Sustainable Production and Consumption 58 (2025) 319–333

Contents lists available at ScienceDirect

Sustainable Production and Consumption

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

Thousand cuts: a realistic route to decarbonise the UK cement and concrete sector by 2050

Madeline C.S. Rihner [a,*], Hisham Hafez [b], Brant Walkley [a], Phil Purnell [b], Michal Drewniok [b]

[a] School of Chemical, Materials and Biological Engineering, University of Sheffield, Sheffield S1 3JD, UK
[b] School of Civil Engineering, University of Leeds, Leeds LS2 9LG, UK

ARTICLE INFO

Editor: Prof. Raymond Tan

Keywords:
Decarbonisation strategies
2050 targets
CCUS
Demand reduction
Low carbon cement
Circular economy

ABSTRACT

To meet net-zero CO_2 targets by 2050, the United Kingdom (UK)'s cement and concrete sector must implement decarbonisation strategies of different readiness levels and effectiveness. These strategies have been presented thoroughly in UK and European Union decarbonisation roadmaps. However, it is challenging to predict, with confidence, whether the UK's 2050 net-zero targets are achievable. This study aims to balance the expectations placed on low-maturity (LM) and high-maturity (HM) strategies such as utilising a lower clinker factor and the use of carbon capture technologies respectively to determine a realistic route in which the UK can reach net-zero targets through a decomposition analysis of each strategy. The sector's carbon emissions were determined by performing a material flow analysis and life cycle assessment. The results showed that by 2050, 11 $MtCO_{2eq}$/yr is expected to be emitted in 2050 under the business-as-usual scenario. HM strategies have an abatement potential of 4.2 $MtCO_{2eq}$/yr, while LM strategies are expected to abate 3.4 $MtCO_{2eq}$/yr. However, LM strategies are limited by industry's willingness to shift from current practices, while the implementation of HM strategies are impeded by financial and resource constraints. Accordingly, it is improbable for the sector to meet UK net-zero carbon targets with confidence unless the yearly concrete demand is reduced by 40 %. To enable the maximum potential of reusing the UK's building stock, direct public incentives, shifts in economic models and policy frameworks are needed.

1. Introduction

The concrete sector accounts for as much as 8 % of all global anthropogenic carbon dioxide (CO_2) emissions, stemming mostly from the production of cement, one of the essential constituents in concrete (Lehne and Preston, 2018). In 2021, the United Kingdom's (UK) cement and concrete sector produced roughly 7 Mt of CO_2 emissions, or 9 % of the country's manufacturing emissions (Drewniok et al., 2023). With the global threat of climate change constantly increasing due to rising greenhouse gas emissions, 196 countries under the Paris Agreement have pledged to achieve net-zero carbon emissions by 2050 (Busch et al., 2022). In accordance with this agreement, the UK passed an amendment to the 2008 Climate Change Act in 2019 requiring the country to shift its target from an 80 % reduction in greenhouse gas emissions from 1990 levels to net-zero emissions by 2050 (UK House of Parliament, 2019, Mcgarry et al., 2022). Accordingly, the UK government's independent statutory, the Climate Change Committee (CCC), unveiled its 6th carbon budget in 2020 which provides the cumulative reduction in GHG emissions over the period leading to 2050 (Emmerling et al., 2019). With current carbon emissions serving as a baseline value, the CCC outlined that a 40 % reduction is needed by 2030, a 20 % reduction is needed by 2040, and a 100 % reduction is needed by 2050.

Under these new guidelines, several decarbonisation roadmaps have been endorsed by the UK government aiming to achieve a net-zero cement and concrete sector by 2050 (WSP and DNV-G, 2015). In 2015, the UK Department for Business, Energy, and Industrial Strategy issued roadmaps for six 'foundation industries', including the cement and concrete sector, to achieve net-zero by 2050 through several

Abbreviations: BAU, Business-as-usual; CCC, Climate Change Committee; CCUS, Carbon capture, utilisation, and storage; EU, European Union; GGBS, Ground granulated blast furnace slag; HM, High-maturity; IEA, International Energy Agency; LCA, Life cycle assessment; LM, Low-maturity; MFA, Material flow analysis; MPA, Mineral Products Association; SCMs, Supplementary cementitious materials; TMRL, Technology and market readiness level; UK, United Kingdom.

* Corresponding author.
E-mail addresses: mcsrihner1@sheffield.ac.uk (M.C.S. Rihner), h.hafez@leeds.ac.uk (H. Hafez), b.walkley@sheffield.ac.uk (B. Walkley), p.purnell@leeds.ac.uk (P. Purnell), m.p.drewniok@leeds.ac.uk (M. Drewniok).

https://doi.org/10.1016/j.spc.2025.06.010
Received 18 February 2025; Received in revised form 7 June 2025; Accepted 22 June 2025
Available online 24 June 2025
2352-5509/© 2025 The Authors. Published by Elsevier Ltd on behalf of Institution of Chemical Engineers. This is an open access article under the CC BY license (http://creativecommons.org/licenses/by/4.0/).

proposed pathways (WSP and DNV-G, 2015). One of the main objectives established in these roadmaps was a policy framework that would allow UK manufactures to achieve net-zero targets without offshoring production (Hammond, 2022). To assess potential decarbonisation pathways to 2050, a simplified modelling framework using feasible strategies was developed by analysts; the results of which yielded a wide range of uncertainties (Griffin et al., 2014). For the cement and concrete sector, it was found that the 'balanced net-zero' pathway produced the highest degree of certainty, with a projected 70 % and 90 % reduction in emissions by 2030 and 2040 respectively compared to 2018 levels. In 2020, this same pathway was also assessed by the CCC and was found to be "realistically achievable" (Hammond, 2022).

The route to 2050 is comprised of two main strategies: low-maturity and high-maturity; the definition of each being adopted from the International Energy Agency (IEA)'s technology and market readiness level (TMRL) measurement system. While this system is similar to other technology readiness level scales (measuring one to nine), two additional levels were added (ten and eleven) to account for market readiness (IEA, 2021a). Accordingly, strategies with a TMRL below ten are considered low-maturity (LM), while those with TMRL greater than nine are considered high-maturity (HM). There are 16 decarbonisation strategies commonly listed in UK and European Union (EU) cement and concrete decarbonisation roadmaps, but only 12 were seen as fit for the scope of this study. The rationale of exclusion for the remaining strategies are noted in Table 1.

The 12 remaining strategies are divided into LM (carbon capture, utilisation, and storage (CCUS), electrification of clinker production, the use of non-clinker binders, and the recycling of waste concrete into binders) and HM (using low-carbon electricity and low-carbon fuels, shifting to more energy-efficient clinker kilns, using less clinker in cement, optimising the structural design of concrete structures and the mix design of concrete products, and reducing the over-specification of concrete). Despite being classified as a HM decarbonisation strategy according to the TMRL scale, the ability to preserve existing buildings through reuse and refurbishment is defined separately in this study as a circularity solution. The reason for this classification is due to the strategy's unique role in achieving net-zero as the only strategy that is able to reduce the demand for new cement and concrete products. The definition and classification of all 12 strategies is illustrated in Fig. 1.

The IEA states that HM strategies will always exhibit market precedence over LM strategies, regardless of their scale. The probability of achieving the aspired-for market penetration for HM strategies is high once a strategy has reached the "predictable growth" phase (IEA, 2021a). In contrast, LM strategies are dictated by technological, economic, and socio-political barriers that hinder a strategy's first year of introduction and its ability to achieve intended market penetration. Despite this uncertainty, the carbon abatement expected from LM strategies in the CCC 6th carbon budget is significantly larger than that expected from HM strategies (Hammond, 2022). The same disproportional contribution of LM strategies to the decarbonisation potential by 2050 is also seen in most cement and concrete roadmaps issued to date. A comprehensive and detailed study by Pamenter and Myers (2021) reviewed several UK and EU cement and concrete sector decarbonisation roadmaps and analysed the role of each stakeholder along the value chain in enabling the transition to net-zero. It was concluded that it is feasible for the UK cement and concrete sector to achieve net-zero by 2050 by reducing 72 % of emissions via LM strategies (28 % from alternative binders, 23 % from kiln electrification, and 21 % from CCUS) and 28 % of emissions via HM strategies. In contrast, the Mineral Products Association (MPA) in the UK attributes a 60 % decarbonisation potential value to CCUS (MPA, 2020b). This variability between decarbonisation potential values attributed to each strategy was also reported in a meta-analysis of six cement and concrete decarbonisation roadmaps in the UK and EU by Marsh et al. (2023), where it was concluded that there is a 50–80 % variability in the decarbonisation potential associated with LM strategies.

The scope of this paper explores the different routes in which the UK cement and concrete sector may achieve net-zero emissions by 2050 through the implementation of decarbonisation strategies and circular economy principles. The latter providing a solution that is able to reduce carbon emissions across the entire supply chain by reducing material demand and waste through the reuse of existing structures (Yang et al., 2023). To achieve this, a first of its kind model for evaluating decarbonisation potential was created. The first objective is to assess the current business-as-usual (BAU) by benchmarking the sector's annual consumption volumes and embodied carbon values. The literature established that there is an absence of clarity as to the realistic degree of decarbonisation potential projected in 2050. Therefore, the second objective is to calculate the decarbonisation potential from LM, HM, and circular strategy implementation by 2050. The third objective is to compare two decarbonisation potential pathways that are defined by the likelihood of strategy implementation to the net-zero pathway outlined by the UK CCC.

Table 1
Excluded roadmap cement and concrete sector decarbonisation strategies.

Decarbonisation strategy	Description of strategy	Rationale for exclusion
Material substitution	Using alternative materials such as steel and timber	This strategy has not been proven to achieve environmental or economic savings when compared to concrete (D'amico et al., 2021).
Decarbonisation of transportation	Reducing fossil fuel consumption by utilising more sustainable transport methods such as electric vehicles	The system boundary for this study excludes transportation. In addition, transportation emissions within the cement and concrete sector are generally insignificant compared to those associated with production (Mcgrath et al., 2012).
Re-carbonation	Utilising use and end-of-life concrete as a form of carbon capture	These strategies fall outside the scope of the study given that they are a function of both exposure and operational conditions of concrete during its service life.
Leveraged thermal mass	Reduce operational carbon emissions generated from heating and cooling	

2. Methodology

The methodology followed for this study, along with the flow of the data, is best described in Fig. 2. All data utilised in this study comes from secondary, literary sources. In order to assess the decarbonisation potential of any given strategy, the business-as-usual (BAU) scenario must first be determined. Therefore, stage 1 assesses the baseline 2025 supply volumes with a material flow analysis and the carbon intensity values with a life cycle assessment. Further information regarding both of these applied methods are detailed in Section 2.1.1 and Section 2.1.2, respectively. Using these values, the BAU emissions in 2050 were determined based on the predicted increase cement and concrete demand. The second stage critically analyses the decarbonisation potential of both LM and HM strategies. Decarbonisation strategies that have existing market precedence are classified as HM, and those that do not as are classified as LM. In addition to these two strategy types, a circularity strategy was also evaluated which considers the reuse and refurbishment of existing building stock. Since the scope of this study is limited to cement and concrete consumption only, reuse and refurbishment of other building materials was not considered. This study also assumes that the concrete does not require significant repair; therefore, the embodied carbon associated with any repairs to extend a building's service life is minimal compared to new construction. Lastly, using these

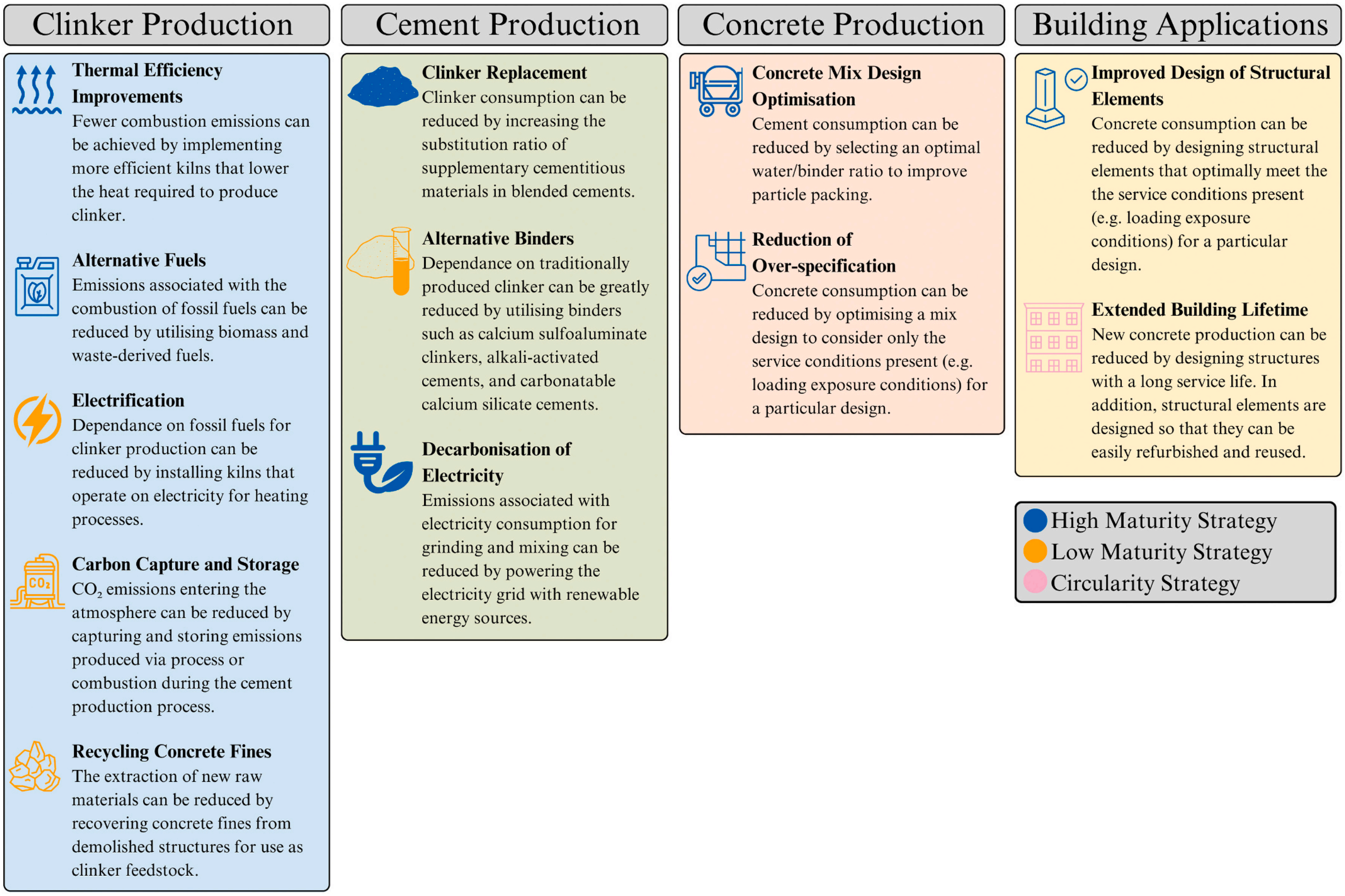

Fig. 1. Cement and concrete sector decarbonisation strategies found in UK and EU roadmaps (adapted from Marsh et al. (2023)).

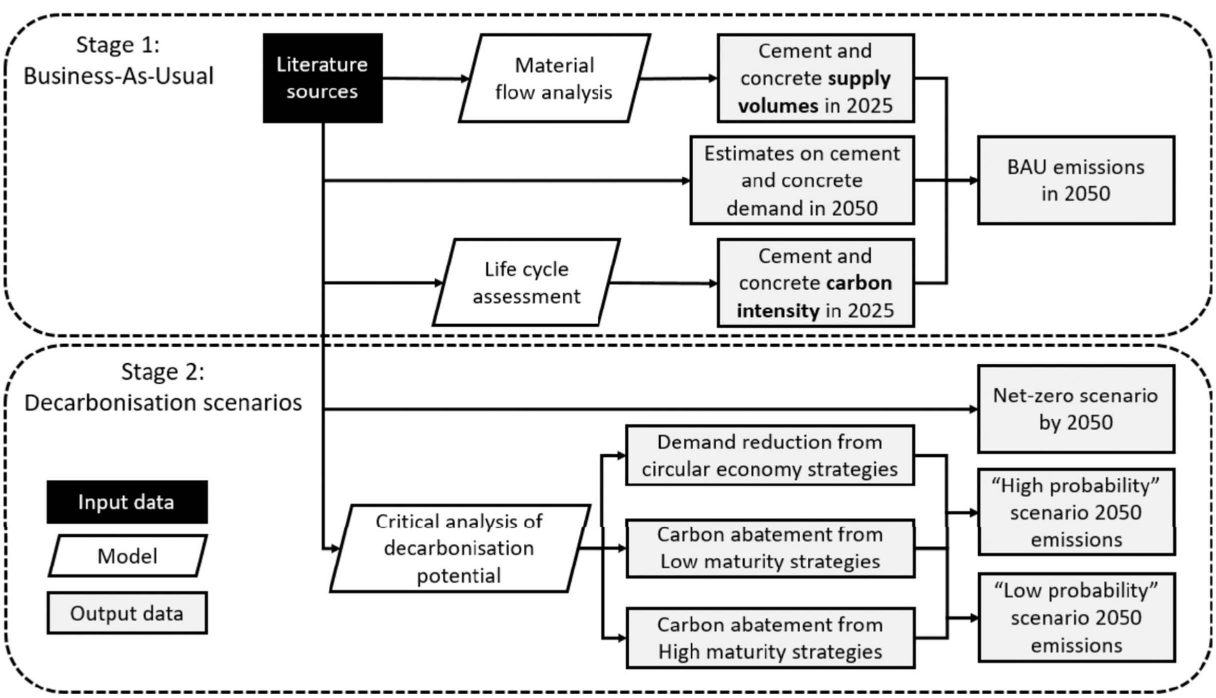

Fig. 2. Methodology flowchart.

values, two different pathways to net-zero with different probabilities were evaluated and compared to the CCC's net-zero pathway. While this study examines the UK as a case study, the methodology outlined can be applied to any geographical region given data availability.

2.1. Business-as-usual

The BAU carbon emissions for any product at its market level is defined by the volume of product that is produced and its carbon intensity as expressed in Eq. 1.

$$\text{BAU carbon emissions}\ \left(\mathrm{MtCO_{2eq}}/\mathrm{yr}\right) = \text{Product annual consumption}\ (\mathrm{Mt}/\mathrm{yr})^{*}\ \text{Product carbon intensity}\ \left(\mathrm{kgCO_{2eq}}/\mathrm{kg}\right) \tag{1}$$

Within the cement and concrete sector, carbon emissions are usually assessed at three distinct market levels: clinker, cement, and concrete. Clinker is the primary constituent in ordinary portland cement (OPC). While not typically sold as a product, it is examined individually due to the high carbon emissions associated with its production. The cement market level is defined as all cementitious materials including OPC and supplementary cementitious materials (SCMs). While cement is most often produced for use in concrete products, it is also used in non-concrete applications such as mortars and soil stabilisers. The concrete market level encompasses a range of different concrete products including ready-mixed, precast, and blocks. Concrete used for infrastructure, pavements, and roads are included in the market-level analysis of this study, but the reduction in concrete demand through reuse is limited to its application in buildings (e.g. residential, office, retail). As noted in Eq. 2, the BAU carbon emissions arising from the UK cement and concrete sector is a function of both concrete and cement for non-concrete applications. When evaluating the carbon emissions of large complex systems such as industrial sectors, a methodological approach that combines material flow analysis and life cycle assessment has been found to yield more robust and transparent results when compared to utilising these methods independently (Meglin et al., 2022). Therefore, at each market level in the cement and concrete sector, the consumption volumes and the carbon intensity values were benchmarked by conducting a material flow analysis and life cycle assessment, respectively.

$$\begin{aligned}&\text{BAU cement and concrete sector carbon emissions in 2025}\ \left(\mathrm{MtCO_{2eq}}/\mathrm{yr}\right)\\&\quad=\left(\text{Average concrete carbon intensity}\ \left(\mathrm{kgCO_{2eq}}/\mathrm{kg}\right)^{*}\ \text{Annual concrete consumption}\ (\mathrm{Mt}/\mathrm{yr})\right)\\&\quad+\left(\text{Average cement carbon intensity}\ \left(\mathrm{kgCO_{2eq}}/\mathrm{kg}\right)^{*}\ \text{Annual cement consumption}\ (\mathrm{Mt}/\mathrm{yr})^{*}\ \text{Cement consumption for other uses}\ (\%)\right)\end{aligned} \tag{2}$$

Between 2025 and 2050, the demand for cement and concrete products is expected to increase as a rising global population leads to an increased demand for new buildings and infrastructure. Therefore, by multiplying the benchmarked 2025 cement and concrete sector carbon emissions by the expected increase in cement and concrete demand, the BAU carbon emissions in 2050 can be determined. This calculation is expressed in Eq. 3.

$$\begin{aligned}&\text{BAU cement and concrete sector carbon emissions in 2050}\ \left(\mathrm{MtCO_{2eq}}/\mathrm{yr}\right)\\&\quad=\text{BAU cement and concrete sector carbon emissions in 2025}\ \left(\mathrm{MtCO_{2eq}}/\mathrm{yr}\right)^{*}\ \text{Demand for cement and concrete}\ (\%)\end{aligned} \tag{3}$$

2.1.1. Material flow analysis- 2025 cement and concrete market

To examine the physical flow of products within the UK cement and concrete sector, a material flow analysis (MFA) was conducted. This method utilises a mass balance approach to ensure that all input flows are equal to the output flows (Meglin et al., 2022). By quantifying these values, the annual consumption volumes can be benchmarked to determine the sector's carbon emissions as noted in Eq. 2. The system boundary for this MFA is defined as 'cradle-to-gate' and encompasses the consumption of raw materials for the production of clinker, cement, and concrete products (both produced and imported) in addition to flow of these finished products for building applications. All data was taken from publicly available UK industrial reports and secondary literature sources (MPA, 2023; UN Comtrade, 2023). Due to data availability, the year of analysis selected for this MFA was 2022 and it was assumed that this data closely reflects the cement and concrete sector's consumption values in 2025. All product flows are measured in megatons per year.

At the clinker market level, the clinker consumption value was taken as the sum of clinker produced and imported. Since the clinker amount in imported cement is not straightforwardly calculated nor reported, it was assumed to be equal to CEM I (OPC). The cement market level encompasses both cement (OPC) and supplementary cementitious materials (SCMs). The total cement consumption value (produced and imported) was taken as the total clinker and additions (e.g. gypsum) consumed. SCMs are materials that exhibit cementitious properties that are used to create lower carbon blended cements. These materials include, but are not limited to, ground granulated blast furnace slag (GGBS), fly ash, and limestone fines. As previously mentioned, cement is used in both concrete and non-concrete applications; therefore, the percentage of cement used for each application type was also determined. The concrete market level encompasses cement market level products and all remaining concrete constituents (e.g. water, aggregates, and chemical admixtures). To determine this, an average concrete mix design was assumed. A water to cement ratio of 0.50 was taken to estimate the total water consumed for concrete production, and the total chemical admixtures consumed was assumed to be 1 % of the total cement content. The total amount of aggregates consumed was calculated by subtracting the concrete consumption volume by the volume of all other concrete constituents consumed.

In addition to determining the total concrete consumption value, the percentage of concrete used for building applications was also determined. Concrete consumption for building applications is defined by material intensity and demand. Therefore, the percentage of concrete used for building applications can be calculated using Eq. 4.

$$\text{Concrete consumption used for buildings per year } (\%) = \left(\text{Concrete material intensity in buildings } (\text{kg/m}^2)^{*} \text{ Demand for concrete buildings } (\text{km}^2)\right)/\text{Annual concrete consumption } (\text{Mt/yr}) \tag{4}$$

Material intensity of concrete use in buildings represents the amount of concrete specified per unit of building floor area to fulfil structural and non-structural requirements. While the material intensity for infrastructure and other projects is often regarded as a constant due to its durability-bound strategic nature, the material intensity of concrete in buildings is often subject to potential reductions within decarbonisation strategies (Marsh et al., 2023). The minimum material intensity for each typology of concrete units (e.g. a low-rise building) could be calculated based on the code requirements depending on the project specifications (e.g. exposure conditions and serviceability requirements) (Cabeza et al., 2021). However, values vary widely on a case-by-case basis. A recent industry accepted model created by Drewniok et al. (2023) shows that, for the same building typology, the specified material intensity varies between 700 and 1400 kg per gross floor area of a building. Therefore using the values from Drewniok et al. (2023), the market average material intensity of new construction was determined (SI, Table 2). The values utilised for the MFA are summarised in Table 1 of the SI.

$$\text{Average concrete carbon intensity } \left(\text{kgCO}_{2\text{eq}}/\text{kg}\right) = \text{Average cement carbon intensity } \left(\text{kgCO}_{2\text{eq}}/\text{kg}\right)^{*} \text{ Binder content } (\text{kg/kg}) + \sum \text{Carbon intensity of the non}-\text{binder components } \left(\text{kgCO}_{2\text{eq}}/\text{kg}\right)^{*} \text{ Non}-\text{binder content } (\text{kg/kg}) \tag{5}$$

2.1.2. Life cycle assessment -2025 cement and concrete market

To benchmark the carbon intensity values at all three market products in the UK cement and concrete sector, life cycle assessment (LCA) methodology was utilised. The ISO14040 and ISO14044 specifies the four main stages of an LCA: goal and scope definition, inventory data collection, impact assessment, and interpretation (ISO, 2006a; ISO, 2006b). The goal of this LCA is to assess the carbon intensity associated with the production of three products: clinker, cement, and concrete. For this study, two market scenarios (UK and global) and one best practice scenario was assessed. An LCA scope is mainly defined by three parameters: system boundary, functional (or declared) unit, and allocation procedure. The three most common system boundary types for an LCA are 'cradle-to-gate', 'cradle-to-grave', and 'cradle-to-cradle'. Due to the uncertainty regarding each product's upstream process, a 'cradle-to-gate' system boundary was selected, which typically includes raw material extraction (stage A1), the transportation of those raw materials to the factory (stage A2), and the manufacturing of the product itself (stage A3) (Hafez et al., 2019). While cement and concrete sector LCA studies most often select this boundary type due to the manufacturing stage resulting in the highest environmental impact, the boundary for this study is limited to processing and production (stage A3) due to the uncertainty regarding the locations of production (Rihner et al., 2025). A detailed illustration of this system boundary can be found in supplementary information (SI, Fig. 1). In line with other market decomposition analyses that examine multiple products, a mass declared unit (e.g. one kilogram) was selected for each product assessed (e.g. clinker, cement, concrete). A declared unit was selected over a functional unit as functional units are used only when comparing specific products that serve the same purpose. To accurately account for the environmental impact of material by-products such as GGBS, data was selected which utilised the economic allocation method (MPA, 2025).

The second stage of an LCA is inventory data collection, for which this model presents a top-down approach using reports (ERMCO, 2020; MPA, 2020a; IEA, 2022), databases (Wernet et al., 2016), secondary literature sources. This approach was selected given that each market product is a key constituent within another market product (e.g., clinker is used to produce cement, cement is used to produce concrete). As a result, the carbon intensity of one market product is dependent on the carbon intensity of another. Calculations were performed in Excel using Eqs. 5–7. For the impact assessment stage, embodied carbon ($kgCO_{2eq}$/kg) was the only environmental impact indicator analysed in line with the study's goal. As expressed in Eq. 5, the average carbon intensity of concrete is defined by its binder and non-binder components.

Binder content is a function of the total volume of paste required to bind the non-cementitious components of the mix. It is a property of a

Table 2
Summary of the inventory data used to determine cement and concrete sector carbon intensity values (References: Shen et al., 2015; IEA, 2021b; IEA, 2023; Summerbell, 2018; MPA, 2020a; IEA, 2022; ESO, 2023; Ember, 2024; Drewniok et al., 2023; Wernet et al., 2016; ERMCO, 2020; Monteiro et al., 2017).

Market Level	Variable	Unit	UK Value	Reference	Global Value	Reference
Clinker	Clinker process emissions	kgCO2eq/kg	0.49	Shen et al. (2015)	0.49	Shen et al. (2015)
	Energy intensity in clinker production	GJ/t clinker	3.7	IEA (2021b)	3.55	IEA (2023)
	Carbon intensity of fuel mix	kgCO2eq/MJ	0.07	Summerbell (2018)	0.09	Summerbell (2018)
	Average clinker carbon intensity	**kgCO2eq/kg**	**Calculated**			
Cement	Clinker factor	-	0.7	MPA (2020a)	0.71	IEA (2022)
	Average SCM carbon intensity	kgCO2eq/kg	0.13	Calculated (SI Table 6)	0.13	Calculated (SI Table 6)
	Electric energy required for indirect processes	kWh/kg	0.11	Summerbell (2018)	0.1	IEA (2022)
	Carbon intensity of national electricity grid	kgCO2eq/kWh	0.2	ESO (2023)	0.44	Ember (2024)
	Average cement carbon intensity	**kgCO2eq/kg**	**Calculated**			
Concrete	Carbon intensity of the non-binder components	kgCO2eq/kg	0.007	Drewniok et al. (2023)	0.006	Ecoinvent; Wernet et al. (2016)
	Binder content	kg/kg	0.14	ERMCO (2020)	0.14	Calculated; (Monteiro et al., 2017, IEA, 2022)
	Average concrete carbon intensity	**kgCO2eq/kg**	**Calculated**			

concrete's mix design and therefore dependent on the gradation of the dry components, the water: binder ratio, and the required concrete properties such as slump and strength (Damineli et al., 2010). Fundamentally, it is the mass ratio of cementitious materials compared to the total concrete. The average binder content ratio is calculated by dividing the total volume of cement used in concrete by the volume of concrete produced (SI, Table 3). The mass ratio values for the remaining non-cementitious components of a concrete mix (water, coarse aggregates, fine aggregates, and water-reducing admixtures) were calculated and multiplied by each component's respective carbon intensity value (SI, Table 5). Since cement is also utilised in non-concrete applications, its average carbon intensity was calculated separately using Eq. 6.

$$\begin{aligned}&\text{Average cement carbon intensity}\left(\mathrm{kgCO_{2eq}}/\mathrm{kg}\right) = \text{Average clinker carbon intensity}\left(\mathrm{kgCO_{2eq}}/\mathrm{kg}\right)^{*}\ \text{Clinker factor}\\&\quad + \text{Average SCMs carbon intensity}\left(\mathrm{kgCO_{2eq}}/\mathrm{kg}\right)^{*}\ (1 - \text{Clinker factor})\\&\quad + \text{Electric energy required for indirect processes}\ (\mathrm{MJ}/\mathrm{kg})^{*}\ \text{Carbon intensity of national electricity grid}\left(\mathrm{kgCO_{2eq}}/\mathrm{MJ}\right)\end{aligned} \tag{6}$$

Table 3
Values for current global best practice of HM strategies (References: IEA, 2018; Summerbell, 2018; Ember, 2024; MPA, 2025; Scrivener et al., 2018; BSI, 2023).

Market Level	Variable	Value	Unit	Reference
Clinker	Carbon Intensity of Fuel Mix	0.058	kgCO2eq/MJ	IEA (2018)
	Energy Intensity of Clinker Production	3	GJ/t clinker	Summerbell (2018)
Cement	Carbon Intensity of National Electricity Grid	0.041	kgCO2eq/kWh	Switzerland; (Ember, 2024)
	Average SCM Carbon Intensity	0.08	kgCO2eq/kg	MPA (2025)
	Clinker Factor	0.5	-	Scrivener et al. (2018)
Concrete	Binder Content	0.12	kg/kg	BSI (2023)

Clinker factor represents the mass ratio of the amount of clinker in cement. While this value is often reported (IEA, 2022), sometimes it is not. In the case of the UK cement and concrete sector, SCMs are typically mixed alongside other constituents at ready-mix concrete plants and precast factories (Mcgrath et al., 2012). As a result, the clinker factor was calculated for this study by dividing the total amount of SCMs and clinker consumed (produced and imported) by the total cementitious material consumed. The average SCMs carbon intensity was determined taking the sum of the product market share for each SCM determined from the MFA and multiplying it by its respective carbon intensity value (SI, Table 6). The third and fourth parameters, electric energy required for indirect processes and carbon intensity of the national electricity grid, accounts for the embodied carbon created due to the indirect electric energy needed for grinding and mixing at various stages of cement manufacturing. The final variable, average carbon intensity of clinker, was calculated using Eq. 7.

Table 4
Summary of the decarbonisation strategies, interventions, and the impacted variables.

Market Level	Strategy	Strategy Type	Intervention	Parameter Affected
Clinker	Carbon Capture and Storage (CCUS)	LM	CCUS at clinker production plant	Clinker process emissions
	Recycling of Concrete Fines	LM	Use of concrete fines as clinker replacement	Average clinker carbon intensity
	Electrification	LM	Producing clinker using electricity	
	Alternative Fuels	HM	Biomass share in the fuel mix	Carbon intensity of fuel mix
	Thermal Efficiency Improvements	HM	More energy efficient clinker kilns	Energy intensity of clinker production
Cement	Decarbonisation of Electricity	HM	Lower carbon electricity grid	Carbon intensity of national electricity grid
	Alternative Binders	LM	Alkali-activated industrial waste	Average cement carbon intensity
	Clinker Replacement	HM	SCMs as clinker replacement	Clinker factor
Concrete	Reduction in Over-specification	HM	Performance-based concrete specifications	Binder Content
	Concrete Mix Design Optimisation	HM	Particle packing and use of chemical admixtures	
Building Applications	Improved Design of Structural Elements	HM	More efficient structural design	Concrete material intensity in buildings

$$\text{Average clinker carbon intensity}\left(\text{kgCO}_{2\text{eq}}/\text{kg}\right)=\text{Clinker process emissions}\left(\text{kgCO}_{2\text{eq}}/\text{kg}\right)$$
$$+\text{Energy intensity in clinker production}\ (\text{MJ/kg})^{*}\ \text{Carbon intensity of fuel mix}\left(\text{kgCO}_{2\text{eq}}/\text{MJ}\right) \quad (7)$$

The process emissions for clinker production represent the chemically bound CO_2 released during the calcination of calcium carbonate ($CaCO_3$) and magnesium carbonate ($MgCO_3$) in the feed meal (Shen et al., 2014). These values are subject to the chemical composition of the raw materials, but an average value was taken for both the UK and global market (Shen et al., 2015). For the same raw meal, the energy intensity values for clinker production primarily depends on the type of kiln used (John, 2020). Modern kilns are classified as wet, semi-dry, and dry. Dry kilns are the most energy efficient option at 3.40GJ/t and wet kilns are the least at 5.29GJ/t (Sahoo et al., 2022). Globally, 80 % of all cement kilns are dry (Tkachenko et al., 2023), however in the UK, the share of kiln types used in production is reported as 27 % semi-dry and 73 % dry (MPA, 2019). The carbon intensity of the fuel mix is dependent on the fuel type and amount used within a given fuel mix. This value was calculated by multiplying the percentage of each fuel used in a mix by its respective carbon intensity (Summerbell, 2018). A summary of the key parameters and the inventory data used for the developed method is shown in Table 2.

While the input parameters noted in Table 2 present values that accurately represent each geographical market assessed, these values are impacted by economic and regulatory factors. When best available technologies and techniques are implemented for a production process, optimised values are achieved. These 'best practice' values act as theoretical minimum targets that have been proven to be attainable for a given process (Schorcht et al., 2013). Therefore, the final scenario considers the currently available best practice to analyse the variability present between the current norm in a given market and what is realistically achievable in any geographical region. Given that there is no variance in carbon intensity for the UK and global markets, this scenario analysis also acts as an uncertainty analysis as part of the interpretation step of the LCA.

Table 3 summarises the best practice values considered. According to IEA (2009), the best thermal efficiency practice, and hence the energy intensity, can be achieved by implementing a dry manufacturing process with a preheater and pre-calciner alongside using raw materials with high burn ability. An optimum and realistic fuel mix design for clinker production is comprised of 60 % alternative fuels and biomass. The increased usage of both of these materials in a mix design decreases fossil fuel consumption and therefore lowers the carbon intensity value for the fuel mix (IEA, 2018). For the clinker replacement strategy, a market-average clinker factor of 0.50 was selected as the best practice (Scrivener et al., 2018). Given current material availability, the optimised mix design assumes that the non-clinker components in the mix would be comprised of gypsum (5 %), limestone filler (15 %), GGBS (15 %), and calcined clays (15 %) (SI, Table 7). As a result of the country's reliance on zero-carbon sources such as nuclear and hydropower, Switzerland was found to have the lowest national grid carbon intensity (IEA, 2024). Reducing the binder content is feasible through concrete mix design optimisation by using inert fillers, chemical admixtures, and improving dry mix particle packing (Zunino, 2023). For this study, the best practice binder content was determined by multiplying the maximum minimum binder content in each strength class currently specified in UK concrete standards (BSI, 2023) by the respective market share for each strength class (ERMCO, 2020) (SI, Table 4).

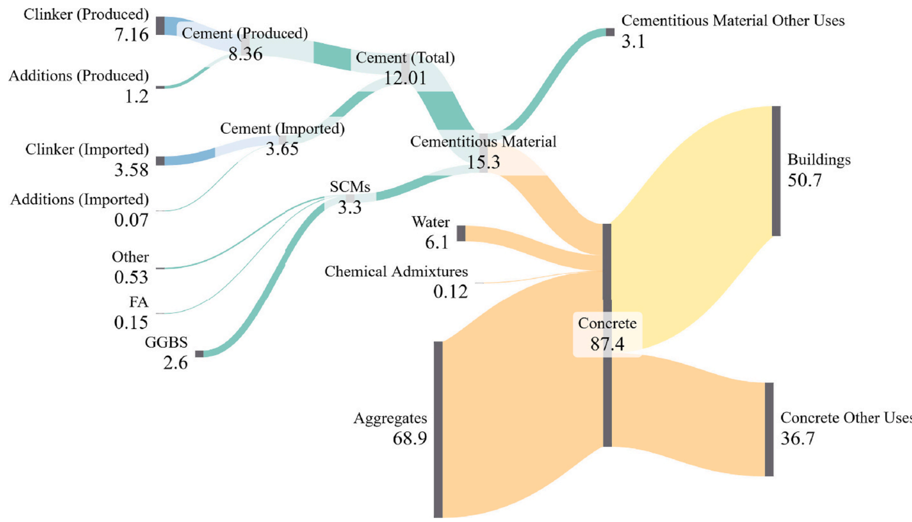

Fig. 3. Material flow diagram of UK cement and concrete consumption (Mt/yr), 2022.

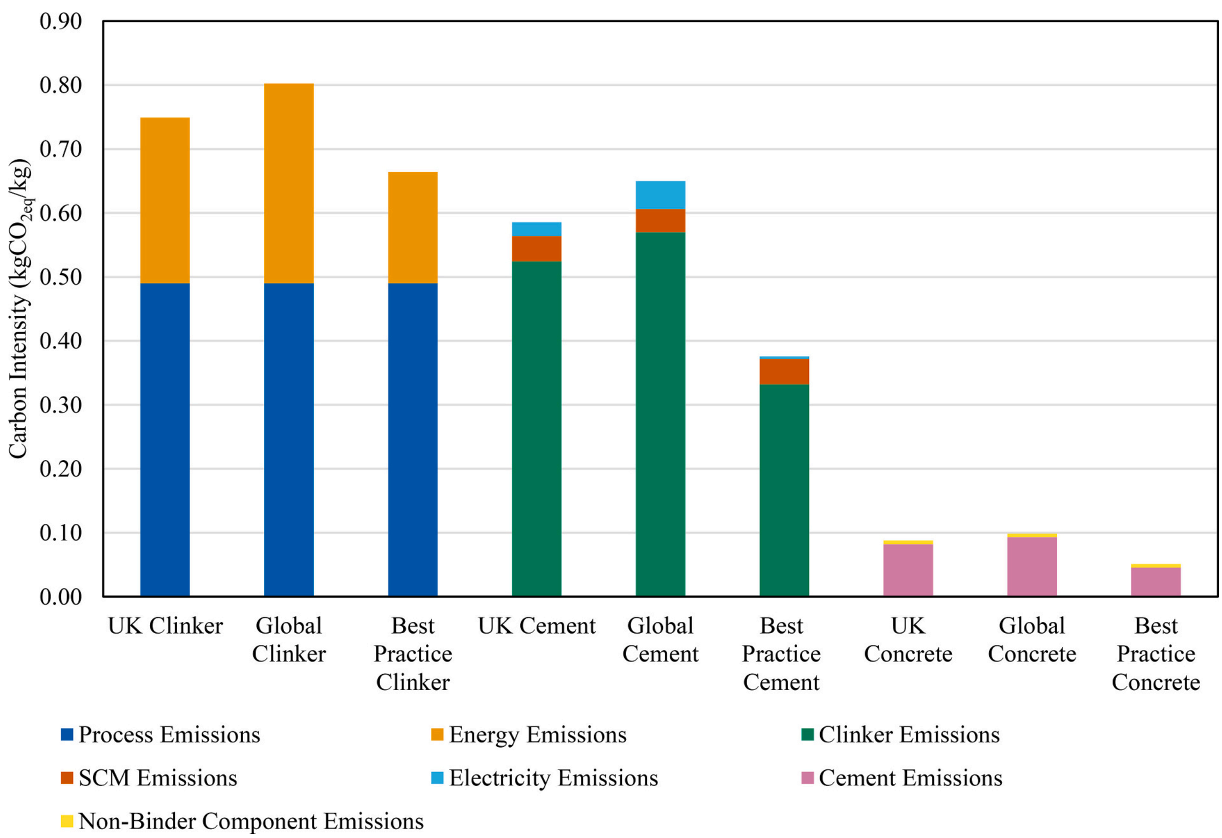

Fig. 4. Carbon Intensity at each market level (UK, global, best practice).

2.2. Decarbonisation scenarios by 2050

The second stage of the methodology critically analyses the decarbonisation potential of twelve decarbonisation strategies outlined in various UK and EU cement and concrete sector decarbonisation roadmaps. As previously mentioned, decarbonisation strategies can be classified into two main categories: LM and HM. In this study, four LM and seven HM strategies were analysed. For each decarbonisation strategy, there is a corresponding intervention defined as an innovation or adaptation that aims to reduce the carbon footprint of a given product (Bernstein and Hoffmann, 2015). As noted in Table 4, each intervention effects a parameter used to calculate the carbon intensity value at each market level (Eqs. 5–7).

To determine the decarbonisation potential in 2050 for any LM or HM intervention, Eq. 8 is proposed where the benchmarked carbon intensity and annual consumption values for a specific product (e.g.

clinker, cement, concrete) is multiplied by the strategy's decarbonisation reduction potential.

$$\text{Strategy decarbonisation potential in 2050 } (\text{MtCO}_{2eq}/\text{yr}) = \text{Product carbon intensity } \left(\text{kgCO}_{2eq}/\text{kg}\right)^{*} \text{ Strategy reduction potential beyond the current average } (\%)^{*} \text{ Product annual consumption } (\text{Mt/yr}) \tag{8}$$

The reduction potential beyond the current average values were taken from five cement and concrete decarbonisation roadmaps reviewed by Marsh et al. (2023). When examining the reduction potential for a given decarbonisation strategy across all roadmaps, a high level of discrepancy (>50 %) is evident, particularly with LM strategies. The high uncertainty level associated with these strategies is due to the immaturity of the current market and significant upfront costs associated with their implementation (Johnson et al., 2021). Conversely, the decarbonisation potential of HM strategies can be calculated with certainty due to their existing market precedence. This greater certainty however does not result in lower variability; often technological, economical, and current standardised practices effect the decarbonisation potential of these implemented strategies globally. For example, in the case of energy consumption, material inputs and machinery currently implemented can impact the ability to achieve low energy intensity values (Worrell et al., 2007). Given the variability and uncertainty of different strategy types, this study assumes two scenarios, each corresponding to a likelihood of LM and HM strategy implementation and the corresponding decarbonisation potential of each, namely: high and low.

2.2.1. High-probability decarbonisation scenario

Under the high probability scenario, it is assumed that the minimum reduction potential percentage reported for LM and HM decarbonisation strategies in the outlined UK and EU roadmaps will be achieved. The probability of this occurring for HM strategies is particularly high as the reduction potential predicted in the roadmaps is built upon pre-existing values from current strategy implementation. While LM strategies do not have existing market precedence, it is heavily emphasised across the roadmaps that market intervention is certain within the coming years. Significant financial and capital investment from government organisations (Department for Energy Security and Net Zero, 2024) and industrial stakeholders (Heidelberg Materials, 2024) particularly for CCUS illustrate the growing interest in the quick implementation of these strategies. For these reasons, the probability of achieving the minimum reduction potential for LM strategies is likely.

However, an emissions gap between the baseline BAU in 2050 and the applied minimum reduction potential from all LM and HM decarbonisation strategies may be present, especially as the demand for new buildings and infrastructure increases. Thus, to hit 2050 targets, some reduction in demand for new construction will likely be required through the implementation of circularity strategies that consider the reduction of material demand through the preservation (refurbishment and reuse) of existing building stock. Therefore, the reduction in concrete demand required is defined as the remaining carbon savings needed to achieve net-zero after all other decarbonisation strategies have been implemented. Under this scenario, it was assumed that concrete could not be substituted by any other building material. Eq. 9 expresses this reduction in demand required as a percentage. The total decarbonisation potential carbon savings in 2050 is equal to the carbon savings from all applied LM and HM decarbonisation strategies when the minimum reduction potential is considered and the BAU cement and

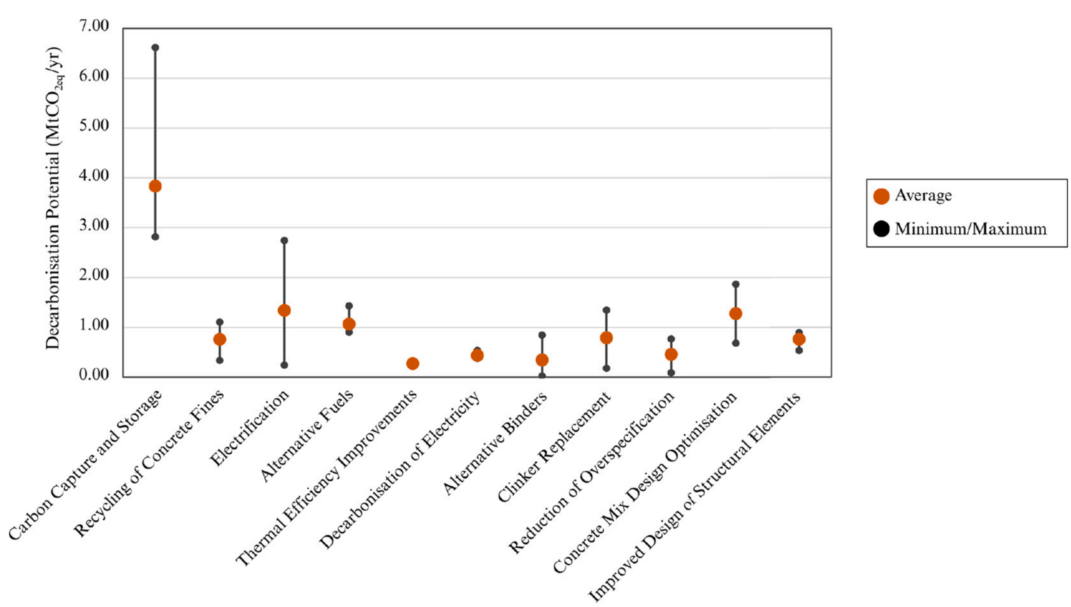

Fig. 5. Decarbonisation potential in 2050 by strategy utilising values reported in five UK and EU roadmaps

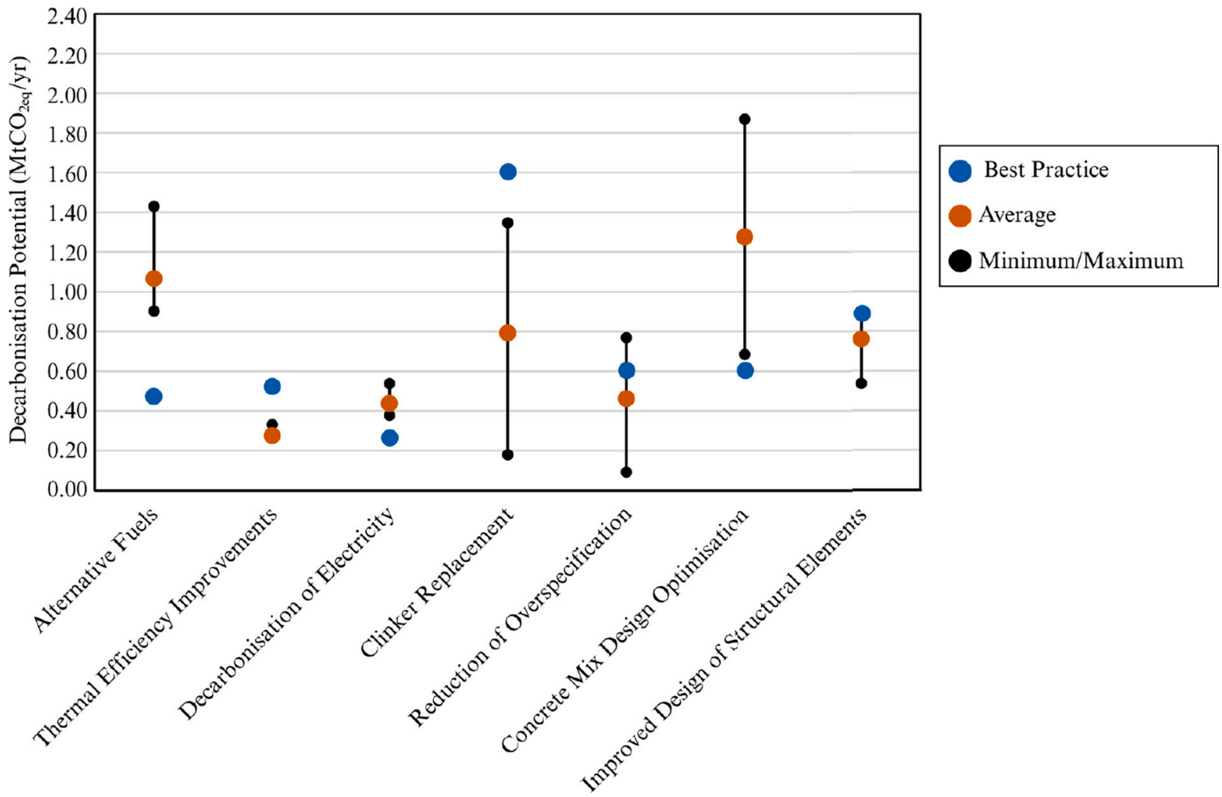

Fig. 6. Decarbonisation potential in 2050 of HM strategies (roadmap values vs. best practice values).

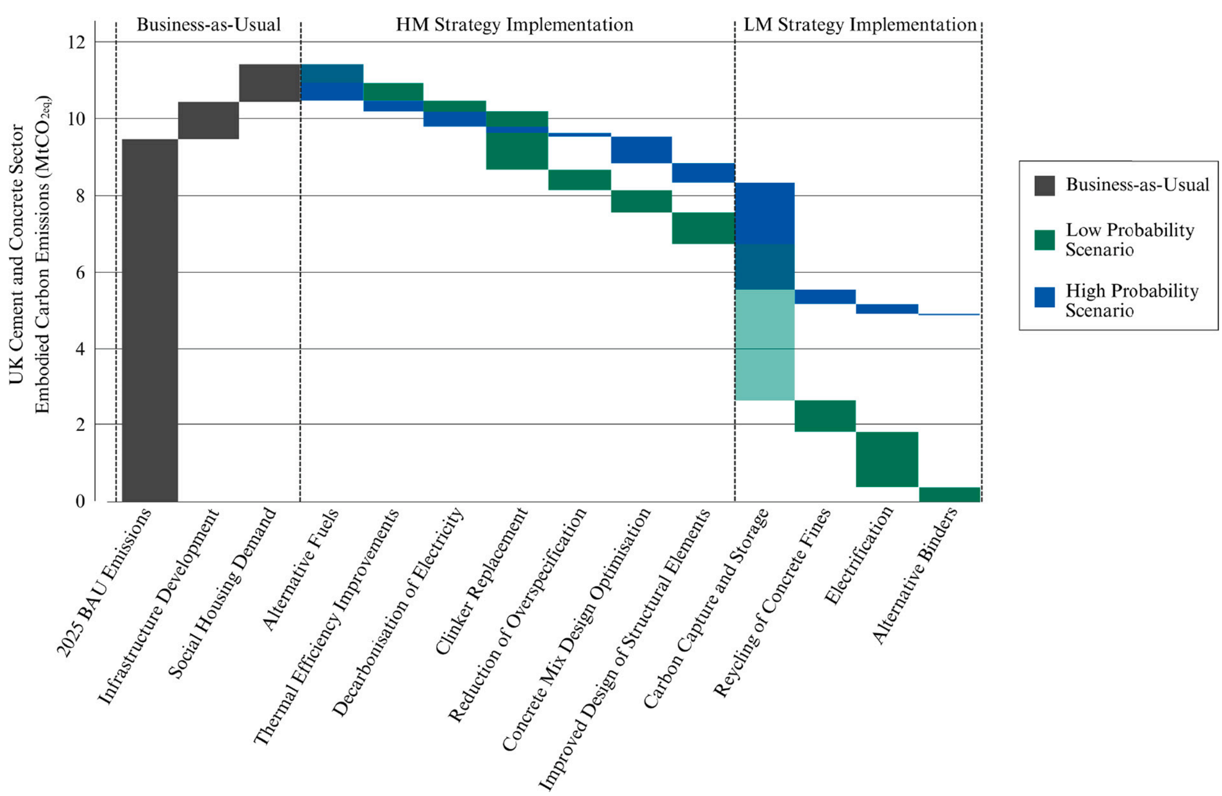

Fig. 7. The business-as-usual emissions from cement and concrete consumption in the UK by 2050 and the combined decarbonisation potential values from this study's analysis of low and high probability of strategy implementation.

concrete sector carbon emissions in 2050 was calculated using Eq. 3.

$$\text{Reduction in concrete demand (\%) in 2050} = \big(\text{BAU cement and concrete sector carbon emissions in 2050 } (MtCO_{2eq}/yr) - \text{total decarbonisation potential carbon savings in 2050 } (MtCO_{2eq}/yr)\big) / \text{BAU cement and concrete sector carbon emissions in 2050 } (MtCO_{2eq}/yr) \quad (9)$$

2.2.2. Low-probability decarbonisation scenario

Under the low probability scenario, it is assumed that the currently available global best practice for each HM decarbonisation strategy is fully implemented; therefore, the highest rate of decarbonisation from these strategies is achieved. The best-practice carbon intensity values noted in Table 3 are applicable to HM strategies where their intervention occurs at either the clinker, cement, or concrete market level. However, one strategy, improved design of structural elements, exists at the building application level and does not have a corresponding carbon intensity value. This decarbonisation strategy impacts the concrete material intensity in buildings, which is a parameter used to determine the percentage of concrete consumed in buildings per year as noted in Eq. 3. For this study, the best practice concrete material intensity in buildings is conservatively projected to equal to 843 kg/m^2 (Hafez et al., 2024a; Drewniok et al., 2023) (SI, Table 2).

Given the financial and regulatory limitations of fully implementing state-of-the-art optimised technologies, the likelihood of the complete implementation of HM best-practices are low. One example of this is in regards to the current carbon intensity of the national electric grid. A widespread, rapid transition to an energy mix that provides the same low-carbon energy as the best practice case in Switzerland would be challenging in most countries given the high costs to implement these technologies fully. Another example is in regards to a kiln's thermal efficiency. While the best-practice value and therefore lowest energy consumption reaches 3 GJ/t clinker, this value was only achieved by 10 % of all operating kilns evaluated (IEA, 2009). While all optimal, best-practice values are achievable, significant investments must be made enable implementation. Since the carbon intensity of one market product will effect another, the change in carbon intensity caused by the implementation of each decarbonisation strategy was compounded to create an optimised best practice across all sector market levels. Much like the high-probability scenario, an emissions gap between the baseline BAU in 2050 and the applied best practice cases for all HM decarbonisation strategies may be present. Assuming no demand reduction is required, Eq. 10 was used to determine reduction potential required by LM strategies under the low probability scenario to reach net-zero.

$$\text{Reduction required by LM strategies } (MtCO_{2eq}/yr) = \text{BAU cement and concrete sector carbon emissions in 2050 } (MtCO_{2eq}/yr) - \text{Total HM decarbonisation potential } (MtCO_{2eq}/yr) \quad (10)$$

3. Results and discussions

3.1. Market-level benchmarking: material flow analysis results

Using the data gathered for the MFA, a material flow diagram was created to benchmark current cement and concrete consumption volumes as seen in Fig. 3. At the clinker market level, it was found that 10.7 Mt/yr was consumed for cement production. At the cement market level, 15.3 Mt of cementitious materials are consumed annually with cement accounting for 78 % of all cementitious materials. While the UK currently has a sufficient supply of FA and limestone fines, the country relies on GGBS imports to meet current demand (UN Comtrade, 2023). Therefore, to determine the amount of GGBS consumed, it was assumed that since most, if not all, GGBS produced in the UK goes towards the cement sector (Alberici et al., 2017), its consumption value was estimated as the sum of GGBS produced in the UK and the total amount of GGBS imported into the UK. The amount of GGBS produced annually was calculated utilising a blast furnace slag to pig iron tonne ratio of 0.28:1.0 (Curry, 2018). As reported by Drewniok et al. (2023), approximately 20 % of all cementitious materials is used for other uses which includes mortars and soil stabilisers. The majority of cementitious materials produced were used in concrete products including ready-mixed, precast, and blocks. The total amount of concrete consumed in the UK for both building and other applications was taken as the sum of precast and ready-mixed concrete consumed as reported by Drewniok et al. (2023). Utilising Eq. 4, it was determined that 58 % of all ready-mix concrete consumed was for use in building applications. From this, the annual consumption of concrete for use in buildings was found to be 50.7 Mt, with the remaining concrete going towards other applications such as infrastructure, pavements, and roads.

3.2. Market-level benchmarking: life cycle assessment results

Using LCA methodology, the embodied carbon of each product in the cement and concrete sector was calculated for the UK and global markets, in addition to the current global best practice. As shown in Fig. 4, the general trend found that for all UK market levels, the carbon intensity values are slightly lower compared to the global averages. The global average clinker carbon intensity value is roughly 6.5 % higher compared to the UK value, and the global cement and concrete carbon intensity values were both found to be 10 % and 13 % higher compared to the UK values, respectively. These differences are attributed to the global carbon intensity of the fuel mix being 29 % higher compared to UK value at the clinker market level and the global electrical grid carbon intensity at the cement market level value being 120 % higher compared to the UK value. As expected, the best practice scenario provided the lowest carbon intensity values across all market levels. The carbon intensity of a best practice concrete mix is 42 % less than that of the average concrete mix consumed in the UK. This signifies the importance of HM decarbonisation strategy implementation, such as the higher use of SCMs to replace clinker, further optimisation of binder content, and switching to a decarbonised electricity grid. Table 8 in the SI summarises the UK cement and concrete sector's BAU embodied carbon, annual consumption, and carbon emission values.

Table 5
Concrete volumes required to be produced in 2050 to achieve the decarbonisation potential.

Strategy	Decarbonisation potential ($MtCO_{2eq}$/yr)	Carbon intensity of LM intervention ($kgCO_{2eq}$/kg)	Reference	Concrete volumes (Mt/yr)
Recycling of concrete fines	0.82	0.0593	Dunant and Allwood (2024)	28.40
Electrification	1.44	0.0597	Marsh et al. (2023)	50.98
Alternative binders	0.34	0.0447	Nikravan et al. (2023)	7.78

3.3. Decarbonisation potential in 2050

The minimum, maximum, and average reduction potential values for each decarbonisation strategy were selected from the UK and EU roadmaps reported by Marsh et al. (2023). Using these values, the decarbonisation potential for each strategy was calculated using Eq. 8. While Marsh et al. (2023) reported all reduction values on a concrete market level, this study has opted to report the estimated reductions for each strategy on their corresponding market level. As seen in Fig. 5, CCUS provides the greatest decarbonisation potential followed by electrification, with a maximum 62 % and 25 % reduction in BAU clinker market level carbon emissions, respectively. Both of these LM strategies have a much higher variance however compared to other HM and even other LM strategies.

While the probability of achieving the minimum decarbonisation potential of HM strategies values shown in Fig. 5 is high, the lower probability implementation of the best practice for each HM strategy was also investigated. As shown in Fig. 6, the decarbonisation potential values based on best practices exceeds the average from roadmaps significantly. The only three exceptions are the decarbonisation of electricity, use of alternative fuels, and concrete mix design optimisation. This discrepancy is likely due to EU roadmaps, and therefore EU countries, exhibiting higher biomass capabilities than what is currently achievable in the UK. For example, the EU roadmap projects a shift to 80 % biofuel use in clinker production (Favier et al., 2018), while a study by Stamford and Azapagic (2014) notes that the optimal consumption of biomass without CCUS in the electric grid is 25 % due to the risk associated with the wider implementation of biomass use in energy production in terms of abundance and carbon intensity compared to natural gas. Similarly, a recent study concluded that the UK shift to zero-carbon electricity grids would also imply an unrealistic level of use of biomass (Hafez et al., 2024b). For concrete mix design optimisation, the discrepancy is likely due to decarbonisation roadmaps using a higher binder content that exceeds current code limitations in addition to not accounting for differences in concrete strength classes.

To assess the total decarbonisation potential of all strategies, the BAU carbon emissions in 2050 was calculated by multiplying the benchmarked 2025 BAU cement and concrete sector carbon emissions (Eq. 2) by the expected demand in 2050. According to the Global Cement and Concrete Association, the global demand for cement and concrete is expected to increase by roughly 43 % from 2025 to 2050 (GCCA, 2025). Examining the UK market however suggests a lower demand compared to global value. Mcgarry et al. (2022), notes that the demand for concrete for construction of new infrastructure in the UK between now and 2050 will increase by 10 % compared to 2023 values. Another 10 % total increase (0.5 % per year) was also forecasted to account for the need for social housing (Drewniok et al., 2023). Accordingly, the demand for concrete is assumed to increase linearly by 20 % from 2025 to 2050. This assumption is in line with a market analysis report published by Climate Group ConcreteZero (2024) which assumed that the UK demand for concrete is expected to increase at half the rate of GDP growth (1.2 % per year). Fig. 7 illustrates the reduction in 2050 BAU emissions though the implementation of all decarbonisation strategies under both the high and low probability scenarios.

3.3.1. High probability scenario

For the high probability scenario, it is apparent that the 3.07 $MtCO_{2eq}$ and 3.43 $MtCO_{2eq}$ expected to be abated through the implementation of HM and LM strategies respectively does not allow the UK cement and concrete sector to achieve net-zero by 2050. A gap of at least 4.88 $MtCO_{2eq}$ is expected to remain unabated, which accounts for

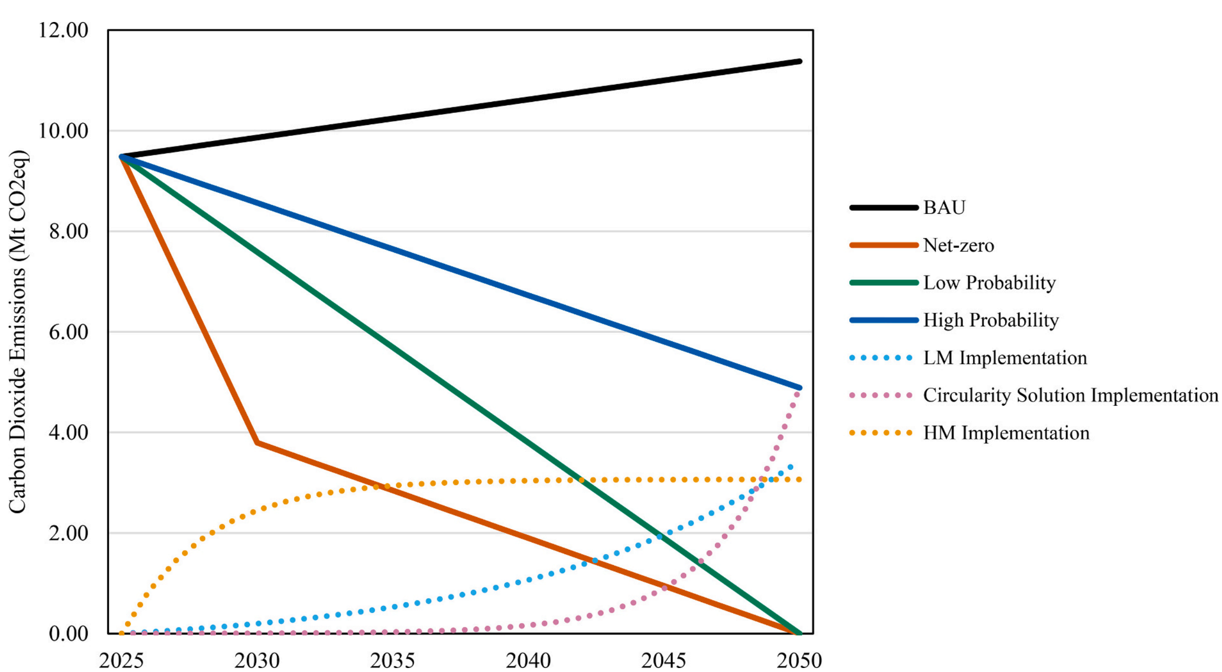

Fig. 8. Cumulative carbon budget to 2050 implementing combined HM and LM strategies.

roughly half of the sector's 2025 emissions. Since the underlying assumption of the high probability scenario is that concrete could not be substituted by timber or steel, the only abatement solution left is to reduce the demand of concrete through the implementation of circularity solutions for buildings. As per the method shown in Eq. 9, to achieve net-zero the building demand in 2050 must be reduced by 43 %. A study by Wu et al. (2023) highlighted that the Chinese cement and concrete sector must also extend the service life of current buildings to achieve net-zero by 2050. Despite it not necessarily falling in the scope of this study, there are several enablers to the success of reusing buildings in the UK. An exemplary case study is the refurbishment of the Triton Square London office building, which was originally constructed in 1998, but underwent a deep retrofit that was completed in 2021 (Robertson and Sturel, 2021). In addition to the established environmental impact savings of reusing buildings, there is evidence on the significant potential cost savings as well (Eberhardt et al., 2019). A recent framework was developed in the UK (BS 8001:2017) that advocates for a system-level approach to exploring circularity throughout the whole value chain of building construction (Pacheco et al., 2024).

3.3.2. Low probability scenario

Under the low probability scenario, the HM best practice strategies are expected to reduce the 2050 BAU carbon emissions by 41 %. Therefore, to achieve net-zero emissions by 2050, the remaining 59 % must be abated through the implementation of LM strategies. Using the average reduction potential reported across the collected UK and EU roadmaps, a ratio was taken to determine the decarbonisation potential of each strategy respective to the total amount of carbon emissions that must be abated through LM strategy implementation. From this, it was determined that CCUS has the highest reduction potential followed by electrification. Although there are enabling factors present that could aid in the full implementation of LM strategies (e.g. optimising operations, financial, and policy incentives), there is a risk that they would not be fully implemented sector wide in time for the 2050 carbon reduction targets due to technological, social, and economic barriers. This technical limitation is described as the differential performance of decarbonisation strategies at each market level or application compared to the benchmark. An example of this is the use of belite-based cements, for which the carbon footprint is lower compared to OPC but would cause a similar reduction in the 28-day compressive strength (Naqi and Jang, 2019). In this study, this performance-based limitation is ignored since all low-carbon solutions are assumed to have a comparable technical performance to the benchmark. LM strategies are more likely to face social barriers to market penetration such as resistance to major changes (e.g. updating standards) due to their disruptive nature to the market norm. For example, the current Eurocode for structural concrete design does not yet accommodate the use of alternative binders (only considers clinker-based cements) (Qian et al., 2022). Economic barriers also inhibit the market penetration potential of a decarbonisation innovation. An increase in the selling price of lower-carbon cement products may occur as a result of costly technology in the case of CCUS or due to the scarcity of resources such as the industrial waste availability for use in alkali activated cements. For example, the current estimated cost for carbon abatement using CCUS technologies ranges between \$50–100/kgCO_{2eq} (Kearns et al., 2021). A recent study predicted the implementation of CCUS for clinker production would cause cement prices to reach £150/t in the UK; a 15 % increase compared to the current market price (Dunant and Allwood, 2024; ONS, 2021). Ideally, the gap in price would be organically narrowed through the implementation of carbon taxes. However, the current £10/t of cement UK carbon tax levy is only limited to fossil fuel use and does not extend to include process emissions from clinker production (Grover et al., 2016). In addition, there is emerging evidence supporting that increasing carbon taxes might not necessarily result in higher carbon abatement on a sector level (Abrell et al., 2019; Gugler et al., 2023).

Even under a scenario where full implementation of LM strategies is achieved, the probability of achieving the production capacity to reach the required carbon abatement for each strategy is low. In the case of CCUS, the maximum expected capacity for cement production by 2050 is 1.9 Mt$CO_{2eq.}$/yr; 54 % less than the capacity required under the low probability scenario (Hafez et al., 2024b). For each of the remaining LM strategies, specific concrete production volumes must be met in 2050 to achieve their required decarbonisation potential values under the low probability scenario. To determine each strategy's required production volume, the strategy's decarbonisation potential was divided by the difference between the previously calculated UK concrete intensity value and the carbon intensity of the new technology. Table 5 highlights these input values and the calculation results for each LM strategy. From this, it can be concluded that between the three LM strategies, 87.2 Mt of concrete must be produced in 2050 to achieve the required decarbonisation potential; 0.2 % less than the 2025 consumption volume. To meet this required production volume under this unrealistic scenario, traditional concrete manufacturing processes must be fully replaced by the four outlined LM strategies. The result of this transition would render HM strategies obsolete, negating 4.7 MtCO_{2eq} of achievable decarbonisation potential.

3.4. Cumulative decarbonisation potential by 2050

As explained in Deutch (2020), the reduction in global temperature to mitigate global warming is proportional to the logarithmic change in atmospheric CO_2 concentration. Given that cumulative reductions of the net-zero pathway were based on a 2020 baseline that is assumed in this study to subsist till 2025, a logarithmic decay pathway is required to meet the net-zero targets by 2050. Fig. 8 presents the cumulative decarbonisation potential by 2050 for the low probability pathway, the high probability pathway, and the UK's net-zero pathway outlined by the CCC. Table 9 in the SI further details the calculated carbon savings and emissions for all scenarios considered. Between both the high and low probability scenarios and the net-zero pathway, there is a cumulative gap that grows from 2025 to 2050. This gap results in both probability scenarios creating cumulative carbon emissions that will result in a higher atmospheric increase of carbon, further reducing the effectiveness of mitigation scenarios. Under the high probability scenario specifically, the incremental increase in carbon emissions present between this pathway and the net-zero pathway highlights the need for the immediate implementation of LM strategies, optimised HM strategies, and an overall reduction in concrete demand through the reuse of existing building stock. The rate at which each decarbonisation strategy can be implemented however is dependent on the strategy type considered. Even though HM strategies have existing market precedence and therefore lower initial capital investment, the transition to the best practice case is limited by the willingness to shift from current practices. Alternatively, LM strategies require much higher capital investment which may result in a slower uptake in industry. The implementation of circularity strategies is also subject to several barriers including social, economic, and legislative. In the UK specifically, barriers to material reuse that were identified include cost, availability/storage, a lack of client demand, and poor integration (Densley Tingley et al., 2017). To overcome these barriers, study by Giorgi et al. (2022) highlighted the need for direct public incentives and policy frameworks that focus on breaking down these barriers and to promote circular economy thinking.

4. Conclusions

This study critically analysed the potential of the UK cement and concrete sector to achieve net-zero carbon targets in 2050. Decarbonisation strategies laid out in several UK and EU roadmaps were then examined with the objective to model their combined decarbonisation potential under two probability scenarios. These scenarios were then compared to the UK CCC's 6th carbon budget net zero pathway. The conclusions of the study are as follows:

1. The average decarbonisation potential for HM interventions such as clinker replacement, binder content reduction, and structural optimisation in published roadmaps underestimate the associated carbon abatement by 60–100 % compared to best practices. In the UK, the carbon intensity of cement and concrete consumption is 42–55 % higher than the best practice. While best practice implementation in the UK may be impeded by industry's willingness to shift from current practices, immediate implementation is recommended to tap into a potential reduction of 4.7 million tonnes of CO_2 per year.
2. Implementation of best practice HM strategies would result in a 41 % reduction in carbon emissions, requiring the remaining 59 % to be abated through the implementation of LM strategies or reduction in material demand. The financial and resource constraints on the implementation of LM strategies such as CCUS, electrification, and alternative binder is detrimental to their decarbonisation potential by 2050. Given their combined carbon abatement potential of 3.4 million tonnes of CO_2 per year however it is imperative that these strategies are implemented as soon as possible.
3. The only way to meet the net-zero target by 2050, with confidence, is the reduction of concrete demand by 43 % through the implementation of circular economy principles. Hence we need to overcome economic, legislative, and social barriers by shifting economic models, creating new policy frameworks, and presenting direct public incentives to implement this strategy quickly and effectively.

While this study aimed to challenge the carbon abatement potential reported in the literature in order to balance the expectations placed on the two main types of decarbonisation strategies, one gap in the methodology followed is that these 'certain' carbon abatement values do not account for the uncertainties associated with the carbon emissions of the studied variables. Hence, the main objective of a follow-on study would be to collect evidence, based on expert-opinion and market research, to validate the market penetration and decarbonisation potential expected of each LM and HM intervention in the UK cement and concrete sector enroute till 2050. While low-carbon alternatives to concrete such as natural fibres and earth-based materials are seen as a sustainable solution in many geographic regions, it was not considered a viable option for the UK due to the volumes required and uncertainty surrounding its environmental impact due to transport. Since this was a region dependent assumption however, it is recommended that future studies examine the impact of the reuse and refurbishment of other building materials in line with local material availability.

CRediT authorship contribution statement

Madeline Rihner: Writing – review & editing, Writing – original draft, Visualization, Methodology, Formal analysis, Data curation, Conceptualization. **Hisham Hafez:** Writing – review & editing, Writing – original draft, Supervision, Project administration, Methodology, Formal analysis, Data curation, Conceptualization. **Brant Walkley:** Writing – review & editing, Supervision, Funding acquisition. **Phil Purnell:** Writing – review & editing, Supervision, Funding acquisition. **Michal Drewniok:** Writing – review & editing, Supervision.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgments

This work was supported by the EPSRC grant 'TransFIRe' Ref. EP/V054627/1, as well as an EPSRC Doctoral Training Partnership PhD Scholarship to Madeline Rihner from The Energy Institute at The University of Sheffield (EP/T517835/1).

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.spc.2025.06.010.

References

Abrell, J., Kosch, M., Rausch, S., 2019. How Effective Was the UK Carbon Tax? — A Machine Learning Approach to Policy Evaluation. CER-ETH - Center of Economic Research (CER-ETH) at ETH Zurich.

Alberici, S., De Beer, J., Van Der Hoorn, I.S., Maarten, 2017. In: Department for Business, E. I. S (Ed.), Fly Ash and Blast FURANCE Slag for Cement Manufacturing. BEIS.

Bernstein, S., Hoffmann, M., 2015. The Politics of Decarbonization: A Framework and Method.

BSI, 2023. BS EN 8500-1:3023 Concrete- Complementary British Standard to BS EN 206.

Busch, P., Kendall, A., Murphy, C.W., Miller, S.A., 2022. Literature review on policies to mitigate GHG emissions for cement and concrete. Resour. Conserv. Recycl. 182, 106278.

Cabeza, L.F., Boquera, L., Chàfer, M., Vérez, D., 2021. Embodied energy and embodied carbon of structural building materials: worldwide progress and barriers through literature map analysis. Energ. Buildings 231, 110612.

CLIMATE GROUP CONCRETEZERO, 2024. Decarbonising the UK Concrete Industry: Closing the Emissions Gap. The Climate Change Organisation.

Slag—iron and steel. In: Curry, K.C., INTERIOR, U. S. D. O. T (Eds.), 2018. 2018 Minerals Yearbook. U.S. Geological Survey.

D'amico, B., Pomponi, F., Hart, J., 2021. Global potential for material substitution in building construction: the case of cross laminated timber. J. Clean. Prod. 279, 123487.

Damineli, B.L., Kemeid, F.M., Aguiar, P.S., John, V.M., 2010. Measuring the eco-efficiency of cement use. Cem. Concr. Compos. 32, 555–562.

Densley Tingley, D., Cooper, S., CulleN, J., 2017. Understanding and overcoming the barriers to structural steel reuse, a UK perspective. J. Clean. Prod. 148, 642–652.

Department for Energy Security and Net Zero, 2024. Contracts Signed for UK's First Carbon Capture Projects in Teesside gov.uk.

Deutch, 2020. Is Net Zero Carbon 2050 Possible? Joule 4 (11), 2237–2240. https://doi.org/10.1016/j.joule.2020.09.002.

Drewniok, M.P., Azevedo, J.M.C., Dunant, C.F., Allwood, J.M., Cullen, J.M., Ibell, T., Hawkins, W., 2023. Mapping material use and embodied carbon in UK construction. Resour. Conserv. Recycl. 197, 107056.

Dunant, C.F., Allwood, J.M., 2024. What investments in material production are needed to achieve net-zero construction in the UK by 2050? J. Clean. Prod. 464, 142709.

Eberhardt, L., Birgisdottir, H., Birkved, M., 2019. Potential of circular economy in sustainable buildings. In: IOP Conference Series: Materials Science and Engineering, 471, 092051.

EMBER, 2024. Energy Institute - Statistical Review of World Energy [Online]. Available: https://ourworldindata.org/grapher/carbon-intensity-electricity [Accessed April 15th 2024].

Emmerling, J., Drouet, L., Wijst, K.-I.V.D., Vuuren, D.V., Bosetti, V., Tavoni, M., 2019. The role of the discount rate for emission pathways and negative emissions. Environ. Res. Lett. 14, 104008.

ERMCO, 2020. European Ready-Mixed Concrete Industry Statistics 2019. European Ready Mixed Concrete Organization.

ESO, 2023. Carbon Intensity Dashboard [Online]. National Grid. Available: https://www.nationalgrideso.com/future-energy/our-progress-towards-net-zero/carbon-intensity-dashboard [Accessed 2024].

Favier, A., De Wolf, C., Scrivener, K.L., Habert, G., 2018. A Sustainable Future for the European Cement and Concrete Industry: Technology Assessment for Full Decarbonisation of the Industry by 2050. European Climate Foundation.

GCCA, 2025. Societal Demand for Cement and Concrete [Online]. Available: https://gccassociation.org/concretefuture/societal-demand-for-cement-and-concrete/ [Accessed].

Giorgi, S., Lavagna, M., Wang, K., Osmani, M., Liu, G., Campioli, A., 2022. Drivers and barriers towards circular economy in the building sector: stakeholder interviews and analysis of five European countries policies and practices. J. Clean. Prod. 336, 130395.

Griffin, P.W., Hammond, G.P., Norman, J.B., 2014. Prospects for emissions reduction in the UK cement sector. Proceedings of the Institution of Civil Engineers - Energy 167, 152–161.

Grover, D., Shreedhar, G., Zenghelis, D., 2016. The Competitiveness Impact of a UK Carbon Price: What Do the Data Say? : ESRC Centre for Climate Change Economics and Policy and Grantham Research Institute on Climate Change and the Environment.

Gugler, K., Haxhimusa, A., Liebensteiner, M., 2023. Carbon pricing and emissions: causal effects of Britain's carbon tax. Energy Econ. 121, 106655.

Hafez, H., Kurda, R., Cheung, W.M., Nagaratnam, B., 2019. A systematic review of the discrepancies in life cycle assessments of green concrete. Appl. Sci. 9, 4803.

Hafez, H., Bajić, P., Aidarov, S., Malija, X., Drewniok, M., Purnell, P., Tošić, N., 2024a. Parametric study on the decarbonization potential of structural system and concrete mix design choices for mid-rise concrete buildings. Mater. Struct. 57, 85.

Hafez, H., Drewniok, M.P., Velenturf, A.P.M., Purnell, P., 2024b. A resource-bound critical analysis of the decarbonisation roadmaps for the UK foundation industries by 2050. Environments 11, 153.

Hammond, G.P., 2022. The UK industrial decarbonisation strategy revisited. Proceedings of the Institution of Civil Engineers - Energy 175, 30–44.

Heidelberg Materials, 2024. New Carbon Capture Trial at Ketton Cement Works.
IEA, 2009. Carbon Emissions Reductions up to 2050. International Energy Agency.
IEA, 2018. Technology Roadmap Low-Carbon Transition in the Cement Industry. Paris.
IEA, 2021a. ETP Clean Energy Technology Guide [Online]. Available: https://www.iea.org/articles/etp-clean-energy-technology-guide [Accessed].
IEA, 2021b. Thermal Specific Energy Consumption Per Tonne of Clinker in Selected Countries and Regions, 2018 [Online]. International Energy Agency. Available: https://www.iea.org/data-and-statistics/charts/thermal-specific-energy-consumption-per-tonne-of-clinker-in-selected-countries-and-regions-2018 [Accessed April 10th 2024].
IEA, 2022. Tracking Cement [Online]. Available: https://www.iea.org/energy-system/industry/cement [Accessed April 15th 2024].
IEA, 2023. Global Thermal Energy Intensity of Clinker Production by Fuel in the Net Zero Scenario, 2010–2030 [Online]. Available: https://www.iea.org/data-and-statistics/charts/global-thermal-energy-intensity-of-clinker-production-by-fuel-in-the-net-zero-scenario-2010-2030.
IEA, 2024. Switzerland [Online]. Available: https://www.iea.org/countries/switzerland [Accessed].
ISO, 2006a. International Organization for Standardization (ISO) 14040, Environmental Management - Life Cycle Assessment - Principles and Framework. British Standards Online.
ISO, 2006b. International Organization for Standardization (ISO) 14044, Environmental Management - Life Cycle Assessment - Requirements and Guidelines. British Standards Online.
John, J.P., 2020. Parametric studies of cement production processes. Journal of Energy 2020, 4289043.
Johnson, O.W., Mete, G., Sanchez, F., Shawoo, Z., Talebian, S., 2021. Toward climate-neutral heavy industry: an analysis of industry transition roadmaps. Appl. Sci. 11, 5375.
Kearns, D., Liu, H., Consoli, C., 2021. Technology Readiness and Costs of CCS. Global CCS Institute.
Lehne, J., Preston, F., 2018. Making Concrete Change: Innovation in Low-carbon Cement and Concrete. Chatham House. The Royal Institute of International Affairs.
Marsh, A., Dillon, T., Bernal, S., 2023. Cement and concrete decarbonisation roadmaps – a meta-analysis within the context of the United Kingdom. RILEM Technical Letters 8, 94–105.
Mcgarry, H., Martin, B., Winslow, P., 2022. Delivering low carbon concrete for network rail on the Routemap to net zero. Case Studies in Construction Materials 17, e01343.
Mcgrath, T., Nanukuttan, S., Basheer, P.A.M., Long, A., Owens, K., Doherty, W., 2012. Embodied energy and carbon footprinting of concrete production and use. In: Proceedings of the 3rd International Conference on the Durability of Concrete Structures, ICDCS 2012.
Meglin, R., Kytzia, S., Habert, G., 2022. Regional circular economy of building materials: environmental and economic assessment combining Material Flow Analysis, Input-Output Analyses, and Life Cycle Assessment. J. Ind. Ecol. 26, 562–576.
Monteiro, P.J.M., Miller, S.A., Horvath, A., 2017. Towards sustainable concrete. Nat. Mater. 16, 698–699.
MPA, 2019. Options for switching UK cement production sites to near zero CO2 emission fuel: Technical and financial feasibility. In: ASSOCIATION, M.P. (Ed.), Feasibility Study for the Department for Business Energy and Industrial Strategy. Mineral Products Association, Cindar Ltd, VDZ.
MPA, 2020a. Profile of the UK Mineral Products Industry, 2020 edition.
MPA, 2020b. UK Concrete and Cement Industry Roadmap to Beyond Net Zero.
MPA, 2023. In: MINERAL PRODUCTS ASSOCIATION (Ed.), MPA Cement Stats Data Collation Template BH Collation F v3.
MPA, 2025. In: MINERAL PRODUCTS ASSOCIATION (Ed.), Fact Sheet 18 Embodied CO2e of UK Cements.
Naqi, A., Jang, J.G., 2019. Recent progress in green cement technology utilizing low-carbon emission fuels and raw materials: a review. Sustainability 11, 537.
Nikravan, M., Firdous, R., Stephan, D., 2023. Life cycle assessment of alkali-activated materials: a systematic literature review. Low-carbon. Mater. Green Constr. 1 (1). https://doi.org/10.1007/s44242-023-00014-6.
ONS, 2021. Inflation and Price Indices: 2351000000 Cement [Online]. Office of National Statistics. Available: https://www.ons.gov.uk/economy/inflationandpriceindices/timeseries/juv6/mm22 [Accessed August 9th 2024].
Pacheco, D.A.D.J., Rampasso, I.S., Michels, G.S., Ali, S.M., Hunt, J.D., 2024. From linear to circular economy: the role of BS 8001:2017 for green transition in small business in developing economies. J. Clean. Prod. 439, 140787.
Pamenter, S., Myers, R.J., 2021. Decarbonizing the cementitious materials cycle: a whole-systems review of measures to decarbonize the cement supply chain in the UK and European contexts. J. Ind. Ecol. 25, 359–376.
Qian, Z., Lantsoght, E.O., Lukovic, M., 2022. A critical review on structural behavior of alkali-activated concrete beams. In: 14th Fib International PhD Symposium in Civil Engineering. Rome.
Rihner, M.C.S., Whittle, J.W., Gadelhaq, M.H.A., Mohamad, S.N., Yuan, R., Rothman, R., Fletcher, D.I., Walkley, B., Koh, L.S.C., 2025. Life cycle assessment in energy-intensive industries: cement, steel, glass, plastic. Renew. Sust. Energ. Rev. 211, 115245.
Robertson, A., Sturel, E., 2021. 1 Triton Square, London – low-carbon development through reuse of an existing building. The Structural Engineer 99, 30–35.
Sahoo, N., Kumar, A., Samsher, 2022. Review on energy conservation and emission reduction approaches for cement industry. Environmental Development 44, 100767.
Schorcht, F., Kourti, I., Scalet, B.M., Roudier, S., Sancho, L.D., 2013. Best Available Techniques (BAT) Reference Document for the Production of Cement, Lime and Magnesium Oxide: Industrial Emissions Directive 2010/75/EU (Integrated Pollution Prevention and Control).
Scrivener, K., Martirena, F.B., Shashank, Maity, S., 2018. Calcined clay limestone cements (LC3). Cem. Concr. Res. 114, 49–56.
Shen, L., Gao, T., Zhao, J., Wang, L., Wang, L., Liu, L., Chen, F., Xue, J., 2014. Factory-level measurements on CO2 emission factors of cement production in China. Renew. Sust. Energ. Rev. 34, 337–349.
Shen, W., Cao, L., Li, Q., Zhang, W., Wang, G., LI, C., 2015. Quantifying CO2 emissions from China's cement industry. Renew. Sust. Energ. Rev. 50, 1004–1012.
Stamford, L., Azapagic, A., 2014. Life cycle sustainability assessment of UK electricity scenarios to 2070. Energy Sustain. Dev. 23, 194–211.
Summerbell, D.L., 2018. Environmental Performance Improvement in the Cement Industry. University of Cambridge.
Tkachenko, N., Tang, K., Mccarten, M., Reece, S., Kampmann, D., Hickey, C., Bayaraa, M., Foster, P., Layman, C., Rossi, C., Scott, K., Yoken, D., Christiaen, C., Caldecott, B., 2023. Global database of cement production assets and upstream suppliers. Scientific Data 10, 696.
UK House of Parliament, 2019. In: DEPARTMENT FOR BUSINESS ENERGY AND INDUSTRIAL STRATEGY (Ed.), The Climate Change Act 2008 (2050 Target Amendment) Order 2019.
UN Comtrade 2023. United Nations.
Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B., 2016. The ecoinvent database version 3 (part I): overview and methodology. Int. J. Life Cycle Assess. 21, 1218–1230.
Worrell, E., Price, L., Neelis, M., Galitsky, C., Zhou, N., 2007. World Best Practice Energy Intensity Values for Selected Industrial Sectors. Lawrence Berkeley National Laboratory.
WSP, DNV-G, 2015. In: DEPARTMENT OF ENERGY AND CLIMATE CHANGE AND THE DEPARTMENT FOR BUSINESS, I. A (Ed.), Industrial Decarbonisation & Energy Efficiency Roadmaps to 2050: Cement.
Wu, T., Ng, S.T., Chen, J., Cao, Z., 2023. More intensive use and lifetime extension can enable net-zero emissions in China's cement cycle. Resour. Conserv. Recycl. 198, 107144.
Yang, M., Chen, L., Wang, J., Msigwa, G., Osman, A.I., Fawzy, S., Rooney, D.W., Yap, P.-S., 2023. Circular economy strategies for combating climate change and other environmental issues. Environ. Chem. Lett. 21, 55–80.
Zunino, F., 2023. A two-fold strategy towards low-carbon concrete. RILEM Technical Letters 8, 45–58.