the contribution of transport ranged from 2.1 to 22.4% depending on the origin of aggregates (e.g. recycled or natural), the transport distance, the structural application, and the impact category (Serres et al., 2016).

LIFE CYCLE ENVIRONMENTAL IMPACTS SCORECARD — CONCRETE

	EXTRACTION	PRODUCTION	CONSTRUCTION	SERVICE LIFE	RECOVERY (BACKFILL)	RECYCLING	LANDFILL
GWP	2	1	Unspecified	Unspecified	3	3	3
AP	1	1	Unspecified	Unspecified	2	2	2
EP	2	1	Unspecified	Unspecified	3	3	3
ODP	2	1	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified
ADP	2	1	Unspecified	Unspecified	3	3	3
ETP	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified
HTP	2	2	Unspecified	Unspecified	1	1	Unspecified
PODP	2	1	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified
PM	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified
LO	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified

Numbers in red: Scoring based on whether diesel oil is consumed by the wheel loader at the facility

Figure 5-9 Scoring that indicates the contribution of concrete production, use and management against all LCA impacts (value 1 indicates the highest contribution). The absence of information on specific stages (e.g. construction and service life) results in score values that compare only stages with existing data. Boxes with red values indicate that findings are controversial. Abbreviations: Global warming potential (GWP); Acidification potential (AP); Eutrophication potential (EP); Ozone layer depletion potential (ODP); Abiotic depletion potential (ADP); Ecotoxicity potential (ETP); Human toxicity potential (HTP); Photochemical ozone creation potential (POCP); Particulate matter (PM); Land occupation (LO).

As shown in **Figure 5-9**, the stage of recovery (as EoL option) may have a small contribution to the net environmental impacts. Nonetheless, it was reported that in the HTP impacts, the recovery stage can contribute to ca. 34% HTP, if diesel oil is consumed by the wheel loader at the facility (Kua and Kamath, 2014). A detailed list of the results on environmental impacts of concrete materials as received reporting basis can be found in the **Appendix** (**Table A.2**). It is worth noting that there are no score values 4, or 5, on the scorecard.

5.2.2. Bricks

In the UK, the majority of bricks produced (ca. 96%) are manufactured from clay (MPA, 2013) and therefore the life cycle of clay-based bricks constitutes a considerable contribution to environmental impacts in the UK construction sector. Notwithstanding the need to explore these impacts, recent literature evidence on this aspect is very limited.

LCA studies have shown that the binder content can play a critical role in the environmental impacts of bricks (Seco et al., 2018). A cradle-to-gate LCA study of four clay-based brick products developed in the UK typically consisting of lower Oxford clay (90-92% w/w) stabilised with hydrated lime (3-8% w/w), GGBS (7% w/w) and PC (3-10%) looked into the variations in the environmental impacts of bricks produced under different mixtures/ proportions (Marcelino-Sadaba et al., 2017). The clay-based bricks produced had a compressive strength, set by the British Standard for concrete masonry units (BS 6073-2:2008), within 4-8 MPa, with the highest in GGBS-containing bricks. The environmental impacts of the four types of clay-based bricks are presented in **Figure 5-10**. As observed in **Figure 5-10**, clay-based bricks (92% w/w) stabilised with lime (8% w/w) have the worst environmental performance against most LCA impacts (AP, ADP, ETP, HTP and POCP) compared to the other types of bricks, due to the lower amount of clay used in the other three samples (90% vs 92%). The bricks that contained GGBS were the most environmentally-friendly products. Unsurprisingly, the bricks with the PC content had higher contribution to GWP, ODP, and EP mostly related to cement production. The use of waste and by-product materials (e.g. GGBS) for partial and whole replacement of traditional binder (e.g. lime and Portland cement) may lead to significant environmental benefits (Marcelino-Sadaba et al., 2017, Seco et al., 2018).

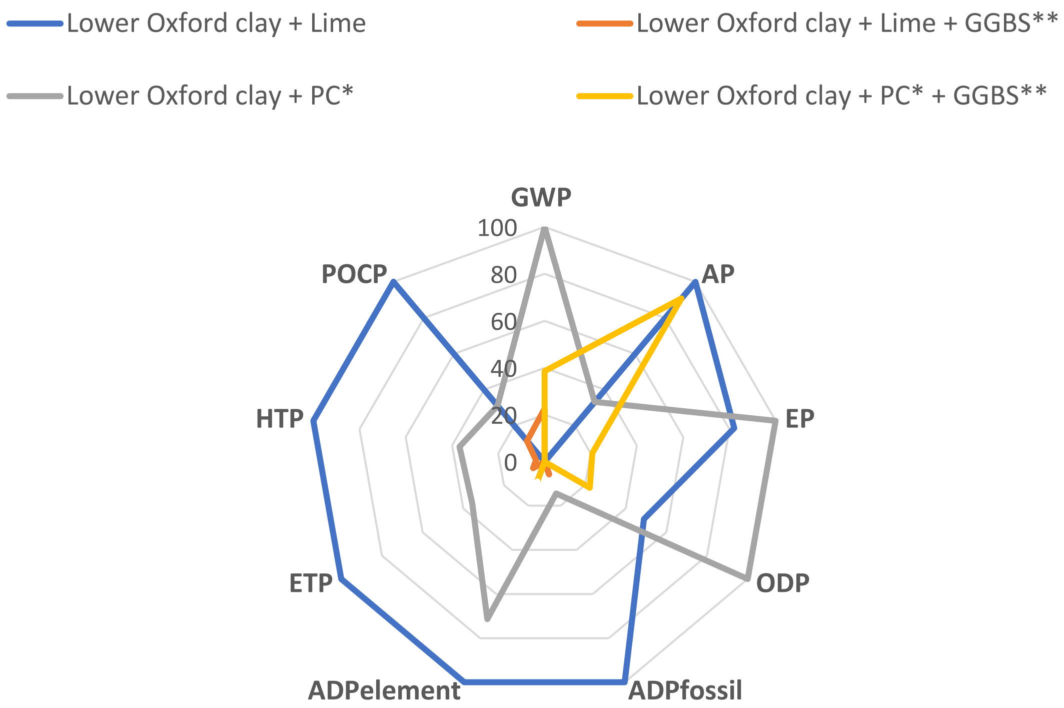

Figure 5-10 Normalised environmental impacts of common clay-bricks per unit strength in the UK – stablished with combinations of Portland cement, hydrated lime, Blast-furnace slag, and MgO – with respect to GWP, AP, EP, ODP, ADP, ETP, HTP, and POCP arising from cradle-to-gate LCA considering production of raw materials (e.g. aggregate extraction and PC production), transportation, and brick production. Abbreviations: PC: Portland cement; GGBS: ground granulated blast-furnace slag. Note: Min-max normalisation applied to brick samples for each impact to visualise the sample with the greatest environmental burdens. Adapted by Marcelino-Sadaba et al. (2017).

The results from the UK-based study on bricks were compared with an LCA study of three marl (or calcareous) clay-based bricks in Spain, and a common fired clay brick in terms of GWP and ADP_{fossil}, as presented in **Figure 5-11** (Marcelino-Sadaba et al., 2017). The superiority of marl-based bricks compared to lower Oxford clay-based bricks in terms of low environmental impacts can be seen in **Figure 5-11**, indicating that the soil characteristics in different regions (here at country level) can play a pivotal role in the environmental impacts of brick production. **Figure 5-11** demonstrates that unfired clay-based bricks have lower environmental impacts compared to fired clay bricks, and therefore the applicability and limitations of the use of unfired systems has to be identified resulting in the adoption of complementary rather than competing actions between the fired and unfired systems. The environmental footprint of clay bricks can be significantly reduced through the minimization of transport distance of raw materials with the use of locally available resources, whilst the utilisation of alternative fibre materials (e.g. sawdust and/or wheat straw) can lead further to a considerable improvement of environmental performance of the end-product in terms of GWP, AP, ADP, ETP, HTP, and POCP (Christoforou et al., 2016). The life cycle environmental impacts of ceramic brick production could be also reduced by focusing on the filtration of fine particles since their emissions released during combustion constituted the main contributor to human health indicators, while attention should also be placed on the transportation steps investigating alternative measures such as use of biofuels and/or shipment by boat or train (de Souza et al., 2016).

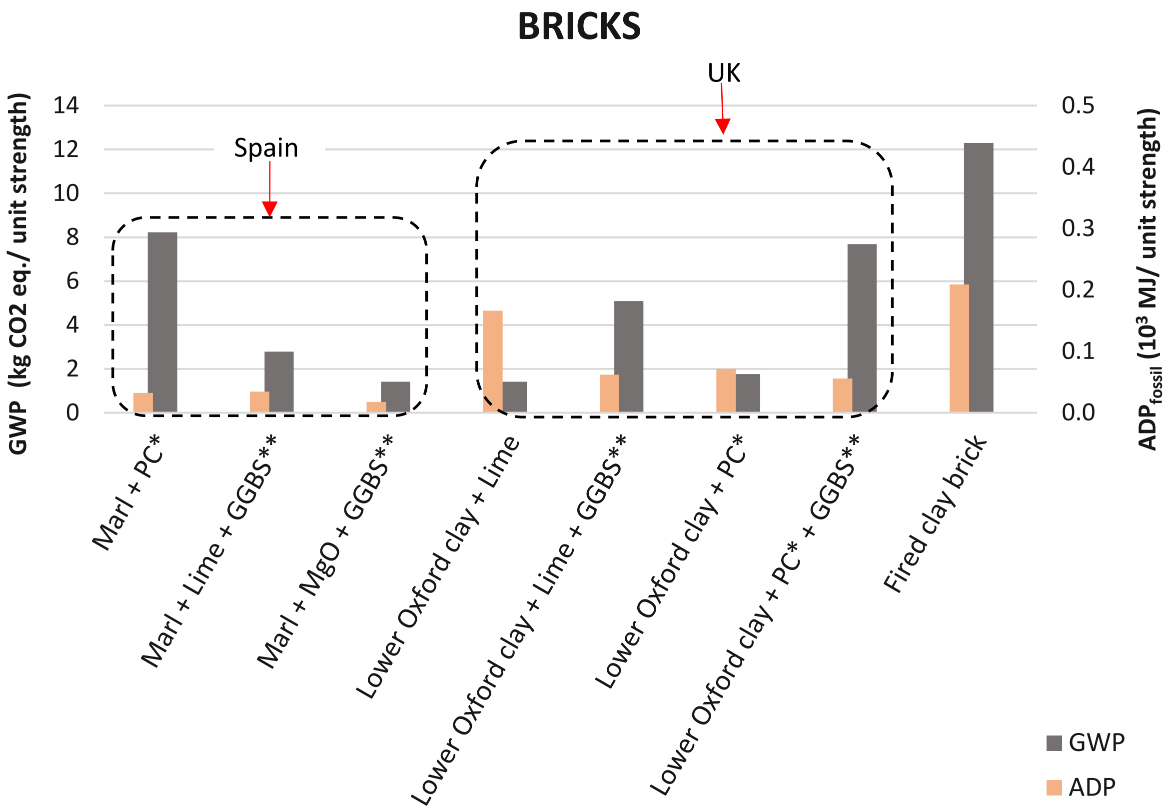

Figure 5-11 Environmental impacts of clay-bricks stablished with combinations of Portland cement, hydrated lime, Blast-furnace slag, and MgO, expressed per unit strength, with respect to GWP and ADP_{fossil} arising from cradle-to-gate LCA considering production of raw materials (e.g. aggregate extraction and PC production), transportation, and brick production. Abbreviations: PC: Portland cement; GGBS: Ground granulated blast-furnace slag. Adapted by Marcelino-Sadaba et al. (2017).

Limited information was identified regarding the contribution of each stage of the life cycle of brick system to total impacts. **Figure 5-12** illustrates the contribution of each stage for fired clay bricks at each stage of their life cycle using a scoring system with values ranging from 1 to 5 (score value 1 indicates the highest environmental impact, and score value 5 indicates the lowest contribution to environmental impacts).

LIFE CYCLE ENVIRONMENTAL IMPACTS SCORECARD — CLAY BRICKS

	EXTRACTION	PRODUCTION	CONSTRUCTION	SERVICE LIFE	RECOVERY (BACKFILL)	RECYCLING	LANDFILL
GWP	2	1	Unspecified	Unspecified	Unspecified	Unspecified	3
AP	2	1	Unspecified	Unspecified	Unspecified	Unspecified	3
EP	2	1	Unspecified	Unspecified	Unspecified	Unspecified	3
ODP	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified
ADP	2	1	Unspecified	Unspecified	Unspecified	Unspecified	3
ETP	1	1	Unspecified	Unspecified	Unspecified	Unspecified	2
HTP	2	1	Unspecified	Unspecified	Unspecified	Unspecified	3
PODP	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified
PM	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified
LO	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified	Unspecified

Figure 5-12 Scoring that indicates the contribution of clay bricks production, use and management against all LCA impacts (value 1 indicates the highest contribution). The absence of information on specific stages (e.g. construction and service life) results in a scoring based only on existing data. Landfill was considered as EoL management option due to the absence of information on the contribution of other EoL options. Abbreviations: Global warming potential (GWP); Acidification potential (AP); Eutrophication potential (EP); Ozone layer depletion potential (ODP); Abiotic depletion potential (ADP); Ecotoxicity potential (ETP); Human toxicity potential (HTP); Photochemical ozone creation potential (POCP); Particulate matter (PM); Land occupation (LO)

The stage of drying and firing was reported as the greatest contributor to the total energy requirements of brick production (e.g. 87%), followed by clay mixing and forming (e.g. 8%), while extraction, transportation and EoL (e.g. landfill) ranged at relatively low levels across the full life cycle (e.g. 5%) (Kua and Kamath, 2014). As with other impacts (such as GWP, AP, EP, and HTP) the contribution of extraction, production, distribution, use, and landfill as an EoL option across the full life cycle, was found under different percentages, although the most prevalent was the stage of production (de Souza et al., 2016). A detailed list of the results on environmental impacts of clay bricks as received reporting basis can be found in the Appendix (**Table A.2**).

5.3. Construction, demolition and excavation activities and waste

Construction, demolition and excavation activities give rise to a large volume of waste materials; CD&E waste. This waste stream is generated by the economic activities of construction, maintenance, and demolition, and/or deconstruction of buildings, transport networks and other structures. It is an extremely heterogeneous waste stream, the amount and composition of which can vary widely depending on the type of structure, and activities carried out on-site. Frequently, this waste stream is characterised as inert, because the majority of materials present in it are excavated soil (topsoil, and subsoil and rock, also known as spoil) and aggregates, which are considered to have a low environmental impact upon disposal.

Amongst the key activities that may take place at the construction stage, e.g., site clearing, dewatering, excavation, pit support and backfilling, soil excavation can contribute the biggest amount of carbon emissions (Devi and Palaniappan, 2017). In soil excavation, the energy use accounts for 14-89 MJ m^{-3} and 19-135 MJ m^{-3} including transportation (excavation and transport of soil); depending on several technological and operational parameters (Devi and Palaniappan, 2017). In terms of technological parameters, the mobile equipment used during soil excavation such as excavators, scrapers, cranes, bore/drill rigs, tracks, rubber-tired loader, etc. is an important source of air pollution, due to fuel consumption that may vary depending on the capacity of equipment, fuel quality, operation conditions (e.g. engine cycles), engine maintenance, and engine technology (Wang et al., 2016, Devi and Palaniappan, 2017). However, research on related emissions is missing, indicating the imperative importance of their quantification at local level (Wang et al., 2016).

In relation to operational parameters, excavation depth and volume, location, work duration, weather conditions, soil type and slope, traffic conditions, power source, operator's experience, and distances play also a critical role (Devi and Palaniappan, 2017). The depth of excavation and the morphology of soil provide different environmental footprint depending on the excavation resistance (Devi and Palaniappan, 2017). For example, one study reported that the excavation of clay induces higher environmental impacts (energy use and CO_2 emissions) than the excavation of sand and gravel, sandy clay and loam (Lewis and Hajji, 2012), while rock excavation has higher impact on GWP than excavation of clay, sand and soft soil (Forsythe and Ding, 2014, Devi and Palaniappan, 2017). Evidence also demonstrated that there is a positive correlation of the site slope with GWP (Forsythe and Ding, 2014). Improved control and monitoring of on-site carbon and dust emissions from excavation activities, and the increase in the duration of excavation-related projects (to ensure that the processes are carried out with care to prevent dust emissions), need to be carefully considered to improve the environmental performance of soil excavation (Giunta, 2020).

Topsoil excavation (i.e. the stripping of a layer with a thickness range of 15-30 cm) (Giunta, 2020), can result in high PM emissions which were found to be at 5.7 kg km^{-1} with the fine particulate matter (PM_{10}) representing ca. 3.42 kg km^{-1} (Giunta, 2020). For example, in motorway construction, topsoil excavation can be a significant contributor (ca. 38.9-87.8%) to emissions of fine particulate matter (PM_{10}) than deep excavation and construction depending on the processes applied to worksite (e.g. rock crushing, machinery used and concrete used in construction) (Giunta, 2020). Specifically, in the stage of topsoil excavation for road construction, the most influential factor is the transit of tracks in unpaved roads (up to 80%) followed by the excavation of topsoil (up to 50%) depending on the function of worksite, while processes related to trucks download and equipment operation are lower (up to 14%) (Giunta, 2020). In the next stage of deep excavation and road construction, evidence indicated that fine emissions were higher at stages of crushing attributed to tertiary crushing (5-25 mm) and screening, followed by transit of tracks and loading of excavated materials in tracks (Giunta, 2020).

5.3.1. Excavated soil and rock

Topsoil is a finite resource, and can be used in the landscaping of construction process, but first it needs to be removed from the construction site. The reason is that topsoil and subsoil may include plant roots (from trees and other plants) and organic material in varying stages of decomposition, and as a result it has to be removed in order: i) to prevent later attempts of plant growth that may damage the structure, and/ or affect its stability; 2) to prevent moisture evaporation that could also damage the structure. In regards to the former, even the smallest of plant life can exert considerable pressure on the construction materials used (whether it is road construction, pavements or other structures), which could create ruptures. Topsoil is generally soft and compressible and retains moisture to sustain plant growth during dry periods, which is why it is necessary for it to be removed from construction sites. Subsoil has a distinctly different structure to topsoil and usually contains a higher clay content. Plant roots penetrate through this layer and thus it has to also be removed to ensure stability and longevity of the structure. A typical soil profile is shown in **Figure 5-13**.

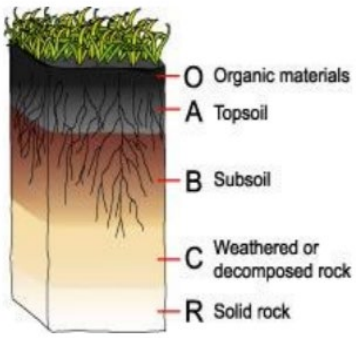

Figure 5-13 A typical soil profile (Adopted from: Mine (2014))

To gain an insight into the environmental impacts of excavated topsoil (O-B in Figure 5-13) and spoil (C and R in Figure 5-13), we looked at the impacts arising from their excavation (i.e. removal), and their end of life fate. Site clearance for construction activities necessitates the removal of vegetation, as well as the excavation of topsoil and subsoil. Topsoil and subsoil excavation results in morphological changes, and modification of the natural drainage. Due to this change, soil erosion might be increased, whilst it may reduce soil fertility and productivity *in situ*, adversely affecting plant growth. The use of heavy equipment can destroy the natural structure of the soil, which in turn can reduce water infiltration, reduce the aeration of the soil and decrease the moisture retention capacity of the soil. Compaction also makes soils difficult to work with and can lead to considerable soil erosion.

It must be noted that topsoil or soils are regarded as waste, and moreover, they are unfit for disposal to inert landfill due to their biologically active nature. For spoil (rock) to be disposed to inert landfills, it has to undergo treatment in specialised facilities. This is to ensure that the waste soil will not produce a hazardous leachate and does not contain a significant organic content, sulphate concentration, or any other matter that is likely to give rise to environmental pollution or harm to human health (GEA, 2020, EA, 2010, DEFRA, 2009). This follows the waste duty care of practice, and particularly the Waste Acceptance Criteria (WAC), which the producer must comply to when disposing of the soil waste. There are options under which clean soils can be moved from one site to another such as the U1 waste exemption (allows the use of up to 1kt of uncontaminated soils and stones in construction projects, as a 'non-waste') and the CL:AIRE DoWCoP (allows the use of clean soils from one site to another, following the development of a Materials Management Plan (**MMP**)[4]). Generally, clean soil resources generated by excavation activities can be re-used on site for landscaping (topsoil), or where practicable to form embankments (subsoil). This necessitates soil to be stripped and stockpiled to prevent excessive damage and degradation.

However, construction activities are generally carried out as one operation, which results in the mixing of topsoil and subsoil. This can have adverse impacts on the topsoil disposal/ management pathways. For instance, it can:

- contaminate soil as a result of accidental spillage or the use of chemicals, which means that it will have to be disposed as a hazardous waste material in landfills;

[4] MMP: demonstrates that the use of the material will not pose a risk to human health or the environment, it is suitable for use and its intended use is legal; the MMP is reviewed by a CL:AIRE Qualified Person, who then completes and submits an online declaration that signifies the approval of transferring soils. The CL:AIRE Qualified Person's fee and an administration fee is payable to CL:AIRE, which is based on the quantity of material being transferred (up to 5,000m3 is free of charge; above that the fee is £10 per 1,000m3 i.e. for 6,000m3 the fee is £60)

- mix topsoil with subsoil, diluting its nutrient rich quality, and potential end-use, which means that it may be used in landfill restoration, restoration of spoil tips, etc. or other landscaping applications;
- mix topsoil with construction waste or contaminated materials, leading to disposal on landfills.

Furthermore, the mishandling of soils can have an adverse effect on their properties (e.g. fertility, permeability, and ecological diversity), and it may also increase the risk of flooding and off-site discharges if illegally, or not properly, disposed of (DEFRA, 2009). It may also lead to human health impacts due to PM and dust emissions. For instance, Li et al (2010) found that dust accounted for 27% of total impacts, emphasising that techniques able to decrease the construction dust can effectively mediate the environmental burdens arising from site cleaning, excavation and backfill (Li et al., 2010). Water spray can lower dust emissions and can provide significant benefits in respect to natural resources preservation (Li et al., 2010).

Soil mishandling can impact on its disposal / management pathways, yet the greatest effects could occur during its storage and stockpiling. Stockpiling of soils can take place in landfills or dedicated areas; for the latter there is no evidence, and therefore we are left to assume that the most common stockpiling site is landfills. Stockpiling the soil for long periods of time can lead to the destruction of the soil's structure and nutrient availability. It must be highlighted that, as consistently supported in different studies, the most common disposal/ management pathway of excavated soil and rock is landfilling, followed by backfilling application, recycling (e.g. aggregates for use in new products, or as cover in landfills and quarries), as well as reuse on-site, or in restoration activities in other construction projects (Magnusson et al., 2015). **Figure 5-14** provides an insight into the disposal/management pathways of excavated soils in England.

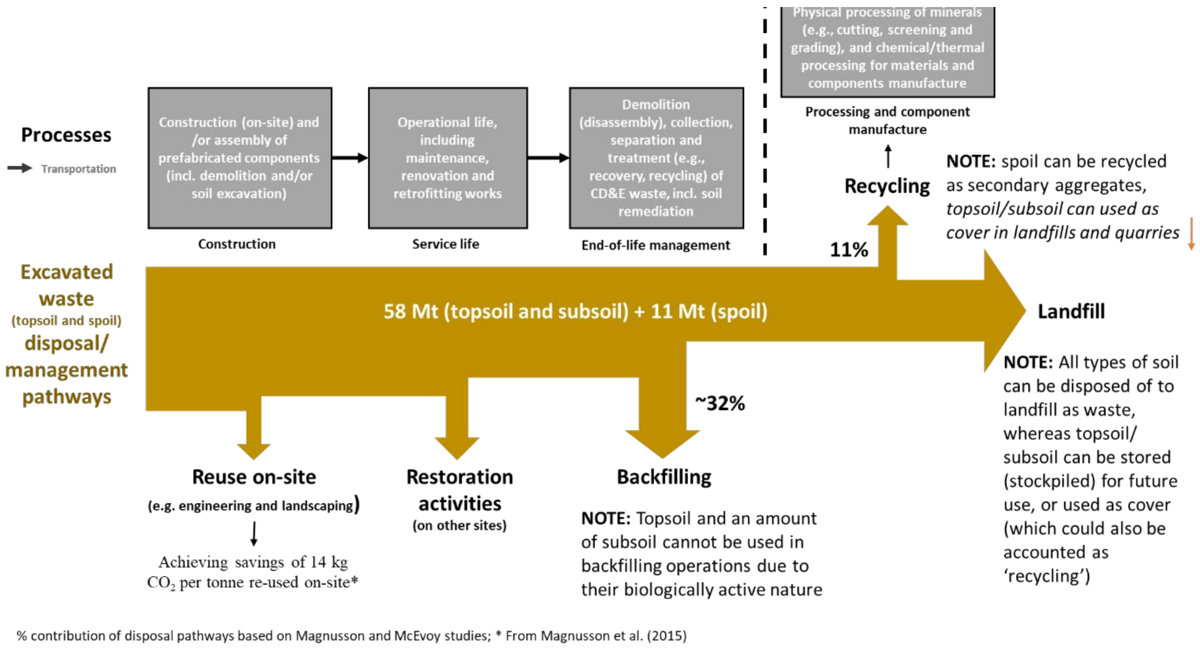

Figure 5-14 Excavated soil and rock disposal/ management pathways in the UK in 2016. The percentage contribution to disposal pathways was difficult to decipher; any percentages presented in the figure are estimates of soil waste sent to final treatment at recycling and/or other recovery operation including backfilling reported in the literature.

The tonnage shown in **Figure 5-14** are based on the UK waste estimates in 2016 (DEFRA, 2020). It is common for the excavated waste to be accounted for as part of construction and demolition waste. This makes it difficult to estimate the quantities of excavated soil and rock in the UK, or even get proxies from other areas, due to the lack of such information from the existing scientific literature (McEvoy et al., 2004, Magnusson et al., 2015).

Regarding the environmental impacts of disposal / management pathways, several studies support that the reuse of excavated soil and rock on-site or for restoration purposes contributes the least to environmental impacts, due to the reduction in transport, and an associated reduction in air pollutants (Haas et al., 2020). For example, Magnusson et al. (2015) stated that the reuse of excavated soil and/ or rock reuse can reduce CO_2 emissions from fuel savings to about 4 kt CO_2-eq for 700, 000 m^3 of excavated material reused on site. Similarly, Chittoori et al. (2012), who described the cost and environmental benefits of reusing excavated soil within a pipeline construction project reported that reuse on-site reduced the soil and rock management costs and climate impacts by 85%.

Nonetheless, reuse of soil on-site or in other projects could require the use of a stabilisation agent, i.e. an additive that leads to chemical reactions that stabilizes soft soil and enhances its geotechnical properties when used as fill materials. Cement and lime stabilization are commonly used in soil treatment, which could result in an increase in the environmental impacts associated with additive materials production, but other more sustainable binder materials such as fly ash and sewage sludge ash can also be mixed with the soil (Magnusson et al., 2015). The latter has been found to provide additional benefits, such as binding heavy metals and reducing potential leaching of those to the soil and groundwater.

Another management pathway is soil stockpiling in landfills, which could cause soil compaction. This can in turn lead to a loss of soil structure, damage the soil's physical, chemical and biological condition, cause organic material and nutrients leaching, and result in contamination (England Highways, 2014). For instance, when soil is stockpiled for longer than two weeks, and is not properly aerated, anaerobic conditions could occur in the centre of the pile that may lead to chemical and biological changes, and methane (**CH_4**) emissions, a GHG that is 23 times more potent than CO_2 emissions in a 100-year horizon. The stockpiling of soil for more than a year can lead to irreversible damages, whereas potential erosion could result in a pollution to the local environment, particularly during wet weather events. Nonetheless, compaction at the landfill sites can also prevent water and air infiltration from the surface, and thus decrease heavy metal leaching potential and limit oxidization (Katsumi, 2015).

A quantitative life cycle assessment of environmental impacts of earthwork construction including most common unit processes such as site cleaning, dewatering, excavation, pit support (e.g. soil nailing walls and slope protection piles) and backfilling in China (about 60,000 m2), categorised into ecosystems, natural resources and human health damage, were examined (Li et al., 2010). Authors reported that human health damage accounted for 27% of total impacts due to construction dust indicating that techniques able to decrease the construction dust can effectively mediate the environmental burdens arising from site cleaning, excavation and backfill (Li et al., 2010). In relation to unit processes, pit support process accounted for 59.4% contribution to total impacts due to the large consumption of steel – reported as the greatest contributor among other ancillary materials (50%) – resulting in a significant amount of resource and energy consumption and pollutant discharge during its manufacturing process and therefore its restricted consumption may provide considerable environmental benefits (Li et al., 2010). The next most significant contributors to total impacts after pit support were excavation (18.3%), site cleaning (12.3%) and backfill (7.5%), while health damage contained the greatest portion (83%) of impacts for these three processes indicating the need for construction dust control (e.g. water-spray) (Li et al., 2010). From the perspective of ancillary materials, the contribution of energy and fuel consumption to total impacts accounted for less than 10% and therefore the alteration of construction equipment into more environmentally friendly might not induce any significant environmental benefit (Li et al., 2010).

5.3.2 Construction and demolition waste (CDW)

Insights into the impact of CDW management along the full life cycle in the construction sector were obtained by studies that conducted LCA from three different perspectives: i) cradle-to-grave LCA of buildings although in most cases extraction and production stage was not included; ii) LCA of virgin/natural versus recovered/recycled mixtures of construction materials; and iii) LCA of different CDW management options. Literature evidence related to these perspectives is summarised in **Table 5-3**.

Table 5-3 LCA case studies in CDW management across the life cycle stages.

Material	*Life cycle stage - Focus*	*Country*	*Main finding*	*Considered impact of leaching*	*Ref.*
Inert CDW (soil, concrete, ceramics)	Integrated landfill, recovery (as in backfilling), recycling under different ratios	Brazil	The increase of recycling rate tends to reduce the environmental impacts, but the transport distance for recycling should not exceed the transport distance for landfill more than 312%	No	(Penteado and Rosado, 2016)
Aggregates of CDW (stone, sand and soil)	Landfill vs recycling (used in concrete structures)	India	Recycling is a better CDW management option than landfilling in terms of GWP and material recovery, but environmental trade-offs arise from the energy intensive recycling process in terms of eutrophication and mineral resource scarcity	No	(Jain et al., 2020)
Mineral CDW (concrete and masonry debris)	Landfill vs recovery (unbound aggregates)	Denmark	Recovery is a better CDW management option than landfilling for non-toxic impact categories, while recovery (as in backfilling) has higher impacts than landfilling for toxic categories due to higher leaching of oxyanions	Yes	(Butera et al., 2015a)
Mineral CDW (concrete and masonry debris)	Recovery (unbound material for road construction or landfill) vs extraction (extraction from gravel pit and quarry excavation)	Denmark	Leaching emissions at the end-of-life are significant for toxicity impacts and should not be neglected by cradle-to-grave LCA studies	Yes	(Butera et al., 2015a)

Aggregates	Recycling (for concrete) & recovery (for road construction as unbound material) vs gravel extraction	Italy	Recycling/ recovery is not as sustainable as expected due to environmental loads arising from transportation, while the adoption of an integrated facility that produces both recycled and virgin aggregates may promote sustainability	Yes	(Faleschini et al., 2016)
Mineral industrial waste	Recovery (filling material in road and earth constructions)	Germany	Heavy metals were retarded by 100% by layer of 2m clay/silt subsoil indicating the fundamental role of subsoil. The major challenge of leaching consideration in LCA frameworks is the high variability in micro-scale technical and geographical factors	Yes	(Schwab et al., 2014)
Coarse (used for concrete production) and fine (used for backfilling) aggregates	Landfill vs Recovery (road base and filling materials in road construction) vs recycling (used for concrete production)	Belgium	Environmental impacts at the end-of-life can be reduced by 36-59% with recovery of CDW waste compared to landfilling depending on the efficiency of sorting strategy during recycling	No	(Di Maria et al., 2018)
CDW	Current CDW waste management system (79% w/w recovery rate) vs two alternative scenarios (75% and 74% w/w recovery rate through different operations)	Finland	The system produced environmental benefits in terms of GWP and ADP excluding the critical role of transportation, but still behind the target of recycling 70% as specified by the EU Waste Framework Directive (2008/98/EC) indicating the need of major changes within the system	No	(Dahlbo et al., 2015)
CDW (EC code 17 09 04 (80&w/w) AND 17 01, 17 03 02, 17 08 02 (20% w/w))	Current CDW waste management system (19.8% landfilling and 80.2% recovery) vs landfill vs best-case scenario (90% recycling and 10% recovery)	Italy	The system induced higher impacts than the savings arising from natural resource consumption mainly due to transportation	Yes	(Borghi et al., 2018)
Commercial building	Production vs construction vs use (operation and maintenance) vs end-of-life (with 10% landfilling)	UK	Use was greatest contributor to CO_2 emissions including operation (45.6%) and maintenance (21.8%), followed by production (31%), end-of-life (1.1%), and construction (0.5%)	No	(Brooks et al., 2021)

			indicating that the adoption of clean technologies and low carbon strategies may lead to reduction of CO_2 emissions by 22%		
Common types of houses: detached, semi-detached and terraced	Construction vs use vs end-of-life (25% reuse, 68% recycling and 8% landfill)	UK	In terms of GWP, the contribution of use was 90%, followed by construction with 9%, and end-of-life with 1%, while a similar pattern observed for other impacts excluding ODP where construction was the main contributor	No	(Cuéllar-Franca et al., 2012)
Residual building	Recovery vs overall life cycle	Italy	Recycling of reinforced steel may lead to considerable environmental savings accounted for 65% in particulate matter, 89% in GWP, and 73% in mineral extraction, while recycling of aggregates induced significant savings in land occupation and non-renewable energy potential	No	(Vitale et al., 2017)

Using the information collected in **Table 5-3** we were able to construct **Figure 5-15** that summarises the contribution of CDW to environmental impact categories, at each stage of their life cycle using a scoring system with values ranging from 1 to 5 (score value 1 indicates the highest environmental impact, and score value 5 indicates the lowest contribution to environmental impacts). Note that **Figure 5-15** refers to a mixture of construction materials with the greatest part consisting of aggregates (e.g. concrete, masonry, ceramics, and soils). It is worth noting that the scoring system developed was based on the information collected from our literature review. Due to a general lack of distinction between recovery and recycling management options, the same score values were allocated in both options. Moreover, it is worth noting that in case where information collected is limited to one, or two stages of the life cycle, the score values allocated were 1 and 2 indicating which of the two stages (for which we have information), contributes the most to the specific environmental impact. For example, in the LCA category: PM, we found evidence on the end-of-life options (landfill, and recovery / recycling) and therefore score values ranged from 1 to 2. We also found that the stage of extraction was omitted in LCA studies of buildings and therefore extraction was excluded from scoring to avoid misinterpretations. Nonetheless, based on our insights form the evidence gathered we were able to provide a qualitative comparison with the EoL stage.

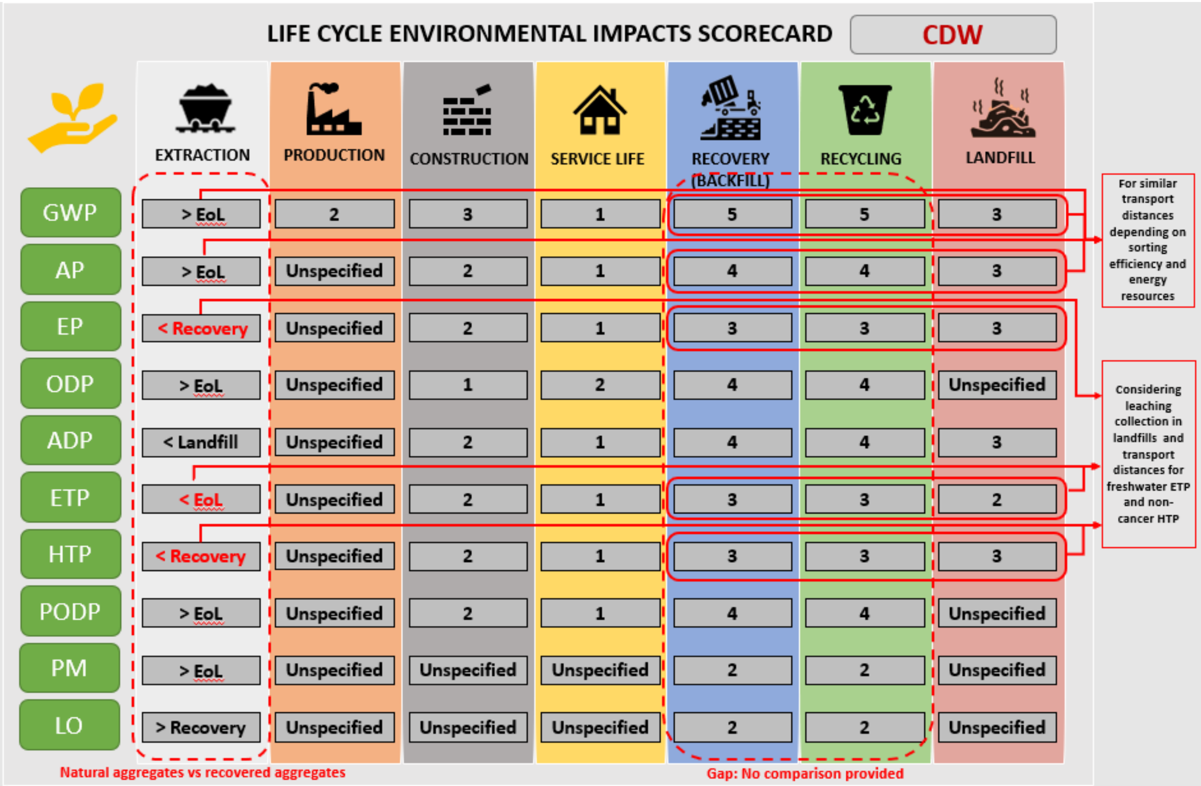

Figure 5-15 Scoring of stages of life cycle of CDW materials – a mixture of construction materials mainly consisting of aggregates – that indicate their contribution to prevalent environmental impacts across the full life cycle as received reporting basis under certain conditions. The absence of information on specific stages (e.g. production) results in score values that compare only stages with existing data. The stage of extraction was compared only with EoL management options due to the absence of information on the contribution of extraction across the full life cycle, while recovery and recycling scored with the same value due to lack of comparative evidence. Boxes filled in red indicate that findings are controversial. Abbreviations: Global warming potential (GWP); Acidification potential (AP); Eutrophication potential (EP); Ozone layer depletion potential (ODP); Abiotic depletion potential (ADP); Ecotoxicity potential (ETP); Human toxicity potential (HTP); Photochemical ozone creation potential (POCP); Particulate matter (PM); Land occupation (LO); End-of-Life (EoL)

Figure 5-15 aimed to represent the UK residential sector (Brooks et al., 2021, Cuéllar-Franca and Azapagic, 2012), whilst information was also obtained from other non UK-based studies especially for comparing EoL options with extraction stage. The usefulness of these findings and figure is justified by the fact that the housing sector in the UK represents 72% of the building stock, which is comprised of semi-detached (28%), detached (16%) and terraced (28%) houses. The total GWP of the building sector is estimated at 132 million tonnes of CO_2-eq. per year , with semi-detached, terraced and detached houses contributing by 38.6%, 34.1% and 27.3%, respectively (Cuéllar-Franca and Azapagic, 2012).

Role of EoL in LCA of buildings

As shown in **Figure 5-15**, service life is the greatest contributor to the majority of environmental impacts (GWP, AP, EP, ADP, ETP, HTP, and POCP) due to electricity consumption over the building lifespan, which can be over 50 years. Next in order is on-site construction, followed by the production stage of materials such as bricks, concrete, mortar. At the stage of construction, impacts are mostly related to the high consumption of clay bricks and lime mortar indicating that the reuse of bricks and recycled aggregates could provide important environmental benefits (e.g. 7% reduction of toxicity impacts and POCP) (Cuéllar-Franca and Azapagic, 2012). These findings are based on two cradle-to-grave LCA studies in the building sector in the UK; one focused on a commercial building including office spaces in Liverpool (Brooks et al., 2021), and the other, on the most common building types in the UK such as detached, semi-detached, and terraced houses (Cuéllar-Franca and Azapagic, 2012). In regards to ODP, on-site construction was found to have a greater contribution (in the ODP) than the use stage, and this is due to the use of expanded polystyrene insulation material, of which production is responsible for the release of ozone depleting agents such as hydrochlorofluorocarbons (**HCFCs**) (Cuéllar-Franca and Azapagic, 2012). In both studies, the EoL stage has a negligible contribution to the environmental impacts compared to the other stages, particularly for GWP (ca. 1%). This could be due to the fact that both studies assumed that ca. 10% w/w of total CDW goes to landfill, and the rest (90%) is reused, recycled and/or recovered (Cuéllar-Franca and Azapagic, 2012, Brooks et al., 2021).

Even though it is considered common practice for the service life of the building to be accounted in the life cycle environmental impact, it would be superlative if this was added for a time-period of one year as this would allow a sound comparison, as often materials used/ wasted from maintenance and restoration activities are unaccounted for; leading to false comparisons. Nonetheless, the substitution of conventional materials with sustainable alternatives (low carbon strategies) at the stage of construction (e.g. recycled steel, lightweight brick, and mortar with 40% PFA) has the potential to lead to a ca. 12% reduction in the embodied carbon (Brooks et al., 2021). Furthermore, the recycling of reinforced steel may lead to a 65% reduction in PM, 89% reduction in GWP, and 73% reduction in minerals extraction, and the recycling of aggregates could also result to significant savings in PM, and preservation of natural resources with respect to the EoL stage (Vitale et al., 2017).

Aggregate extraction vs CDW management in LCA

In **Figure 5-15**, the stage of extraction was compared with CDW management options following evidence from two comparative LCAs of natural/virgin aggregates (e.g. gravel, rocks and sand) versus recycled/recovered aggregates (e.g. concrete, masonry, ceramics, and soils) (Faleschini et al., 2016, Butera et al., 2015). The extraction stage was greater contributor than EoL stages to most impacts (ca. 70-95%) excluding toxicity impacts, such as freshwater eutrophication and non-cancer HTP in which recovery of CDW was contributing more. In regards to ADP, the landfilling of CDW ranked the highest, whereas freshwater ETP was high in both landfill and recovery due to oxyanion leaching (As, V and

Sb) and phosphate emissions (Butera et al., 2015). However, for the same impact categories (EP, HTP, and ETP), a study conducted in Italy reported that extraction of natural aggregates induced slightly higher impacts than recovery and recycling (Faleschini et al., 2016). For that reason, the boxes of these categories (EP, HTP, and ETP) in **Figure 5-15** were filled with red colour indicating that the relationship is controversial.

The interrelationship between natural aggregate extraction and CDW management is complex and depends on several factors including the source of natural aggregates (Butera et al., 2015), efficiency of recovery strategies (Gálvez-Martos et al., 2018, Di Maria et al., 2018), transport distance, and technical and geographical factors (e.g. waste/material composition, construction type and soil properties) (Schwab et al., 2014). For example, one LCA study conducted in Denmark reported that the crushing of CDW for recovery had greater contribution to environmental impacts (up to 30%) than the environmental savings from their use (due to avoided extraction of natural aggregates (5-15%)). This is due to the abundant nature of natural gravel (in Denmark) that did not require quarry excavation and crushing of rocks; meaning that savings from natural aggregates substitution were lower (ca. 10%) (Butera et al., 2015).

To increase the savings obtained by the recycling / recovery of aggregates, several actions are widely recommended at regional scale, such as: i) increase in the demand of recycled aggregates in the market through the enforcement of green procurement laws, dissemination of their technical properties and performance, and the restriction of quarrying activities; ii) improvements in the quality of recycled aggregates by adopting efficient sorting processes (e.g. selective demolition before recycling) and innovative technologies, and by incentivizing the authorization of recycling plants powered by electricity; iii) minimisation of transport distance; and iv) reduction of landfilling through imposition of higher disposal taxes and/or banning the disposal of recyclable fractions (Borghi et al., 2018). These recommendations are in agreement with Gálvez-Martos et al. (2018) recommendations on reducing environmental impacts and improving resource efficiency in the construction sector in Europe.

LCA of CDW management options

Even though there is a lack of comparative evidence between recovery (as in backfilling) and recycling of CDW, we were able to compare the contribution of these options to the environmental impacts against the landfilling of CDW (**Figure 5-15)**. Dahlbo et al. (2015) assessed the performance of a common CDW management system in Finland including a series of mass flows and treatment operations for each waste fraction (metal, concrete and mineral waste, wood, and miscellaneous waste) for different recovery options (recycling (38%), energy (35%), backfilling (6%)) and landfilling (21%). Results showed that the current CDW management system produced environmental benefits in terms of GWP excluding transportation (-360 kg CO_2-eq per CD&E waste tonne), but the role of each EoL option in environmental savings was not provided.

Increasing the recovery rate of inert CDW in the construction sector tends to decrease the environmental footprint of construction materials especially in terms of GWP, ADP and land use change; at a variable degree depending on the plant efficiency, transport distance, and energy resources used in recycling (Jain et al., 2020, Penteado and Rosado, 2016, Butera et al., 2015, Gálvez-Martos et al., 2018). An LCA study that compared CDW the landfilling, recovery (as in backfilling) and recycling with different sorting techniques in Belgium, found that the recovery or recycling of CDW compared to landfilling could reduce the environmental impacts by 36-59% depending on the efficiency of sorting strategy (Di Maria et al., 2018). In the same study, it was suggested that selective demolition for the recovery of wood and metals can lower the environmental impacts due to a higher quality (free of impurities) of recovered CDW (Di Maria et al., 2018). The beneficial role of selective demolition was also reported in a previous LCA study that investigated the environmental impacts of a building in South Italy using an on-site (selective demolition, collection, sorting) and off-site (material and energy recovery processes and landfill) management plan for the main CDW streams (plastics, steel, copper, aggregates, glass) (Vitale et al., 2017). Results showed that selective demolition may increase the quality and quantity of construction materials sent to treatment facilities for recovery and disposal (Vitale et al., 2017).

Butera et al. (2015) reported that recovery referring to the construction and operation of crushing provided negligible contribution to total impacts associated with the CDW management with the maximum participation in $ADP_{element}$ (ca. 4.5%) mostly due to the consumption of steel for the crushing machinery, while capital goods for landfilling had a more considerable contribution to $ADP_{elements}$ and HTP (ca. 25-30%) due to steel consumption, and diesel production and combustion. The low contribution of capital goods to total impacts of CDW management (EoL) suggests the construction of smaller decentralised crushing facilities and/or on-site crushing activities so as to minimise transport distances (Butera et al., 2015).

However, environmental trade-offs may arise from the recycling process in terms of EP, ETP, and HTP (Jain et al., 2020, Butera et al., 2015, Faleschini et al., 2016), due to the lower leaching per tonne of CDW disposed of to landfills (L/S ratio) within a 100-year timeframe including the leachate collection treatment in landfills (Butera et al., 2015). Therefore, cradle-to-grave LCA studies of construction materials should consider the related emissions from leaching production at the end-of-life phase, especially for toxicity impacts (Butera et al., 2015). This statement was also reported by Schwab et al. (2014) indicating the need for quantitative consideration of environmental impacts of long-term leaching in LCA of CDW waste, taking into account site-specific soil geography and substance-specific fate characteristics for the design of waste management strategies in the construction sector. A subsequent end-of-life system needs to be expanded in LCA of construction materials – this of 'leaching from graveyard to grave' (Schwab et al., 2014).

Transportation in CDW management: a critical LCA factor

Environmental impacts related to transportation is one of the most highly researched and discussed processes in the life cycle environmental assessment of inert materials used in the construction sector (Faleschini et al., 2016, Jain et al., 2020, Butera et al., 2015, Penteado and Rosado, 2016, Di Maria et al., 2018). The transport distance can significantly affect the percentage contribution of each stage in the environmental impact categories across the construction value chain (Penteado and Rosado, 2016, Jain et al., 2020, Vitale et al., 2017).

Comparing the production of virgin materials with the recycling of the same materials as stand-alone processes might demonstrate that recycling is not as sustainable as expected mostly due to transportation-related issues (Jain et al., 2020, Faleschini et al., 2016). The sustainability of CDW can be significantly compromised by increasing the transport distance (Faleschini et al., 2016). For example, for most impacts (e.g. GWP, POCP, particulate matter, AP, EP, and ADP), the contribution of transportation of CDW from construction site to waste management facilities (e.g. crushing facilities) can account for 40-50% of total impacts (distance: 30 km) related to the stages of end-of-life management, while the negative contribution of avoided transportation of natural aggregates (e.g. from gravel pit to the construction site) accounted for 30–40% (distance: 50 km) (Butera et al., 2015). This finding indicated that the distance of CDW transportation is better not to exceed 90% of the transport distance for natural aggregates to ensure environmental savings (Butera et al., 2015).

In addition to this, one study argued that recycling of inert CDW is a better alternative than landfill only if the transport distance for recycling does not exceed this for landfilling more than 312% (Penteado and Rosado, 2016). The importance of transport distance was also highlighted by Di Maria et al. (2018) indicating the optimal location for facilities of CDW recovery is defined by the local demand for aggregates and environmental impacts caused by long distance transport.

Several CDW management measures can be implemented to deal with the transport distance with some of them reported in LCA studies: i) implementation of CDW sorting at construction sites (on-site facilities) so that to avoid the transport of refused materials to sorting and recycling facilities (Butera et al., 2015, Penteado and Rosado, 2016); ii) adoption of registered and licensed areas by municipalities for delivering, sorting and temporary storage of small volumes of CDW (Penteado and Rosado, 2016); iii) use of mobile recycling units (e.g. centralized trucks fleet, which may be converted into electric vehicles) (Penteado and Rosado, 2016) and/or decentralized recovery facilities (e.g. crushing) located nearby main CDW generation regions to deal with the dispersion of generation sources (Penteado and Rosado, 2016, Butera et al., 2015); iv) promoting the connection between stakeholders (e.g. recyclers and constructors) by localising the recovery facilities across the regional territory (Borghi et al., 2018); v) increase the share of renewable resources in energy consumption during recycling of CD&E waste to trade-off the impact of transportation on GWP (Jain et al., 2020, Faleschini et al., 2016, Borghi et al., 2018). In relation to the latter measure, the beneficial role of renewable resources used for the energy intensive recycling process is affected by the association of resources with land use (e.g. biomass, solar, wind, etc.).

6. A CONSISTENT APPROACH FOR COMPARING DIFFERENT MATERIALS

Attempts to compare the environmental performance of construction materials (of inert nature) have been based on experimentation, modelling, the use of data from the Ecoinvent, GaBi and other LCA databases, or they have been focused only on different parts of the construction value chain. In regards to the latter, the majority of LCA studies adopted a *cradle-to-gate* approach, or placed increased focused on the downstream part of system, i.e. waste management. As a result, there has been little consistency between the data collected by the different studies. Due to the lack of sound evidence and the impracticalities of generating comprehensive comparative data on construction materials' environmental performance across the construction value chain, we had to use alternative ways of analysis. By integrating the results from the evidence review, we developed a scoring method to validate our observations and suggestions based on existing data. We demonstrated that it is possible with the use of this method to depict materials' life cycle environmental performance in a useful manner.

Here, we propose that the use of a consistent method can be particularly useful in comparing the environmental performance of materials. To demonstrate, we present in **Figure 6-1** a comparison between concrete and clay bricks. This qualitative illustration could constitute the basis of a consistent approach to use in comparing different materials along the value chain. As shown in **Figure 6-1**, there are many blind spots across the life cycle of both concrete and clay-based bricks. These blind spots preclude a clear understanding of where inefficiencies occur in the system, and should be urgently addressed to ensure a holistic, and comprehensive assessment. We acknowledge that there is an inherent sensitivity of environmental impacts accounting due to variations in raw materials used in the production of both concrete and clay-based bricks, which makes this comparison particularly interesting. The processing technologies employed, the purpose use of these materials in the construction sector and their recovery after the assets end-of-service life come to add to the complexities of getting some robust estimates on their environmental performance. Nonetheless, the scoring system can be an agile method to depict a range of figures pertaining to different materials, whilst still providing a useful comparison and insight into the blind spots. Once data becomes available, the scoring values can be adjusted reflecting exactly how the new data provide an improved depiction of the system.

We also find this approach to be quite powerful in comparing different mixtures of materials used in making the same product, or scenarios of where improvements can be made in the system. Notwithstanding its usefulness, the analysis of the environmental performance of materials by itself is only part of the picture. To be able to see the big picture, environmental analyses should be accompanied by the economic, social, and technical performance of materials; only then we would be able to perform sound decision-making processes and develop powerful management strategies. This holistic, system-based approach can help construction sector can become more resource efficient and productive, and move towards circular economy solutions.

LIFE CYCLE ENVIRONMENTAL IMPACTS SCORECARD

Concrete (upper value) / Clay bricks (lower value)

	EXTRACTION	PRODUCTION	CONSTRUCTION	SERVICE LIFE	RECOVERY (BACKFILL)	RECYCLING	LANDFILL
GWP	2 2	1 1	Unspecified Unspecified	Unspecified Unspecified	3 Unspecified	3 Unspecified	3 3
AP	1 2	1 1	Unspecified Unspecified	Unspecified Unspecified	2 Unspecified	2 Unspecified	2 3
EP	2 2	1 1	Unspecified Unspecified	Unspecified Unspecified	3 Unspecified	3 Unspecified	3 3
ODP	2 Unspecified	1 Unspecified	Unspecified Unspecified	Unspecified Unspecified	Unspecified Unspecified	Unspecified Unspecified	Unspecified Unspecified
ADP	2 2	1 1	Unspecified Unspecified	Unspecified Unspecified	3 Unspecified	3 Unspecified	3 3
ETP	Unspecified 1	Unspecified 1	Unspecified Unspecified	Unspecified Unspecified	Unspecified Unspecified	Unspecified Unspecified	Unspecified 2
HTP	2 2	2 1	Unspecified Unspecified	Unspecified Unspecified	1 Unspecified	1 Unspecified	Unspecified 3
PODP	2 Unspecified	1 Unspecified	Unspecified Unspecified	Unspecified Unspecified	Unspecified Unspecified	Unspecified Unspecified	Unspecified Unspecified
PM	Unspecified Unspecified	Unspecified Unspecified	Unspecified Unspecified	Unspecified Unspecified	Unspecified Unspecified	Unspecified Unspecified	Unspecified Unspecified
LO	Unspecified Unspecified	Unspecified Unspecified	Unspecified Unspecified	Unspecified Unspecified	Unspecified Unspecified	Unspecified Unspecified	Unspecified Unspecified
Evidence	Rock > Clay > Gravel & Sand	Cement production > Clay firing	Only for buildings as a mixture of construction materials		Only for C&D waste as a mixture of construction waste materials		
Main influential factors	• Soil morpho-logy; • Equipment; • Duration; • Site conditions	Fired vs dried bricks; Cement proportion; Aggregate source; Product composition	Materials; Technologies; Practices (e.g. imports, policies)	Lifespan of materials; Operational carbon;	• Transport distance; • Efficiency of recovery processes/ MMC; • Quality of recovered/ recycled waste materials; • Leaching production		

Need for databases and site-specific indicators

Figure 6-1 Scoring of stages of life cycle of construction materials (e.g. concrete and clay bricks) that indicate their contribution to prevalent environmental impacts across the full life cycle as received reporting basis. The absence of information on specific stages results in score values that compare only stages with existing data. Literature evidence on comparison amongst materials and critical factors that need consideration is also given. Abbreviations: Global warming potential (GWP); Acidification potential (AP); Eutrophication potential (EP); Ozone layer depletion potential (ODP); Abiotic depletion potential (ADP); Ecotoxicity potential (ETP); Human toxicity potential (HTP)l; Photochemical ozone creation potential (POCP); Particulate matter (PM); Land occupation (LO); End-of-Life (EoL)

As shown in **Figure 6-1**, at the stage of extraction, evidence indicated that rock excavation has higher impacts than clay excavation (Forsythe and Ding, 2014, Devi and Palaniappan, 2017) followed by gravel and sand extraction (Lewis and Hajji, 2012). Nonetheless, additional factors such as soil morphology (e.g. soil type (Devi and Palaniappan, 2017) and slope (Forsythe and Ding, 2014)), mobile equipment (Wang et al., 2016, Devi and Palaniappan, 2017), activity duration (Giunta, 2020, Devi and Palaniappan, 2017) and other site-related conditions (e.g. location, weather, excavation depth, volume of extracted soil, etc. (Devi and Palaniappan, 2017)) are critical parameters that may affect the environmental performance of materials at the extraction stage.

At the stage of production, cement production is a more energy intensive process than clay firing (1450 ºC vs 950 ºC) (de Souza et al., 2016)). This is justified by comparing the embodied carbon of concrete (range: 0.03-0.56 kg CO_2-eq. kg^{-1}) with that of clay bricks (range: 0.179-0.354 kg CO_2-eq. kg^{-1}) in **Table 5-2** indicating the wider range of the former (concrete) due to variable proportions of cement used in production. Concrete's lower embodied carbon limit denotes that the comparison at this stage must be interpreted with caution as the types of raw materials (e.g. clay, gravel, sand, crushed rocks, clay, limestone, recycled aggregates) and proportions of cement substitutes used, such as fly ash, GGBS, and MgO, in the final product can affect the environmental performance of the end-material. This is in line with Kua and Kamath (2014), who reported that the replacement of concrete by fired clay bricks (10-30%) in Singapore does not necessarily reduce the environmental impacts. Another key factor in comparing the environmental performance of materials, is their technical specifications (e.g. compressive strength and volume composition) that may demand specific types of materials and mixtures. For example, clay unfired bricks are considerably more preferable than fired bricks from the environmental point of view, but the fired bricks are better in terms of technological performance (Marcelino-Sadaba et al., 2017). This highlights that the production and use of a standardised material might not result to desirable environmental footprint, and a tailored approach to structures and contexts might be needed.

Moving beyond the cradle-to-gate activities, the system becomes more complex especially as different materials are bound together to construct the desired structure, e.g. a building. During on-site construction, concrete can be aptly used in many different ways (e.g. in foundations, in tiles, in blocks, etc.) and bricks are widely used in wall building and can be layered using cement mortar. A way to control or even reduce the environmental impacts is to use a mixed methods approach that employs the use of modern methods of construction (**MMCs**) (e.g. modular construction, and manufacture of precast sections), the use of alternative materials (e.g. timber, geopolymer concrete, adobe and/or unfired bricks, or use of lime mortar instead of cement), and the use of new technologies (e.g. building information modelling) to improve on-site construction practices, and reduce waste (Giesekam et al., 2014). In the stage of service life, the operational performance of buildings can be improved by extending the lifespan and reducing the operational carbon strongly affected by the stage of construction.

At the EoL management stage, the system becomes even more complicated not only because of the heterogeneity of CD&E waste but also due to the high sensitivity of environmental impacts quantification and efficiency to local-scale conditions such as transport distance (e.g. between landfill and recovery facility), efficiency of recovery processes, quality of recovered/recycled aggregates and leaching production. For example, on-site selective demolition proved to increase the sorting efficiency and therefore the quality of recovered waste (Di Maria et al., 2018, Vitale et al., 2017, Estanqueiro et al., 2018). The partial or overall replacement of energy consumption by renewable resources in recycling facilities is able to increase the environmental savings arising from the respective EoL option (Jain et al., 2020, Faleschini et al., 2016, Borghi et al., 2018). Sufficient quality of recycled/recovered aggregates is a prerequisite to avoid additional amount of raw materials and resources in the stage of production depending on the purpose use (e.g. bound or unbound material) and therefore to receive environmental savings along the supply chain (Knoeri et al., 2013, Colangelo et al., 2018). In addition, leaching production during CDW management can induce impacts especially for toxic categories affecting subsoil and groundwater (Butera et al., 2015), although many studies did not examine the factor of leaching considering that CDW are inert materials. However, to our knowledge, there is no clear evidence which CDW materials are prone to leaching or producing an ecotoxic leachate that is likely to give rise to environmental pollution or harm to human health.

7. CONCLUSIONS

This report has described the environmental life cycle impacts of inert/ less reactive materials produced, used and managed in the UK, based on data collected from a rapid, yet comprehensive, evidence literature review. The prevalence of CD&E waste in total waste arisings in the UK, meant a construction sector focus. Through examination of the recent LCA studies on inert/less reactive materials used in the construction sector, we developed a consistent approach to depicting, and comparing their environmental impacts, looking at the entire system, and created a scoring system to support a holistic and integrated analysis.

Ambiguities in the legislation and the lack of reliable and accurate data creates many evaluation challenges. The use of LCA may be useful in tracking the environmental performance of materials across their life cycle, but the discrepancy in the processes and activities included in the LCA analyses carried out by different case studies, can lead to varied, and often misleading, insights. To allow comparison between LCA case studies there is a need to normalise: 1) the processes that are to be included in the system boundaries, and 2) the functional unit used. Better insights can be obtained when unit strength of construction materials is selected as functional unit rather than volume units (e.g. $1m^3$) which, has gained more attention by the global literature.

Undeniably, there is a sheer complexity in assessing the environmental performance of the construction sector activities across the full value chain. This is due to a range of parameters that tremendously affect the environmental performance of inert/ less reactive materials used by the sector. These parameters include: the composition, quality and quantity of materials extracted and processed; transportation distances (especially of voluminous, low-value materials such as aggregates and soils); regional differences in geomorphology and techniques used; and construction practices followed. Adding to this complexity, is the lack of good inventory on the types of materials used in structures, and the methods of construction, and deconstruction/ demolition. The latter could play an important role in promoting resource efficiency and eliminating waste.

Moreover, data deficiency on inert / less reactive materials' environmental performance over their full life cycle, can create intended and unintended siloed approaches to addressing sustainability in the construction sector, which end up preventing effective decision-making. At present, the growing focus on embodied carbon appears to divert attention from other potentially critical environmental impacts, such as AP, and HTP, and from the EoL fate of supposedly inert/ less reactive waste materials generated in the construction sector. This highlights that there are many blind spots in the inert/less reactive materials' life cycles in the UK, particularly in soil waste. With this being the largest waste stream generated annually, better scrutiny over its entire life cycle is urgently needed for addressing pollution and improving resource efficiency.

There isn't an optimum EoL solution that would reduce the environmental impacts of materials used. First, and foremost, there needs to be a reduction in the material throughput in the construction sector as this would reduce significantly environmental impacts across all stages of the construction sector's value chain. Second, there is a multitude of parameters that come into play when assessing the environmental impacts of inert /less reactive waste materials; therefore, improved assessment comes with an improved understanding of all those parameters that come into play at the selection, extraction, processing/manufacture, distribution, installation, use and end-of-life management stages of materials. Third, there is "no one-size fits all approach" in the construction sector, which reinforces our argumentum that to improve the CDW management system, there needs to be a good understanding of the system as a whole.

The scoring system developed herein can be useful to depicting the multitude of data, and the lack thereof, at each stage of the materials life cycle. It can be a means to attaining a holistic, integrated view of the system, which in turn can generate useful insights and allow the identification of inefficiencies in the system. Such a consistent approach should be highly encouraged in the assessment of the environmental performance of all materials, and should be accompanied with corresponding assessments on other sustainability domains (economic, social, and technical). This would comprehensively inform policy and decision-making processes, and contribute towards the adoption of sustainable circular economy solutions.

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APPENDIX

Table A.1. Cradle-to-gate embodied carbon of the majority of materials used in construction projects in the UK as obtained from ICE database V3.3 (Circularecology, 2021)

Material category	Material type	Embodied carbon (kg CO2 eq./ kg)
Aggregates and Sand	general UK, mixture of land won (64.2% w/w), marine (8.3% w/w), secondary and recycled (27.5% w/w), bulk, loose	0.0075
	general, virgin mixture of land won (89% w/w) and marine (11% w/w), bulk, loose	0.0049
	from virgin land won resources, bulk, loose	0.0044
	from virgin marine resources, bulk, loose	0.0090
	from recycled resources, no heat treatment, bulk, loose	0.0061
	from recycled resources, with heat treatment, bulk, loose	0.1188
	expanded clay, bulk, loose	0.3932
	expanded foamed glass, bulk, loose	0.2776
	from secondary resources, bulk, loose	0.0633
	mixture of recycled and secondary resources, bulk, loose	0.0142
Aluminium	Aluminium General, European Mix, Inc Imports	6.6687
	Aluminium sheet, European Mix, Inc Imports	6.5812
	Aluminium foil, European Mix, Inc Imports	7.4690
	Aluminium extruded profile, European Mix, Inc Imports	6.8252
	Aluminium, cast, European Mix, Inc Imports	6.7152
	Aluminium General, Worldwide	13.0555
	Aluminium sheet, Worldwide	12.9559
	Aluminium foil, Worldwide	13.7669
	Aluminium extruded profile, Worldwide	13.1764
	Aluminium cast, Worldwide	13.1869
	Aluminium General, European Mix, Inc Imports	6.6687
	Aluminium, produced in Europe	5.5839
	Aluminium General, Worldwide	13.0555
	Aluminium, North American	5.6516
	Aluminium, Africa	12.4057
	Aluminium, China	14.5936
	Aluminium, Japan	10.6245
	Aluminium, Middle East	10.8098
	Aluminium, Oceania	12.7521

	Aluminium, Other Asia	15.8691
	Aluminium, Russia	5.5483
	Aluminium, South America	8.3190
	Aluminium, South Korea	11.9469
Asphalt	Asphalt, 3% (bitumen) binder content (by mass)	0.0501
	Asphalt, 3.5% binder content	0.0511
	Asphalt, 4% binder content	0.0522
	Asphalt, 4.5% binder content	0.0532
	Asphalt, 5% binder content	0.0542
	Asphalt, 5.5% binder content	0.0553
	Asphalt, 6% binder content	0.0563
	Asphalt, 6.5% binder content	0.0573
	Asphalt, 7% binder content	0.0584
Bricks	General (Common Brick)	0.2130
	A Single Brick	0.4537
Cement	General (UK average)	0.8321
	Average CEM I, Ordinary Portland Cement (OPC)	0.9120
	CEM II-A-S - 13% GGBs	0.8028
	CEM II/B-S - 28% GGBs	0.6716
	CEM II/A-P - 13% natural pozzolanic ash	0.7980
	CEM II/B-P 28% natural pozzolanic ash	0.6612
	CEM II/A-V - 13% fly ash siliceous	0.7979
	CEM II/B-V - 28% fly ash siliceous	0.6610
	CEM II/A-W - 13% fly ash calcareous	0.7979
	CEM II/B-W - 28% fly ash calcareous	0.6610
	CEM II/A-L - 13% limestone	0.7995
	CEM II/B-L- 28% limestone	0.6644
	CEM II/A-LL - 13% limestone	0.7995
	CEM II/B-LL - 28% limestone	0.6644
	CEM II/A-M - 16% cement replacement	0.7736
	CEM II/B-M - 28% cement replacement	0.6663
	CEM III/A - 50.5% GGBS	0.4748
	CEM III/B - 73% GGBS	0.2780
	CEM III/C - 88% GGBS	0.1468
	CEM IV/A - 23% cement replacement	0.7067
	CEM IV/B - 46% cement replacement	0.5014
	CEM V/A - 24% GGBS and 24% cement replacement	0.4887
	CEM V/B - 36% GGBS and 36% cement replacement	0.2836
Mortar	Mortar (1:3 cement:sand mix)	0.1831

	Mortar (1:4)	0.1494
	Mortar (1:5)	0.1266
	Mortar (1:6)	0.1101
	Mortar (1:½:4½ Cement:Lime:Sand mix)	0.1571
	Mortar (1:1:6 Cement:Lime:Sand mix)	0.1425
	Mortar (1:2:9 Cement:Lime:Sand mix)	0.1267
Concrete admixtures	General concrete admixtures – Average of data collected	1.6662
	Concrete admixtures – Air entrainers, Europe	0.5270
	Concrete admixtures – Hardening Accelerators	2.2800
	Concrete admixtures – Plasticisers and Superplasticisers	1.8800
	Concrete admixtures – Retarders	1.3100
	Concrete admixtures – Set Accelerators	1.3300
	Concrete admixtures – Water Resisting Admixtures	2.6700
Ceramics	General	0.7000
	Fittings	1.1400
	Sanitary Products	1.6100
	Tiles and Cladding Panels	0.7800
Clay	General (Simple Baked Products)	0.2400
	Tile	0.4800
	Vitrified clay pipe DN 100 & DN 150	0.4600
	Vitrified clay pipe DN 200 & DN 300	0.5000
	Vitrified clay pipe DN 500	0.5500
Concrete (by strength class) - In-situ	General	0.1034
	GEN 0 (6/8 MPa)	0.0654
	GEN 1 (8/10 MPa)	0.0899
	GEN 2 (12/15 MPa)	0.0971
	GEN 3 (16/20 MPa)	0.1042
	20/25 MPa	0.1120
	25/30 MPa	0.1190
	28/35 MPa	0.1260
	32/40 MPa	0.1382
	35/45 MPa	0.1487
	40/50 Mpa	0.1591
	PAV1	0.1260
	PAV2	0.1382
Concrete (by strength class) - CEM I	GEN 0 (6/8 MPa)	0.0704
	GEN 1 (8/10 MPa)	0.0972
	GEN 2 (12/15 MPa)	0.1050

	GEN 3 (16/20 MPa)	0.1127
	RC 20/25 (20/25 MPa)	0.1209
	RC 25/30 (25/30 MPa)	0.1286
	RC 28/35 (28/35 MPa)	0.1362
	RC 32/40 (32/40 MPa)	0.1495
	RC 35/45 (35/45 MPa)	0.1609
	RC 40/50 (40/50 MPa)	0.1723
	PAV1	0.1362
	PAV2	0.1495
Concrete (by strength class) - Portland Limestone Concrete-14% Limestone	GEN 0 (6/8 MPa)	0.0614
	GEN 1 (8/10 MPa)	0.1542
	GEN 2 (12/15 MPa)	0.0905
	GEN 3 (16/20 MPa)	0.0966
	RC 20/25 (20/25 MPa)	0.1040
	RC 25/30 (25/30 MPa)	0.1108
	RC 28/35 (28/35 MPa)	0.1174
	RC 32/40 (32/40 MPa)	0.1288
	RC 35/45 (35/45 MPa)	0.1403
	RC 40/50 (40/50 MPa)	0.1529
	PAV1	0.1169
	PAV2	0.1293
Concrete (by strength class) - 35% natural pozzolanic ash	GEN 0 (6/8 MPa)	0.0557
	GEN 1 (8/10 MPa)	0.0757
	GEN 2 (12/15 MPa)	0.0814
	GEN 3 (16/20 MPa)	0.0872
	RC 20/25 (20/25 MPa)	0.0941
	RC 25/30 (25/30 MPa)	0.0998
	RC 28/35 (28/35 MPa)	0.1059
	RC 32/40 (32/40 MPa)	0.1171
	RC 35/45 (35/45 MPa)	0.1241
	RC 40/50 (40/50 MPa)	0.1326
	PAV1	0.1056
	PAV2	0.1173
Concrete (by strength class) - 15% Cement Replacement-Fly Ash	GEN 0 (6/8 MPa)	0.0639
	GEN 1 (8/10 MPa)	0.1282
	GEN 2 (12/15 MPa)	0.0961
	GEN 3 (16/20 MPa)	0.1031
	RC 20/25 (20/25 MPa)	0.1110

	RC 25/30 (25/30 MPa)	0.1180
	RC 28/35 (28/35 MPa)	0.1261
	RC 32/40 (32/40 MPa)	0.1391
	RC 35/45 (35/45 MPa)	0.1485
	RC 40/50 (40/50 MPa)	0.1590
	PAV1	0.1259
	PAV2	0.1393
Concrete (by strength class) - 30% Cement Replacement-Fly Ash	GEN 0 (6/8 MPa)	0.0566
	GEN 1 (8/10 MPa)	0.1435
	GEN 2 (12/15 MPa)	0.0855
	GEN 3 (16/20 MPa)	0.0915
	RC 20/25 (20/25 MPa)	0.0989
	RC 25/30 (25/30 MPa)	0.1050
	RC 28/35 (28/35 MPa)	0.1131
	RC 32/40 (32/40 MPa)	0.1251
	RC 35/45 (35/45 MPa)	0.1327
	RC 40/50 (40/50 MPa)	0.1419
	PAV1	0.1127
	PAV2	0.1255
Concrete (by strength class) - 40% Cement Replacement-Fly Ash	GEN 0 (6/8 MPa)	0.0523
	GEN 1 (8/10 MPa)	0.0708
	GEN 2 (12/15 MPa)	0.0761
	GEN 3 (16/20 MPa)	0.0814
	RC 20/25 (20/25 MPa)	0.0878
	RC 25/30 (25/30 MPa)	0.0931
	RC 28/35 (28/35 MPa)	0.0986
	RC 32/40 (32/40 MPa)	0.1089
	RC 35/45 (35/45 MPa)	0.1154
	RC 40/50 (40/50 MPa)	0.1233
	PAV1	0.0983
	PAV2	0.1092
Concrete (by strength class) - 25% Cement Replacement-Blast furnace slag	GEN 0 (6/8 MPa)	0.0551
	GEN 1 (8/10 MPa)	0.1029
	GEN 2 (12/15 MPa)	0.0812
	GEN 3 (16/20 MPa)	0.0870
	RC 20/25 (20/25 MPa)	0.0942
	RC 25/30 (25/30 MPa)	0.1001
	RC 28/35 (28/35 MPa)	0.1073

	RC 32/40 (32/40 MPa)	0.1204
	RC 35/45 (35/45 MPa)	0.1294
	RC 40/50 (40/50 MPa)	0.1383
	PAV1	0.1076
	PAV2	0.1204
Concrete (by strength class) - 50% Cement Replacement-Blast furnace slag	GEN 0 (6/8 MPa)	0.0406
	GEN 1 (8/10 MPa)	0.0918
	GEN 2 (12/15 MPa)	0.0584
	GEN 3 (16/20 MPa)	0.0622
	RC 20/25 (20/25 MPa)	0.0679
	RC 25/30 (25/30 MPa)	0.0720
	RC 28/35 (28/35 MPa)	0.0779
	RC 32/40 (32/40 MPa)	0.0888
	RC 35/45 (35/45 MPa)	0.0952
	RC 40/50 (40/50 MPa)	0.1015
	PAV1	0.0782
	PAV2	0.0888
Concrete (by strength class) - 70% Cement Replacement-Blast furnace slag	GEN 0 (6/8 MPa)	0.0340
	GEN 1 (8/10 MPa)	0.0437
	GEN 2 (12/15 MPa)	0.0468
	GEN 3 (16/20 MPa)	0.0497
	RC 20/25 (20/25 MPa)	0.0527
	RC 25/30 (25/30 MPa)	0.0556
	RC 28/35 (28/35 MPa)	0.0584
	RC 32/40 (32/40 MPa)	0.0634
	RC 35/45 (35/45 MPa)	0.0677
	RC 40/50 (40/50 MPa)	0.0719
	PAV1	0.0584
	PAV2	0.0634
Average UK Cement:Sand:Aggregate (by volume and cementous content)	1:1:2	0.1918
	1:1.5:3	0.1431
	1:2:4	0.1152
	1:2.5:5	0.0852
	1:3:6	0.0852
	1:4:8	0.0688
CEM I Cement:Sand:Aggreg	1:1:2	0.2088
	1:1.5:3	0.1555
	1:2:4	0.1250

ate (by volume and cementous content)	1:2.5:5	0.1052
	1:3:6	0.0921
	1:4:8	0.0741
CEM I per m3 concrete	100 kg CEM I per m3 concrete	0.0520
	120	0.0598
	140	0.0675
	160	0.0753
	180	0.0830
	200	0.0908
	220	0.0985
	240	0.1063
	260	0.1141
	280	0.1219
	300	0.1297
	320	0.1376
	340	0.1454
	360	0.1532
	380	0.1610
	400	0.1687
	420	0.1765
	440	0.1843
	460	0.1921
	480	0.1998
	500	0.2076
Average UK cement per m3 concrete	100 kg Average UK cement per m3 concrete	0.0486
	120	0.0557
	140	0.0628
	160	0.0698
	180	0.0769
	200	0.0840
	220	0.0910
	240	0.0981
	260	0.1052
	280	0.1124
	300	0.1195
	320	0.1267
	340	0.1339
	360	0.1410

	380	0.1481
	400	0.1552
	420	0.1623
	440	0.1694
	460	0.1764
	480	0.1835
	500	0.1906
By Cementitious Content with pfa (30% replacement rate)	100 kg total cementitious content per m3 concrete	0.0405
	120	0.0459
	140	0.0514
	160	0.0568
	180	0.0622
	200	0.0677
	220	0.0731
	240	0.0785
	260	0.0840
	280	0.0895
	300	0.0951
	320	0.1006
	340	0.1061
	360	0.1116
	380	0.1170
	400	0.1225
	420	0.1280
	440	0.1334
	460	0.1389
	480	0.1443
	500	0.1498
By Cementitious Content with pfa (40% replacement rate)	100 kg total cementitious content per m3 concrete	0.0366
	120	0.0413
	140	0.0460
	160	0.0506
	180	0.0553
	200	0.0599
	220	0.0646
	240	0.0693
	260	0.0740
	280	0.0787

	300	0.0835
	320	0.0883
	340	0.0930
	360	0.0977
	380	0.1024
	400	0.1071
	420	0.1118
	440	0.1165
	460	0.1212
	480	0.1258
	500	0.1305
By Cementitious Content with pfa (50% replacement rate)	100 kg total cementitious content per m3 concrete	0.0336
	120	0.0376
	140	0.0417
	160	0.0457
	180	0.0498
	200	0.0538
	220	0.0579
	240	0.0619
	260	0.0661
	280	0.0702
	300	0.0743
	320	0.0785
	340	0.0826
	360	0.0867
	380	0.0908
	400	0.0949
	420	0.0989
	440	0.1030
	460	0.1071
	480	0.1112
	500	0.1153
By Cementitious Content with pfa (70% replacement rate)	100 kg total cementitious content per m3 concrete	0.0262
	120	0.0288
	140	0.0313
	160	0.0339
	180	0.0365
	200	0.0391

	220	0.0416
	240	0.0442
	260	0.0469
	280	0.0495
	300	0.0522
	320	0.0548
	340	0.0575
	360	0.0601
	380	0.0627
	400	0.0653
	420	0.0679
	440	0.0705
	460	0.0731
	480	0.0757
	500	0.0783
Precast concrete products	precast concrete pipe, DN600 unreinforced per kg	0.1462
	precast concrete paving (Blocks, Slabs, Channels and Kerbs)	0.1324
	precast concrete beams and columns -steel reinforced with world average steel	0.2490
	As above but with European recycled steel rebar	0.1939
Precast concrete blocks	concrete block, medium density solid, average strength, per kg	0.0931
	concrete block, high density solid, average strength, per kg	0.0931
	AAC concrete block	0.2800
Soil	General (Rammed Soil)	0.0240
	Cement stabilised soil (5% cement content)	0.0610
	Cement stabilised soil with (6% cement and 2% lime).	0.0840
	GGBS stabilised soil (8% GGBS and 2% lime)	0.0470
	Fly ash stabilised soil (8% fly ash and 2% lime).	0.0410
Steel	Steel, UO Pipe	3.0200
	Steel, Tin-free Electrolytic Chrome Coated Steel Sheet - Tin-free (ECCS)	2.8900
	Steel, electrogalvanized steel	3.0300
	Steel, welded pipe	2.7800
	Steel, Organic coated sheet	3.0600
	Steel, Tinplate	2.8500
	Steel, finished cold-rolled coil	2.7300
	Steel, hot-dip galvanized steel	2.7600
	Steel, Plate	2.4600

	steel, Cold Rolled Coil	2.5300
	Steel, pickled hot-rolled coil	2.4200
	Steel, Wire rod	2.2700
	Steel, Hot Rolled Coil	2.2800
	Steel, Rebar	1.9900
	Steel, Section	1.5500
	Steel, Engineering steel	1.2700
	Steel, global seamless tube	2.1300
Stone	General	0.0790
	Granite	0.7000
	Limestone	0.0900
	Marble	0.1300
	Marble tile	0.2100
	Sandstone	0.0600
	Shale	0.0020
	Slate	0.007-0.063
Timber (excluding carbon storage)	Timber - Average of all data - No Carbon Storage	0.4928
	Timber, Chipboard - No Carbon Storage	0.4002
	Timber, Closed panel timber frame system - No Carbon Storage	0.4525
	Timber, CLT - No Carbon Storage	0.4373
	Timber, Fibreboard - No Carbon Storage	0.7153
	Timber, Glulam - No Carbon Storage	0.5121
	Timber, Hardboard - No Carbon Storage	0.8152
	Timber, Hardwood - No Carbon Storage	0.3056
	Timber, Laminate - No Carbon Storage	0.6978
	Timber, Laminated strand lumber - No Carbon Storage	0.5041
	Timber, Laminated veneer lumber - No Carbon Storage	0.3898
	Timber, MDF - No Carbon Storage	0.8565
	Timber, Open panel timber frame system - No Carbon Storage	0.3452
	Timber, OSB - No Carbon Storage	0.4551
	Timber, Parquet - No Carbon Storage	0.8112
	Timber, Particle Board - No Carbon Storage	0.6643
	Timber, Plywood - No Carbon Storage	0.6815
	Timber, Softwood - No Carbon Storage	0.2626
	Timber, Wood I-Beam - No Carbon Storage	0.4833
	Timber, Wood-plastic composite - No Carbon Storage	1.4400
Timber including carbon storage)	Timber - Average of all data - Including Carbon Storage	-1.0309
	Timber, Chipboard - Including Carbon Storage	-1.1207

	Timber, Closed panel timber frame system - Including Carbon Storage	-1.1016
	Timber, CLT - Including Carbon Storage	-1.2041
	Timber, Fibreboard - Including Carbon Storage	-0.8632
	Timber, Glulam - Including Carbon Storage	-0.8957
	Timber, Hardboard - Including Carbon Storage	-0.8240
	Timber, Hardwood - Including Carbon Storage	-1.2860
	Timber, Laminate - Including Carbon Storage	-0.5804
	Timber, Laminated strand lumber - Including Carbon Storage	-1.0841
	Timber, Laminated veneer lumber - Including Carbon Storage	-1.2466
	Timber, MDF - Including Carbon Storage	-0.6437
	Timber, Open panel timber frame system - Including Carbon Storage	-1.2690
	Timber, OSB - Including Carbon Storage	-1.0473
	Timber, Parquet - Including Carbon Storage	-0.8130
	Timber, Particle Board - Including Carbon Storage	-0.8150
	Timber, Plywood - Including Carbon Storage	-0.9331
	Timber, Softwood - Including Carbon Storage	-1.2919
	Timber, Wood I-Beam - Including Carbon Storage	-1.0499
	Timber, Wood-plastic composite - Including Carbon Storage	0.5800
Glass	Glass, General, per kg	1.4370
	Glass, Glazing, Double, per kg	1.6256
	Glass, Glazing triple, per kg	1.7470
	Glass, Toughened, per kg	1.6672
	Glass, Multi layer safety, filled core, fire resistant, toughened, per kg	2.0818
	Glass, Multi layer safety, unfilled, per kg	1.5555
	Glass, sky light or roof, with frame, per kg	3.1019

Table A.2. Raw quantitative data of LCA case studies of materials included in the report (bricks, materials and aggregates – natural and recycled) across the life cycle stages.

Material	**LCA[1] boundary condition**	**Country**	**LCA scenarios**	**GWP[2] (kg CO_2 eq.)**	**AP[3] (kg SO_2 eq.)**	**EP[4] (kg Phosphate eq.)**	**ODP[5] (kg R_{11} eq.)**	**ADP_{fossil}[6] (MJ)**	**$ADP_{element}$[6] (kg Sb eq.)**	**ETP[7] (kg DCB eq.)**	**HTP[8] (kg DCB eq.)**	**POCP[9] (kg ethyl -eq.)**	**Embodied carbon (kg CO2 eq.)**	**Functional unit**	**Ref.**
Adobe bricks	Cradle-to-site	Cyprus	On-site with locally available soil and transported wheat straw*	1.76E-03	1.52E-05	8.31E-07	8.72E-13	4.90E-02		8.17E-02	1.95E-04	5.84E-07	1.76E-03	1 kg of adobe brick	(Christoforou et al., 2016)
Adobe bricks	Cradle-to-site	Cyprus	On-site with locally available soil and transported sawdust*	1.70E-03	1.46E-05	9.44E-07	8.84E-13	4.79E-02		7.64E-02	1.85E-04	2.60E-07	1.70E-03	1 kg of adobe brick	(Christoforou et al., 2016)
Adobe bricks	Cradle-to-site	Cyprus	On-site with transported soil and transported wheat straw*	5.41E-03	4.39E-05	7.88E-06	2.22E-12	9.93E-02		1.12E-01	3.37E-04	-1.15E-05	5.41E-03	1 kg of adobe brick	(Christoforou et al., 2016)
Adobe bricks	Cradle-to-site	Cyprus	On-site with transported soil and transported sawdust*	5.30E-03	4.29E-05	7.88E-06	2.21E-12	9.75E-02		1.06E-01	3.24E-04	-1.17E-05	5.30E-03	1 kg of adobe brick	(Christoforou et al., 2016)
Adobe bricks	Cradle-to-site	Cyprus	In factory with transported soil and transported wheat straw*	1.29E-02	1.03E-04	2.44E-05	5.00E-12	2.03E-01		1.75E-01	6.28E-04	-3.65E-05	1.29E-02	1 kg of adobe brick	(Christoforou et al., 2016)

Adobe bricks	Cradle-to-site	Cyprus	In factory with transported soil and transported sawdust*	1.28E-02	1.02E-04	2.44E-05	5.00E-12	2.01E-01		1.69E-01	6.15E-04	-3.66E-05	1.28E-02	1 kg of adobe brick	(Christoforou et al., 2016)
Ceramic brick walls	Cradle-to-grave	Brazil		3.20E+01				3.90E+02						1 m2 of exterior wall with a lifespan 40 years	(de Souza et al., 2016)
Concrete brick walls	Cradle-to-grave	Brazil		6.40E+01				6.80E+02						1 m2 of exterior wall with a lifespan 40 years	(de Souza et al., 2016)
Cast-in-place reinforced concrete	Cradle-to-grave	Brazil		9.40E+01				1.10E+03						1 m2 of exterior wall with a lifespan 40 years	(de Souza et al., 2016)
Ceramic brick walls	End-of-life	Brazil	Consider only trasportation to lanfill	2.9				6.10E+01						1 m2 of exterior wall with a lifespan 40 years	(de Souza et al., 2016)
Concrete brick walls	End-of-life	Brazil	Consider only trasportation to lanfill	3.8				8.10E+01						1 m2 of exterior wall with a lifespan 40 years	(de Souza et al., 2016)

Cast-in-place reinforced concrete	End-of-life	Brazil	Consider only trasportation to lanfill	5				1.10E+02					1 m2 of exterior wall with a lifespan 40 years	(de Souza et al., 2016)
Clay bricks	Cradle-to-gate	Spain	Marl-based stabilised with PC*	8.23E+00	2.02E-02	2.38E-03	5.33E+02	3.24E+01	1.38E-07	2.34E-01	5.35E-03	3.96E-07	Expressed per unit strength	(Marcelino-Sadaba et al., 2017)
Clay bricks	Cradle-to-gate	Spain	Marl-based stabilised with lime and GGBS**	2.79E+00	1.24E-02	9.06E-04	1.32E+02	3.42E+01	6.65E-08	1.49E-01	3.46E-03	2.11E-07	Expressed per unit strength	(Marcelino-Sadaba et al., 2017)
Clay bricks	Cradle-to-gate	Spain	Marl-based stabilised with Mg-oxide and GGBS**	1.42E+00	4.80E-03	4.72E-04	6.54E+01	1.79E+01	3.29E-08	5.92E-02	1.50E-03	2.85E-08	Expressed per unit strength	(Marcelino-Sadaba et al., 2017)
Clay bricks	Cradle-to-gate	UK	Lower Oxford clay-based stabilised with lime	1.41E+00	8.30E-02	4.62E-03	6.80E+02	1.66E+02	3.76E-07	9.64E-01	2.11E-02	2.15E-06	Expressed per unit strength	(Marcelino-Sadaba et al., 2017)
Clay bricks	Cradle-to-gate	UK	Lower Oxford clay-based stabilised with lime and GGBS**	5.09E+00	2.44E-02	1.73E-03	2.40E+02	6.20E+01	1.28E-07	2.87E-01	6.64E-03	4.75E-07	Expressed per unit strength	(Marcelino-Sadaba et al., 2017)
Clay bricks	Cradle-to-gate	UK	Lower Oxford clay-based stabilised with PC*	1.77E+01	4.39E-02	5.25E-03	1.14E+03	7.16E+01	3.05E-07	5.02E-01	1.16E-02	8.46E-07	Expressed per unit strength	(Marcelino-Sadaba et al., 2017)

Clay bricks	Cradle-to-gate	UK	Lower Oxford clay-based stabilised with PC* and GGBS**	7.69E+00	7.76E-02	2.46E-03	4.40E+02	5.57E+01	1.50E-07	2.47E-01	6.08E-03	2.56E-07	Expressed per unit strength	(Marcelino-Sadaba et al., 2017)
Typical fired clay brick	Cradle-to-gate	Typical PC*-based concrete block	Common material controls	1.23E+01	6.33E-03	1.29E-03	2.09E+02	2.09E+02	2.76E-07	1.56E-01	1.88E-03	1.20E-08	Expressed per unit strength	(Marcelino-Sadaba et al., 2017)
Ordinary PC* concrete (PC+gravel+sand)	Cradle-to-grave	Singapore	Conventional production (plus clay extraction)	1.07E-01	1.00E-03	0.00E+00		1.67E+00			2.60E-02		1 kg of concrete	(Kua and Kamath, 2014)
30%-coarse recycled PC*concrete (PC+rec.concrete+gravel+sand)	Cradle-to-grave	Singapore	30% replacemnet of natural aggregates by recycled concrete waste - production	9.20E-02	1.00E-03	0.00E+00		1.54E+00			2.50E-02		1 kg of concrete	(Kua and Kamath, 2014)
Clay brick	Cradle-to-grave	Singapore	Conventional production (plus clay extraction)	2.07E-01	2.00E-03	0.00E+00		4.90E+00			8.00E-02		1 kg of clay brick	(Kua and Kamath, 2014)
Natural aggregates (crushed limestone)	Cradle-to-site	Portugal	Extraction-production	1.54E+01	1.08E-01	2.64E-02	2.18E-06	2.46E+02	1.05E-01	2.91E+03	5.53E+00	2.79E-03	No functional unit	(Estanqueiro et al., 2018)
Recycled coarse aggregates (CDW)	Cradle-to-gate	Portugal	Recycling (fixed plant)-	2.44E+01	1.33E-01	3.34E-02	4.37E-06	4.51E+02	1.93E-01	4.15E+03	8.13E+00	4.16E-03	No functional unit	(Estanqueiro et al., 2018)

			production											
Recycled coarse aggregates (CDW)	Cradle-to-gate	Portugal	Recycling mobile plant)-production	2.05E+01	1.14E-01	2.85E-02	3.76E-06	3.86E+02	1.66E-01	3.46E+03	7.26E+00	3.57 E-03	No functional unit	(Estanqueiro et al., 2018)
Ordinary PC* concrete (CEM I+gravel+sand)	Cradle-to-gate	France	20-mm concrete with identical strength	4.44E+02	8.60E-01	2.20E-01	2.87E-05	2.14E+03	1.64E+00			1.30 E-01	Same mechanical strength	(Serres et al., 2016)
100% recycled PC* concrete (CEM I+rec.gravel+rec.sand)	Cradle-to-gate	France	20-mm concrete with identical strength	3.35E+02	1.22E+00	1.30E-01	2.01E-05	1.39E+03	8.70E-01			7.00 E-02	Same mechanical strength	(Serres et al., 2016)
100%-coarse recycled PC* concrete (CEM I+rec.gravel+sand)	Cradle-to-gate	France	20-mm concrete with identical strength	3.79E+02	1.08E+00	1.70E-01	2.10E-05	1.60E+03	1.19E+00			1.00 E-01	Same mechanical strength	(Serres et al., 2016)
Ordinary PC* concrete (CEM II+gravel+sand)	Cradle-to-gate	France	8-mm with identical volume of granular skeleton	4.69E+02	7.50E-01	2.30E-01	2.96E-05	2.22E+03	1.70E+00			1.40 E-01	Same volume composition	(Serres et al., 2016)
100% recycled PC* concrete (CEM II+rec.gravel+rec.sand)	Cradle-to-gate	France	8-mm with identical volume of granular skeleton	3.57E+02	9.00E-01	1.30E-01	1.65E-05	1.29E+03	9.00E-01			7.00 E-03	Same volume composition	(Serres et al., 2016)
100% recycled PC* concrete (CEM II+rec.terracotta tiles)	Cradle-to-gate	France	8-mm with identical volume of granular skeleton	3.66E+02	9.80E-01	1.50E-01	2.67E-05	1.87E+03	9.90E-01			6.00 E-02	Same volume composition	(Serres et al., 2016)

Limestone-PC* concrete (PC+limestone+sand)	Cradle-to-cradle	China	Conventional production (plus extraction)	4.03E+02	1.21E+03	1 m^3 of concrete	(Ding et al., 2016)
50%-coarse recycled limestone-PC* concrete (PC* +rec.concrete+limestone+sand)	Cradle-to-cradle	China	50% replacemnet of natural aggregates by recycled concrete waste without shaping (removing the hardened mortar from recycled aggregates) - production	4.06E+02	1.21E+03	1 m^3 of concrete	(Ding et al., 2016)
50%-coarse recycled PC* concrete (PC*+rec.concrete+limestone+sand)	Cradle-to-cradle	China	50% replacemnet of natural aggregates by recycled concrete waste with shaping - production	4.17E+02	1.23E+03	1 m^3 of concrete	(Ding et al., 2016)

100%-coarse recycled PC* concrete (PC*+rec.concrete+sand)	Cradle-to-cradle	China	100% replacemnet of natural aggregates by recycled concrete waste without shaping - production	4.06E+02	1.22E+03	1 m^3 of concrete	(Ding et al., 2016)
100%-coarse recycled limestone-PC* concrete (PC*+rec.concrete+sand)	Cradle-to-cradle	China	100% replacemnet of natural aggregates by recycled concrete waste with shaping - production	4.29E+02	1.25E+03	1 m^3 of concrete	(Ding et al., 2016)
Ordinary PC* concrete (PC*+gravel+sand)- steam-cured	Cradle-to-gate	China	Conventional production (Steam-cured) plus extraction	4.19E+02		1 m^3 of concrete	(Huang et al., 2019)
Ordinary PC* concrete (PC*+gravel+sand)- CO2-cured	Cradle-to-gate	China	Conventional production (CO2 mineral carbonation-cured) plus extraction	3.64E+02	2.88E+03	1 m^3 of concrete	(Huang et al., 2019)

5% Wollastonite-PC* concrete (PC*+gravel+Wollastonite+sand)- CO2-cured	Cradle-to-gate	China	5% of total aggregates was Wollastonite (CO2 mineral carbonation-cured) plus extraction	2.92E+02	2.61E+03	1 m^3 of concrete	(Huang et al., 2019)
5% MgO-PC* concrete (PC*+gravel+MgO+sand)- CO2-cured	Cradle-to-gate	China	5% of total aggregates was MgO (CO2 mineral carbonation-cured) plus extraction	4.10E+02	2.95E+03	1 m^3 of concrete	(Huang et al., 2019)
5% limestone-PC* concrete (PC*+gravel+limestone+sand)- CO2-cured	Cradle-to-gate	China	5% of total aggregates was limestone (CO2 mineral carbonation-cured) plus extraction	4.21E+02	4.12E+03	1 m^3 of concrete	(Huang et al., 2019)
25% calcium silicate concrete (gravel+calcium silicate+sand)- CO2-cured	Cradle-to-gate	China	25% of total aggregates was calcium silicate (CO2 mineral carbonation-cured) plus extraction	3.03E+02	2.56E+03	1 m^3 of concrete	(Huang et al., 2019)

100%-coarse recycled PC* concrete (PC*+rec.concrete+sand)-CO2-cured	Cradle-to-gate	China	100% of coarse aggregates was recycled concrete (CO2 mineral carbonation-cured) plus extraction	4.54E+02		4.37E+03	1 m^3 of concrete	(Huang et al., 2019)
25% Slag-PC* concrete (PC*+gravel+GGBS**+sand)- CO2-cured	Cradle-to-gate	China	25% of total aggregates was GGBS** (CO2 mineral carbonation-cured) plus extraction	3.16E+02		2.55E+03	1 m^3 of concrete	(Huang et al., 2019)
Ordinary PC* blend	Cradle-to-gate	US	Self-consolidating concrete (SSC) mixtures made under varying proportions	5.69E+02	1.35E+00		1 m^3 of concrete	(Celik et al., 2015)
PC* -fly_ash-limestone_powder (85-0-15)	Cradle-to-gate	US	Self-consolidating concrete (SSC) mixtures made under varying proportions	4.87E+02	1.15E+00		1 m^3 of concrete	(Celik et al., 2015)

PC* -fly_ash-limestone_powder (75-0-25)	Cradle-to-gate	US	Self-consolidating concrete (SSC) mixtures made under varying proportions	4.34E+02	1.03E+00	1 m³ of concrete	(Celik et al., 2015)
PC* -fly_ash-limestone_powder (70-30-0)	Cradle-to-gate	US	Self-consolidating concrete (SSC) mixtures made under varying proportions	4.12E+02	9.74E-01	1 m³ of concrete	(Celik et al., 2015)
PC* -fly_ash-limestone_powder (50-50-0)	Cradle-to-gate	US	Self-consolidating concrete (SSC) mixtures made under varying proportions	3.11E+02	7.31E-01	1 m³ of concrete	(Celik et al., 2015)
PC* -fly_ash-limestone_powder (55-30-15)	Cradle-to-gate	US	Self-consolidating concrete (SSC) mixtures made under varying proportions	3.33E+02	7.84E-01	1 m³ of concrete	(Celik et al., 2015)

PC* -fly_ash-limestone_powder (45-40-15)	Cradle-to-gate	US	Self-consolidating concrete (SSC) mixtures made under varying proportions	2.82E+02	6.63E-01	1 m^3 of concrete	(Celik et al., 2015)
PC* -fly_ash-limestone_powder (35-50-15)	Cradle-to-gate	US	Self-consolidating concrete (SSC) mixtures made under varying proportions	2.32E+02	5.44E-01	1 m^3 of concrete	(Celik et al., 2015)
PC* -fly_ash-limestone_powder (25-60-15)	Cradle-to-gate	US	Self-consolidating concrete (SSC) mixtures made under varying proportions	1.83E+02	4.27E-01	1 m^3 of concrete	(Celik et al., 2015)
PC* -fly_ash-limestone_powder (55-20-25)	Cradle-to-gate	US	Self-consolidating concrete (SSC) mixtures made under varying proportions	3.32E+02	7.83E-01	1 m^3 of concrete	(Celik et al., 2015)

PC* -fly_ash-limestone_powder (45-30-25)	Cradle-to-gate	US	Self-consolidating concrete (SSC) mixtures made under varying proportions	2.81E+02	6.61E-01								1 m^3 of concrete	(Celik et al., 2015)
PC* -fly_ash-limestone_powder (35-40-25)	Cradle-to-gate	US	Self-consolidating concrete (SSC) mixtures made under varying proportions	2.31E+02	5.42E-01								1 m^3 of concrete	(Celik et al., 2015)
PC* -fly_ash-limestone_powder (25-50-25)	Cradle-to-gate	US	Self-consolidating concrete (SSC) mixtures made under varying proportions	1.82E+02	4.26E-01								1 m^3 of concrete	(Celik et al., 2015)
Natural aggregates (crushed limestone)	Cradle-to-gate	China	Extraction-production	3.27E+00	1.97E-02	3.01E-03					2.84E-02	2.12 E-03	1 m^3 of concrete	(Guo et al., 2018)
Recycled coarse aggregates (waste concrete)	Cradle-to-gate	China	Recycling - production	5.10E+00	1.94E-02	2.78E-03					2.61E-02	2.16 E-03	1 m^3 of concrete	(Guo et al., 2018)
Natural fine aggregates (sand)	Cradle-to-gate	China	Extraction (from river)-production	1.79E+00	1.49E-02	1.47E-03					1.42E-02	8.13 E-04	1 m^3 of concrete	(Guo et al., 2018)

Limestone-PC concrete (PC+limestone+sand)	Cradle-to-gate	China	Extraction-production	3.24E+02	1.49E+00	1.59E-01				1.51E+00	2.89 E-01	1 m^3 of concrete	(Guo et al., 2018)
75%-coarse recycled limestone-PC concrete (PC+rec.concrete+limestone+sand)	Cradle-to-gate	China	75% replacemnet of natural aggregates by recycled concrete waste - production	3.07E+02	1.25E+00	1.19E-01				1.13E+00	1.75 E-01	1 m^3 of concrete	(Guo et al., 2018)
Hollow concrete block	Acquisition of raw materials, processing and manufacturing	UK	Produced by Portland cement and sand	1.04E+01	2.90E-01							1 m2 of partion wall system	(Broun and Menzies, 2011)
Clay brick	Acquisition of raw materials, processing and manufacturing	UK	Produced by cement, lime and sand	2.55E+01	6.10E-01							2 m2 of partion wall system	(Broun and Menzies, 2011)
Hollow concrete block	Maintenance	UK	Produced by Portland cement and sand	3.48	8.40E-02							3 m2 of partion wall system	(Broun and Menzies, 2011)
Clay brick	Maintenance	UK	Produced by cement, lime and sand	2.192	5.30E-02							4 m2 of partion wall system	(Broun and Menzies, 2011)
Hollow concrete block	Demolition	UK	Produced by Portland cement and sand	4.15E-01	1.50E-02							5 m2 of partion wall system	(Broun and Menzies, 2011)

Clay brick	Demolition	UK	Produced by cement, lime and sand	3.60E-01	1.20E-02	6 m2 of partion wall system	(Broun and Menzies, 2011)

[1]LCA: Life cycle assessment; [2]GWP: Global warming potential; [3]AP: Acidification potential; [4]EP: Eutrophication potential; [5]ODP: Ozone layer depletion potential using as equivalent the chlorofluorocarbon-11; [6]ADP: Abiotic depletion potential; [7]ETP: Ecotoxicity potential; [8]HTP: Human toxicity potential; [9]POCP: Photochemical ozone creation potential; [*]Ranges depend on the type of fibre additive (wheat straw or sawdust); **GGBS: Ground Granulated blast-furnace slag