Assessment of decarbonisation strategies for the UK cement and concrete sector within the context of energy intensive industries

By: Madeline Rihner

Registration Number: 210204142

Supervisors:

Dr. Brant Walkley

Dr. Hisham Hafez

Professor Lenny Koh

A thesis submitted in partial fulfilment of the requirements for the degree of Doctor of Philosophy

School of Chemical, Materials, and Biological Engineering

University of Sheffield

October 31st, 2025

ABSTRACT

Cement and concrete are fundamental to the UK's built environment, yet their production is intrinsically carbon intensive, accounting for 1.5% of the country's CO_2 emissions (MPA, 2020). To comply with global and national net-zero targets, the UK cement and concrete sector is implementing and exploring a wide range of different decarbonisation strategies. While government and industry decarbonisation roadmaps provide an optimistic outlook into the future of sustainable cement and concrete, there is significant uncertainty regarding the feasibility of achieving net-zero targets. Life cycle assessment (LCA), the most commonly used methodology to assess a process or product's environmental impact, is fragmented despite standardisation. Decarbonisation roadmaps heavily rely on strategies that have little existing market precedence. Cross-sectoral interdependencies, such as those between steel and cement, have also introduced additional uncertainty as materials critical to cement decarbonisation may no longer be available.

The aim of this study is to assess how the UK cement and concrete sector can realistically achieve Net Zero 2050 by reviewing current standardised LCA practices in academic literature, identifying viable decarbonisation pathways, and analysing cross-sector impacts. Findings indicate that net-zero by 2050 is achievable but not guaranteed. Greater methodological and data transparency in LCA studies is essential to ensure its effectiveness as a decision-making tool. Low-maturity decarbonisation strategies could abate up to 3.4 Mt CO_2/yr but face significant financial and resource barriers, while optimising existing strategies could abate 4.7 Mt CO_2/yr. Together, these approaches would still require a 43% reduction in material demand for the sector to reach net-zero. Cross-sector collaboration is also critical. The UK steel sector's transition to greener production routes could lower its carbon footprint by 84% but at the cost of slag cement shortage, increasing the sector's reliance on imports. To avoid cross-sector decoupling, the UK cement sector must match the steel sector's rate of decarbonisation.

ACKNOWLEDGEMENTS

After four years, it is crazy to think my PhD journey is coming to an end. During this time, I have learned a great deal not only about cement, but also about myself. Of course, none of this would have been possible without the incredible people who I met and who have supported me. First and foremost, I would like to thank John Provis for helping me secure this PhD opportunity. Without him, I would not have discovered this PhD topic which was so perfectly aligned with my research interests. I would also like to thank Energy Institute for awarding me the EPSRC Doctoral Training Partnership PhD Scholarship and the Grantham Centre for Sustainable Futures, who provided additional financial support during my PhD.

Of course, a PhD experience is defined by not just the research topic but also the incredible supervisors who help guide you. I want to express my sincere gratitude to my PhD supervisors who have been there since the beginning: Brant Walkley and Lenny Koh. Brant's unwavering support and encouragement, both for my research and for my well-being, during PhD allowed me to grow as both a researcher and a person. Lenny's support, advice, and connections helped me build a strong foundation for my research, for which I am truly thankful for. Of course my PhD would not have come together the way it has without the guidance and support of Hisham Hafez who joined on as my secondary supervisor in 2023 after a chance encounter at a conference in Thailand. Hisham's guidance has been invaluable and has allowed me to greatly expand my knowledge in this field. I am deeply thankful for his time and insights he has offered over the course of the project.

To the waffle lovers, Jacob and Mahmoud, it has been an absolute pleasure collaborating with you both! I am truly thankful for their support and insightful contributions. A special thank you to my colleagues in SMASH for being such a wonderful, encouraging, and fun research group. I specifically want to thank Micael for being so kind and supportive at conferences, during tutorial sessions, and throughout my PhD. I also would like to thank Shubham for his care and encouragement during my PhD; I will miss performing the concrete song with you at outreach events!

I am so thankful for my friends outside of the university who have provided me with so much support in their own ways, specifically Cleo, Louise, Sarah, and Lexie showing that friendship truly is magic. Lastly, I want to thank my Mom and Papa for their love, support, and encouragement throughout this journey. Un immense merci à Mamie et Papi pour avoir toujours pris soin de moi, pour leur amour, leur patience et leur soutien pendant toutes ces années de doctorat *(Translation: A huge thank you to Mamie and Papi for always taking care of me, for their love, patience and support throughout all these years of doctoral studies).*

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

DECLARATION

I, the author, confirm that the Thesis is my own work except where work that has formed part of jointly authored publications has been included. The contribution of the candidate and the other authors to this work has been explicitly indicated below. The candidate confirms that appropriate credit has been given within the thesis where reference has been made to the work of others. I am aware of the University's Guidance on the Use of Unfair Means (https://www.sheffield.ac.uk/study-skills/assessment/academic-integrity/academic-integrity). This work has not been previously been presented for an award at this, or any other, university.

Chapters 4, 5, and 6 are comprised entirely of three published works developed over the course of my PhD. The publications arising from the thesis are:

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, *Renewable and Sustainable Energy Reviews*, Volume 211, https://doi.org/10.1016/j.rser.2024.115245.

Rihner, M.C.S., Hafez, H., Walkley, B., Purnell, P., & Drewniok, M. (2025) Thousand cuts: a realistic route to decarbonise the UK cement and concrete sector by 2050, *Sustainable Production and Consumption*, Volume 58, 2025, Pages 319-333, https://doi.org/10.1016/j.spc.2025.06.010.

Whittle, J.W., Rihner, M.C.S., Hafez, H., Espinosa., R.E., Fletcher, D.I., Walkley, B., Koh, L.S.C. (2026) From symbiosis to scarcity: Evaluating disruption associated with decarbonisation to circular waste materials between the UK cement and steel sectors, Resources, Conservation and Recycling, Volume 224, 2026, https://doi.org/10.1016/j.resconrec.2025.108560

A detailed account of my contributions to each publication is provided in the preface preceding each Results and Discussion chapters.

1. INTRODUCTION

The growth of urban development can be attributed to the invention of concrete. It is a material that has been utilised throughout human history, from the ancient Roman Pantheon that has stood the test of time to the modern day structural marvel of the Burj Khalifa. The material's high compressive strength and versatility has allowed it to be applied to many different forms of construction, especially when used in combination with reinforcing steel. The material is also highly durable, being resistant to weathering and fire. All of these properties have made concrete the most used construction material in the world, with approximately 30 billion tons of concrete being consumed globally each year (Monteiro et al., 2017). This consumption is only expected to increase in the future, as a rising global population leads to the subsequent demand for new buildings and infrastructure. While concrete construction is an attractive choice to meet this demand, sustainability concerns have emerged due to the substantial carbon footprint associated with the material's production. Figure 1 shows the main material and process flows associated with concrete production.

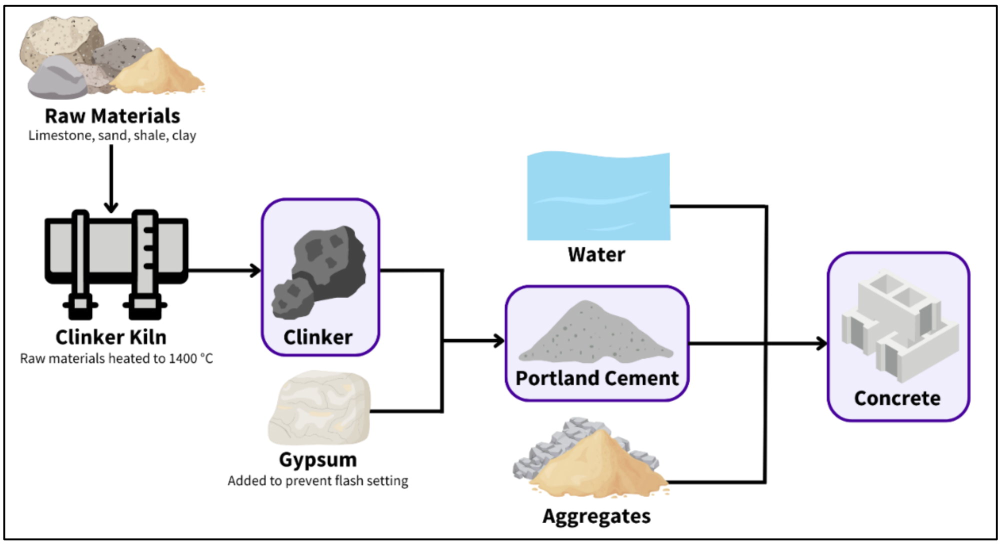


Figure 1: Simplified schematic of concrete production

The largest contributor to concrete's carbon emissions originates from one of its most essential constituents: cement. Cement, when combined with water, forms a glue that binds the fine and course aggregates together to make concrete. Its manufacturing process accounts for as much as 8% of all global anthropogenic carbon dioxide (CO_2) emissions, with 90% of these emissions originating from the production of clinker, the main constituent in traditional cement produced by the heating finely ground raw materials (primarily limestone, along with sand, shale and clay) in a cement kiln at approximately 1500°C (Lehne and Preston, 2018). The carbon impact from cement production is made all the more significant due to the amount consumed annually. In 2022, roughly 4.2 Gt of cement was produced globally, resulting in the release of 1.6 billion metric tons of CO_2 (IEA, 2022). To combat climate change, in 2015 the Paris Agreement was signed by 196 nations in an effort to limit global warming to below 2°C above pre-industrial levels (UNFCCC, 2016). In response to this initiative, the United Kingdom (UK) amended its own Climate Change Act of 2008 in 2019 which required the country to reach net-zero emissions by 2050 (UK House of Parliament, 2019). As one of the UK's most energy-intensive sectors, the cement and concrete sector has faced significant pressure to decarbonise in order to meet these targets.

To effectively target emission hotspots within the cement and concrete sector, its carbon emissions must first be quantified accurately and precisely. The most common method used to determine a product or sector's carbon footprint is life cycle assessment (LCA). LCA is a systematic analysis that evaluates the environmental impacts of a given product throughout its life cycle from the extraction of raw materials to its final disposal (ISO 14040, 2006). Although LCA is a methodology standardised by the International Organisation for Standardisation (ISO 14040 and ISO 14044) and the European Norm (EN 15804), its implementation remains fragmented which has resulted in unreliable and incompatible findings. This is particularly evident across cement LCA studies where concerns have been raised regarding data quality and transparency, functional unit selection, and inconsistent rules for by-product allocation. Consequently, the value of LCA as a key strategic decision making methodological framework has been questioned.

Despite these limitations, the use of LCA has allowed for the quantitative evaluation of different decarbonisation strategies. A decarbonisation strategy is defined as a plan or action that can be implemented to reduce the CO_2 emissions associated with a specific material or process. Within the cement and concrete sector, several decarbonisation strategies have either been implemented or are currently being developed for wide-scale adoption (IEA, 2021). Strategy implementation within the sector may occur at the clinker, cement, or concrete production stage, however it is necessary for the sector to implement decarbonisation strategies concurrently in order to achieve net-zero targets. The rate at which a given decarbonisation strategy can be implemented is determined not only by technological advancement but also by market readiness, which signifies the strategy's social, economic, and political viability. Accordingly, decarbonisation strategies can be classified according to their level of maturity. High-maturity strategies, such as using low-carbon electricity and fuels and shifting to more energy-efficient clinker kilns, have a defined market precedence which have allowed the UK cement and concrete sector to reduce emissions by 53% since 1990 (MPA, 2020). Conversely, low-maturity strategies, such as carbon capture, utilisation and storage and the electrification of clinker production, can significantly reduce the sector's emissions, however their implementation is currently limited due to economic, political, and technological barriers. Given the variation in market readiness among decarbonisation strategies, decarbonisation roadmaps were developed to quantify, prioritise, and coordinate strategy implementation. These strategic political frameworks are published by both government institutions (Climate Change Committee, 2020) and industrial organisations (MPA, 2020) and provide an overview on how key carbon targets will be reached enroute to Net Zero 2050. Within these roadmaps however, carbon abatement is often attributed disproportionally to low-maturity strategies, resulting in carbon targets that are often viewed with considerable uncertainty (Hammond, 2022).

Cross-sectoral interdependencies across other energy-intensive industries has also introduced additional uncertainty regarding the feasibility of achieving net-zero climate targets. As part of the UK steel sector's strategy to rapidly decarbonise, blast furnaces used in traditional steel production are currently being replaced by electric arc furnaces which utilise electric energy to melt scrap steel to create new products in lieu of fossil fuels and virgin raw materials (Jozepa, 2025, British Steel, 2024). While a sustainable and efficient solution for the steel sector, this shift presents unintended consequences to cement sector decarbonisation efforts as blast furnace slag, a key by-product produced during the traditional steelmaking process, is no longer produced. Blast furnace slag, when ground and granulated, can serve as a partial replacement for clinker when used as a supplementary cementitious material or as a full replacement when used as an alkali-activated material, allowing for significant reductions in clinker consumption and associated carbon emissions (Aslani et al., 2023). With the sector's current reliance on imported blast furnace slag to meet demand, this transition will result in the material becoming increasingly unsustainable and economically unviable (Alberici et al., 2017, Comtrade, 2025). This unintended consequence may stagnate decarbonisation efforts in the cement sector, further calling into question the feasibility of achieving net-zero by 2050.

Overall, the escalating pressure for energy-intensive sectors, such as the cement and concrete sector, to decarbonise has resulted in reduced transparency, overly optimistic assumptions, and fragmented approaches. Therefore, this thesis aims to address three primary research questions:

- Question 1: How can the application of LCA methodology be improved to provide a transparent, robust, and consistent representation of direct sectoral emissions across different energy-intensive industries to support more effective decision-making?

- Question 2: What approaches enable high-certainty prediction of net-zero pathways in the UK cement and concrete sector, considering differing levels of market readiness and technological maturity across decarbonisation strategies?
- Question 3: To what extent does the UK steel sector's shift to the electric arc furnace production route affect decarbonisation efforts in the UK cement and concrete sector through its impact on GGBS availability, given their existing cross-sectoral dependency?

By addressing these three research questions, this thesis aims to investigate how the UK cement and concrete sector can realistically achieve net-zero by 2050. The core results and discussion section of this thesis focuses on decarbonisation foundations, pathways, and synergies, providing a holistic assessment of decarbonisation strategies for the UK cement and concrete sector within the context of energy-intensive industries. Figure 2 illustrates the organisational structure of the thesis. The thesis consists of 7 chapters with chapters 3, 4, 5, and 6 serving as the results and discussion sections. The numbering, figures, tables, and references found in chapters 4, 5, and 6 are self-contained to keep the style consistent with the published versions.

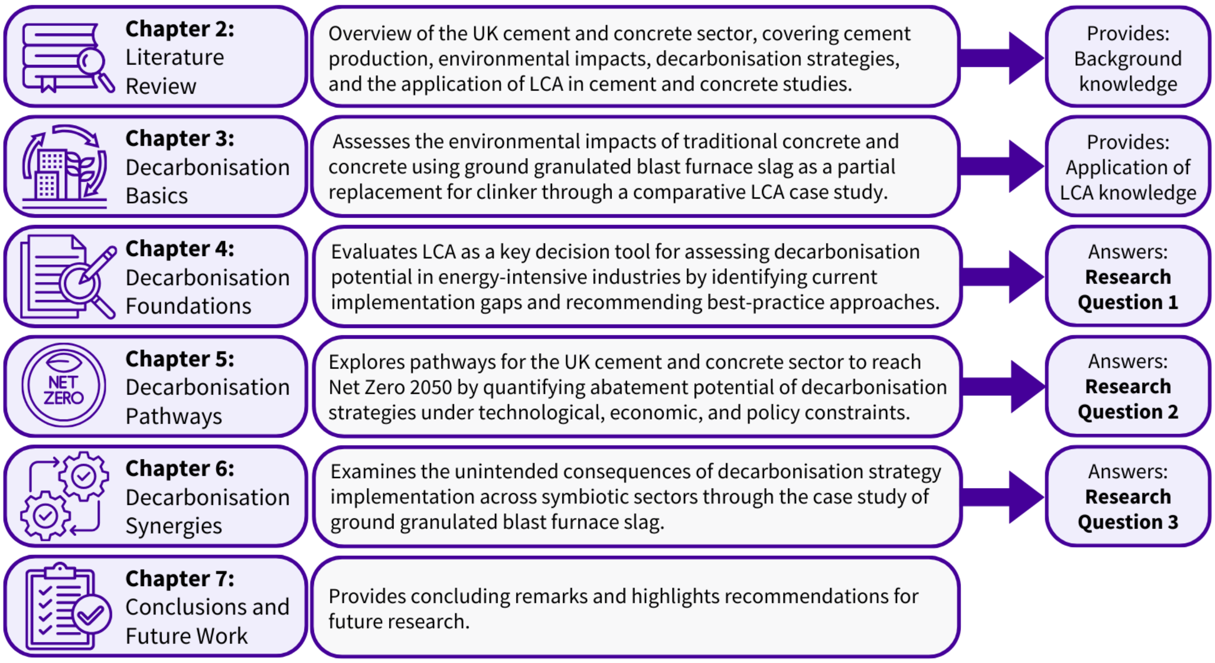


Figure 2: Visual overview of thesis structure

2. LITERATURE REVIEW

2.1 Understanding Cement in the UK Context

Cement is generally defined as an adhesive material capable of binding different material fragments together to form one cohesive mass (Bye, 2011). The British Standard European Norm (BS EN) 197-1:2006 defines cement more specifically as a hydraulic binder, that when combined with water, undergoes a series of chemical interactions that allows the material to set, harden, and retain its strength and stability even if the material is fully submerged in water (BS EN 197-1, 2019). To increase the durability and strength of this binder (typically referred to as cement paste), it can be combined with both fine and course aggregates to create concrete.

2.1.1 History of Cement Production and its Current Use

The earliest evidence of humans utilising hydraulic cements in its modern understanding dates back to ancient Egypt. The Great Pyramid was constructed with the aid of a cementitious mortar that was created by burning gypsum and combining it with sand and water (Blezard, 2004). The ancient Greeks would alternatively burn limestone to create lime mortar, as gypsum mortar would prove unusable in the Mediterranean climate (Lucas, 1934). In addition, the Greeks were the first to incorporate volcanic ash with lime to form an even stronger mortar. Using this knowledge, the ancient Romans discovered that the volcanic ash located near the village of Pozzuoli next to Mt. Vesuvius could be used to create high quality pozzolan-lime cement (Van Oss, 2005, Blezard, 2004). The modern use of the term "pozzolan" to describe a siliceous material that exhibits cementitious properties when combined with lime and water originates from this period. This cement would be used extensively in Roman construction with some structures such as the Pantheon and Colosseum still standing today (Kosmatka et al., 2002). The fall of the Roman Empire in 476 AD brought forth a decline in cement usage for several centuries, with many techniques used to produce high quality cement mixes seemingly becoming lost until the 12th century (Blezard, 2004). Interest in the understanding and use of hydraulic cementitious properties did not re-emerge until the 18th century, when John Smeaton discovered that impure limestone containing traces of clay produced superior hydraulic cementing properties. This discovery allowed for cement manufacturers to consistently produce high quality, finely grounded natural cement, further increasing its use (Blezard, 2004, Kosmatka et al., 2002).

The invention of modern day cement, labelled as ordinary or traditional Portland cement, is attributed to Joseph Aspdin who filed a patent for the artificial stone in 1824 (Aspdin, 1824). The name was chosen due to the colour and strength similarities the material shared with Portland stone quarried on the Isle of Portland on the coast of Dorset (Blezard, 2004). While the patent specified the main constituents used including crushed calcined limestone, clay, and water, many specific details regarding the material's manufacturing process were left out, likely as a way to prevent competing businesses from utilising his techniques (Blezard, 2004). In 1845, Isaac C. Johnson would discover and publicly reveal that the secret behind Portland cement's high quality was the higher kiln temperatures that resulted in the formation of clinker, a glassy material that contained hydraulic di-calcium silicate and tri-calcium silicate that forms the primary constituent in Portland cement (Blezard, 2004, Kosmatka et al., 2002). This information allowed for the Portland cement's manufacturing process to be continuously improved, eventually resulting in today's modern manufacturing process. While the material would be used throughout the late 19th and early 20th centuries, the production of Portland cement began to see a massive increase globally during the second half 20th century (Blezard, 2004). As populations continued to grow, countries began seeing the binder and its use in concrete as a preferred construction material due to the abundance of raw materials available for its manufacture, its low cost of production, and the convenience of its use as seen in the production of ready mix concrete (Biernacki et al., 2017). Today, concrete is one of the most used construction materials in the UK with 90 Mt consumed each year (MPA, 2020).

2.1.2 Current UK Supply of Cement

Currently the UK has eleven operating kiln sites, with the largest production site located in Hope, England. Figure 3 presents the total amount of cement consumed in the UK each year from 2013 to 2023. In 2016, cement production peaked at 9.4 Mt, however since then material production has been on a downward trend, reaching its lowest level in 2023. Over the past decade, exports of cement have essentially ceased, as all production is utilised domestically. On average, 22% of the cement consumed in the UK is imported each year, with imports doubling from 15% to 32% over the decade. During this same time period, domestic cement production began to steadily decrease, implying that the UK has become increasingly reliant on imports to meet demand. In 2023, the UK imported 3.6 Mt of cement, primarily sourced from Spain, Ireland, and France likely due to their close proximity to the UK (Comtrade, 2025).

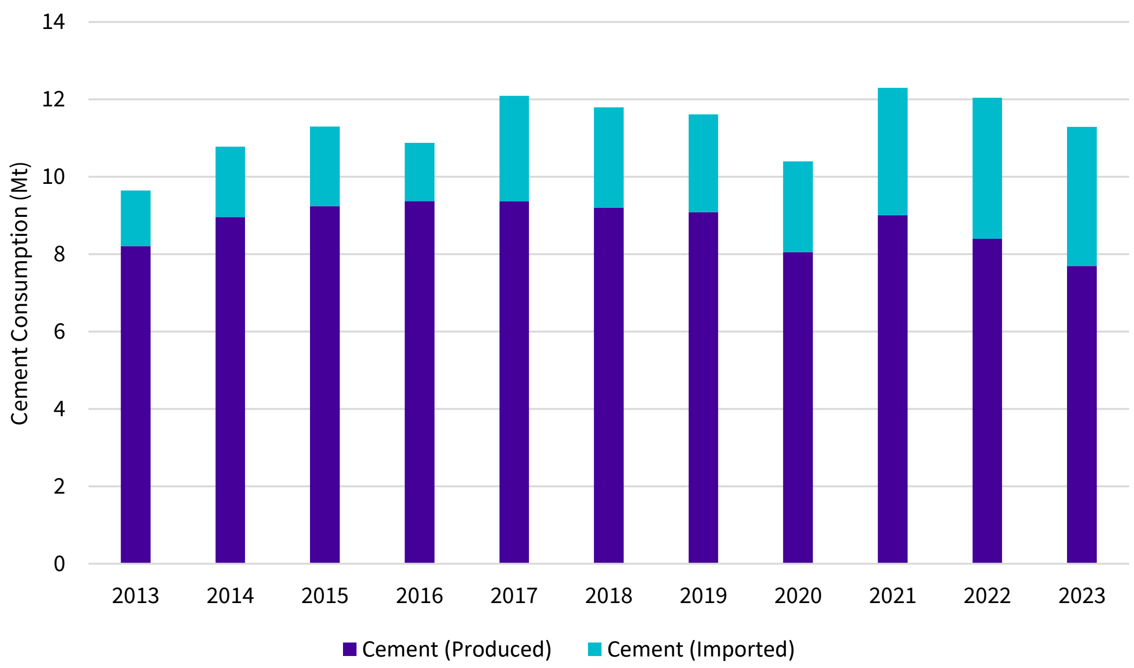


Figure 3: UK manufactured and imported cement (2013-2023) (MPA, 2023)

The sector's current reliance on imports is only expected to grow over the coming decades. In 2014, the British Geological Survey with the aid of the Mineral Product Association and other industry contacts was able to compile information about the current raw material quarry reserve life at several UK cement plants. Table 1 illustrates this information alongside the cement clinker capacity per year at each site. While the longest-running sites will be able to continue operating until 2042, many others will encounter raw materials shortages before that time (BGS, 2014). With new cement plants typically requiring a large upfront investment of at least £250 million along with additional millions of pounds of annual upkeep costs, it is in the best interest of the UK cement sector to optimise the current raw material supply found at these plants (BGS, 2014).

Table 1: Reserve life of UK quarries adjacent to cement plants (BGS, 2014)

MPA	Plant	Cement clinker capacity (Thousand tonnes/year)	Reserve Life (Years)
Staffordshire	Cauldon	900	> 40 years
Vale of Glamorgan (South Wales)	Aberthaw	500	> 30 but < 40 years
East Lothian (Scotland)	Dunbar	900	> 30 but < 40 years
Tyrone (Northern Ireland)	Cookstown	480	> 40 years
Derbyshire	Tunstead	1095	> 30 but < 40 years
Rutland	Ketton	1390	> 10 but < 20 years
Lancashire	Ribblesdale	750	> 20 but < 30 years
Flintshire (North Wales)	Padeswood	820	> 30 but < 40 years
North Lincolnshire	South Ferriby	750	> 40 years
Warwickshire/Bedfordshire	Rugby - Southam Quarry	1500	> 30 but < 40 years
Warwickshire/Bedfordshire	Rugby - Kensworth Quarry		> 20 but < 30 years
Peak District National Park	Hope	1300	> 20 but < 30 years

2.1.3 Clinker and Cement Composition

The composition of modern Portland cement clinker is defined in BS EN 197-1:2011 as a "hydraulic material which shall consist of at least two-thirds by mass of calcium silicates, the remainder consisting of aluminium and iron containing clinker phases and other compounds" (BS EN 197-1, 2019). Clinker consists primarily of four major constituents: calcium oxide (CaO), silica (SiO_2), alumina (Al_2O_3), and ferrous oxide (Fe_2O_3). In addition, trace amounts of other oxides can be found including sulphur (SO_3) and magnesia (MgO) (Bye, 2011). The percentage range of these oxide elements are listed in Table 2.

Table 2: Composition of Portland cement clinkers (Lavagna and Nisticò, 2023)

Constituent	Percentage
CaO	58-67%
SiO_2	16-26%
Al_2O_3	4.0-8.0%
Fe_2O_3	2.0-5.0%
MgO	1.0-4.0%
SO_3	0.1-2.5%

CaO, or lime, is the most prevalent constituent in clinker (Jackson, 2004, Manning et al., 2019). The main source for this constituent is limestone; a calcareous material that contains both calcium and magnesium. While the presence of some magnesium is acceptable in the raw mix, the total amount of the constituent may not exceed 5% (BS EN 197-1, 2019). This limit has become standard in the UK to reduce the possibility of excessive expansion caused by the hydraulic reaction between magnesium oxide (MgO) and water producing magnesium hydroxide ($Mg(OH)_2$). This volumetric expansion results in the formation of internal cracking, which decreases the overall durability and robustness of the material (Li et al., 2021, Rehsi, 1983). It should be noted that some limestone such as dolomites are not considered feasible for use in cement due to its high MgO content (BGS, 2014).

Figure 4 illustrates the locations of limestone deposits in the UK suitable for clinker production. The most commonly used limestone types in the UK are Carboniferous and Jurassic limestone due to their high degree of purity and low porosity (BGS, 2014, BGS, 2005). Carboniferous limestone deposits are seen most notably in the Peak District of Derbyshire in South Yorkshire and the Lake District in the north-west of England. In addition, there are sizable deposits of Carboniferous limestone in the north and south of Wales. Unlike the other parts of the UK, Scotland's limestone deposits are scarce. The main deposit found in the south east of Scotland is home to the region's only cement plant in Dunbar (BGS, 2014). Jurassic limestone is found mainly along a large belt that stretches from the south west of England through the country's midlands and up to the north-eastern Yorkshire coast. Despite the length of this belt, the deposits themselves are quite thin when compared to the Carboniferous limestone deposits. Cretaceous chalk, while more porous compared to Cretaceous limestone, is also used extensively. This chalk is found mainly in the south and south-east of England, but also on the coast of the East Midlands and Yorkshire. Figure 4 also highlights the close proximity of cement manufacturing plants to limestone deposits which allows for easy quarry access to reduce not only cost but also transportation related emissions (BGS, 2014).

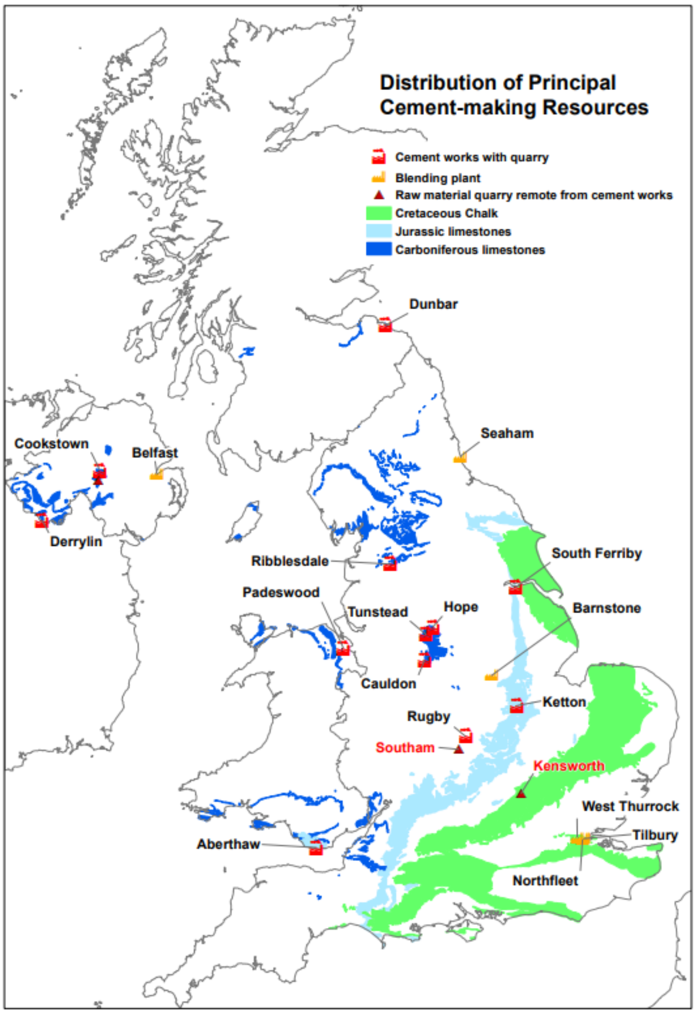


Figure 4: Limestone resources in the UK and cement plant locations (BGS, 2014)

For the remaining constituents of the raw material mix, clay and shale are the main raw material sources utilised to account for the silica, alumina, and iron oxides present in the composition of clinker (Kosmatka et al., 2002). In the UK, these argillaceous components needed for the manufacture of Portland cement clinker are abundant. As a result, quarries containing these argillaceous materials can typically be found in very close proximity to cement plants (BGS, 2014). Additional raw materials may also be added to help improve the quality of cement is needed. This includes raw materials such as iron ore and sand which can provide an additional iron oxide and alumina to a raw mix respectively (Kosmatka et al., 2002). Based on the stoichiometry of the calcining reaction, Manning et al. (2019) determined the amount of raw materials needed to produce one ton of clinker. It was calculated that 1.16 tons of limestone and 0.39 tons of shale are needed per ton of clinker given a proportion of 75% limestone and 25% shale. Between 2019 and 2023, an average of 7.14 Mt of clinker was produced in the UK each year. Using this information, it can be deduced that the approximately 8.3 Mt of limestone and 2.8 Mt of shale is used in UK clinker production annually (Idoine et al., 2025).

Since clinker constitutes 90–95% of ordinary Portland cement's raw material composition, the chemical composition of cement closely mirrors that of clinker. Table 3 highlights the typical composition of ordinary Portland cement.

Table 3: Composition of Portland cement (Ogirigbo, 2016)

Constituent	Percentage
CaO	60-67%
SiO_2	19-23%
Al_2O_3	3.0-7.0%
Fe_2O_3	1.5-4.5%
MgO	0.5-2.5%
SO_3	2.5-3.5%

The key composition difference between clinker and Portland cement is the increased presence of sulphite originating from the addition of gypsum. Gypsum is the primary constituent added to clinker during the fine grinding process to aid in the regulation of the material's setting time (Bye, 2011). Typically a 5% addition of the cement's mass is added, however this value could be less depending on the amount of tricalcium aluminate (C_3A) present in the clinker, the clinker's fineness, and the amount of sulphite (SO_3) present in both the clinker and gypsum (Zhang, 2011). Excessive amounts of SO_3 in the gypsum can cause the cement's strength to decrease as a result of material instability (Zhang, 2011). In the cement industry, natural gypsum is preferred over the use of synthetic gypsum, an industrial by-product. While both can be utilised, the synthetic gypsum's higher moisture content and faster reaction time makes its use less desirable (BGS, 2006). The East Midlands in England is where the majority of gypsum quarries can be found, however additional quarries can be found in the East Sussex, Staffordshire, and Cumbria (BGS, 2006). In the past couple of decades however, the availability of natural gypsum in the UK has decreased significantly. As a result, the UK has begun importing the material from other European countries such as Spain meet the country's demand (Comtrade, 2025).

2.2 Decarbonising UK Cement Production: Targeting Emission Sources

Although cement is fundamental to modern construction, its carbon-intensive manufacturing processes poses a challenge to the future of sustainable infrastructure. Figure 5 presents the total carbon emissions originating from UK cement production from 1950-2023. Since the 1950, the UK cement industry has emitted 470 Mt of CO_2 emissions. Between 1950 and 1970, carbon emissions increased sharply as a result of post-war construction demand. Emissions peaked in 1973 at 9.9 Mt of CO_2, followed by a general decline before peaking again in 1989 likely due to increased material demand. Since then, emissions originating from cement production have been on a general decline.

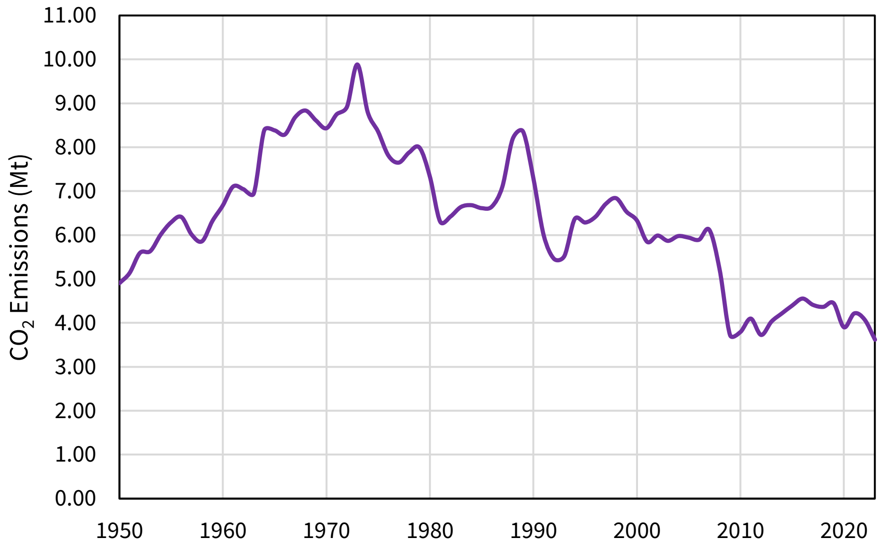


Figure 5: UK carbon emissions originating from cement production (Our World in Data, 2023)

While the UK cement sector has reduced its carbon emissions by 50% compared to 1970s levels, as of 2022 it still accounts for 9% of the country's manufacturing emissions (Li and Unluer, 2025, ONS, 2024). The main emission sources from cement production can be classified into three categories: process emissions, combustion emissions, and indirect emissions (Griffin et al., 2014). The following sub-sections explain each emission category, its contribution to the sector's overall carbon footprint, and the decarbonisation strategies that have been implemented or are being explored to reduce these emissions.

2.2.1 Process Emissions

Accounting for more than 50% of the cement sector's total carbon output, process emissions are the result of one key stage of the cement manufacturing process: calcination (Lehne and Preston, 2018). Calcination is defined as the process in which the crushed and proportioned raw materials are heated in a rotary kiln to form clinker, the primary constituent in cement. The primary chemical reaction that occurs is defined by Equation 1, where the heat present in the kiln causes the decomposition of $CaCO_3$ (i.e. limestone) into CaO and CO_2. This reaction typically occurs around 900°C; however, kiln temperatures must reach upwards to 1500°C to ensure that the key mineral phases in clinker are produced.

$$CaCO_3(s) \rightarrow CaO\ (s) + CO_2\ (g)$$

Equation 1

Decarbonising process emissions remains the sector's most significant challenge. In recent years, carbon capture and storage (CCS) technology has emerged as the leading solution. There are several different capture technologies, however the most mature solution is post-combustion capture using amine-based solvents. This capture process begins by passing the flue gas from the rotary kiln into an absorber containing an amine solution. The CO_2 present in the flue gas chemically binds to the amine in the solution, allowing for the remaining gases to escape. The CO_2 rich amine solution is then pumped into a stripper where the solution is heated to separate the CO_2 gas from the amine. The amine solution is then recycled back into the absorber, allowing the pure CO_2 gas to be dried, compressed, and stored (Xue et al., 2017). The benefit of the post-combustion pathway is the ability to retrofit existing plants with this technology, however this implementation comes at the cost of lower plant efficiency due to the increased energy demand required for amine regeneration (Wang and Song, 2020). Another popular carbon capture solution in the cement sector is oxy-fuel combustion. This CCS approach replaces the air in the rotary kiln with pure oxygen which results in the generation of a flue gas consisting of only water vapour and CO_2 (Faria et al., 2022). While the removal of other elements typically present in flue gas allows for a more simplified CO_2 separation process compared to the post-combustion process, installing this technology would require extensive retrofitting of existing kilns. This is due to the fact that the use of pure oxygen produces a higher internal kiln temperature which results in the rapid degradation of kiln components (Gerbelová et al., 2017). In addition, the high energy required to produce pure oxygen has been found to reduce a plant's energy efficiency by as much as 12% when compared to traditional cement plants (García-Luna et al., 2022). Calcium looping is another CCS technology that has also been explored for widespread implementation. The flue gas produced from the kiln is transported to a carbonator where CaO absorbs the CO_2 present in the gas to form $CaCO_3$. The $CaCO_3$ produced is then released in a calciner where it undergoes the same calcination reaction. The resulting CaO is then recycled back to the carbonator where the process is repeated (Arias et al., 2017). While limestone (and therefore CaO) is abundant, the effectiveness of recycled CaO decreases over time which results in the need to add new virgin material (Ozcan et al., 2013). In addition, the energy required for the calciner process may result in fuel consumption that is two to three times higher compared to a traditional cement plant. Alternative energy generation however such electrification or oxy-combustion may off set this increased energy demand (De Lena et al., 2017, Liu et al., 2023).

While CCS technologies have been proven to be effective at smaller scale pilot plants, they have yet to be fully implemented due to high upfront and operational costs. To counteract these costs, complementary approaches to CCS, such as carbon capture and utilisation (CCU), have been explored. This process enables captured CO_2 to be converted into commercially valuable products including chemicals (Yoon et al., 2025), fuels, and polymers (Baena-Moreno et al., 2019). One of the most common CCU methods that has been explored for use in the cement sector is CO_2 mineralisation. This method enables captured carbon to be permanently stored as $CaCO_3$ during concrete batch mixing, curing, or through carbonation (Zajac et al., 2022).

As noted in Figure 6, the UK cement sector has a carbon capture potential of approximately 12 Mt CO_2 per year. If CCS technology becomes fully implemented, process emissions released during calcination could be reduced by up to 74% (Marsh et al., 2023). Despite limited government investments into the technology over the past decade, in 2024 the UK government announced a £21.7 billion grant (applied over the next 25 years) has been approved to aid in the funding of CCS implementation (UK GOV, 2024, BBC, 2017). Even with this investment, implementation is still impeded by technical challenges, regulation concerns, public perception, and the need for cross-sector infrastructure integration (Shourideh and Yasseri, 2024, Zhang et al., 2024).

Figure 6: UK cement industry CO_2 emissions with CCS potential (Danaci and Bui, 2022)

2.2.2 Combustion Emissions

Accounting for roughly 40% of total CO_2 emissions produced from cement manufacturing, combustion emissions arise from the burning of fuels during clinker production (Lehne and Preston, 2018). The rotary kiln accounts for the largest fuel consumption in the production process as an internal temperature of approximately 1450 °C is needed to achieve clinkerisation (MPA, 2019). Figure 7 illustrates the average fuel mix used in the UK for clinker production. In 2024, fossil fuels accounted for 46% of the UK clinker fuel mix; a nearly 23% decrease compared to the average fuel mix in 2014 (DECC, 2014). Within the fossil fuel share of the fuel mix, coal still remains the predominant type. From 2014 to 2024, the UK saw a decrease in annual coal production by 99% with only 0.11 Mt of the material being produced in 2024 (BEIS, 2020). Currently, the UK relies on coal imports from Colombia, the European Union, and South Africa with these exporters representing 80% of the total coal imports in 2024 (UK Government, 2025). While this is the case, the amount of coal being consumed annually has decreased significantly as a result of an ever increasing interest in alternative, lower carbon fuels. In 2024, only 2.4 Mt of coal was consumed in the UK; an 96% decrease from 2014 (BEIS, 2020). This shift in the cement sector has resulted in an increase use of bio and non-bio based waste-derived fuels. By using alternative biofuels, not only is landfill waste reduced, but also CO_2 emissions generated during clinker production. Over the past decade, the amount of waste-derived fuels used in the UK clinker fuel mix has increased by 33%. In 2024, waste-derived fuels accounted for 54% of the fuel mix, with 25% originating from biomass (MPA, 2025). The most common biomass is solid recovered fuel (SRF) which is made from commercial and domestic waste. Additional alternative fuel sources include tyres, waste solvents, and meat and bone meal (DECC, 2014). While it is possible to increase the percentage of biomass used within the fuel mix, additional uptake may be limited due to supply constraints caused by reduced land availability for biomass harvesting (Pamenter and Myers, 2021).

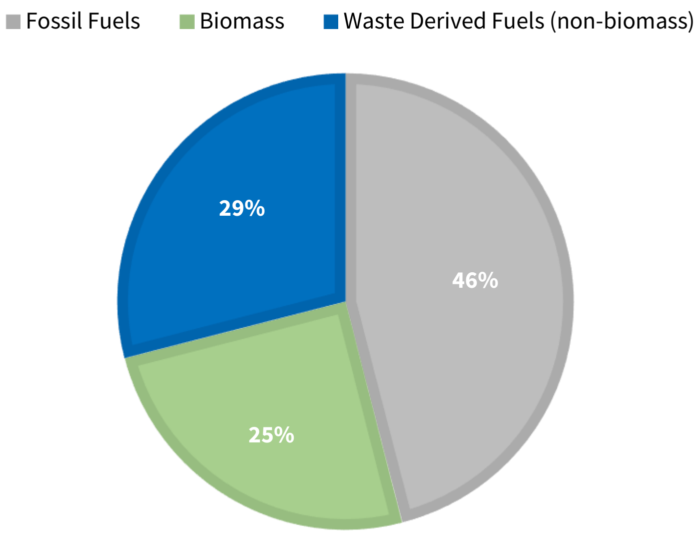


Figure 7: Fuels used in the UK for clinker production (MPA, 2025)

Over the years, thermal efficiency improvements have been implemented as an effective decarbonisation strategy to reduce the amount of fuel consumed, and therefore combustion emissions, arising from clinker production. One example of this that has been widely adopted in the UK is the use of dry kilns over wet kilns. By using dry kilns, heat consumption can be reduced by 35% (Liu et al., 2015). The introduction of pre-heating cyclones and pre-calcining kilns have also significantly improved thermal efficiency by allowing the raw meal to undergo pre-calcination through the utilisation of waste gases generated by the rotary kiln (MPA, 2019). Waste hot gas generated by the kiln preheater can also be transferred to the roller grinder mill to dry the raw mix before the calcination process begins (Bye, 2011). Furthermore, waste heat from the clinkerisation process can be recovered for electricity generation, providing up to 21% of the total electrical energy needed for a typical cement plant (Poggianti et al., 2024). By maintaining the insulation lining inside the kiln, thermal efficiency can be further improved by reducing heat loss (Zeng et al., 2023). Cool air from the clinker cooling process can also be recycled and reclaimed to partially dry raw meal (Marenco-Porto et al., 2023). By implementing thermal efficiency improvements, the carbon emissions associated with clinker production can be reduced by as much as 3.7% (Marsh et al., 2023). The key barrier to the implementation of these thermal efficiency improvements is material and equipment durability as insulation is prone to degradation over time due to wear. This results in frequent equipment downtime which may lead to increased operational costs. A lack of available infrastructure and space limitations were also highlighted as barriers to waste heat recovery system adoption (Christodoulides et al., 2022).

Another innovative solution that has been explored to further reduce combustion related emissions is electrification. By electrifying the rotary kiln, the cement sector's dependence on fossil fuels can be reduced, resulting in emissions savings of up to 30% (Marsh et al., 2023). There are several proposed electrification pathways in which clinkerisation can be achieved. One promising solution are thermal torches which utilise electricity to heat compressed gas (typically CO_2) through ionisation. The ionisation process results in the formation of plasma; an electrically charged ionised gas capable of reaching temperatures of 5,000 °C (Heidelberg Materials, 2025b). Another option is resistive electrical heating which utilises electrified plates inside the kiln to generate and transfer heat through either convection, conduction, or radiation (Antunes et al., 2022). While retrofitting existing kilns with this technology is possible, difficulty regarding efficient heat transfer within the rotary kiln (Antunes et al., 2022), challenges with material degradation due to the high temperatures required for clinkerisation (Quevedo Parra and Romano, 2023), and the initial and operational investment required may limit the technology's widespread adoption (Volaity et al., 2025). The effectiveness of electrification is also dependent on the capacity and energy mix of the local electric grid. In the UK, concerns have been raised about the burden caused by increased electricity usage from energy-intensive processes. To account for this, significant grid infrastructure upgrades would need to be implemented including access to more continuous and reliable green energy sources (Volaity et al., 2025).

2.2.3 Indirect Emissions

Accounting for 10% of the sector's carbon footprint, indirect emissions are those that do not arise directly from on-site fuel combustion or the calcination process. This includes emissions originating from upstream activities such as raw material extraction and transportation as well as electricity consumption for grinding and mixing. Since most cement plants are located next to quarries that contain the necessary raw materials for cement production, transportation impacts are relatively small. Emissions originating from the material extraction process itself is also minimal, accounting for roughly 3% of the cement sector's overall carbon emissions (Wernet et al., 2016). Therefore, largest proportion of indirect emissions arises from the electricity required for the mixing and grinding of raw materials.

The first mixing and grinding stage occurs when the excavated raw materials (e.g. limestone, shale, and sand) are transported to a primary crusher where the materials are crushed to a size of around 125mm. These stones are then transported by a vibrating screen to a secondary crusher where the stones are reduced to a size of 20mm (Kosmatka et al., 2002). After properly proportioning the crushed raw materials to achieve the optimum chemical composition, the raw mix is fed into a roller grinder mill which crushes and grinds the mix into a fine powder. Grinding also occurs after clinkerisation as the clinker is transported to a grinding mill and blended with gypsum to produce Portland cement. Figure 8 illustrates the electricity generation sources for the UK grid in 2024. Currently the two largest generation sources in the UK are natural gas and wind, however the UK has committed to fully decarbonise the country's electricity grid by 2035 by investing in offshore wind and nuclear energy (UK GOV, 2021). As part of this transition, the UK has also reduced coal consumption for energy generation from 101 TWh in 2014 to 2.7 TWh in 2024 (IEA, 2024). Despite this shift, concerns have been expressed regarding the ability to meet the ambitious 2035 target. Helm (2023) notes that the 2035 goal is implausible given the current policies in place. To realistically meet this goal increased government intervention would be needed which may result in a larger financial burden on UK taxpayers (Helm, 2023).

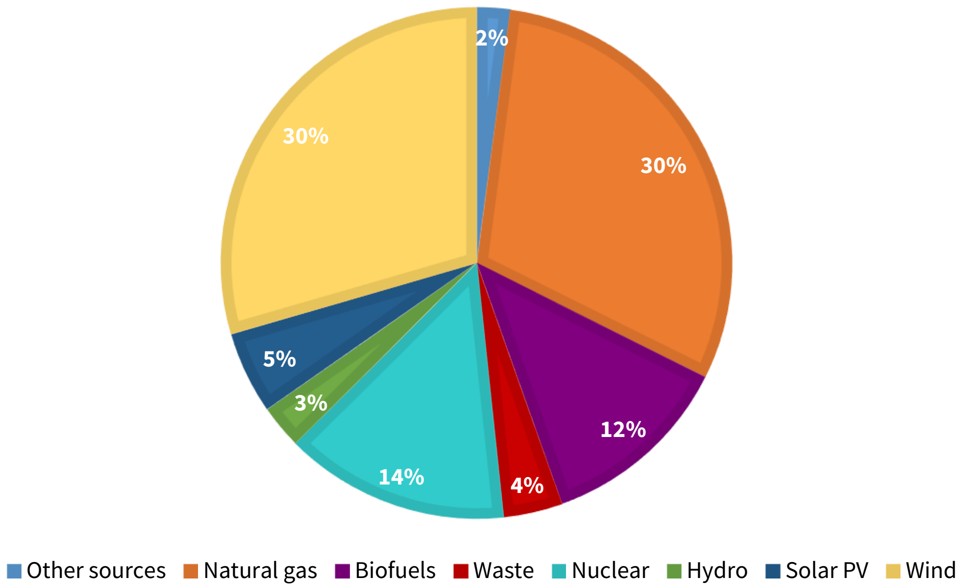


Figure 8: UK electricity generation sources, 2024 (IEA, 2024)

2.3 Decarbonising UK Cement Production: Reducing Clinker Demand

While the cement sector's energy intensive nature greatly contributes to its high carbon footprint, the growing demand for the material has further resulted in increased sectoral emissions. Given this, solutions have been implemented or are currently being explored to reduce the demand for new clinker and therefore Portland cement. One novel way to reduce this demand is by utilising end-of-life concrete. By crushing, sieving, and treating recycled concrete, fines can be separated for use as clinker feedstock to provide an emissions reduction of up to 12% (Marsh et al., 2023). This replacement is possible due to the calcium silicate hydrate product (C-S-H) and un-hydrated alite and belite which can be activated in new cement mixes. A study by De Brabandere et al. (2025) found that a 10% partial replacement with concrete fines in a Portland cement mix had a compressive strength comparable to a traditional Portland cement, however the mix was found to have reduced durability in freeze-thaw conditions in addition to increased drying shrinkage. Oksri-Nelfia et al. (2016) however found that a substitution up to 25% could be used without effecting key properties such as early hydration and compressive strength. The performance of recycled concrete fines can be further enhanced through carbonation (Zhang et al., 2023) or mechanical activation (Rajczakowska et al., 2025). The key limitation to this decarbonisation strategy is the variability associated with recycled concrete as the performance of the material is effected by exposure conditions, age, and the composition of the original concrete mix (Shin et al., 2025). While end-of-life concrete is readily available and a circular solution, its low substitution rates caused by performance limitations impacts the effectiveness of this strategy.

Another solution to reduce clinker demand is by using supplementary cementitious materials (SCMs) as a partial replacement for clinker in blended cement mixes or as a full replacement for clinker as a precursor in alkali activated cements. Both of these strategies are outlined in further detail in Section 2.3.3 and 2.3.4, respectively. There are several types of SCMs used for low-carbon cement production, including silica fume, fly ash, ground granulated blast furnace slags (GGBS), limestone fines, and calcined clays. While GGBS is currently the most used SCM in the UK, there is increased interest in more abundant SCMs such as calcined clays (Scrivener et al., 2018). Therefore, the scope of SCMs in this thesis is limited to GGBS and calcined clays. Section 2.3.1 and Section 2.3.2 highlight the production process and availability of both of these materials in the UK, respectively.

2.3.1 Ground Granulated Blast Furnace Slag

Slag is defined as an industrial by-product formed during the smelting and refining process of ferrous and non-ferrous metallic ores in a furnace. Due to its versatile nature, iron ore has become one of the most consumed metals globally (Brown et al., 2019). Nearly all iron ore mined and smelted is further refined into steel, a carbon-iron alloy, to create robust and durable buildings and infrastructure. The traditional steel manufacturing process is classified as the blast-furnace, basic oxygen furnace route (BF-BOF) which includes the smelting of iron ore in a blast furnace and the further refinement of iron in a basic oxygen furnace to produce steel.

The blast furnace production process is illustrated in Figure 9. The production process begins by adding iron ore, coke and limestone into a blast furnace that has been heated to 1500-1600°C by hot air blasts. This heated air reacts with the carbon present in the coke to create CO_2 which then reacts with the carbon molecules to form carbon monoxide. The reaction that occurs between the carbon monoxide and iron oxide present in the raw iron ore results in the formation of CO_2 and molten iron. During this process, the limestone is broken down into calcium oxide, which causes impurities present in the iron ore to be removed (Oge et al., 2019). The molten iron produced by these reactions settles to the bottom of the blast furnace while the less dense molten slag rises to the top. The molten slag can be cooled in one of two ways: air cooled or water quenched. While air cooled blast furnace slag does not exhibit cementing properties, the material can be used as an alternative aggregate product for concrete production. For blast furnace slag to exhibit cementing properties, the material must be rapidly quenched in warm water. After the material has been cooled using this method, the granulated slag can then be ground into a fine glass powder to create GGBS (Robinson et al., 2007).

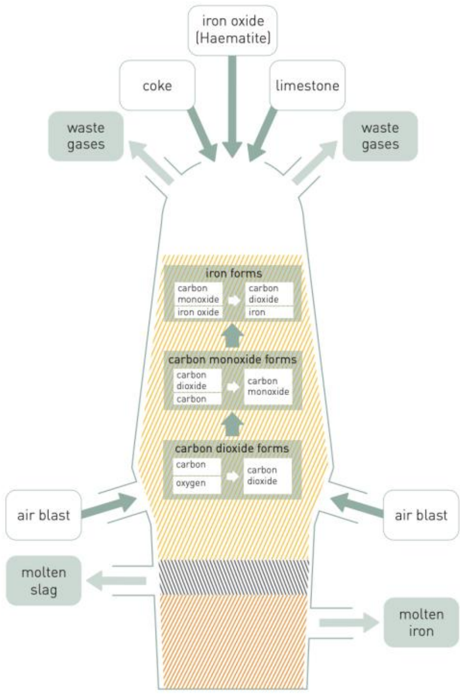


Figure 9: Blast furnace process (Robinson et al., 2007)

Figure 10 illustrates the amount of blast furnace slag consumed in the UK from 2000 to 2024. To determine the average amount of GGBS produced annually, a ratio of 0.28 ton of slag produced per 1.0 ton of pig iron was taken (Curry, 2018). Over the past decade, the UK has seen a continuing decrease in domestic iron production and therefore a decrease in GGBS production. This downward trend has only intensified in recent years as the UK steel sector has begun to transition away from BF-BOFs to electric arc furnaces; an alternative, more sustainable manufacturing method that refines and recycles scrap steel, reducing the sector's dependence on virgin raw materials. This method however does not produce GGBS as a by-product, further limiting the local supply of the material (Alberici et al., 2017).

To account for the shortage of GGBS and satisfy demand, UK has been relying on imported GGBS. Over the past decade, the UK has imported an average of 1 Mt of GGBS annually, with imports exceeding the total amount of GGBS produced domestically since 2023. Countries that are in close proximity to the UK such as the Netherlands, France, Spain, and Germany have consistently been some of the UK's largest exporters for GGBS. The Netherlands specifically has been one of the UK's top three GGBS exporters for the past decade (Comtrade, 2025). While this is the case, in the past five years, Japan has become one of the UK's largest GGBS exporters. Given the distance between Japan and the UK, it is likely that the CO_2 emissions associated with the material's transportation may outweigh the benefits of its use in low carbon cements.

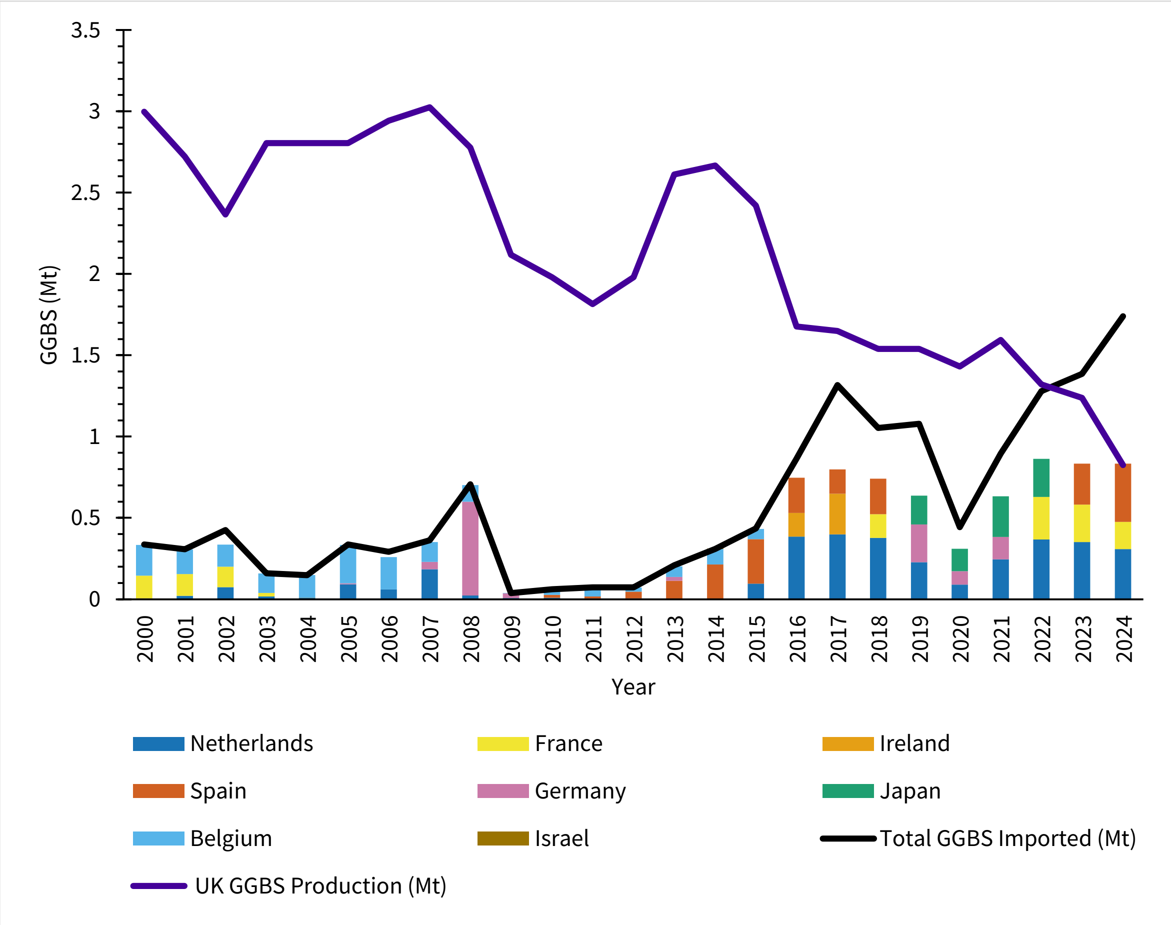


Figure 10: UK GGBS consumption, produced and imported stock, 2000-2024 (Comtrade, 2025, WSA, 2025)

2.3.2 Calcined Clays

In the UK, clays suitable for calcination are found in the southwest of England, particularly in Cornwall (BGS, 2009). To extract the clay, granite quarry walls as blasted to expose the raw material. To separate the clay from the surrounding granite, high pressure water jets are used. The slurry that is created from this extraction process is collected at the base of the quarry where the material is then pumped into a spiral classifier. To refine the raw material, machines separate and remove the sand and gravel present in the slurry. The refined slurry is then transferred into thickener tanks to begin the dewatering process. The thickened underflow is pumped into slurry storage tanks where the material then undergoes filter pressing to reach an 18-30% moisture content. To further remove excess moisture, the pressed clay is dried in a rotary dryer. This final process results in a fine powder having a typical moisture content around 11% (Reeves et al., 2006). In order for the clay to exhibit cementitious properties, the raw material must undergo calcination in rotary kilns at a temperature between 600-800°C (Fernandez et al., 2011).

Figure 11 presents the total amount of kaolin produced and exported in the UK since 2010. During this time period, the UK produced an average of 1 Mt of kaolin annually, with approximately 90% of all kaolin produced being exported for use in paper, ceramics, and specialty clays markets (BGS, 2009). While in the past few years there has been a decline in the total amount of kaolin produced, it was reported in 2009 that the UK has a kaolin quarry reserve life of 50 years (BGS, 2009). What is unknown however, is the impact this material's use in the cement sector would have on the UK supply chain. If large quantities of the material are to be used as an SCM, it is likely that production of kaolin would have to greatly increase in order to meet the demand in this new sector. This may result in the UK's current supply at these quarries with be depleted much quicker than originally estimated. While this may be the case, it should be noted that the kaolinite quality needed for cement production is lower than that required for the paper and ceramic industries (Scrivener et al., 2018). Outside of high-purity kaolinite, local clay types such as London clay has also been found to have good potential for used as an SCM (Dhandapani et al., 2023).

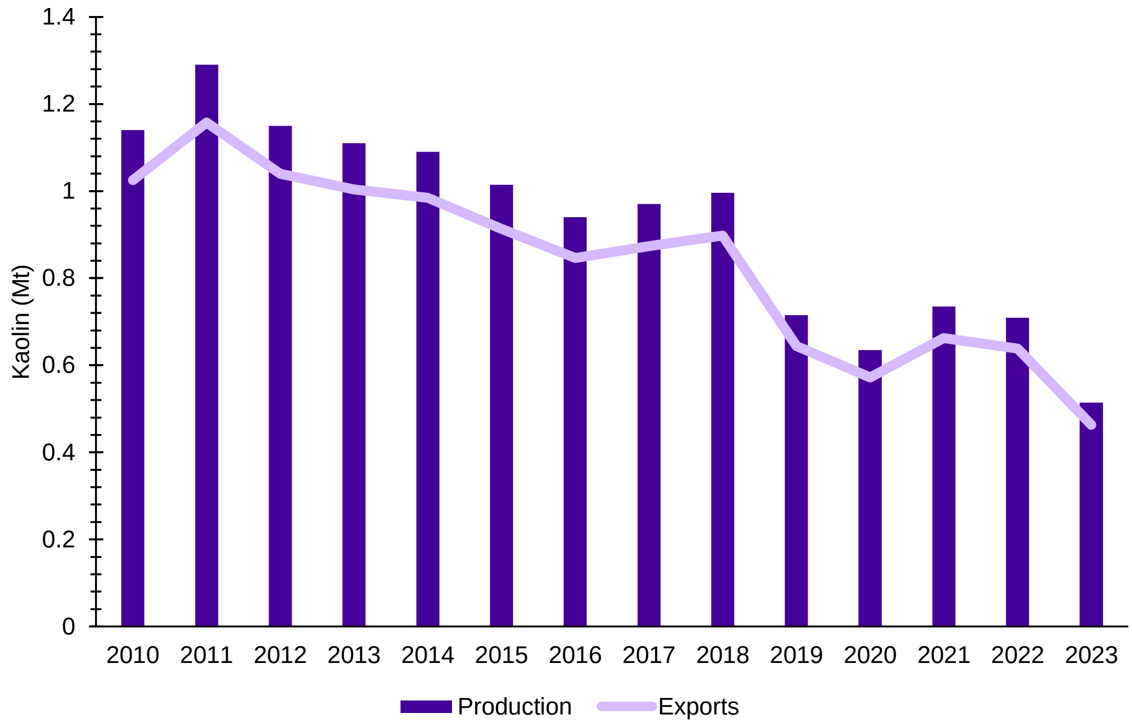


Figure 11: Annual UK manufactured and exported kaolin, 2010-2023 (Bide et al., 2024, Bide et al., 2018, Idoine et al., 2015)

2.3.3 Blended Cements

Blended Portland cements are defined as cements that utilise SCMs (such as GGBS and calcined clays) as a partial replacement for Portland cement clinker (Glavind, 2009). Blended Portland cements are manufactured in two main ways. The first method is by intergrinding both the SCM and Portland cement clinker in a grinding mill, which allows for the two materials to be grinded and blended simultaneously. The second and preferred method requires grinding the SCM and Portland cement clinker separately and then blend them together either on site in a mixer or at a cement factory (Higgins, 2009). Cements blended in a mixer, the UK's preferred method, has several benefits over blending in a factory including increased proportion flexibility, greater consistency, and reduced material transportation (Higgins, 2009, Lewis et al., 2003). British Standard European Norm standard (BS EN) 197-1:2011 Cement part 1: Composition, specifications, and conformity criteria for common cements provides a notation for five main cement types. These main types are then classified further based on the percentage of constituents present in the mix as noted in Table 4. While CEM II Portland slag cement allows up to 35% substitution with GGBS, the UK standard permits replacement levels of up to 95% in CEM III blast-furnace slag cement (BS EN 197-1, 2019). This high replacement value is not allowed for any other SCMs in blended cements, with 55% being the maximum substitution percentage allowed for calcined clays.

Table 4: GGBS and calcined clay blended cement types and their compositions (BS EN 197-1, 2019)

Main Cement Types	Notation	Clinker (%)	GGBS (%)	Natural Calcined Pozzolana (%)
CEM II: Portland Slag Cement	CEM II/A-S	80-94	6-20	-
	CEM II/B-S	65-79	21-35	-
CEM II: Portland-Pozzolana Cement	CEM II/A-Q	80-94	-	6-20
	CEM II/B-Q	65-79	-	21-35
CEM III: Blast Furnace Cement	CEM III/A	35-64	36-65	-
	CEM III/B	20-34	66-80	-
	CEM III/C	5-19	81-95	-
CEM IV: Pozzolanic Cement	CEM IV/A	65-89	-	11-35
	CEM IV/B	45-64	-	36-55
CEM V: Composite Cement	CEM V/A	40-64	18-30	18-30
	CEM V/B	20-38	31-49	31-49

One major factor that results in the effectiveness of SCMs is their composition. The calcium aluminosilicate hydrate gel binding phase, or C-A-S-H gel present in SCMs is very similar to the C-S-H present in hydrated Portland cement which allows these materials to be used as a cement alternative (Mohamed et al., 2020). Figure 12 illustrates differences in chemical composition between Portland cement, GGBS, and calcined clays. While these materials share the same chemical constituents, the proportions of each is varied. The primary constituent found in Portland cement and GGBS is calcium oxide, followed by silica. Due to GGBS typically has a slightly lower calcium oxide content and silica content compared to Portland cement, the material exhibits a lower degree of hydraulic reactivity when compared to Portland cement (Islam et al., 2011).

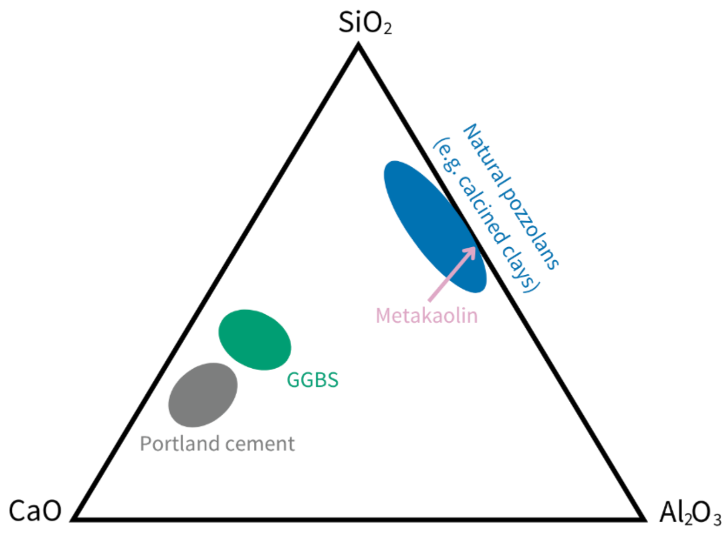


Figure 12: SCM ternary phase diagram, GGBS and calcined clays (Amran et al., 2021)

Compared to GGBS, calcined clays exhibit a broader compositional range due to the mineral variability. The two most common clay types used are kaolinitic clays and illitic clays. Kaolinitic clays have a simple 1:1 octahedral and tetrahedral sheet structure, which when calcined, forms a highly reactive pozzolan. Illitic clays have a more complex 2:1 octahedral and tetrahedral sheet structure which results in an effective but often less reactive SCM when compared to kaolinitic clays (Overmann et al., 2024). The most reactive calcined clay is metakaolin, a clay that is comprised of primarily pure kaolinite. While the principle constituent in Portland cement is calcium oxide, silica is typically the primary constituent in calcined clays. The alumina content present in calcined clays is also much higher compared to traditional Portland cement, with pure kaolin having a higher alumina content compared to other calcined clays due to mineral structure differences (Overmann et al., 2024). While both low-grade kaolinitic and illitic clays were found to exhibit cementitious properties, kaolinitic clays were found to reach calcination at a lower temperature (around 650–900 °C) when compared to illitic clays (above 850 °C) due to the additional iron oxide present in the latter (Msinjili et al., 2019).

In addition to reducing clinker consumption, using SCMs has been found to alter and often improve the performance of cement. Although the high silica content present in blended cements results in a longer setting time and decreased early age strength gain, these cements typically exhibit higher compressive strength when compared to Portland cement alone (Piemonti et al., 2021, Astutiningsih et al., 2018). Blended SCM cements have also been found to improve a concrete's resistance to sulphate-chloride attacks when compared to Portland cement (Abbas et al., 2010, Osborne, 1991).

Alongside traditional blended calcined clay cements, limestone calcined clay cement (LC^3) has shown to be another promising low carbon cement alternative. The difference between blended calcined clay cements and LC^3 is the mix's constituents: a typical LC^3 blended cement consists of 50% clinker, 30% calcined clay, 15% limestone, and 5% gypsum. The inclusion of limestone in the mix has been found to react positively to the high alumina content present in pure kaolin clays (Scrivener et al., 2018). LC^3 was also found to have comparable flexural and tensile strength, higher compressive strength, and improved chloride resistance when compared to traditional Portland cement (Scrivener et al., 2018).

2.3.4 Alkali Activated Cements

Alkali activated cements are a novel low carbon alternative that makes use of the aluminosilicate-rich nature of SCMs such as GGBS and calcined clays to fully replace Portland cement in a given sample. The formation of this solid binder is achieved by utilising SCMs as an aluminosilicate precursor in combination with an alkali activating agent. The reaction that occurs between the alkaline activator and precursor results in the dissolution of the precursor which creates a hardened binder with properties similar to that of Portland cement (Provis and Bernal, 2014). While not as widely used in industry when compared to blended Portland cements, utilising alkali activated materials have the potential to reduce CO_2 emissions by 9% by reducing the sector's dependency on Portland cement clinker (Marsh et al., 2023).

There are several different types of alkali activators that can be used to initiate the reaction of the precursor including alkali silicates, alkali hydroxides, alkali carbonates, and alkali sulphates. The most commonly used alkali metals in alkali activation are sodium and potassium. While activators containing potassium are typically more expensive compared to those containing sodium, the use of potassium based activators have been found to have greater workability and lower viscosity when compared to sodium based activators (Provis, 2009). Precursors are classified in two main systems based on the primary reaction product that is formed when combined with an alkali activator. The main factor that effects which gel structure is the most dominate is the calcium content present in the precursor (Provis and Bernal, 2014). For example, GGBS is classified as a high calcium alkali activated system due to its formation of calcium aluminosilicate hydrate (C-A-S-H) gel as its main hydrate product. In contrast, metakaolin is classified as a low calcium alkali activated system due to its formation of sodium aluminosilicate hydrate (N-A-S-H) gel as its main hydrate product and are therefore classified as geopolymers (Provis, 2018). When compared to traditional Portland cement, alkali activated cements have been found to have better overall performance (Duxson et al., 2007) including higher sulphate attack resistance (Bakharev et al., 2002), improved strength performance after exposure to high temperatures, and better bond performance with steel reinforcement when compared to traditional Portland cements (Ding et al., 2016). Despite these benefits, safety concerns have been raised regarding the handling of alkali activators on construction sites due to their corrosive nature (Yang et al., 2023). While alkali activated materials do provide a more sustainable option compared to traditional Portland cement, the energy intensive production process for the most commonly used activators generates additional carbon emissions that may offset the sector's carbon savings (Umer et al., 2024).

2.4 Validating Decarbonisation Strategies: Life Cycle Assessment

To accurately validate the implementation of a decarbonisation strategy in a given sector, the sector's baseline carbon emissions must first be quantified. The main methodological approach used to assess a product or sector's carbon emissions is life cycle assessment (LCA). LCA is a methodology used to evaluate a wide range of environmental impacts throughout a product's entire life cycle. It has also been defined as a key decision making tool in which environmental changes and improvements can be prioritised to best optimise a system given current financial and technological resources (Jolliet et al., 2015).

The LCA methodological framework was first standardised by the International Organisation for Standardisation (ISO) in late 1990s with the publication of the ISO 14040 series, comprising of ISO 14040:1997 (principles and framework) , ISO 14041:1998 (goal and scope definition and inventory analysis), ISO 14042:2000 (impact assessment), and ISO 14043:2000 (interpretation). In 2006, the standards were revised, and ISO 14041, 14042, and 14043 were consolidated into one new standard: ISO 14044:2006. This standard established the requirements and guidelines for LCA, while ISO 14040:2006 was updated and streamlined to focus on the framework's core principles (ISO 14044, 2006, ISO 14040, 2006). During this period, ISO 14025:2006 was also published which introduced formal standards for Environmental Product Declarations (EPDs) which are publicly accessible and verifiable documents that report the environmental impact results of a product (ISO 14025, 2006). This standard also established the requirement that EPDs must be developed in accordance with Product Category Rules (PCRs) to ensure comparability across similar products. While the first EPD and PCRs were published in the late 1990s, their widespread adoption did not emerge until the 2010s due to the increased demand for sustainability reporting (Kokulu et al., 2025, Toniolo et al., 2019). This was particularly the case in the construction sector as green building certifications such as LEED and BREEAM became commonplace (Gelowitz and McArthur, 2016, Awadh, 2017). In 2007, the ISO released ISO 21930 which provided further guidance on creating EPDs for construction products (ISO 21930, 2017). To improve the comparability of EPDs across the European construction sector, in 2012 the European Committee for Standardisation published EN 15804 (2012). In accordance to both of these standards, PCR 2019:14 for construction products was released (PCR 2019:14, 2025). This document provides specific rules regarding how the standardised LCA framework should be applied when developing EPDs for all construction products, including cement and concrete. Over time, standards and PCR guidelines have undergone several updates however the core framework for construction products has remained largely unchanged.

For the purposes of this thesis, the main methodological framework is based on the requirements outlined in ISO 14040 and ISO 14044, with PCR 2019:14 being drawn upon when appropriate. This approach is consistent with cement and concrete LCA academic literature, which views the methodological approach as a flexible and adaptable framework rather than one that is applied strictly (Olsson et al., 2024). In addition, full compliance with EPD standards in academic studies is typically not feasible due to limited access to primary industrial data and formal third-party verification processes (ISO 14025, 2006). Sections 2.4.1 to 2.4.4 provide details regarding each step of the LCA framework in the context of cement and concrete LCAs including goal and scope definition, inventory analysis, impact assessment, and interpretation as illustrated in Figure 13.

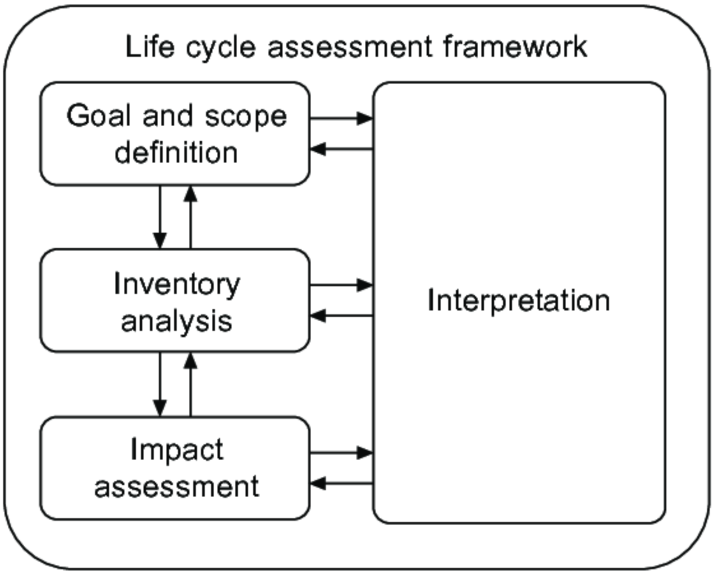


Figure 13: LCA framework (Ouellet-Plamondon and Habert, 2015)

2.4.1 Goal and Scope

The first step in the LCA framework is defining a study's goal and scope. In accordance with ISO 14040, a well-defined LCA goal must clearly state the purpose of conducting the study and for whom the study being conducted for (ISO 14040, 2006). An LCA scope describes the detail of the study and is defined by several key elements as presented in Figure 14.

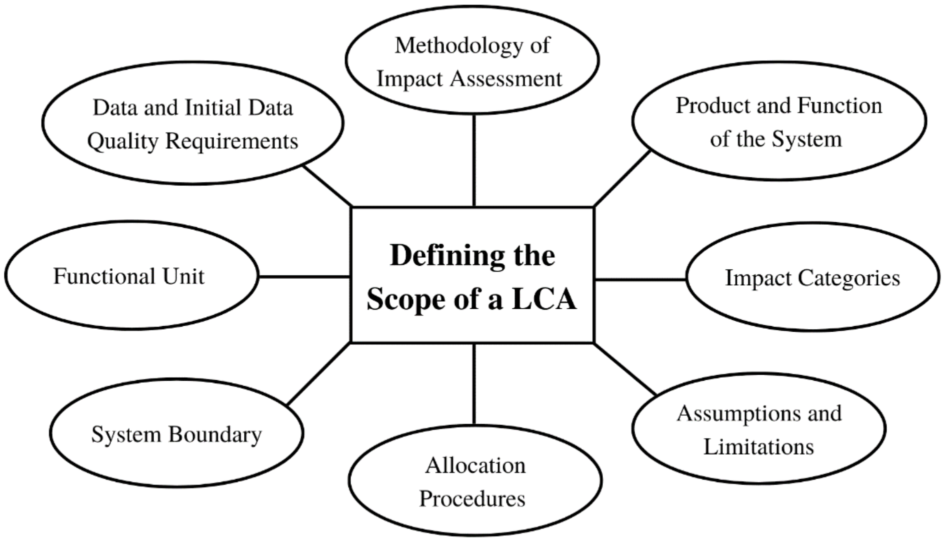


Figure 14: Defining the scope of a LCA

2.4.1.1 Functional Unit

A functional unit is defined by ISO 14040 as a "quantification of identified functions of a product" where the unit "provide(s) a reference to which the inputs and outputs are related" (ISO 14040, 2006). When functionality of a product assessed is unknown or not applicable for construction products, a declared unit may be used instead of a functional one for 'cradle-to-gate' studies if "the product or material is physically integrated with other products during installation so they cannot be physically separated from them at end of life, the product or material is no longer identifiable at end of life as a result of a physical or chemical transformation process, and the product or material does not contain biogenic carbon" (PCR 2019:14, 2025). These conditions hold true for cement, as the material will typically be combined with aggregates to create concrete which cannot be separated at its end of life due to a physical transformation process. For these reasons, mass or volume based declared units are typically selected for cement LCAs. Feiz et al. (2015) however argues that by omitting the use and end of life stages in the analysis, a mass based unit may be seen as a functional unit for cement LCAs as the functionality between each cement type could be seen as its use in concrete production. In the case of concrete LCA studies, the primary function of the material would be to provide a certain strength or durability requirement. It is particularly crucial that the functional equivalence is taken into consideration particularly when conducting a comparative analysis between two or more products (Panesar et al., 2017, Brimachombe and Parsons, 2022).

2.4.1.2 System Boundary

A system boundary is used to specify which life cycle stages, processes, material flows, and energy flows are assessed in a given LCA (ISO 14040, 2006). The most simplistic system boundary type is 'cradle-to-gate'. This boundary includes the extraction of raw materials needed to produce the product, transportation of these materials to the production site, and the manufacturing process of the product. A 'cradle-to-grave' further expands the 'cradle-to-gate' boundary by including the use and disposal stages of the finished product. The final boundary type, 'cradle-to-cradle', applies circular economy principles by assessing the impacts of recycling or reusing the product at its end of life. These system boundaries are illustrated in Figure 15.

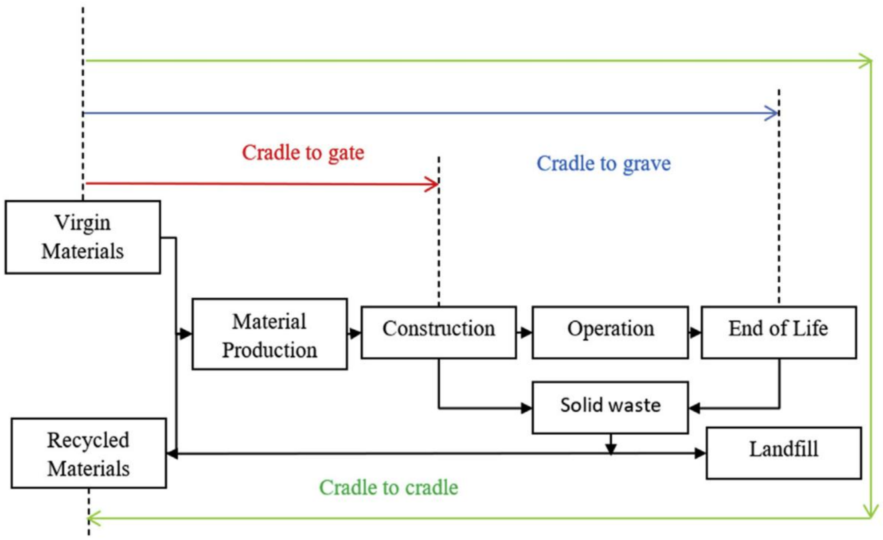


Figure 15: System boundaries for LCAs (Saberian et al., 2022)

For cement and concrete LCAs, the most commonly used system boundary in academic literature is 'cradle-to-gate', particularly since the production phase of a concrete LCA typically accounts for the greatest environmental impact (Colangelo et al., 2018). Some studies provide justification for this system boundary selection assuming that the use and end of life phase would be similar for all concrete mixes analysed (Teixeira et al., 2016). 'Cradle-to-grave' or 'cradle-to-cradle' boundaries are typically not considered as a structure's service life and maintenance requirements often vary significantly. To accurately conduct a study with one of these boundary types, the LCA's scope and functional unit must be specific to a given design (Feiz et al., 2015, Chen et al., 2010). In addition, carbon uptake of concrete structures due to carbonation would need to be assessed in the use and end of life stages. While this aspect can be used to reduce the carbon footprint of concrete, the material's life span, strength, and post demolition time should be considered as they directly impact the amount of CO_2 uptake that can occur (Possan et al., 2016).

2.4.1.3 Allocation

ISO 14040 defines allocation as "partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems" (ISO 14040, 2006). For cement and concrete LCAs, careful consideration must be made for the allocation of industrial waste SCMs such as GGBS within the cement sector. ISO 14044 specifies that allocation should be avoided when possible by either dividing unit processes into sub-processes or by expanding the product system through a process known as system expansion (ISO 14044, 2006). System expansion is an approach that identifies the product displaced by a by-product and accounts for this by subtracting the environmental burdens associated with the production of that displaced product. In the context of cement and concrete LCAs, industrial by-products such as GGBS would displace Portland cement clinker. Since iron is the primary product of the blast furnace process however, system expansion is typically applied by the steel sector where the avoided impacts from using GGBS are credited (World Steel Association, 2014). Cement LCAs therefore often treat GGBS as a low or zero-burden material to avoid double counting avoided emissions if system expansion were applied again (Habert, 2013). While this cut-off approach is simple and does not attribute upstream impacts to the by-product, it may underrepresent emissions that should be accounted for in cement LCAs. Historically, blast furnace slag was classified as a waste product despite the material's use as an effective SCM. As a result of the material's growing market, in 2007 the UK's Environmental Agency's Waste Protocols Projects declared that blast furnace slag must be treated as a by-product as opposed to a waste product (Robinson et al., 2007). This decision was further solidified in the 2008 European Union Directive (EU, 2008). Since the blast furnace slag is no longer considered a waste product, the environmental impacts associated with the material's production must be considered when conducting a LCA. Since allocation cannot be avoided, ISO14044 specifies that a system's outputs should be partitioned either based on physical relationships such as mass or based on non-physical relationships such as economic value.

2.4.2 Inventory Analysis

The second step in conducting a LCA involves collecting data of relevant processes to create a life cycle inventory (LCI). The inventory analysis step of the LCA evaluates the flow of energy, emissions, and matter across a defined system boundary (Jolliet et al., 2015). The process based methodology, based on the physical flows present within a system boundary, is the most commonly used inventory analysis methodology. To evaluate the LCI for a given product, the flows and processes must first be identified. As previously discussed, the functional unit of a LCA is a reference that is used to evaluate the environmental impacts of a given product in terms of its defined function (ISO

14044, 2006). A functional unit is defined by a number of reference flows. ISO 14040 defines reference flows as a "measure of outputs in a product system required to fulfil the function expressed by the functional unit" (ISO 14040, 2006). Environmental impacts associated with each reference flow are calculated through a series of inputs and outputs in a defined system. Unit processes define the basis of the analysis as the smallest element in which inputs and outputs can be quantified (ISO 14040, 2006). In a unit process, elementary flows serve as a connection between the process and the environment. Input elementary flows, such as material extraction and energy use, enter the unit process while output elementary flows, such as generated emissions, leave it (Jolliet et al., 2015). Unit processes that are used in the production of another unit process are labelled as intermediary flows, and any finished products created by the unit process leave the system as product flows (Jolliet et al., 2015). Figure 16 presents a process flow diagram for the Portland cement manufacturing process. This diagram clearly presents each unit process in the cement manufacturing process (i.e. rotary kiln and preheater) along with the elementary flows and intermediary flows which connect each unit process.

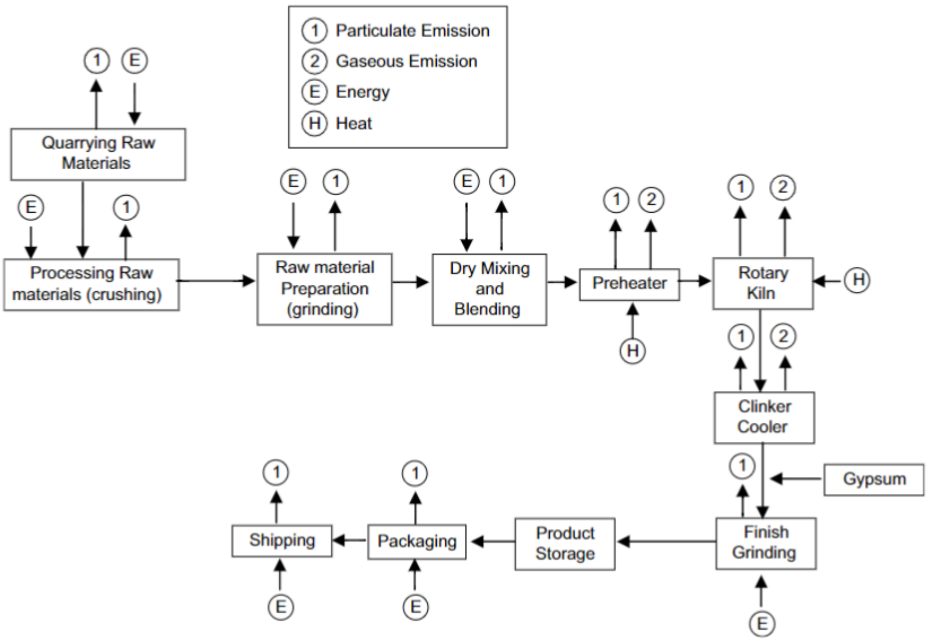


Figure 16: Portland cement process flow diagram (Huntzinger and Eatmon, 2009)

When all of the flows and processes are mapped out and identified, the LCI can now be calculated. Given the selected functional unit, each of the identified intermediary and reference flows must be multiplied by the direct emission and extraction factors for every unit process. These values for each of these unit processes are then summed together to determine the total aggregated emissions and extractions for the entire process (Jolliet et al., 2015). The direct emissions (i.e. CO_2 emissions) and extraction factors (i.e. energy and water consumption) can be found by either in LCA databases, through industry connections, or by conducting experimental work. One of the most widely used LCA databases is Ecoinvent. This data base has a collection of inventory data ranging from energy sources and chemicals to transportation with the ability to sort through the data by geographic location (Wernet et al., 2016).

The quality of an LCA study is dependent on the quality of the data utilised. As noted in ISO 14044, several data quality requirements for the LCI data must be met and specified. These requirements include but are not limited to: selecting data that is temporally and geographically relevant, technologically representative, complete and precise, reproducible, and transparent (ISO 14044, 2006). While this is the case, the quality of some data in popular databases have been questioned. One example relating to the cement and concrete sector is sodium silicate. This common alkali activator used in alkali activated cements is one of these materials in the Ecoinvent database that may result in inaccurate LCA results as this material's data has not been updated since the mid-1990s. In recent years, improvements in sodium silicate production has resulted reduced environmental impacts. These reductions would not be accounted for if the current Ecoinvent dataset is used (McGuire et al., 2011). Another example is how Ecoinvent treats blast furnace slag. Despite the increased use of either mass or economic allocation procedures to assign environmental impacts to the product, the Ecoinvent database still treats this material as recycled cut-off content (Wernet et al., 2016). As a result, using this data significantly underestimates the environmental impacts of using GGBS as an SCM.

2.4.3 Impact Assessment

Conducting a life cycle impact assessment (LCIA) is the third step in the LCA methodology. To create a LCIA, impact categories, category indicators, and characterisation models must first be selected. An impact category is a representation of an environmental issue such as climate change, acidification, or photochemical oxidation (ISO 14040, 2006). It should be noted that these impact categories are typically regarded as midpoint categories as additional environmental impacts can be caused by these observed impacts. As an example, while climate change causes ozone depletion, it can lead to additional environmental impacts such as skin cancer in humans, land degradation, and loss of biodiversity (Jolliet et al., 2015). These additional impacts are known as endpoint impact categories. Compared to midpoint categories, there is typically less certainty in terms of the calculated environmental impacts (Bulle et al., 2019). The category indicator is defined in ISO 14040 as a "quantifiable representation of an impact category" (ISO 14040, 2006). As an example, if climate change is being used as the impact category for an LCA model, its category indicator would be infrared radiative forcing. Characterisation models are used to determine the environmental impacts caused by a category indicator. These models derive characterisation factors which can then be used to convert LCI data into a common unit that represents a category indicator (ISO 14040, 2006). In the case of climate change as the impact category, the characterisation model originates from the Intergovernmental Panel on Climate Change (IPCC) where the characterisation factor is global warming potential (GWP) over a specified time period (typically 100 years) (IPCC, 2022).

Once the impact categories, category indicators, and characterisation factors have been selected, the LCI data is assigned and classified to each impact category. For climate change, GWP classifies several airborne emissions including carbon dioxide, carbon tetrafluoride, methane, and nitrous oxides. For acidification, emissions such as nitrogen oxides, sulphur dioxide, hydrogen chloride, hydrogen fluoride, and hydrogen sulphide would be classified under acidification potential (Brimachombe and Parsons, 2022). The final required step in the LCIA process is characterisation which is the calculation of category indicator results (ISO 14040, 2006). This calculation weighs each emission in the characterisation factor based on their environmental impact in that particular impact category (Jolliet et al., 2015). This calculation results in an equivalency unit value such as carbon dioxide equivalent per kilogram (kg CO_2 eq.) in the case of GWP.

Alongside the required elements needed to conduct an LCIA, ISO 14040 specifies three additional steps that can be taken in an LCIA to produce more accurate results. The first optional element is normalisation. Normalisation a calculation that illustrates the magnitude of an environmental impact of a given functional unit relative to total environmental impact on a global or regional scale. The second optional element, grouping, allows for different impact categories to be classified and ranked based on several factors including emission type and scale (Jolliet et al., 2015). The last optional element weighting. Weighting LCIA results creates a weighted environmental impact using a weighing factors based on either a monetary basis or surveys (Jolliet et al., 2015).

2.4.4 Interpretation

Interpretation, while often listed as the last step in conducting an LCA, should occur at every step of the LCA methodology. ISO 14044 specifies three interpretation checks that should be conducted. The first of these checks is a completeness check which verifies the data collected is relevant, accurate, and of high quality. The second check is a sensitivity check which is used to test the accuracy of a LCA's result by altering assumptions made during the LCA scope step. By changing the assumed functional unit, LCI approach, or allocation procedures used it can be determined how sensitive the results are to change while also producing more robust results. The last check is a consistency check. This check verifies that the LCA results align with the defined goal and scope and any issues present can be identified and corrected (ISO 14044, 2006). Once these checks have all been completed, conclusions can be drawn and recommendations can be made.

3. DECARBONISATION BASICS: Application of Standardised LCA Methodology

To demonstrate the application of the LCA methodology, a comparative concrete LCA case study was conducted to evaluate the impact of utilising GGBS as a partial clinker replacement; a key decarbonisation strategy in the cement and concrete sector. For the purposes of this case study, ISO standards 14040 and 14044 were followed and rules outlined in PCR 2019:14 were drawn upon when appropriate. This decision was based on findings from both Chapter 2 and Chapter 4 which found that most academic LCA studies typically follow the LCA framework outlined in ISO 14040 and ISO 14044 rather than EPD standards (e.g., EN15804 and ISO 21930). This case study is divided into four main subsections: (3.1) goal and scope definition, (3.2) inventory analysis, (3.3) impact assessment, and (3.4) interpretation. The aim of this non-empirical study is to present the LCA knowledge I acquired throughout the duration of the PhD in addition to highlighting limitations of using commercial LCA software.

3.1 Goal and Scope

The goal of this external LCA comparative case study is to assess the environmental impacts associated with the production of two different concrete mix designs in the UK in 2024. To serve as a baseline, the first mix design considered is concrete produced with traditional Portland cement (CEMI). The second mix design assessed is concrete produced with blended cement consisting of 70% Portland cement and 30% GGBS (CEMII). This substitution percentage was selected due to its faster setting time and early strength gain when compared to higher percentages of substitution (Higgins, 2009).

The main audience of this study consists of two groups. The first group is academics who may use this information to further evaluate the sustainability of sustainable cement alternatives. The second group is those in the cement and concrete sector who have an interest in producing sustainable cements. As part of the analysis, both local and imported GGBS were considered. A functional unit of $1m^3$ of concrete (C30/C37) was selected for this study. This volumetric functional unit was selected to ensure functional equivalency between the 100% PC cement and the blended cement. The concrete strength class C30/C37 was selected as a value representative of a conventional concrete class commonly used in the design of reinforced concrete beams and columns (Kars et al., 2024). Using values from literature, average concrete mix designs with compressive strength values ranging between 35-45MPa were selected for each scenario. These values are reported in Table 5.

Table 5: Concrete mix designs for $1m^3$ of concrete (C30/C37) with a compressive strength between 35-45 MPa (Unit: kg/m^3)

Cement Type	Binder	Water	PC	GGBS	Course	Fines	Superplasticiser
100% PC*	355	195	355	0	946	813	0
70% PC, 30% GGBS**	350	175	245	105	998	735	3.5

* (Harrison et al., 2012)

**(Karri et al., 2015, Oner and Akyuz, 2007, Rathnarajan et al., 2017, Younsi et al., 2013, Wu et al., 2001, Dhanya et al., 2018)

A 'cradle-to-gate' system boundary was selected as illustrated in Figure 17. This boundary includes the extraction of raw materials required for cement and concrete production (e.g., limestone, GGBS, gypsum, course and fine aggregates), transportation of these raw materials, and energy required for extraction and production processes (e.g., coal, oil, and electricity).

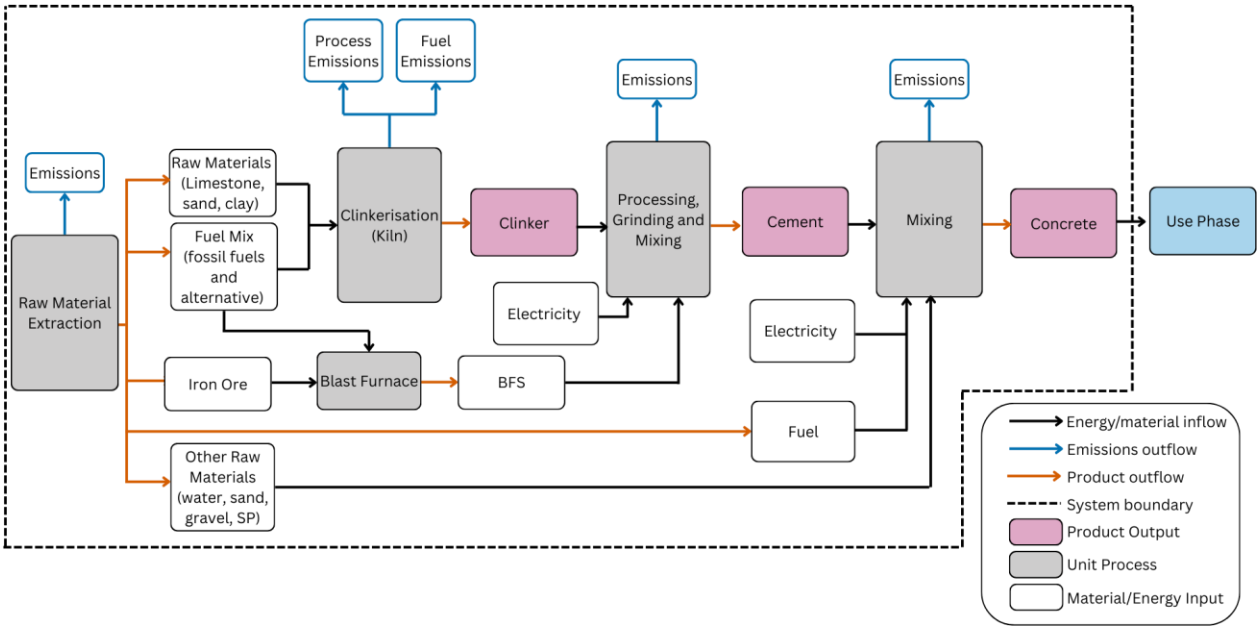


Figure 17: Cradle-to-gate system boundary diagram for concrete production

SimaPro v9.6 was selected to carry out the impact assessment method, and Environmental Footprint 3.1 adapted method was chosen to evaluate several midpoint indicators. This includes acidification, climate change, ecotoxicity (freshwater), particulate matter, eutrophication (freshwater, marine, and terrestrial), human toxicity (cancer and non-cancer), ionising radiation, ozone depletion, photochemical ozone formation, and resource use (water, fossils, and land). As specified in ISO 14044, several data quality requirements for the LCI data must be met and specified. Each of these requirements and how they are considered in the scope of this LCA are noted in Table 6.

Table 6: Data quality check as specified in ISO14044

Data Quality Requirement	**Data Quality Check**
Geographical coverage	As in line with the scope of this study, all unit processes will consider the UK specifically. Where applicable, datasets were regionalised to the UK by utilising UK electricity and energy mixes. If UK data was not available, European data was used. Global data was used solely for processes in which UK or European was not available.
Technology representativeness	The most current and up to date cement production manufacturing processes was considered for this report. This includes selecting data from an energy efficient dry process manufacturing plant which is most commonly found in the UK.

Time related coverage	To ensure current and up to date information regarding the production processes, all EPDs used in this study has been published within the past five years from the publication of this case study. Despite using an Ecoinvent database version that was updated in 2024 (V3.10), several unit processes including cement making, gravel, and sand are out of date according to their respective process descriptions.
Multiple output allocation	During the concrete production process, no by-products are produced. As a result, the flow of all raw materials, emissions, and energy are associated with the production of concrete. Allocation however has to be considered for GGBS which is a by-product of iron production. For the purposes of this case study, economic allocation was considered as specified by the data source selected for GGBS.

3.2 Inventory Analysis

This case study utilised literature data and data available in Ecoinvent 3.10. The cut-off by classification model was selected to provide simplified and transparent results by excluding environmental credits associated with recycling and substitution. Specific data for the main Portland cement production process was taken from the Lafarge Portland Cement (CEM 1, 52.5N) EPD. Data from this EPD was collected at the Cauldon Cement Plant in Staffordshire. Table 7 presents the input data provided in this EPD and the Ecoinvent processes selected for each assembly to create the process 'Portland Cement {UK} Cementmaking'.

Table 7: Product components for Lafarge Portland cement (Lafarge, 2021)

Material	Weight (kg)	Process
Clinker	886	Clinker {UK} Production | Cut-off, U
Gypsum	66	Gypsum, mineral {UK} | gypsum quarry operation| Cut-off, U
Ground Limestone	44	Limestone, crushed, for mill {UK} | production | Cut-off, U
Ferrous Sulphate	3.66	Steel, low-alloyed {GLO} | market for | Cut-off, U
Additives	0.52	Ethylene Glycol {GLO} | market for | Cut-off, U

The clinker dataset was regionalised from 'Europe without Switzerland' to the UK using regionalised data for gypsum, ground limestone, and UK electricity. The gypsum and ground limestone datasets were regionalised from Switzerland to the UK by replacing the Swiss electricity and heat inputs with UK specific datasets from the Ecoinvent database. Water inputs and outputs, in addition to waste disposal, was also regionalised to the UK where applicable. Global market processes were selected for ferrous sulphate and additives due to the uncertainty of where these products are manufactured. This is assumed to be a conservative estimate given the contribution of these two inputs is also relatively small in relation to the other materials required for cement production. The processes for these inputs were selected from the ‘Cement, Portland {CH}| production | Cut-off, U’ process found in Ecoinvent.

Since Ecoinvent does not assign any environmental burden to blast furnace slag under the cut-off method, a proxy inventory dataset was created for GGBS. This dataset utilises EPD data provided by Heidelberg Materials UK which applied economic allocation to determine the material’s environmental impact. While this method is unconventional in LCA studies, using representative flows allows impacts to be modelled that otherwise would not be considered due to data availability. The conversion from impact category to process flow is presented in Table 8, where each impact category represents a representative emissions outflow. Each impact category was divided by regionalised characterisation factors outlined in the Environmental Footprint 3.1 method (European Commission, 2021, Schindler et al., 2025).

Table 8: Proxy emission inventory data based on impact indicator values from Heidelberg Materials UK EPD (Heidelberg Materials, 2025a)

Impact category	Impact Value	Unit	Characterisation Factor (CF)	Emissions Output = Impact Value/CF	Unit
GWP-fossil	0.155	kg CO_2-eq.	1.0	**0.155**	kg CO_2 (fossil)
GWP-biogenic	2.71×10^{-4}	kg CO_2-eq.	1.0	**2.71×10^{-4}**	kg CO_2 (biogenic)
GWP-luluc	5.36×10^{-5}	kg CO_2-eq.	1.0	**5.36×10^{-5}**	kg CO_2 (LULUC)
ODP	$2.69{\times}10^{-9}$	Kg CFC-11 Eq	1.0	**$2.69{\times}10^{-9}$**	kg CFC-11
AP	1.83×10^{-3}	mol H+-Eq	1.187	**1.542×10^{-3}**	kg SO_2
EP-freshwater	$2.88{\times}10^{-5}$	kg P-Eq	1.0	**$2.88{\times}10^{-5}$**	kg P
EP-marine	4.29×10^{-4}	kg N-Eq	1.0	**4.29×10^{-4}**	kg N
EP-terrestrial	4.73×10^{-3}	mol N-Eq	2.318	**2.040×10^{-3}**	kg NH_3
POCP	1.36×10^{-3}	kg NMCOV-Eq	1.0	**1.36×10^{-3}**	kg NMVOC
ADPE	$1.24{\times}10^{-7}$	kg Sb-Eq	1.0	**$1.24{\times}10^{-7}$**	Kg Sb

GWP: Global warming potential, **ODP**: Depletion potential of the stratospheric ozone layer, **AP**: Acidification potential, **EP**: Eutrophication potential, **POCP**: Photochemical Ozone Creation Potential **ADPE**: Abiotic depletion potential

Since the specific country of origin for blast furnace slag (BFS) was not specified, it was assumed that the majority of BFS by weight was produced in the UK. Alongside BFS produced in the UK, a sensitivity analysis was conducted on imported BFS from the Netherlands and Japan; two of the largest BFS exporters. It was also assumed that the grinding and processing of exported BFS occurs at the Heidelberg plant and that the carbon intensity of BFS is the same in the UK as it is in the Netherlands and Japan.

As noted in the Lafarge Portland Cement EPD, the production site considered for this study is the Cauldon cement plant located in Staffordshire. It was assumed that the mixing of blended cement and concrete occurs at this same plant. For GGBS, the production site specified in the Heidelberg Materials EPD is located at Port Talbot in Wales. It was assumed that all imported BFS enters Port Talbot and the transportation distance between the port and the GGBS plant is negligible. Utilising Google Maps, the distance between Port Talbot and the Cauldon cement plant was estimated to be around 300km. For GGBS imported from the Netherlands, the largest European seaport, the Port of Rotterdam, was selected. The Port of Osaka was selected for GGBS imported from Japan as this port specialises in construction material cargo (ceicdata.com, 2023). The distances between these two ports were calculated using an online sea route and distance calculator (sea-distances, 2023). For imported BFS, a 150 km lorry transportation distance from the iron works to the country's port was assumed. Table 9 summarises the land and sea transportation distances for the UK and imported GGBS in addition to the processes selected for each mode of transportation.

Table 9: Land and sea carbon intensity and transportation distances for UK and imported GGBS

Transportation Type	**Process**	**GGBS Transport Distance**		
		United Kingdom	Netherlands	Japan
Land	Transport, freight, lorry > 32 metric ton, EURO5 {RER}| transport, freight, lorry > 32 metric ton, EURO5| Cut-off, U	300km	450km	450km
Sea	Transport, freight, sea, bulk carrier for dry goods {RER}| transport, freight, sea, bulk carrier for dry goods| Cut-off, U	0km	1060km	19900km

Table 10 notes the materials and processes selected for the concrete production process. Mass input values for each material were extracted from literature as noted in Table 7. The electricity process in the gravel and sand datasets were regionalised from Switzerland to the UK. For the blended cement scenario, GGBS was also added as a material input using carbon intensity value noted in the Heidelberg Materials EPD.

Table 10: Material and process inputs for UK concrete production

Materials	Process
Gravel	Gravel, crushed {UK} market for gravel, crushed | Cut-off, U
Sand	Sand {UK} | market for sand | Cut-off, U
Plasticiser	Plasticizer, for concrete, based on sulfonated melamine formaldehyde {GLO} | market for plasticizer, for concrete, based on sulfonated melamine formaldehyde | Cut-off, U
Water	Tap water {Europe without Switzerland}, market for tap water | Cut-off, U
Cement	Portland Cement {UK} Cementmaking (see Table 7)

The energy and electricity input values were extracted from 'Concrete, 37Mpa {CH} concrete production, 37MPa, for civil engineering with cement, Portland | Cut-off, U' and are noted in Table 11. The processes were regionalised to the UK where possible.

Table 11: Energy inputs and process inputs for UK concrete production

Energy	Input	Process
Fuel	0.5 MJ	Diesel, burned in building machine {UK} |market for diesel, burned in building machine| Cut-off, U
Electricity	4.9 kWh	Electricity, medium voltage {UK} | market for electricity, medium voltage| Cut-off, U
Natural Gas	8.8 MJ	Heat, district or industrial, natural gas {UK} | heat and power co-generation, natural gas, conventional power plant, 100MW electrical | Cut-off, U
Heat other than natural gas	9.5 MJ	Heat, district or industrial, other than natural gas {Europe without Switzerland}| heat production, light fuel oil, at industrial furnace, 1MW| Cut-off, U

3.3 Impact Assessment

The GWP for 1m^3 of concrete (C30/C37) made with 100% PC (CEMI) and 70% PC and 30% GGBS (CEMII) was calculated using Environmental Footprint 3.1 method which applies 100-year characterisation factors based on IPCC AR6. Figures 17 and 18 illustrate these results in the form of Sankey diagrams for CEMI and CEMII, respectively. Each flow arrow represents the amount of greenhouse gas emissions generated by a unit process. The thickness of each flow arrow is proportional to the magnitude of the emissions it represents. The white boxes indicate that foreground data were utilised (e.g., data from the Lafarge EPD), while green boxes represent processes that rely on background data from the Ecoinvent database. Table 12 outlines the environmental impacts for several impact categories for CEMI and CEMII using the Environmental Footprint 3.1 method. By dividing the impact results by the respective normalisation factors from the Environmental Footprint 3.1 method in SimaPro, the impacts are expressed in dimensionless normalised units, as shown in Figure 19. This expresses the environmental impact of concrete production relative to the total annual environmental impacts in the EU and enables comparison across different impact categories using a common unit (European Commission JRC, 2025).

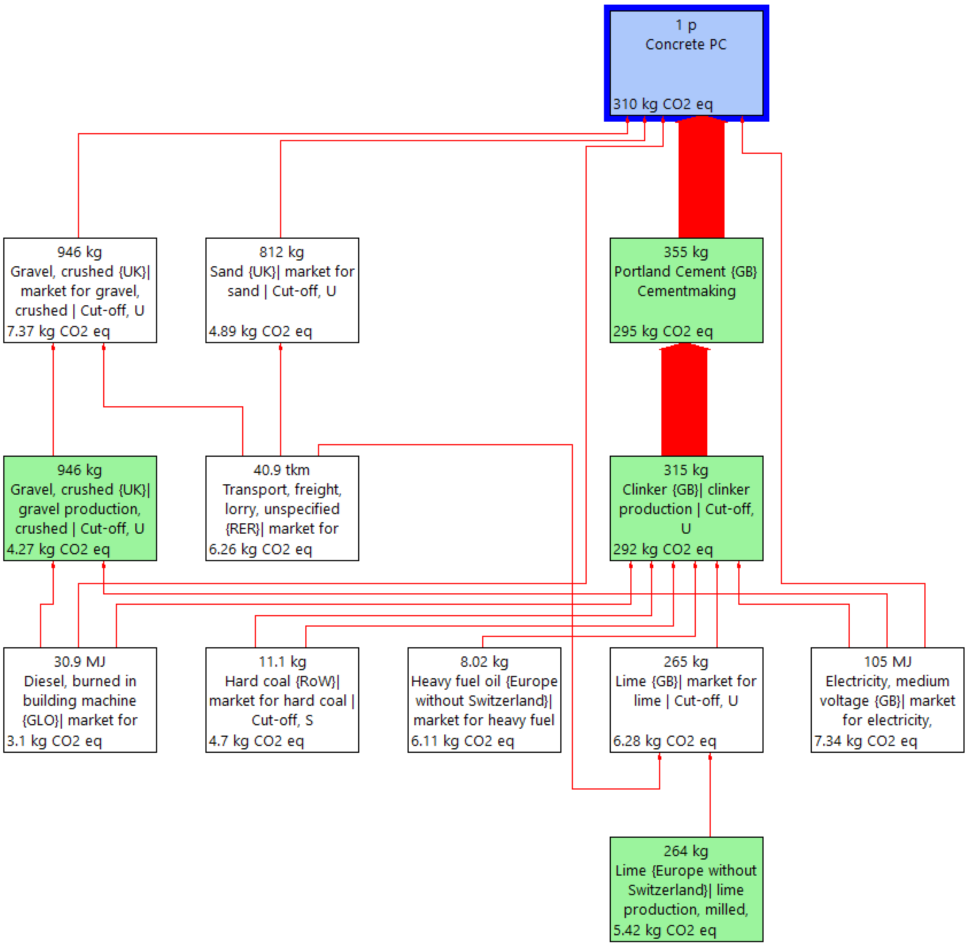


Figure 18: Sankey diagram for 1m^3 of concrete (C30/C37) made with 100% PC using Environmental Footprint 3.1 method characterised to climate change; Cut-off 1%

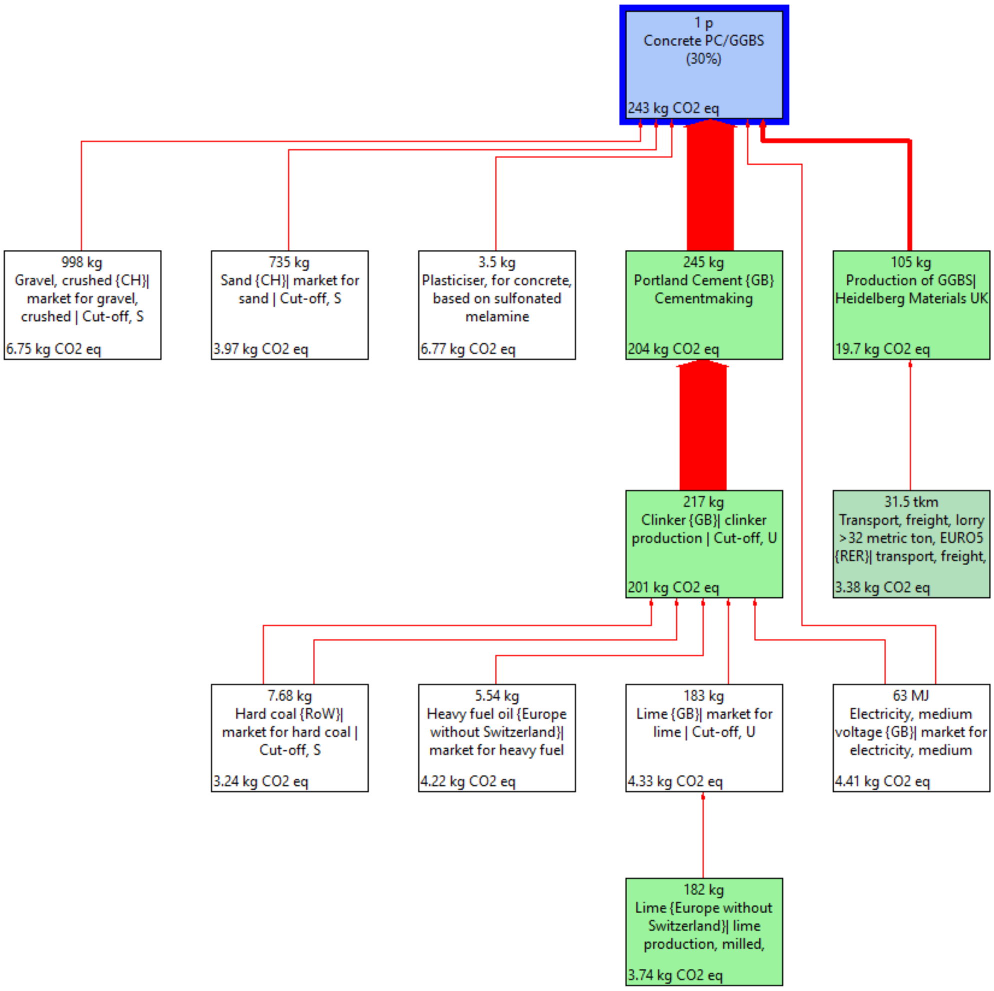


Figure 19: Sankey Diagram for 1m^3 of concrete (C30/C37) made with 70% PC and 30% GGBS using Environmental Footprint 3.1 method characterised to climate change; Cut-off 1%

Table 12: Environmental impact assessment results for 1m^3 of concrete (C30/C37) using Environmental Footprint 3.1 method

Impact category	Concrete (100% PC)	Concrete (70% PC, 30% GGBS)	Unit
Acidification	0.683	1.27	mol H+ eq
Climate change	310	243	kg CO_2 eq
Ecotoxicity, freshwater	270	274	CTUe
Particulate matter	5.07E-06	1.05E-05	disease inc.
Eutrophication, marine	0.203	0.176	kg N eq
Eutrophication, freshwater	0.0188	0.0187	kg P eq
Eutrophication, terrestrial	2.34	2.3	mol N eq
Human toxicity, cancer	6.06E-07	5.46E-07	CTUh
Human toxicity, non-cancer	1.78E-06	1.42E-06	CTUh
Ionising radiation	9.93	7.75E+00	kBq U-235 eq
Land use	469	456	Pt
Ozone depletion	1.44E-06	1.33E-06	kg CFC11 eq
Photochemical ozone formation	0.661	6.81E-01	kg NMVOC eq
Resource use, fossils	1.29E+03	1.11E+03	MJ
Water use	32.6	2.64E+01	m3 depriv.

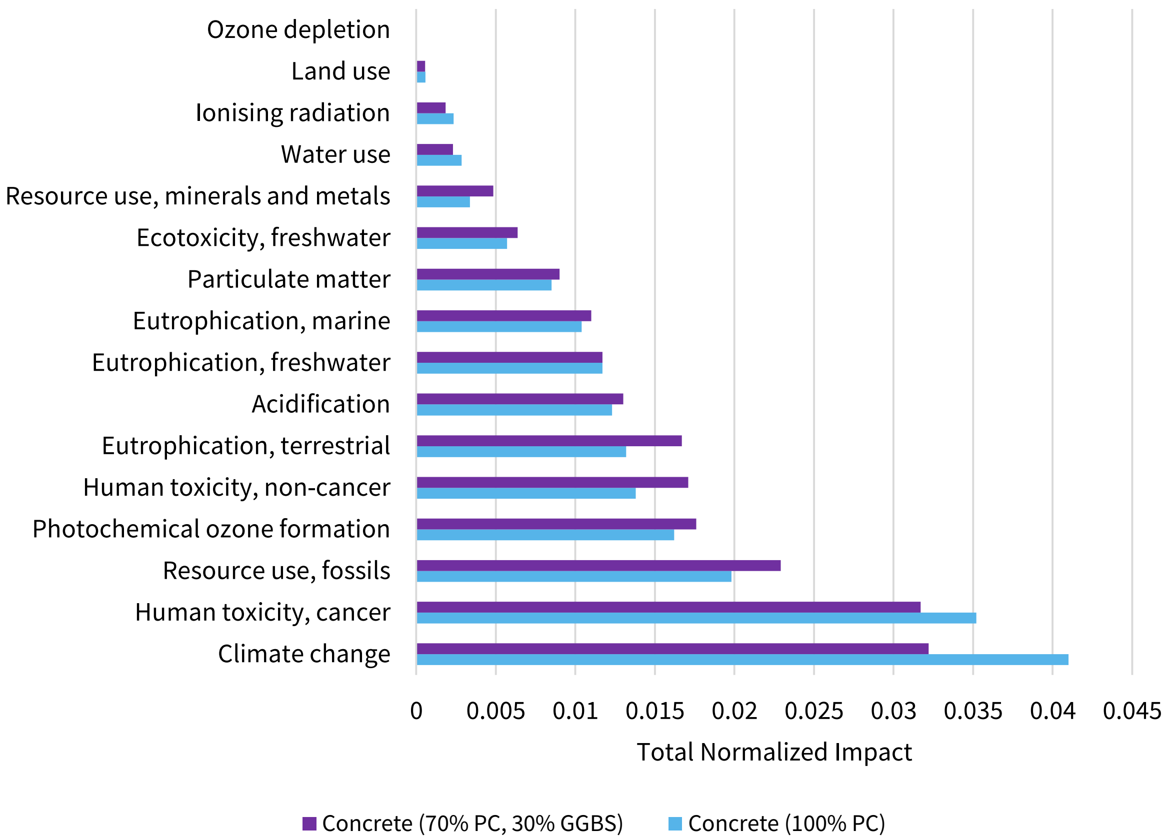


Figure 20: Normalised impacts for 1m^3 of concrete (C30/C37) using Environmental Footprint 3.1

3.4 Interpretation

Examining the raw impacts, it was found that the production of 1 m^3 of concrete (C30/C37) with Portland cement produces 310 kg CO_2eq/kg, with roughly 95% of this impact originating from clinker production. This high impact is due to two main contributing factors. The first is the amount of fossil fuels required to heat the rotary kiln at a high temperature (1450°C). The second factor is due to the chemical reaction limestone undergoes in the rotary kiln which releases CO_2. For these reasons, it is crucial to implement decarbonisation solutions that both optimise the clinkerisation process and reduce the sector's dependency on clinker as a material through the use of alternative cementitious materials. When comparing traditional Portland cement to the blended cement scenario which utilises 30% less clinker, a roughly 22% reduction in the climate change impact category is seen.

For the remaining categories, the 70% PC–30% GGBS concrete showed higher impacts than Portland cement concrete in four categories: particulate matter (107% increase), acidification (86% increase), photochemical ozone formation (3% increase), and freshwater ecotoxicity (1.5% increase). The higher particulate matter impact arises from the energy-intensive grinding during GGBS processing, while the increased acidification results from emissions originating from the blast furnace process. As shown in Figure 20, several impact categories become more significant when normalised against EU reference values. For both scenarios, the climate change impact category represents the most significant environmental impact. Human toxicity (cancer) showed the next highest impact across both concrete mix designs. This results primarily from combustion emissions that release heavy metals such as arsenic, cadmium, and chromium. Additionally, quarrying and crushing of limestone and other raw materials generate silica dust, which further contributes to both cancer and non-cancer human toxicity (e.g., respiratory diseases) (Raffetti et al., 2019). Under normalisation, nine impact categories were found to show higher impacts for the 70% PC–30% GGBS concrete compared to Portland cement concrete, with terrestrial eutrophication, non-cancer human toxicity, fossil fuel resource use, minerals and metals resource use, and photochemical ozone formation having the greatest increase. As previously discussed, large amount of fossil fuels such as coal and fuel oil are consumed during clinker calcination, while the coke required for the blast furnace process contributes further to fossil fuel use. This additional fuel needed for BFS production and processing leads to concrete containing blended GGBS cement having higher impacts in resource use and photochemical ozone formation compared to traditional concrete mixes with Portland cement. Fuel combustion during iron production also generates significant nitrogen oxide emissions, which in turn lead to a higher normalised eutrophication impact (Backes et al., 2021).

With the UK's reliance on imported GGBS to meet market demand, it is crucial to consider the impacts of transportation. Figure 21 notes the climate change impact for three different GGBS transportation scenarios. The first considers if the GGBS was produced in the UK, the second considers if the material was imported from the Netherlands, and the third evaluates if the material was imported from Japan. From the figure it can be concluded that while importing GGBS from the Netherlands does not significantly increase the environmental impacts, importing the material from Japan does increase these impacts by roughly 7%. The climate change impact of importing GGBS from Japan for blended cement was found to be equal to 260 $kgCO_2eq/kg$. Comparing this value to traditional Portland cement production which was found to have a climate change impact of 310 $kgCO_2eq/kg$, it can be concluded that it would be still more sustainable to utilise imported GGBS from Japan as opposed to relying solely on traditional Portland cement.

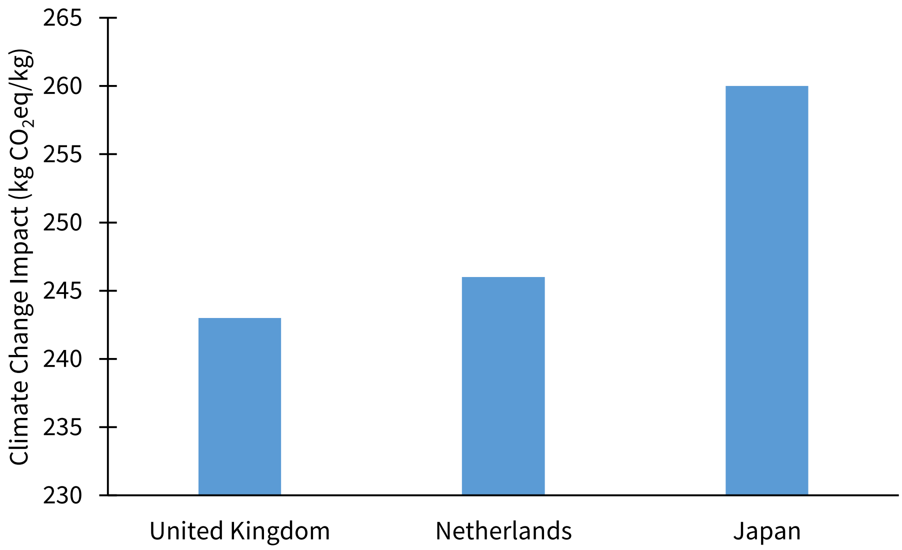


Figure 21: Climate change impact of $1m^3$ of concrete (C30/C37; 70% PC-30% GGBS) with impact of material importation included using Environmental Footprint 3.1

3.5 Conclusions and Recommendations

An LCA was conducted following the ISO 14040 and ISO 14044 standards to evaluate the environmental impacts of concrete production in the UK. A comparative analysis was conducted between two concrete mix designs. The first concrete scenario considered traditional Portland cement and the second scenario considered a blended cement with a 30% clinker substitution with GGBS. A functional unit of $1m^3$ of concrete (C30/C37) using was selected for this cradle-to-gate study. Special attention was paid to the allocation procedure for GGBS as the material is now classified as a by-product. For this reason, economic allocation was chosen as the allocation scenario. To conform to the ISO standards, data quality requirements were evaluated. For this study, data was taken from publically available EPDs and the Ecoinvent database. The impact assessment method selected for this study was the Environmental Footprint 3.1 adapted method.

The LCA results indicate that the most significant impact category is climate change due to the calcination process in the kiln during clinker production. The concrete which utilised a blended cement ratio of 70% PC-30% GGBS was found to have a 22% reduction in carbon emissions when compared to concrete produced with 100% PC. Other notable impact categories include human toxicity and terrestrial eutrophication which primarily result from fossil fuel combustion. With the UK reliant on imported GGBS for cement production, the environmental effect of imported GGBS was evaluated. Comparing the UK's two largest exporters of GGBS, the Netherlands and Japan, it was found that while importing GGBS from the Netherlands had little effect on the total environmental impact, importing the material from Japan made did increase the material's carbon footprint by 7%. While this is the case, it is still a more sustainable solution when compared to relying on concrete produced with traditional Portland cement alone. In conclusion, the use of blended GGBS cement in concrete mixes is recommended over traditional Portland cement alone as an effective way to lower the carbon footprint of concrete production. To further minimise environmental impacts, GGBS should be sourced from neighbouring countries or the UK to reduce transportation distances and associated emissions.

Overall, this case study highlights how LCA methodology can be used to evaluate the effectiveness of decarbonisation strategies, such as clinker replacement, in reducing the carbon footprint of the cement and concrete sector. However, there are several limitations present in the case study. One of these key limitations is SimaPro's black-box approach which reduces data and process transparency. This is particularly the case with the clinker production, as several stages of the production processes (e.g., mixing, grinding, pre-heating, and calcination) are aggregated into one process. Since the clinker process output does not distinguish between process emissions and combustion emissions, the ability to effectively evaluate decarbonisation strategies that focus on these sources separately is limited. Moreover, this aggregated modelling approach makes it difficult to evaluate the combined effects of multiple decarbonisation strategies. Future work should aim to model cement and concrete carbon intensity calculations using more transparent methods. Another key limitation of this case study is the SCM chosen for analysis. With the UK steel sector looking to replace traditional steelmaking with low carbon steel produced with recycled scrap steel, the availability of local GGBS will continue to decline further resulting in the cement sector's reliance on imports to meet demand. By not considering these wider cross-sector changes, this case study overlooks factors that could influence the long-term effectiveness of GGBS as an SCM to reduce sector emissions. Further work is needed to assess the decarbonisation potential of GGBS by assessing current market trends and how these changes may affect the symbiotic relationship between the cement and steel sectors.

4. DECARBONISATION FOUNDATIONS: Insights from Existing LCA Studies

The first article, published in *Renewable and Sustainable Energy Reviews*, critically analyses the current implementation of standardised LCA methodology across four energy intensive industries: cement, steel, glass, and plastics. This analysis was achieved through a systematic literature review in which 256 studies were selected and evaluated based on the LCA framework presented in ISO 14040 and ISO 14044. From this, it was found that LCA application is fragmented across all four sectors, with clear differences arising from the implementation of the methodology including how functional units are defined, how data quality is assessed, and how results should be interpreted. To strengthen the effectiveness of LCA as a critical tool used to advance decarbonisation goals, 'best practice' recommendations were presented. The methodological gaps identified in this study directly influence the transparent LCA modelling approach presented in Chapter 5, as well as the evaluation of GGBS (particularly regarding by-product allocation) on a cross-sector level in Chapter 6. I hereby certify that I led the conceptualisation of the study and led the formal analysis and data curation for the cement and concrete sector. In addition, I co-led the writing of the original draft alongside Jacob W. Whittle and took responsibility of the submission process including responding to comments left by peer-reviewers.

Life Cycle Assessment in energy-intensive industries: cement, steel, glass, plastic

Rihner, M.C.S.[a,h*], Whittle, J.W.[b,h], Gadelhaq, M.H.A.[b,h], Mohamad, S.N.[a,h], Yuan, R.[c,h], Rothman, R.H.[d,h], Fletcher, D.I.[e,h], Walkley, B.[f,h*], Koh, L.S.C.[g,h*]
* Corresponding Authors: Mcsrihner1@sheffield.ac.uk, B.Walkley@sheffield.ac.uk, and S.C.L.Koh@sheffield.ac.uk

a = Department of Chemical and Biological Engineering, University of Sheffield, UK
b = Department of Mechanical Engineering, University of Sheffield, UK
c = Department of Mechanical Engineering, University of Sheffield, UK
d = Department of Chemical and Biological Engineering, University of Sheffield, UK
e = Department of Mechanical Engineering, University of Sheffield, UK
f = Department of Chemical and Biological Engineering, University of Sheffield, UK
g = Management School, University of Sheffield, UK
h = Energy Institute, University of Sheffield, UK

Abstract

Cement, steel, glass, and plastics sectors are at the forefront of industrial decarbonisation and must make effective, evidence-based strategic choices. For the first time, this work analyzes current implementation of life cycle assessment (LCA) methodology, as a key decision lever, across the aforementioned sectors through a critical, systematic literature review of 256 studies. Results reveal differences in how LCA studies are conducted, and that implementation is fragmented. Many studies did not consider functionality when defining a functional unit, and most do not objectively assess data quality. Significant differences also exist regarding the definition of scope and selection of impact categories and interpretations. Therefore, this work provides recommendations for 'best practice' in LCA applied to global industrial sectors, aiding in the development of a consistent and transparent approach to cross-sector LCA implementation. Specifically, functional unit types must be properly defined, cross-sector allocation procedures should be intrinsically linked, a 'cradle-to-cradle' system boundary should be used where possible, and these aspects should be synergistically implemented across sectors where possible. Data quality should be assessed objectively, with greater uniformity in impact assessment methodologies, impact assessment categories and reporting methods. Sensitivity and uncertainty assessments should be completed and reported in line with International Standard Organization (ISO) 14040 and 14044, and a greater focus should be placed on future production processes and technology. This will improve LCA applications and outcomes, allow effective cross-sector comparison, and enhance decision making towards net zero.

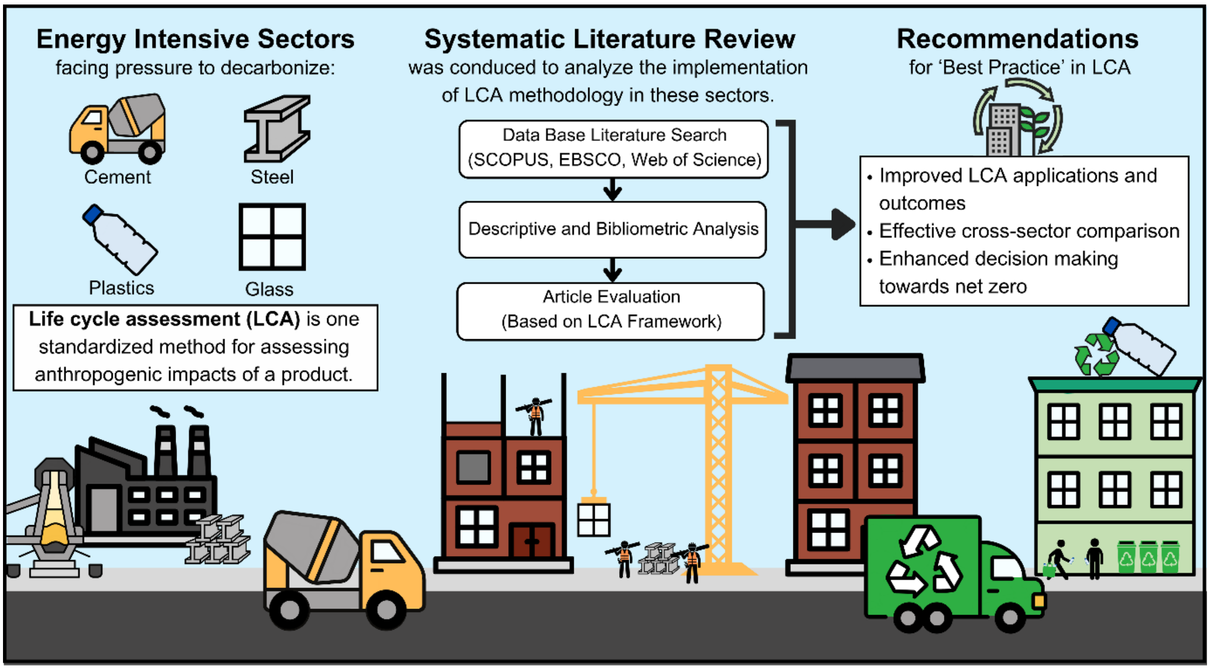
Highlights

- Fragmentation of LCA application in cement, steel, glass and plastics
- Varied functionality, scope, impact categories and data quality in LCA implementation
- Recommendations for 'best practice' LCA in energy-intensive industries
- Consistent and transparent approach required for cross-sector LCA implementation
- Role of comparable LCA for achieving net zero goal and industrial decarbonisation

Keywords

systematic literature review; life cycle assessment; industrial decarbonisation; net zero

Graphical Abstract



Abbreviations

BFS - blast furnace slag

BF-BOF – blast furnace-basic oxygen furnace

CML - Centrum voor Milieukunde Leiden

E-IO - economic input-output

GGBFS - ground granulated blast furnace slag

GWP - global warming potential

IPCC - Intergovernmental Panel on Climate Change

ISO - International Standards Organization

LCA - life cycle assessment

PCR - Product Category Rules

SCM - supplementary cementitious materials

SLR - systematic literature review

UK - United Kingdom

WOS- Web of Science

1. Introduction

Since the Industrial Revolution, the increasing demand for infrastructure and products has resulted in the formation of, and reliance on, several key global industrial sectors that have greatly contributed to the growth of modern society, including: cement, steel, chemicals, paper, and aluminum. In the past two decades, these industries have seen unprecedented growth that will only continue as a rising global population further increases consumer demand [1]. However, this growth has come at the expense of adverse environmental impacts due to the energy intensive nature of these industrial processes [2].

In 2022, it was reported that the global industrial sector accounted for nine gigatons, equivalent to 25%, of all carbon dioxide emissions [1]. Ensuring that all global greenhouse gas emissions decline by 43% by 2030 to reach net zero by 2050 has become an international priority as outlined by United Nations Sustainable Development Goal 13 [3]. A lack of incentive to implement innovative, sustainable solutions in energy intensive industries has until recently been slow due to the absence of new competition for incumbents, and the requirement for large capital expenditure. Furthermore, their position in the supply chain has until recently shielded them from the consumer pressure which drives the environmental and social performance of more publicly visible businesses. However, as a result of recent shifts in expectations of consumers and policy makers, these sectors are now facing profound pressures to reduce their carbon emissions [4]. Several approaches have been adopted to tackle these issues including addressing the need for responsible consumption, production, and innovative sustainable technologies by applying the concept of a circular economy across all industrial sectors, whilst also retrofitting existing infrastructure with cleaner, greener, and more efficient industrial processes; aligning with United Nations Sustainable Development Goals 9 and 12 [5, 6].

An example of this is seen in the United Kingdom (UK), where the UK Research & Innovation council identified six industrial sectors that contribute most to the UK's carbon footprint, with 50 million metric tons of carbon dioxide being emitted annually: cement, metals, glass, paper, ceramics, and chemicals [4]. To bring further attention to carbon emission reductions required in these economically vital industries, which together account for 2.5% of the UK's gross domestic product, these sectors were termed the UK's 'Foundation Industries' [4, 7]. Although the industries that are considered 'foundational' differ between countries, once they are identified, a greater focus can be placed on finding innovative and sustainable solutions to decarbonize while also ensuring economic stability and growth.

There are several standardized methodologies for evaluating the feasibility of decarbonisation including environmental impact assessment [8], environmental management system [9], and carbon footprint of a product [10]. Life cycle assessment (LCA), however, is the most widely used method to provide a general perspective for a given product by evaluating a wide range of environmental impacts throughout its whole life cycle [11]. LCA has also become an increasingly critical lever within industrial product development as a strategic decision making tool, where solutions to address environmental impacts can be prioritised to best optimise a system given current financial, technological, and human resources [12].

The International Standard Organization (ISO) 14040 and 14044 define the LCA framework in the four distinct stages as seen in Figure 1 [11, 13]. The first stage, 'goal and scope', defines and establishes the rules and depth of a study. This includes selecting a functional unit, system boundary, and allocation procedure for a selected product. Stage two, 'inventory analysis', encompasses data collection and data quality evaluation. The third stage, 'impact assessment', aims to calculate the environmental impacts of the defined product using environmental indicators. Stage four, 'interpretation', requires the assessor to draw conclusions based on the analysis and to carry out checks to ensure robust results. While interpretation is often listed as the last step in conducting an LCA, it should occur at every step of the LCA methodology.

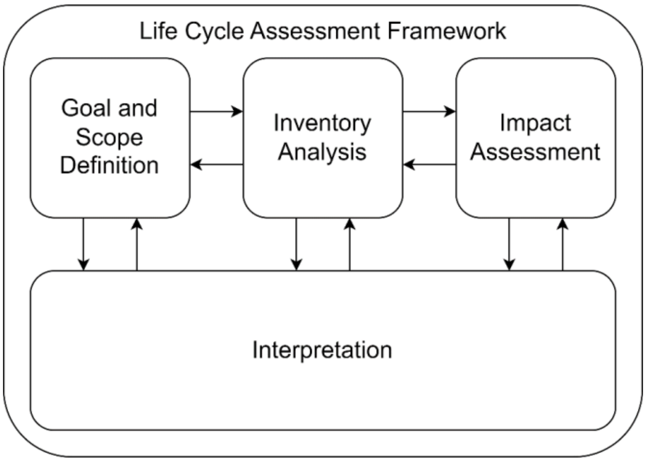


Figure 1: LCA Framework, adapted from [11]

The technological, economic, and social importance of 'foundational' sectors means that they have been extensively studied in isolation through detailed environmental impact analysis of specific processes and technologies as well as individual sector literature review studies [14-17]. However, by assessing existing peer-reviewed LCA studies across multiple sectors, current trends and practices at both a sector and cross-sector level can be identified. Within existing subject studies, this type of cross-sectoral review has not previously been completed. Given the multitude of interconnections and interdependencies between these sectors, each respective industry would benefit from understanding how their counterparts are approaching the challenge of decarbonisation. There are a number of well-established literature review methodologies including narrative, meta-analysis, and meta-synthesis [18]. In this study a systematic literature review (SLR) methodology was selected, as described by Tranfield et al. [19]. This technique adopts a repeatable, replicable, and transparent methodology which captures key studies related to a specific research question. When completed correctly, the study is able to produce reliable findings and conclusions [20].

The goal of this review is to assess and understand how LCA studies are conducted in four global industrial sectors (cement, steel, glass, plastics), analyze the findings to ascertain how standardized LCA methodology is implemented, and generate recommendations for future LCA studies across these industries. Importantly, by analyzing current implementation practices of LCA methodology across the aforementioned sectors through a critical, systematic literature review, new insight is obtained regarding wide scale cross-sector implementation in these energy intensive industries. This is the first study that applies SLR methodology to distil a vast array of research regarding LCA implementation in cement, steel, glass, and plastic sectors holistically, generates a set of overarching conclusions applicable across each of these energy-intensive sectors, and develops a set of key recommendations for 'best practice' in LCA applied to such global industrial sectors. This will improve LCA practice, allow effective cross-sector comparison, and enhance decision making towards net zero.

This study is divided into five sections as follows: section two describes the SLR methodology used to retrieve journal articles, section three reports the bibliometric analysis of the retrieved articles, section four details the findings of the review, section five summarizes the similarities and differences in application of the LCA methodology between each sector, and section six summarizes the key findings and provides recommendations for future LCA implementation.

2. Methodology

2.1. Overall Approach

Due to its prevalence, the process-based approach described by Tranfield et al. [19] and Atansovska et al. [21] was adopted as the overarching strategy for performing this SLR. The process completing this review consists of the following three stages: planning (section 2.2), conducting (sections 2.3 and 2.4), and reporting. The approach is shown by a flowchart in Figure 2.

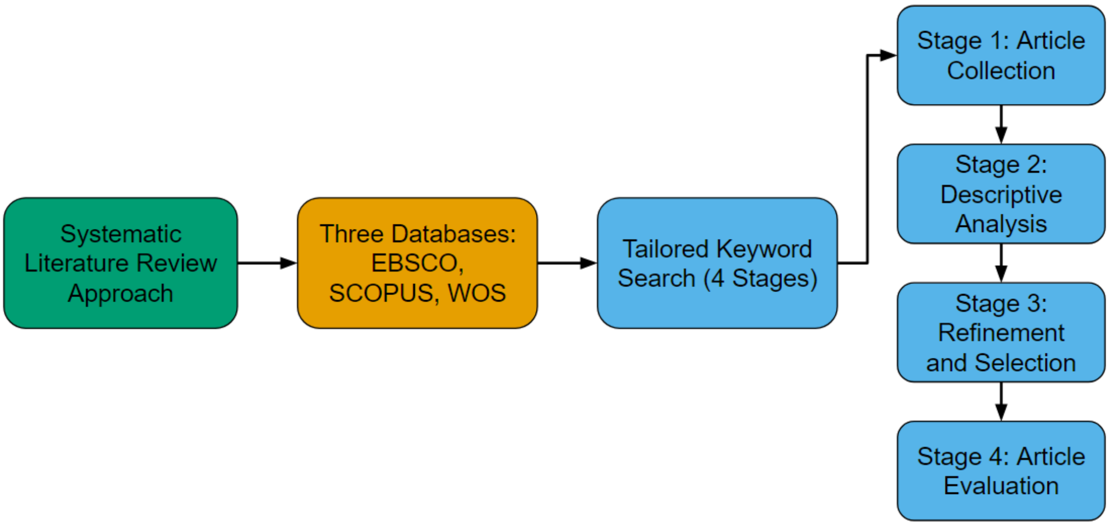


Figure 2: SLR Process Flowchart

2.2. Step 1: Article Collection

2.2.1. Review Scope

This review focused solely on english language, peer-reviewed journal articles. Modern methods of LCA were standardized by ISO 14040 in 1997 [22], but did not become commonplace until 2000 with the publication of impact assessment methodologies [23]. Therefore, to capture the best quality environmental LCA articles which utilize standard methods and terminology, only work published within the last twenty years (2003 until 2023) is reviewed. The key focus of this research is the environmental impact of production and manufacture of feedstock materials within four energy intensive sectors as discussed in section one. Therefore, studies which primarily addressed life cycle costing and social impacts were excluded unless the LCA portion was significant enough to review.

2.2.2. Keyword and Database Selection

The databases that were used to conduct the review were SCOPUS, EBSCO, and Web of Science (WOS). These are considered to be the largest databases that can provide a sufficient number of articles to conduct a systematic review [21]. Several different keyword strings were trialed but ultimately a generic string, sector specific keywords, and associated search parameters were used as shown in Table 1. Sector specific keywords were added to further specify search results in sectors that utilize specific terminology; however, due to the comprehensive array of plastic materials that are considered part of the plastic industry, no specific terminology was required.

Table 1: Keyword String, Sector Specific Terms, and Search Parameters Used

Keyword String			
("Life Cycle Assessment" OR "Life Cycle Analysis" OR "LCA" OR "Life Cycle Impact Assessment") AND (”*Sector Name* Industry” OR “*Sector Name* Production” OR “*Sector Name* Manufactur*” OR “*Sector Specific Terms*”)			
Sector Specific Terms			
Cement	**Steel**	**Glass**	**Plastics**
“Portland Cement Production” OR “Portland Cement Manufactur*”	“Iron Industry” OR “Iron Production”	“Glass Furnace” OR “Glass Melting”	*Not applicable*
Search Parameters			
Database(s)	SCOPUS, EBSCO, WOS		
Publication Year	2003 - 2023		
Article Type	Journal		
Language	English		

2.3. Step 2: Descriptive Analysis

Bibliometric analysis, a type of descriptive analysis, is defined as both a quantitative and qualitative research method which is utilised to evaluate the impact of individual researchers, research clusters, journals, countries, or institutions [21]. It is also a useful method for systematically identifying research trends within different fields of study [24]. To perform this bibliometric analysis, VOSviewer, a software package, was used to visualize and analyze keyword co-occurrence [25].

2.4. Step 3: Article Evaluation

The retrieved articles were reviewed according to review criteria adapted from Atansovska et al. [21] and Bisinella et al. [26]. For this study, the broad groups were refined into three main categories: general, sectoral, and LCA characteristics. The general characteristics section notes information relating to the study’s publication and scope. The sectoral characteristics section was included as a customizable review criteria block by which unique aspects of material production for each sector could be captured. The LCA characteristics section highlights how the LCA methodology was implemented in each study by assessing each category of the LCA framework illustrated in Figure 1. In addition, the standards followed in each study were noted as a quality proxy. This exhaustive, predefined set of review criteria for each criteria group is summarized in Table 2. As required by Tranfield et al. [19], this will keep the review as objective as possible whilst retaining maximum comparative value in the study.

Table 2: Review Criteria used Organized by Criteria Groups

<table>
<tr><th>Criteria Groups</th><th colspan="4">Review Criteria</th></tr>
<tr><td>General Characteristics</td><td colspan="2">• Author(s)
• Journal</td><td colspan="2">• Geographic Location(s)
• Year</td></tr>
<tr><td rowspan="2">Sectoral Characteristics</td><td>Cement</td><td>Steel</td><td>Glass</td><td>Plastics</td></tr>
<tr><td>• Type of Cement
• Alternative Fuel Types
• Supplementary Cementitious Material/ Precursor
• Alkali Activators</td><td>• Production Route
• Final Product</td><td>• Type of Glass
• Manufacturing
• Processes
• Glass Application
• Recyclability</td><td>• Inventory
• Type of Polymer
• Type of Plastic</td></tr>
<tr><td>General Characteristics</td><td colspan="2">• Author(s)
• Journal</td><td colspan="2">• Geographic Location(s)
• Year</td></tr>
<tr><td>Goal and Scope</td><td colspan="2">• Number of Scenarios
• Comparative LCA
• Functional Unit Type
• Functional Unit</td><td colspan="2">• System Boundary
• Substages Considered
• Allocation Procedure
• Type of LCA</td></tr>
<tr><td>Inventory Analysis</td><td colspan="2">• Data Type
• Data Quality Mention/Assessment
• Foreground Data Source</td><td colspan="2">• Background Data Source
• Data Sources
• Databases</td></tr>
<tr><td>Impact Assessment</td><td colspan="2">• LCA Software Used
• Impact Assessment Method</td><td colspan="2">• Midpoint Environmental Indicators
• Endpoint Environmental Indicators</td></tr>
<tr><td>Interpretation</td><td colspan="4">• Sensitivity Analysis Conducted
• Uncertainty Analysis Conducted
• Predictive LCA Conducted</td></tr>
<tr><td>Quality Proxy</td><td colspan="4">• Standards Followed</td></tr>
</table>

3. Descriptive Analysis Results

3.1. Articles Retrieved

The SLR process resulted in a total of 1164 articles across all sectors over the last twenty years, as shown in Table 3.

Table 3: Numerical Summary of Articles Collected following SLR Process

Database/Sector	Cement	Steel	Glass	Plastics	Total
SCOPUS	182	193	38	190	**603**
EBSCO	51	63	23	18	**155**
Web of Science	159	187	19	41	**406**
Total	**392**	**443**	**80**	**249**	**1164**

3.2. Bibliometric Analysis and Keyword Evaluation of Retrieved Articles

In line with the developed SLR process, a bibliometric analysis was performed on the retrieved articles to ensure consistency across each sector of study as well as identify key meta-insights into each sector. Following guidelines established by Van Eck et al. [25] as well as experimentation, the author keyword co-occurrence parameter was set to a minimum of three and five occurrences per keyword as shown in Figure 3 and Figure 4 respectively. The color codes indicate clusters of keywords which are utilised together (i.e., the same keywords are used in different papers). Broadly, these are split into each sector of interest but there are some common clusters which are cross-sector. These two maps denote a good alignment between the keywords returned from the SLR, and the subject areas of interest. Important keywords associated with the production of cement (concrete, Portland cement, and compression strength), steel (blast furnace, electric arc furnace, and iron and steel industry), glass (life cycle assessment, energy, and recycling), and plastics (plastics waste, bioplastics, and recycling) are seen. There is also good alignment with keywords emerging from circular economy and sustainable manufacturing practices (circular economy, sustainable innovation, waste management, and recycling). Furthermore, there are several keywords which have significant cross sector overlap including ‘life cycle assessment’, ‘environmental impact’, and ‘sustainability’.

3.3. Final Taxonomy

Following the bibliometric analysis and keyword evaluation in section 3.2, a large number of duplicate, off-topic, non-compliant, and non-accessible articles were identified and removed. For this study, off-topic papers encompassed all articles that did not perform an LCA or were not relevant to the topic of sustainability studies. Non-compliant studies include studies that:

1. Do not relate to the sector being considered.
2. Do not evaluate production of the main product of the sector.
3. Do not use LCA as the primary methodology to assess environmental impacts.

4. Are literature review studies that evaluate existing LCAs.

Non-accessible articles include studies that could not be accessed or downloaded from their respective databases. This results in 256 unique articles for analysis which were compliant with the criteria and goals of this SLR. The article identification and retrieval flow chart is illustrated in Figure 5.

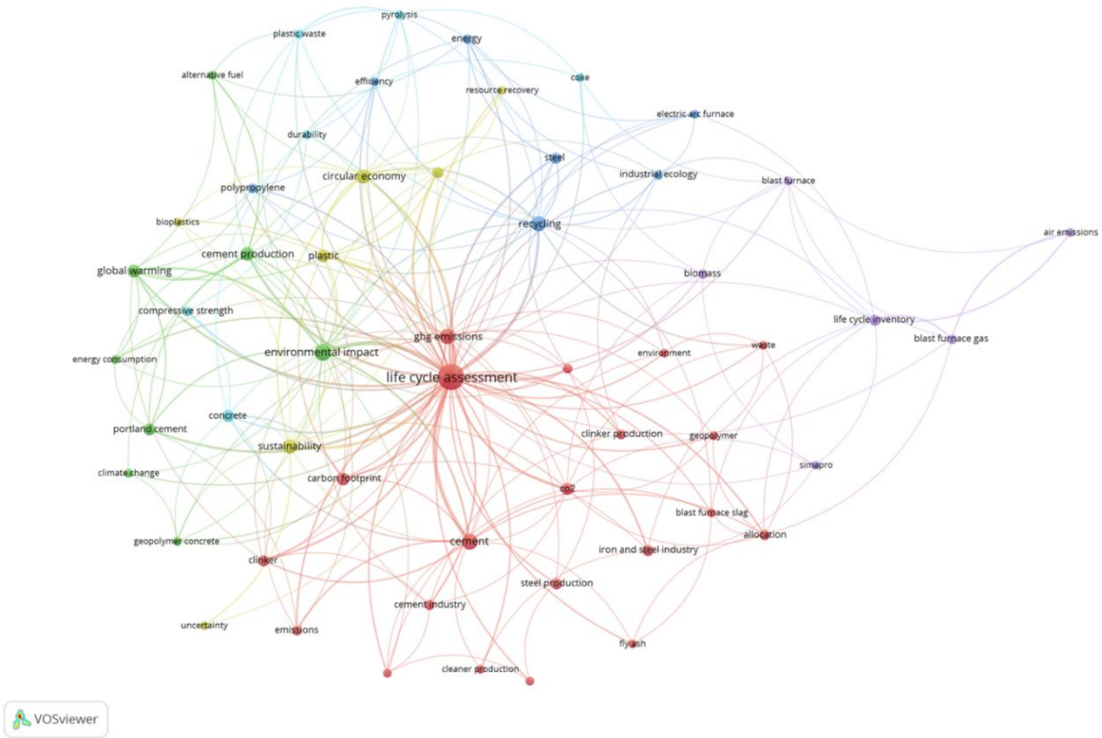


Figure 3: Author keyword co-occurrence map (minimum of 3), with cleaned keywords

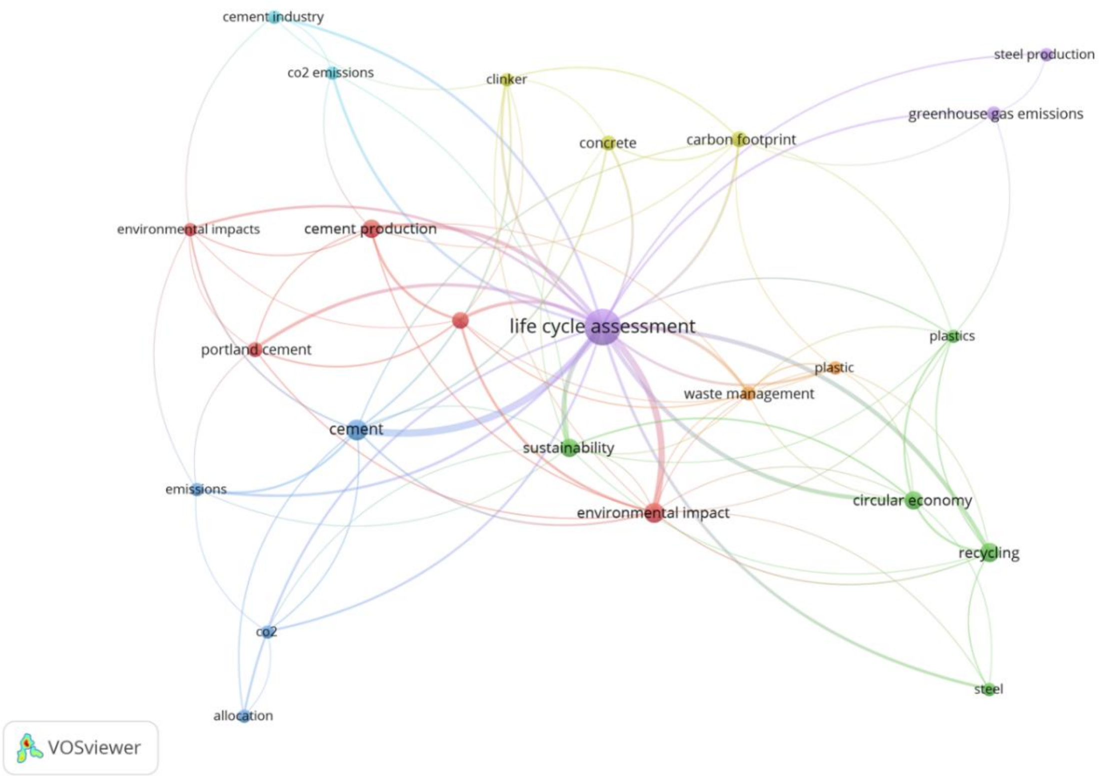


Figure 4: Author keyword co-occurrence map (minimum of 5), with cleaned keywords

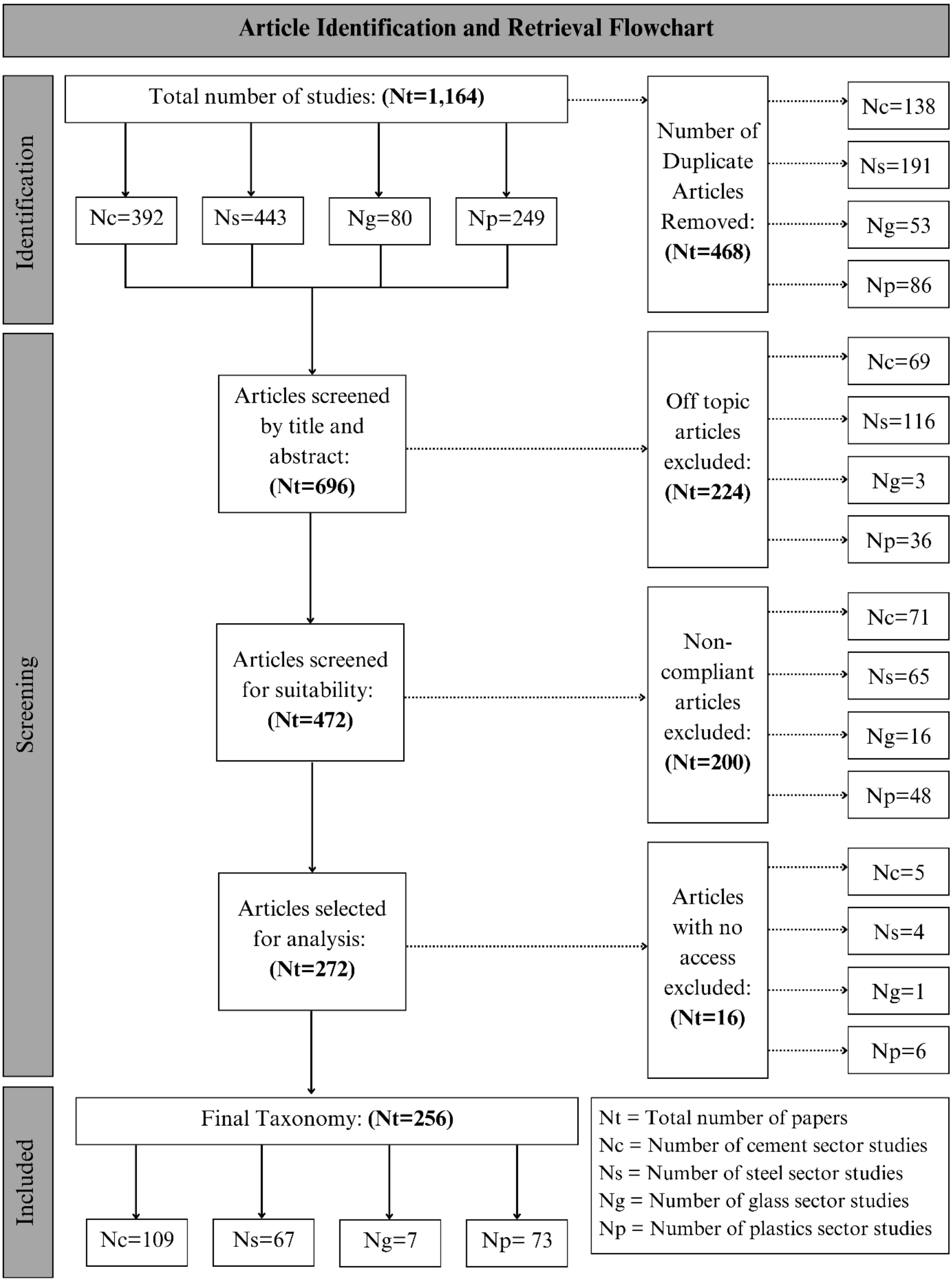


Figure 5: Numerical summary of articles collected following descriptive analysis process (dotted arrows represent number of papers removed during retrieval)

4. Results

4.1. General Characteristics of Studies

Year of Publication: As shown in Figure 6, the application of LCA methodology has been increasing since 2015. There has been a further sharp increase since 2020 which represents 46.5% of the total collected publications since 2003. This phenomenon is particularly visible in the cement and plastics sectors.

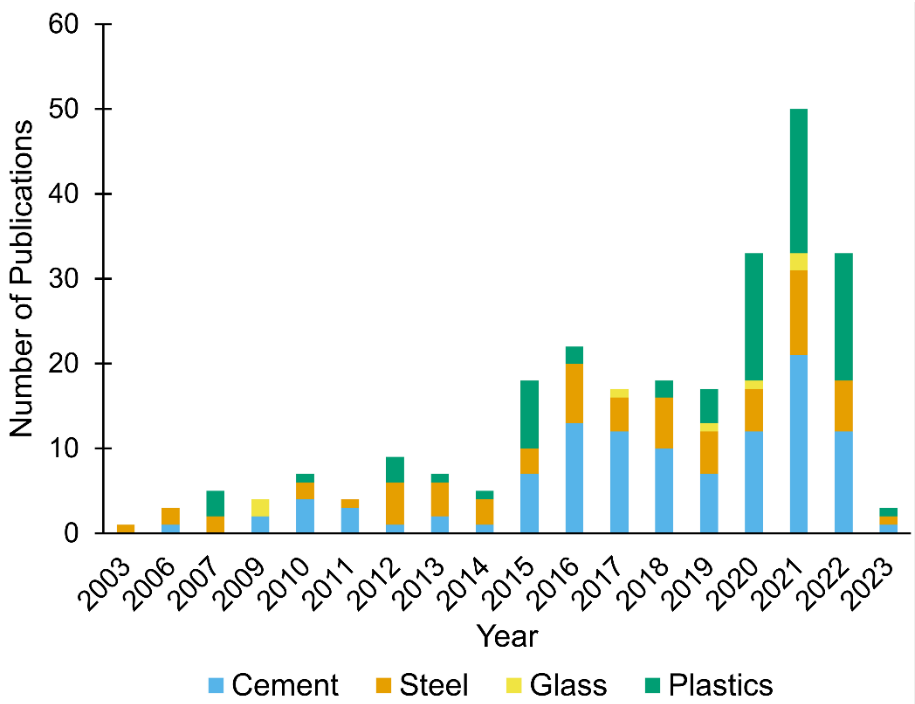


Figure 6: Temporal Assessment summarizing LCA Publications from 2003 to 2023

Geographic Location of Study: Figure 7 highlights the geographic locations evaluated in the studies assessed. It was determined that China is the country with the greatest number of LCA evaluations in the cement, plastics, and steel sectors, likely due to the large economic and industrial growth seen in recent decades [27]. However, glass LCA articles were more focused in Italy due to the importance of glass to the country's economy [28]. The region with the largest number of scenario evaluations was Europe. Excluding studies in Europe that specified a specific country, 33 assessed studies opted to perform an LCA considering Europe as a generalized region; making use of Eurocentric databases such as Ecoinvent. This value includes studies that looked at European Union nations only, Europe, and Europe without Switzerland. In addition to regional evaluations, 11 assessed studies utilised general global data.

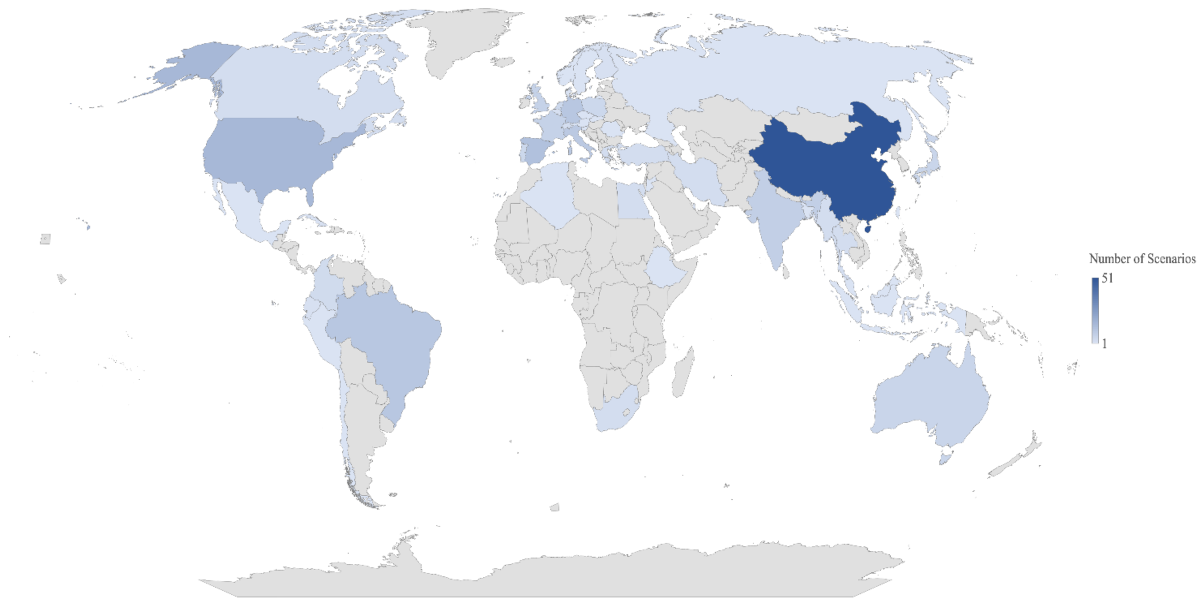


Figure 7: Geographical locations considered for retrieved LCA studies

4.2. Sectoral Characteristics

Cement Sector: As the most commonly used construction binder, Portland cement was evaluated as a scenario in roughly 70% of evaluated studies. While this is the case, only 35% of studies assessed opted to exclusively evaluate Portland cement manufacturing by evaluating current 'business as usual' production processes utilising region specific data [29, 30]. Portland cement was often used as a base scenario to compare alternative cement types such as blended cements or alkali activated cements. Roughly 47% of studies assessed evaluated supplementary cementitious materials (SCMs) for use in blended cements at various substitution levels with fly ash, ground granulated blast furnace slags (GGBFS), and other natural pozzolans the being most assessed SCMs. Three of the selected studies opted to evaluate limestone calcined clay cement (LC^3), as an alternative to blended cements using industrial wastes [31-33]. Approximately 10% of retrieved studies evaluated alkali activated cements, with sodium silicate being the alkali activator that was most selected. In addition to evaluating alternative low carbon cement alternatives, it was found that roughly 20% of studies opted to conduct a comparative LCA to assess the environmental impacts of utilising alternative fuel types including solid recovered fuel and liquid hazardous waste [34, 35]. With the calcination process requiring the largest fuel input, 22 of the studies assessed opted to evaluate the production of clinker as a scenario [36, 37]. In contrast, nine of the selected studies evaluated concrete production by furthering the system boundary to include aggregate and water consumption [38, 39].

Steel Sector: As the dominant steel production route, roughly 75% of assessed studies focused at least partially on blast furnace - basic oxygen furnace (BF-BOF) production. Of these assessed studies, 63% focused exclusively on the BF-BOF production route [40-42]. The remainder conducted a comparative study on multiple routes, including electric arc furnace production [43, 44]. The remaining assessed studies focused on the electric arc production route only [45]. A variety of iron and steel products were selected for the scope. Approximately 28% of all retrieved studies chose to focus on semi-finished products including crude steel [46], slab [47], and liquid steel [48]. The majority of assessed studies evaluated finished steel products including cold rolled coil [49], hot rolled coil [50], rebar [51], and hot rolled beams [52]. A number of retrieved studies focused specifically on intermediate products used within the BF-BOF production route including coke [53], cast iron [54], sinter [55], and iron ore pellets [56]. Whilst others chose to expand the system boundary further and study specialist types of steel products including stainless [57] and galvanized [58] steel. More recent studies tend to include additional scenarios which focus on reducing the environmental impact of production by using innovative technologies. This is across all production routes and includes the use of biomass [59], wood pellet fuel [60], scrap [48], carbon capture technologies [61, 62], and hydrogen fuel [63].

Glass Sector: Out of the seven papers retrieved, four papers focused exclusively on glass production [28, 64-66]. The remaining articles included a glass production LCA as part of a wider system boundary [67-69]. Hollow glass was researched in three papers, with two considering its production [28, 66] and one evaluating production and recycling [69]. Crystal glass was studied in two papers, and mainly focused on the production process [64, 65]. The remainder considered specialist glass including high pressure [67] and borosilicate [68] glass.

Plastics Sector: More than 60% of assessed studies performed an LCA on fossil fuel-based plastics, whilst the remaining focused on understanding the potential of bioplastics as an alternative. Meanwhile, 10% of studies analyzed recycled polymers [70] and other recycling technologies [71]. The most common applications were found to be as feedstock (40%) [72], packaging material (30%) [73], and within modes of transportation (12%) [74]. Certain applications in agriculture [75], household appliances [76], and fashion [77], have received comparatively limited attention. Within Europe, it has been identified that polyethylene terephthalate and polypropylene are the most commonly used polymers despite their known adverse environmental impacts [78]. Therefore, these materials have become a focal point for LCA research with nearly half of assessed studies evaluating them. Limited investigations have been carried out into novel fiber reinforced plastics

[74, 79]. Roughly 68% of assessed studies performed a comparative analysis to evaluate environmental impact differences between virgin and sustainable plastic alternatives [75, 78].

4.3. LCA Characteristics

Table 4 summarises the application of the LCA framework across all assessed studies. A detailed analysis of each sector's application of the LCA framework is presented in the following subsections.

Table 4: Summary of LCA framework application across all assessed studies

Goal and Scope	**Most commonly selected declared/functional unit type**	**Mass-based declared unit**
	Studies that considered the functionality of selected unit (%)	23%
	Most commonly selected allocation procedure	Did not specify/cut-off
	Most commonly selected system boundary	Cradle-to-Gate
	LCA Type	Process based LCA
Inventory Analysis	**Studies that utilised primary data (%)**	45%
	Studies that conducted a data quality assessment (%)	5%
	Most commonly utilised database	Ecoinvent
Impact Assessment	**Most commonly utilised LCA software**	SimaPro
	Top 3 impact methods utilised	CML, ReCiPe, IPCC
	Top 5 midpoint indicators assessed	GWP, ODP, AP, EP, PM
	Studies that assessed endpoint categories (%)	20%
Interpretation	**Studies that conducted a sensitivity analysis (%)**	40%
	Studies conducted an uncertainty analysis (%)	17%
	Primary standard(s) followed	ISO 14040, ISO 14044

4.3.1. Goal and Scope

4.3.1.1. Functional Unit

A functional unit is defined as a "quantification of identified functions of a product" where the unit "provide(s) a reference to which the inputs and outputs are related" [11]. A properly defined functional unit is key when assessing the environmental impacts of a product throughout its whole life cycle. Alternatively, a declared unit can be used instead when the precise function of the material is unknown. A declared unit is defined as "a reference by which product, material, and energy flows are normalized to produce data expressed on a common basis" [80]. As noted in Product Category Rules (PCR) 2019:14, a declared unit can be utilised instead of a functional unit for 'cradle-to-gate' studies if "the product or material is physically integrated with other products during installation so they cannot be physically separated from them at end of life, the product or material is no longer identifiable at end of life as a result of a physical or chemical transformation process, and the product or material does not contain biogenic carbon" [81].

Cement Sector: The most selected functional unit type was found to be a mass-based unit. Over 60% of assessed studies opted for this unit type, with roughly half selecting 'one metric ton of clinker' [82] or 'one metric ton of cement or cement product' [83, 84]. Studies that opted to evaluate the production of concrete typically selected a volume-based unit [85, 86]. These units, however, should be classified as declared units because the precise function of the cement product was not defined. In the retrieved studies, 50% reported using a functional unit despite the material's function not being defined. Only 42% of assessed studies accounted for performance in comparative assessments evaluating more than one cement type [31, 87-89].

Steel Sector: A mass-based unit was the most selected functional unit type, with approximately two thirds of assessed studies selecting 'one metric ton of steel product' (regardless of the eventual product) as the common unit of mass [60, 90, 91]. A number of assessed studies defined their functional unit specific to their data inventory [92, 93]. A single study selected a functional unit type relating to energy [94], whilst another selected a functional unit type relating to volume [95]. These studies have goals which are specifically related to understanding the environmental impact of energy and water consumption respectively. However, these units (mass, volume, and energy) should be classified as declared units because the precise function of the steel product was not defined. As a result, over 80% of retrieved studies reported using a functional unit despite not defining the material's function.

Glass Sector: The most common functional unit selected was a mass-based unit, with most selecting 'one kilogram' of glass produced as the unit of mass. Vinci et al. [28] opted to include a time variable. The remaining two articles did not directly define glass as the assessed product, but still primarily focused on glass production [67, 68].

Plastics Sector: More than 65% of the retrieved studies were found to use mass-based units, choosing either 'one kilogram' or 'one metric ton of polymer product(s)' [96]. Additionally, of the retrieved studies, 14% selected a length unit [97] and 11% selected a volume unit [98]. Furthermore, 10% evaluated mechanical performance, such as carrying capacity, as their functional unit [99].

However, one study chose three different declared units due to the challenges of determining a functional unit for their multifunctional product [71]. It was observed that declared units were often selected but were frequently regarded as functional units [100].

4.3.1.2. System Boundary

A system boundary defines which aspects of a product's life cycle will be included and excluded in the analysis [11]. There are four main system boundaries that can be considered. The first is 'cradle-to-gate', which considers the extraction of any raw materials, the manufacturing process of the product, and any material or product transportation. PCR 2019:14 defines 'cradle-to-gate' as five distinct stages which includes extraction of raw materials [A1], transportation to manufacturing [A2], product manufacturing [A3], transportation to building site [A4] and construction site process [A5] [81]. The second is 'gate-to-gate', which assesses the environmental impacts at the product manufacturing stage (denoted as [A3] for construction products). The third is 'cradle-to-grave', which extends the system boundary to also include the use ([B1-B7] for construction products) and end of life ([C1-C4] for construction products) of a given product. The last is 'cradle-to-cradle' which defines the system boundary as a complete circle by reusing material at the 'end-of-life' stage.

Cement Sector: The most commonly selected system boundary is 'cradle-to-gate', as selected by 95% of assessed studies. This is likely due the material's product stage accounting for the largest environmental impact [101]. Some studies provide justification for this system boundary selection by assuming that the 'use' and 'end-of-life' phase would be similar for any cement mixes analyzed [102]. However, the definition of 'cradle-to-gate' changed based on the scope of the study. Most retrieved studies opted to evaluate the [A1-A3] stages of 'cradle-to-gate' but some did not include the product packaging stage found in [A3] [103, 104], and others opted to not include [A2] [105]. Even fewer considered the final two 'cradle-to-gate' stages, [A4] and [A5]. Two of the retrieved studies that did evaluate these stages assessed the impacts of concrete production [39, 106]. By including all stages of the 'cradle-to-gate' system boundary, a more holistic view can be presented. 'Cradle-to-grave' or 'cradle-to-cradle' is typically not considered due to the complexity of determining the 'use' and 'end-of-life' as reflected in less than 4% of the assessed studies.

Steel Sector: Approximately 75% of retrieved studies selected a 'cradle-to-gate' system boundary, with most opting to evaluate the [A1-A3] stages [41, 107]. Others chose not to include [A2] to avoid double counting within their model [108] and due to the effects regionalization [109]. Furthermore, one study chose to exclude both [A1] and [A2] due to the type of LCA model utilised [110]. None of the retrieved studies elected to evaluate the [A4] and [A5] stages of 'cradle-to-gate', likely due to the site specific nature of these stages [59] and the perceived relatively low impact [108]. Roughly 15% of evaluated studies selected the 'gate-to-gate' boundary type. This was often chosen when the study had a goal related to a specific aspect of the production process [48, 111]. Only 6% of assessed studies elected to evaluate a 'cradle-to-grave' or 'cradle-to-cradle' boundary type, often to explore novel solutions to reduce environmental impact through increased scrap inclusion [112] and carbon capture technologies [113].

Glass Sector: Three studies selected a 'cradle-to-gate' system boundary [64, 67, 68], however only one study explicitly included transportation of raw materials to the manufacturing location [64]. Two studies opted to extend the system boundary to include the 'use' and 'end-of-life' stages [65, 66]. Another study extended this boundary further by including recycling and an analysis of the circular economy [69]. The remaining study did not specify a system boundary, but the study scope implies that a strict 'gate-to-gate' analysis was selected [28].

Plastics Sector: Roughly 47% of retrieved studies adopted a 'cradle-to-gate' system boundary [114, 115], while 40% selected the extended 'cradle-to-grave' boundary [116, 117]. Only one study explored the 'cradle-to-cradle' perspective, emphasizing the potential for recycling and waste valorization [78]. In contrast, five studies selected a 'gate-to-gate' system boundary to examine specific manufacturing phases [71, 118]. Six other studies employed a 'gate-to-grave' boundary, which includes manufacturing and disposal phases [119, 120]. Additionally, three studies left the system boundary unspecified [121-123]. Four studies explored more than a single system boundary (e.g., 'cradle-to-gate' and 'cradle-to-grave') [71, 78, 124, 125].

4.3.1.3. Allocation

Allocation is defined as "partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems" [11]. Although ISO 14040 does not specify a specific procedure for allocation, it does note that allocation through system expansion and avoided burden should be avoided when possible. Instead, it is best to allocate based on physical (e.g., mass) or non-physical (e.g., economic) relationships [13]. The benefit of using a mass allocation procedure is that the value typically remains constant over time, with change only occurring if the product's production process is significantly improved due to technological changes. However, problems may occur if the mass of the by-product being considered is greater than that of the main product. While an economic allocation procedure can take into account the value and therefore the demand of the product being considered, price fluctuations may render the results inaccurate in the future [126].

Cement Sector: Careful consideration must be made for the allocation of SCMs such as GGBFS, fly ash, and other industrial waste materials. Traditionally, these materials have been classified as waste products despite their increased demand within the cement sector. Since 2008 however, BFS has been considered a by-product within the European region [127, 128]. If an industrial waste material is no longer considered a waste product, the environmental impacts associated with it must be considered. It was found that out of the 48 retrieved studies that examined the use of industrial waste materials for use in low carbon cements, over half did not specify an allocation procedure to account for the environmental impacts to the material's production. When an allocation procedure was specified, economic allocation was most often used [129, 130]. The same number of studies opted to evaluate both mass and economic allocation procedures [131, 132]. Additional allocation procedures that were considered for industrial wastes include avoided burden [85], energy [39], and impacts [133].

Steel Sector: The majority of retrieved studies did not specify an allocation procedure used [90, 134, 135] despite the importance of correctly allocating the impact of the several intermediate products produced during the steel production process (e.g., BFS) [136]. Five different allocation procedures were used by the remaining studies: system expansion [94], avoided burden [137], mass [51], energy [138], and economic [138]. The allocation procedure selected varies significantly between studies but is typically influenced by the goal of the study. There appears to be no consensus on which allocation procedure to use, and this can have an impact on the study result.

Glass Sector: Despite the production of potential by-products, allocation procedures were not applied in any of the retrieved studies.

Plastics Sector: More than half of all retrieved studies did not specify the use of any allocation procedure. When used however, it was found that allocation based on physical relationships (e.g., energy and mass) was selected in 30% of studies [71, 139]. Other allocation procedures were used including system expansion [140, 141] and economic [142].

4.3.2. Inventory Analysis

The inventory analysis step of an LCA evaluates the flow of energy, emissions, and matter across a defined system boundary by creating a life cycle inventory [12]. There are three main LCA methodologies that influence how a life cycle inventory is created: process-based, economic input-output (E-IO), and hybrid [143]. The process-based methodology is based on the physical flows present within a system boundary. The E-IO method evaluates a material's environmental impacts based on economic flows between sectors within a product's supply chain. The hybrid LCA methodology is a combination of the process-based and E-IO methodologies [144].

4.3.2.1. Type of LCA

Cement Sector: The process-based methodology was found to be the most commonly used [130]. Three of the assessed studies selected the E-IO method [27, 145, 146]. The hybrid methodology was conducted in two of the retrieved studies, with one evaluating production in China [147] and the other in Australia [129]. The implementation of the process-based approach across most papers is in line with the research questions presented which focus on process specific evaluations.

Steel Sector: Roughly 88% of studies assessed selected the process-based method [148], with a small number taking a E-IO [95] or hybrid approach [149]. Given the focus on material production, this is to be expected. However, it does highlight that it is uncommon for material level analysis to be conducted using novel LCA methodologies.

Glass Sector: All retrieved studies utilised a process-based methodology, highlighting a potential research gap to utilize novel LCA methodologies [66].

Plastics Sector: The process-based methodology was predominantly selected in the retrieved studies [150, 151]. Two studies performed a hybrid LCA to evaluate the wider environmental impacts of the assessed product [152, 153].

4.3.2.2. Data Source

Data sources for LCA studies can be split into two different categories: primary and secondary. Primary data is data collected by the LCA practitioner from either industry or laboratory testing. Secondary data is obtained through publicly available databases, existing publications, or other sources.

Cement Sector: It was found that approximately 43% of retrieved studies utilised primary data sources. A majority of these studies were transparent about where the primary data was obtained, but some noted that the data was confidential and therefore could not be detailed [30, 154, 155]. Only a small number of retrieved studies utilised primary data exclusively [82, 104], with most opting to use a combination of both primary and secondary data sources [156, 157]. Secondary data sources were used in some form in over half of all assessed studies, with roughly 57% relying on solely secondary data [158, 159]. Some of the most widely used databases were found to be

Ecoinvent [160], GaBi [86], European Reference Life Cycle Database [161], United States Life Cycle Inventory [35], and Chinese Life Cycle Database [130].

Steel Sector: Approximately half of retrieved studies utilised primary data sources. A small number of these have detailed the source, which includes major manufacturers such as Corus [53], ArcelorMittal [93], and ThyssenKrupp [136]. However, most studies that reported using a primary data source do not specify the supplier. Most of the assessed studies supplement their foreground data with secondary data, typically from studies or publicly accessible database sources [55, 108, 138]. The remaining studies made use of this secondary data only [162]. A wide range of databases have been used including Ecoinvent [51], WorldSteel [163], GaBi [164], and DEAM [137].

Glass Sector: Only one study relied solely on primary data due its specific ‘gate-to-gate’ system boundary selection [28]. Five studies supplemented the collected primary data with secondary data sources [64-68], while the remaining study opted to use only secondary data [69]. The transparency of primary data collected varied across the retrieved studies, with Pulselli et al. [65] providing the most detailed primary data inventory despite the study’s need to exclude certain data values for confidentiality reasons. A number of databases were utilised including Ecoinvent [66], Environmental Development of Industrial Products [64], and European Reference Life Cycle Database [69].

Plastics Sector: Approximately 60% of the retrieved studies utilised primary data obtained from experimental works [118] or directly from plastic industries [165]. A number of these studies supplemented this data with secondary data from databases such as Ecoinvent [166], GaBi [122], European Reference Life Cycle Database [167], and Chinese Life Cycle Database [168]. A third of assessed studies relied solely on secondary data [169].

4.3.2.3. Software

Cement Sector: SimaPro was found to be the most utilised software package with nearly half of all retrieved studies opting to use it [170, 171]. The second and third most used software were found to be GaBi [172, 173] and OpenLCA [174, 175]. Roughly a quarter of studies did not specify a specific software package [147, 176], however it is likely that many of these studies opted to perform calculations in Excel or in custom software [160, 177].

Steel Sector: Approximately 45% of retrieved studies utilised either SimaPro [41, 107] or GaBi [61, 178]. However, a similar number used an unspecified software package [44]. How the results were presented in some assessed studies means that the software used could be deduced, but this finding would also suggest a significant number used custom solutions. The remainder of studies used a variety of software packages including OpenLCA [56, 179].

Glass Sector: In six of the seven retrieved studies, there was an even split between the GaBi [64-66] and SimaPro [28, 67, 68] software packages. The remaining article opted to select OpenLCA for their analysis [69].

Plastics Sector: SimaPro was found to be the most utilised software package with 35% opting to use it [180]. The second and third most commonly used software packages were found to be GaBi [79] and OpenLCA [181]. Other software selections include Excel [182] and Thinkstep Professional [183]. About 10% of retrieved studies did not specify a specific software package [184].

4.3.3. Impact Assessment

Life cycle impact assessment is defined as the stage at which the "magnitude and significance of the potential environmental impacts for a product system" are evaluated throughout the entire life cycle of a product [11]. At this step, the impact categories, category indicators, and characterization models are selected.

Cement Sector: It was determined that Centrum voor Milieukunde Leiden (CML) was the most often selected impact assessment methodology, with 33 of the retrieved studies selecting this method [83, 131, 155]. The second and third most utilised methodologies were found to be Intergovernmental Panel on Climate Change (IPCC) [32, 185] and ReCiPe [29, 186]. A number of studies opted to select more than one impact assessment methodology to consider a wider range of impact categories [187] or for sensitivity analysis purposes [147]. Less than 13% of retrieved studies did not specify an impact assessment methodology [174, 188]. The most common midpoint indicator reported was global warming potential (GWP), with nearly every study reporting this indicator. The second most reported midpoint indicator was found to be ozone depletion potential with over half of all retrieved studies evaluating this category. Other common midpoint categories that were reported include acidification potential, photochemical oxidation potential, eutrophication potential, and terrestrial ecotoxicity. Endpoint indicators were reported in only a fifth of all retrieved studies, with human health, resources, and ecosystem quality being the most reported categories. To assess endpoint categories, it was concluded that Eco-Indicator [36, 132] and IMPACT [189, 190] were most commonly selected.

Steel Sector: There is a near-even (approximately 20% each) split between the ReCiPe [54], IPCC [191], and CML [45] impact methodologies in the retrieved studies. These were occasionally used in combination with each other [40, 178, 192]. Other impact methodologies utilised include International Reference Life Cycle Database [41], IMPACT [47], Eco-indicator [52], cumulative energy demand [193], and water footprint [194]. However, nearly a quarter of all retrieved studies did not specify an impact methodology [109, 113, 195]. Studies which used the IPCC and CML methodologies tend to focus heavily on GWP, with over 70% reporting this. In contrast, a majority of assessed studies which used the ReCiPe methodology reported all of the midpoint indicators available [196]. Aside from GWP, the most common midpoint indicators reported were particulate matter formation, eutrophication potential, and acidification potential. Regardless of impact methodology, only 12% of retrieved studies reported an endpoint indicator result [58, 60, 179]. The most common endpoint indicators reported were human health, ecosystem quality, and resources. All but two of these studies used the ReCiPe methodology to do this [49, 51].

Glass Sector: The CML impact assessment methodology was selected in five of the seven retrieved studies. In addition to CML, one study opted to also use the ecological scarcity method [69]. The remaining studies selected other methodologies including ReCiPe [28], IPCC [67], and cumulative energy demand [67]. The most frequently assessed midpoint indicators were GWP, acidification potential, eutrophication potential, ozone depletion potential, and photochemical oxidation potential. Only one article opted to calculate endpoint indicators, with human health, ecosystem quality, resources, and climate change being assessed [28].

Plastics Sector: Among the retrieved studies, ReCiPe was found to be the most often selected impact assessment method with approximately 19% of the studies utilising this method [152, 197]. CML was found to be the second most used method [72]. However, around 14% of assessed studies employed a mixed-methodology approach, indicating a preference for combining various methods to comprehensively evaluate environmental impacts [114, 142]. Less than 5% of retrieved studies did not specify the impact methodology [198, 199]. The most commonly assessed impact indicator was found to be GWP [167, 200]. Other reported indicators include freshwater ecotoxicity, particulate matter formation, terrestrial acidification potential. Only 20% of evaluated studies considered endpoint indicators, with the most evaluated categories being human health, ecosystem quality, and resources. To calculate these indicators, ReCiPe was found to be the impact assessment method typically selected [201].

4.3.4. Interpretation

ISO 14044 standard specifies two main interpretation checks that should be conducted [13]. The first check is a sensitivity analysis which is used to test the accuracy of a result by altering assumptions defined in the goal and scope. A second check is an uncertainty analysis which verifies if the data collected is relevant and accurate. This is typically done using a Monte Carlo analysis method. The Monte Carlo method is a probabilistic method to model intricate systems and assess uncertainties by simulating numerous scenario analysis [202]. A third analysis type that can be considered to evaluate possible future scenarios for a given material process is a scenario analysis. This can be done through the theoretical implementation of novel changes to a product's manufacturing processes [26]. Once the interpretation step is completed, conclusions from the results can be drawn and recommendations can be made.

Cement Sector: It was found that roughly 40% of the retrieved studies opted to conduct a sensitivity analysis. The most common sensitivity analysis performed in these studies involved altering key input data parameters (most notably transportation distance) by 5-20% [173, 203]. Altering the allocation procedure was also a common sensitivity analysis check for studies that evaluated by-products for use in low carbon cements [131, 204]. Only 11 studies opted to perform an uncertainty analysis [89], with Monte Carlo simulation being the most used evaluation method [205, 206]. Only eight studies conducted both a sensitivity and uncertainty analysis as part of the study [129, 187, 206]

Steel Sector: Only a quarter of retrieved studies reported completing a sensitivity analysis. The most common method of completing this was by altering the input parameters and reporting the effects on the results. There is no consensus for a standard approach to this, with studies altering parameters by 5% [58], 10% [51], and 20% [207]. Roughly 20% of assessed studies completed an uncertainty analysis, with Monte Carlo simulations being the most common method [192, 208]. Less than 8% of retrieved studies elected to complete both interpretation checks, with frequency increasing in more recent studies [58, 209].

Glass Sector: Three of the seven studies opted to perform a sensitivity analysis, despite the studies not referring to it as such. Methods included changing input parameters such as the amount of recycled glass present in a batch [66, 69] and transportation distance [69]. One study provided the most comprehensive sensitivity analysis by evaluating geographical, economic, and technological parameters such as the amount of energy consumed, process efficiency, and plant life span [67]. None of the retrieved studies performed an uncertainty analysis.

Plastics Sector: It was found that more than half of the retrieved studies conducted a sensitivity analysis [97, 210]. Approximately 26% of studies performed an uncertainty analysis with Monte

Carlo simulation being the most commonly selected method [211]. Roughly 19% of the retrieved studies conducted both a sensitivity and uncertainty analysis [169, 212].

5. Discussion

5.1. LCA Cross Sector Comparisons

5.1.1. Goal and Scope

Across the sectors investigated, in the majority of cases, the terminology relating to functional and declared units is being inconsistently used. This means that the performance of materials is not being accounted for when the functional unit is defined. In the retrieved studies, over two thirds reported using a functional unit without considering the mechanical properties that affect a product's function. As such, a declared unit should be used to describe the assessed product more accurately. While the use of either unit type can be justified, it is important to distinguish the difference between these terms as it can have a direct impact on the results of a study. This is particularly true in cases where functional equivalence must be taken into consideration when conducting a comparative analysis between two or more products [172]. Ultimately, differences between studies, even at this fundamental level, may limit the understanding gained by conducting an LCA. Thus, greater focus needs to be placed on how the units are identified and defined. This can only be done through an objective comparison on the basis of functionality by the use of a properly defined functional unit. As noted, alternatively, a declared unit can be implemented when the material properties or the function is not affected or known.

A good example of the challenge that comes with defining material functionality is found within the cement sector. Using a blended cement mix or an alkali activated cement can result in performance differences (most notably compressive strength) when compared to Portland cement alone [213, 214]. This illustrates the clear mechanical property differences that should be accounted for when conducting an LCA. When considering compressive strength differences in comparative assessments, most studies opted to use a strength ratio modification based on measured compressive strength to normalize environmental impacts [158, 189]. Aside from compressive strength, other parameters that have been selected to evaluate cement functionality include rapid chloride permeability and service life [215]. Some studies opted to define their functional unit by using a combination of parameters to provide a more holistic definition of cement functionality [172, 216]. This type of functional unit is particularly relevant when evaluating the environmental impact of a specific structural element that requires a certain mechanical strength and defined service life [217]. An approach like this is not seen in any of the other sectors, despite an intrinsic link between material function and properties. Within the steel sector, this could be defined by the mechanical properties or by considering a specific product function, such as the wear resistance of rail steels. Similarly, in the glass sector, the mechanical properties of specific products are critical depending on application, and therefore must be considered within the functional unit. Likewise, defining a functional unit for plastic products is a complex task due to the wide range of potential applications. As a result, declared units are typically favored over more complex functional units that define performance. Ultimately, this highlights that environmental impact is influenced by material function, which could produce different environmental impact results when assessed beyond a 'cradle-to-gate' system boundary.

It was found in all retrieved studies the most commonly selected system boundary was 'cradle-to-gate'. Typically, data availability for 'cradle-to-gate' is more abundant and reliable compared to data in downstream stages such as 'use' and 'end-of-life'. Furthermore, this system boundary enables a direct comparison of the environmental impact of different products or materials at the manufacturing phase. This is valuable for industries seeking to identify hotspots, trade-offs, and opportunities for improvement to make informed decisions about their processes and materials. While this is the case, there is a notable lack of studies that extend the system boundary. However, within the plastics industry, the 'cradle-to-grave' boundary is often evaluated due to more prevalent circular economy practices. Despite this, there is a clear need for studies that provide a greater holistic view across the entirety of a material's life cycle to understand the full environmental impacts of a product. Evaluating a product that is ultimately used in a variety of applications with different requirements is challenging unless the study scope and functional unit is very specific [84, 204]. For example, with the primary function of cement being the main ingredient in concrete, carbon uptake of concrete structures due to carbonation would need to be assessed in the 'use' and 'end-of-life' stages. While this aspect can be used to reduce concrete's carbon footprint, the life span, concrete strength, and post demolition time all have to be considered as they directly impact the amount of carbon dioxide uptake that can occur [218]. Similarly, it is difficult to generalize the use of steel as a product due to the variety of potential downstream uses (either as a feedstock, semi-finished, or finished product). This results in a complex boundary that is difficult to define unless a specific product is being evaluated.

There is also a clear challenge on how to tackle allocation, with a variety of procedures used within each sector. The most interesting exponent of this, despite the intrinsic link between them, is in the steel and cement sectors through the production of BFS as a by-product and the use of GGBFS to create blended cements respectively. There is no clear consensus on how BFS (as well as other by-products) should be allocated. This means that both sectors are sometimes allocating environmental impacts within their system boundary, and sometimes outside of it. When allocation was considered, it was found that the cement sector most often utilised economic allocation and the steel sector utilised mass allocation. However, in the glass and plastics sectors, allocation of by-products is rarely considered. This makes current cross-sector comparison challenging and means technological advances cannot be properly assessed. This contradicts the core principles of LCA.

5.1.2. Inventory Analysis

A common thread between studies across each sector is the frequent utilization of the same two basic methodologies: process-based and E-IO. These are both common but have known limitations. Process-based LCAs can be conducted using a process flow, or through a matrix method [219]. This works well for a simple product system, but industrial processes tend to have multiple input and output streams. In this case, the allocation of material flow becomes a challenge due to the large amount of data required to fully satisfy the system boundary. Consequently, the method is known to suffer from error truncation, which can hamper long term decisions for policy making or comparative assessments [143]. E-IO was devised to counter these issues by considering the whole product supply chain within an economy [220]. However, this can suffer from detail limitations due to the scale of data [221] required and the fact that E-IO datasets are not regularly updated [143]. A small number of studies within each sector have understood these limitations and made use of more novel techniques such as hybrid methodology. These may provide a more holistic view on the material's environmental impacts through the evaluation of a wider system boundary. The reasons for this limited uptake could be numerous but is likely due to a lack of method standardization beyond detailed academic studies and limitations presented by the software found to be most commonly used [144, 222].

Across all retrieved studies, there is a mixture of reported data sources. Less than half reported using primary data, with only a small group reporting the data supplier. However, the majority did not include this information, often citing data security concerns. It was also found that raw data is often not published; however, this trend is reversing in recent years with a move toward open access publication [223]. This means there is a concerning lack of transparent primary data being used in studies which are important levers for decision making toward industrial decarbonisation. As noted, many studies also utilize secondary data. However, it is a known issue that large databases are not kept up to date which could create additional issues with study comparison and validation [224].

The quality and accuracy of an LCA is directly related to the quality of the data collected. However, most of the retrieved studies struggled to demonstrate sufficient, objective quality checks on their data; and in most cases, do not mention it at all. In ISO 14044, specific requirements are outlined in regard to data quality, particularly studies that are publicly released and used in comparative assertions. Quality checks are designed to classify data sources based on the relevance and reliability of data and to better understand the uncertainty created by using certain types of data. This includes assessing the time period, geographical coverage, technological coverage, consistency, and reproducibility of the data used in an LCA study among other data quality checks [13, 141]. In the cement sector, only a fifth of the evaluated studies explicitly mentioned the importance of data quality or conducted a qualitative data quality assessment. A small number opted to extend this evaluation further by conducting a quantitative data quality assessment such as conducting a pedigree matrix evaluation [172, 206]. Similarly, in the steel sector, a quarter of studies mentioned the issue of data quality [42, 162]. However, only a fifth of these studies opted to complete a full data quality assessment but typically only using qualitative techniques [40]. Despite the noted importance, a data quality assessment was not conducted in any retrieved glass sector studies. In the plastics sector, roughly 18% of retrieved studies explicitly mentioned data quality, with a small percentage performing a formal data quality assessment utilising a pedigree matrix [116, 141, 169]. As a result, the majority of data is being left unscrutinized. Inevitably, this leaves uncertainty over the reliability of some data (both primary and secondary). This is particularly true within studies which focus exclusively on production sites [41, 65, 104] that should otherwise be the example case for the open publication of data. This issue limits how LCA methods can be used to make informed decisions about the impacts of different technologies and processes. It is important for studies to prioritize data transparency when possible and assess the reliability and credibility of all data sources to ensure accurate findings.

5.1.3. Impact Assessment

Across the retrieved studies, the most utilised impact assessment methodologies were CML, IPCC, and ReCiPe. Some studies elected to use a combination of methodologies to satisfy their study goals. However, there are technical differences between each methodology [23, 225]. One study opted to directly compare different methodologies and assess the outcomes as part of a sensitivity analysis, and found that the results were very similar across all categories included [147]. This suggests that the reason behind choosing a particular methodology is often practitioner driven, and typically only down to meeting study goals. There were also a wide variety of midpoint impact assessment categories evaluated, the most common being GWP. A number of retrieved studies chose to only report categories directly related to climate change, which could be down to an absence standardization [226]. Other common categories varied depending on the sector; for example, the cement, steel, and glass sectors often assessed acidification potential and eutrophication potential, whereas the plastics sector typically assessed freshwater ecotoxicity potential and terrestrial acidification potential. Furthermore, there are two distinct methods of reporting these results: through graphs or numerical values. The latter of these is much clearer but is less common. There is also discussion about how each methodology employs distinct characterization factors to convert emissions or resource use into single scores [227]. These factors are often derived from different scientific models, databases, and assumptions, which leads to differences in single score values for the same processes [228]. Although each of these issues is often down to the sector and study goals, this level of variation in category and reporting choice can make comparative assertions between studies challenging. Furthermore, only a fifth of retrieved studies reported an endpoint impact assessment category. Although midpoint categories are typically more detailed and therefore have a lower overall uncertainty associated with them [229, 230], endpoint categories are crucial to making long-term decisions about product processes. The most common methodologies used to report endpoint categories were Eco-Indicator, IMPACT, and ReCiPe; meaning that some studies changed methodologies between midpoint and endpoint.

5.1.4. Interpretation

Most retrieved studies only completed a variation of qualitative interpretation. That is, making comments on the results found without the associated statistical error and uncertainty. Less than half of all retrieved studies completed a sensitivity analysis which is critical in understanding the accuracy of the results. Even fewer completed an uncertainty analysis which should be used to understand the potential errors in input data. The lack of implementation of these key requirements is concerning. This stage is critical to not only verify that the results align with the defined goal and scope, but also to identify and correct any mistakes. The limited implementation of this stage across all sectors suggests that all sectors are facing the same challenges with fully implementing the interpretation step. In addition, it was determined that only a fifth of all retrieved studies opted to evaluate future scenarios by conducting a predictive LCA [148, 182, 231]. This LCA approach should be examined more thoroughly as industrial decarbonisation becomes critical.

5.1.5. Standards

Many retrieved studies reported using an LCA related standard; typically, either ISO 14040 [11] and/or ISO 14044 [13]. As mentioned, despite this good level of 'on the surface' compliance with the existing framework, a significant number of these studies do not fully follow the standardized methodology. Concerningly, a quarter of all retrieved studies reported using either an unnamed standard or do not mention a standard at all. This mixture of standard compliance, standard use, and underreporting of standard utilization means that cross-sector comparison is stifled at a fundamental level.

Observing how LCAs are conducted in academia compared to industry further highlights key differences in the implementation of the standardized methodology. LCAs created for industry are required to follow a strict set of guidelines to publish the findings as an Environmental Product Declaration (EPD). Environmental Product Declarations are published in accordance with ISO 14040 [11] in addition to PCRs [81] which provide guidelines for each product category. A rigorous third-party review by experts is also required for certification. These requirements, which are notably absent in academically published LCAs, illustrate the lack of robustness present in established practices. Only two retrieved studies followed PCR guidelines [66, 232]. There are efforts to harmonize academic and industrial frameworks. For example, the Partnership for Carbon Transparency (PACT) Pathfinder Framework attempts to integrate existing standards and guidelines to enhance several key criteria including data quality indicators, emission calculation methods, allocation procedures, and decarbonisation incentives [233]. It is critical that harmonized approaches continue to be pursued within the LCA community to enable effective and transparent LCA studies.

5.2. Study Limitations

Whilst the retrieved articles provide a comprehensive review of the energy-intensive sectors of interest, there are limitations within the methodology which may influence the outcome of the study. Achieving a keyword string which is both general yet specific is challenging, and the boundaries chosen may have artificially limited study retrieval; however, the large number of relevant studies accessed would suggest that any artificial limitation on study retrieval is not substantial. While this is the case, it should be noted that there were significantly fewer studies retrieved from Africa and some parts of Asia, likely due to a lack of accessible literature available from those regions that focused on the chosen energy-intensive sectors. As a result, a bias towards studies conducted on western, industrialized nations is apparent. Similarly, the search criteria deliberately limited results to only peer-reviewed journal articles given the large quantity of studies available, and the need to focus on high quality, peer-reviewed research. However, this was observed to be a particular limitation in regard to retrieval of studies focused on the glass sector, and had this criterion not been included, the resulting search criteria may have resulted in more studies for analysis. Furthermore, analysis of the chemical sector was restricted to studies related to plastics. Although a significant portion of the chemical sector is dedicated to plastic production, there are several other product areas which would benefit from detailed analysis. Future research can expand on not just these sectors but also other energy intensive sectors such as paper, ceramics, and other metals. As noted within the methodology, this study deliberately excludes the analysis of impact category values. Although outside the scope of this research, analysis of impact category values would allow future studies to be benchmarked against a wide selection of literature from each energy-intensive industry.

6. Conclusion

This study investigated the implementation of LCA methodology across four, key global energy intensive industries: cement, steel, glass and plastics. An SLR methodology was implemented using a novel keyword string. The challenge of generating a string that captured a wide range of keywords was solved through the use of sector specific terms. This string enabled cross-sector retrieval of studies specifically focused on production of materials within the aforementioned sectors. Following this, studies were objectively inspected through a bibliometric analysis using VOSviewer. Studies that were duplicate, off topic, non-compliant, and non-accessible were removed. The final taxonomy yielded 256 unique journal articles for further analysis.

The findings revealed significant contrast across the four sectors and the implementation of the standardized LCA methodology:

- Goal and Scope: A declared unit of mass was most commonly selected; however, this unit was often incorrectly labeled as a functional unit. Despite being a key aspect of material performance, functionality was rarely defined as part of the functional unit. 'Cradle-to-gate' was the most commonly used type of system boundary. When an allocation procedure was selected for a by-product, it was found that the steel sector most often used mass allocation and the cement sector frequently selected economic allocation. In the remaining sectors, an allocation procedure was typically not specified.
- Inventory Analysis: The process-based method was the most commonly employed LCA type to create a life cycle inventory. Several retrieved studies in each sector did elect to use alternative methodologies including E-IO and hybrid. Primary data sources were used by approximately half of all retrieved studies. However, most either wholly or partially relied on secondary data. Nearly all assessed studies utilised an industry standard software

package, typically SimaPro or GaBi. The number of retrieved studies that carried out data quality checks were significantly limited. Among the few that did assess the quality of their data, the pedigree matrix emerged as the preferred method.

- Impact Assessment: Impact assessments were completed by all assessed studies. A range of methodologies were used to report midpoint indicators, such as CML, IPCC, and ReCiPe. A diverse range of impact assessment categories have been reported within the retrieved studies. Only a fifth of assessed studies calculated an endpoint indicator. The lack of uniform reporting (in terms of methodology, category, and style) makes comparative assertions difficult.
- Interpretation: Less than half of retrieved studies elected to conduct a sensitivity or uncertainty analysis. Many qualitative and quantitative approaches were taken, with no clear consensus on how to complete either interpretation stage.

Ultimately these findings show that a consistent approach to meet the existing LCA standards is lacking. While this study focused on the cement, steel, glass, and plastics sectors, it is expected that this lack of consistency will be similar in other industrial sectors. Therefore, key recommendations are outlined which will allow for greater transparency and comparative value for future LCA studies across all industrial sectors and products:

1. Although different functional unit types are appropriate under the right circumstances, they must be properly defined. This is imperative in comparative assertions; therefore, this aspect of ISO 14040 should be very rigorously observed when conducting a study.
2. It has been identified that allocation procedures for by-products used in two or more sectors should be intrinsically linked. This influences how and where some product impacts are attributed. Therefore, LCA practitioners must seek to properly allocate by-products in a consistent manner; particularly where products enter the life cycle of another product.
3. The majority of studies in energy intensive industries are likely to be focusing on a production site, therefore making use of the ‘cradle-to-gate’ boundary. As the importance of the circular economy principle grows, a ‘cradle-to-cradle’ system boundary would provide a more holistic view of a product’s life cycle.
4. It is clear that energy intensive industries have not always worked synergistically, despite the benefits of systems symbiosis; likely due to the complexity of modern supply chains. To allow for this approach, the system boundary and functional unit of a study must be defined with this in mind. This will allow for cross-sector analysis by combining studies to build a

wider system boundary (e.g., using steelmaking BFS to create low carbon cements and glass flux), whilst retaining the accuracy of single product studies.

5. A high quality LCA can only be completed with high quality data, which is typically from a primary source. However, there is a clear challenge with allowing open access to sensitive industrial data. Therefore, the academic and industrial LCA community must work together to find a solution.
6. Data quality should be assessed objectively. This is paramount to understanding how confident a reader can be in the result of an LCA study. At a minimum, this should be done in line with ISO 14044 by accessing data on factors such as temporal, geographical, and technological relevance qualitatively. However, to be as effective as possible, this should be done using a quantitative technique such as a matrix evaluation. Assessing data in this way enhances effective comparative assertions and helps build public confidence in studies.
7. Greater uniformity in impact assessment methodologies, impact assessment categories and reporting methods would be highly beneficial. There is no standard approach to selecting either a methodology or reporting categories which stifles comparison on a fundamental level.
8. Interpretation is the most critical step in the LCA framework. Sensitivity and uncertainty assessments should be completed and reported as stated in ISO 14040 and ISO 14044. Confidence in results should be reported to enable transparent decision making.
9. The implementation of LCA methodology by academic and industrial practitioners should seek to converge through the improved selection of system boundaries, impact categories, and interpretation methods. This should be done using stringent guidelines taking inspiration from industrial PCRs.
10. A greater focus should be placed on future production processes and technology.

This research has generated several recommendations for improvements on current LCA practice, particularly in energy-intensive industries. While these are only recommendations, effective implementation of current and future LCA methodologies will offer the opportunity to carry out these suggestions. Further work should seek to address limitations and build upon this research by assessing impact category values within each sector to establish benchmark values. This will pave the way for an integrated research direction towards net zero, through effective cross-sector comparison and using the LCA technique as a critical decision lever.

Acknowledgements

M. S. C. Rihner, would like to thank the Engineering and Physical Sciences Research Council (EPSRC), and the Energy Institute at The University of Sheffield, for a DTP PhD Scholarship (EP/T517835/1), and the Grantham Centre for Sustainable Futures (GCSF). J.W. Whittle would like to thank EPSRC through the Advanced Metallic Systems Centre for Doctoral Training (EP/S022635/1) and British Steel Limited. M. H. A. Gadelhaq would like to thank EPSRC and the Energy Institute at The University of Sheffield for a DTP PhD Scholarship (EP/T517835/1). Su Natasha M. would like to thank Majlis Amanah Rakyat Malaysia for Graduate Excellence Program Scholarship (MARA-GrEP) (330408332561) and the GCSF.

The Authors would also like to thank James Marsden, Department of Chemical and Biological Engineering, The University of Sheffield, for his contribution to the cement sector data analysis.

Author Contributions

M.C.S. Rihner: Conceptualization (lead); Data Curation (equal); Formal Analysis (cement sector); Methodology (supporting); Writing- Original Draft Preparation (equal). J.W. Whittle: Conceptualization (supporting); Data Curation (equal); Formal Analysis (steel sector); Methodology (lead); Writing- Original Draft Preparation (equal). M.H.A. Gadelhaq: Conceptualization (supporting); Data Curation (equal); Formal Analysis (glass sector); Methodology (supporting); Writing- Original Draft Preparation (equal). S.N. Mohamad: Conceptualization (supporting); Data Curation (equal); Formal Analysis (plastics sector); Methodology (supporting); Writing- Original Draft Preparation (equal). R. Yuan: Writing- Review and Editing. R.H. Rothman: Writing- Review and Editing. D.I. Fletcher: Supervision; Writing- Review and Editing. B. Walkley: Supervision; Writing- Review and Editing. L.S.C. Koh: Project Administration; Supervision; Writing- Review and Editing.

Declaration of Conflicting Interest

The authors declare that there is no conflict of interest.

References

[1] International Energy Agency. "Industry: Sectoral Overview." International Energy Agency. https://www.iea.org/reports/industry (accessed 8th June, 2023).

[2] United Nations. "United Nations: Climate Action." https://www.un.org/en/climatechange (accessed 8th June, 2023).

[3] United Nations. "United Nations Sustainable Development Goals: Goal 13." https://sdgs.un.org/goals/goal13 (accessed 8th June, 2023).

[4] Innovate UK KTN. "Foundation Industries." UKRI. https://iuk.ktn-uk.org/materials/foundation-industries/ (accessed 8th June, 2023).

[5] United Nations. "United Nations Sustainable Development Goals: Goal 12." https://sdgs.un.org/goals/goal12 (accessed 8th June, 2023).

[6] United Nations. "United Nations Sustainable Development Goals: Goal 9." https://sdgs.un.org/goals/goal9 (accessed 8th June, 2023).

[7] Centre for Process Innovation. "How innovation is reducing the emissions of the UK's foundation industries." https://www.uk-cpi.com/blog/how-innovation-is-reducing-the-emissions-of-the-uks-foundation-industries (accessed 17th October, 2023).

[8] European Commission, "Council Directive 97/11/EC of 3 March 1997 amending Directive 85/337/EEC on the assessment of the effects of certain public and private projects on the environment," Official Journal L, vol. 73, no. 14/03, pp. 0005-0015, 1997.

[9] United States Environmental Protection Agency. "Learn About Environmental Management Systems." US EPA. https://www.epa.gov/ems/learn-about-environmental-management-systems (accessed 10th June, 2023).

[10] ISO 14067:2018 Greenhouse gases — Carbon footprint of products — Requirements and guidelines for quantification, International Organization for Standardization, British Standards Online, 2018.

[11] ISO 14040, Environmental management - Life cycle assessment - Principles and framework, International Organization for Standardization, British Standards Online, 2006.

[12] O. Jolliet, M. Saade-Sbeih, S. Shaked, A. Jolliet, and P. Crettaz, "Environmental Life Cycle Assessment," 2015, doi: 10.1201/b19138.

[13] ISO 14044, Environmental management - Life cycle assessment - Requirements and guidelines, International Organization for Standardization, British Standards Online, 2006.

[14] L. Ren, S. Zhou, T. Peng, and X. Ou, "A review of CO_2 emissions reduction technologies and low-carbon development in the iron and steel industry focusing on China," (in English), RENEWABLE & SUSTAINABLE ENERGY REVIEWS, vol. 143, JUN 2021, Art no. 110846, doi: 10.1016/j.rser.2021.110846.

[15] D. D. Furszyfer Del Rio et al., "Decarbonizing the glass industry: A critical and systematic review of developments, sociotechnical systems and policy options," Renewable and Sustainable Energy Reviews, vol. 155, p. 111885, 2022/03/01/ 2022, doi: https://doi.org/10.1016/j.rser.2021.111885.

[16] S. Griffiths et al., "Decarbonizing the cement and concrete industry: A systematic review of socio-technical systems, technological innovations, and policy options," Renewable and Sustainable Energy Reviews, vol. 180, p. 113291, 2023/07/01/ 2023, doi: https://doi.org/10.1016/j.rser.2023.113291.

[17] A. Marson, M. Piron, F. Zuliani, A. Fedele, and A. Manzardo, "Comparative Life Cycle Assessment in the plastic sector: A systematic literature review," Cleaner Environmental Systems, vol. 9, p. 100119, 2023/06/01/ 2023, doi: https://doi.org/10.1016/j.cesys.2023.100119.

[18] Y. Xiao and M. Watson, "Guidance on Conducting a Systematic Literature Review," Journal of Planning Education and Research, vol. 39, no. 1, pp. 93-112, 2019/03/01 2017, doi: 10.1177/0739456X17723971.

[19] D. Tranfield, D. Denyer, and P. Smart, "Towards a Methodology for Developing Evidence-Informed Management Knowledge by Means of Systematic Review," British Journal of Management, vol. 14, no. 3, pp. 207-222, 2003, doi: 10.1111/1467-8551.00375.

[20] W. Mengist, T. Soromessa, and G. Legese, "Ecosystem services research in mountainous regions: A systematic literature review on current knowledge and research gaps," The science of the total environment., vol. 702, p. 134581, 2020, doi: 10.1016/j.scitotenv.2019.134581.

[21] I. Atanasovska, S. Choudhary, L. Koh, P. H. Ketikidis, and A. Solomon, "Research gaps and future directions on social value stemming from circular economy practices in agri-food industrial parks: Insights from a systematic literature review," Journal of Cleaner Production, vol. 354, 2022, doi: 10.1016/j.jclepro.2022.131753.

[22] ISO 14040:1997 Environmental management — Life cycle assessment — Principles and framework, International Organization for Standardization, British Standards Online, 1997.

[23] ILCD Handbook: Analysis of exsiting Environmental Impact Assessment methodologies for use in Life Cycle Assessment. European Union: Institute for Environment and Sustainability, 2010, p. 115.

[24] M. Duque-Acevedo, L. J. Belmonte-Ureña, F. J. Cortés-García, and F. Camacho-Ferre, "Agricultural waste: Review of the evolution, approaches and perspectives on alternative uses," Global Ecology and Conservation, vol. 22, 2020, doi: 10.1016/j.gecco.2020.e00902.

[25] N. J. van Eck and L. Waltman, "Software survey: VOSviewer, a computer program for bibliometric mapping," Scientometrics, vol. 84, no. 2, pp. 523-538, Aug 2010, doi: 10.1007/s11192-009-0146-3.

[26] V. Bisinella, T. H. Christensen, and T. F. Astrup, "Future scenarios and life cycle assessment: systematic review and recommendations," The International Journal of Life Cycle Assessment, vol. 26, no. 11, pp. 2143-2170, 2021, doi: 10.1007/s11367-021-01954-6.

[27] S. Wu and H. Han, "Sectoral changing patterns of China's green GDP considering climate change: An investigation based on the economic input-output life cycle assessment model," Journal of Cleaner Production, vol. 251, p. 119764, 2020/04/01/ 2020, doi: https://doi.org/10.1016/j.jclepro.2019.119764.

[28] G. Vinci, F. D'Ascenzo, A. Esposito, M. Musarra, M. Rapa, and A. Rocchi, "A sustainable innovation in the Italian glass production: LCA and Eco-Care matrix evaluation," (in English), JOURNAL OF CLEANER PRODUCTION, vol. 223, 2019, doi: 10.1016/j.jclepro.2019.03.124.

[29] T. Z. Tun, S. Bonnet, and S. H. Gheewala, "Emission reduction pathways for a sustainable cement industry in Myanmar," Sustainable Production and Consumption, vol. 27, pp. 449-461, 2021/07/01/ 2021, doi: https://doi.org/10.1016/j.spc.2021.01.016.

[30] F. N. Stafford, F. Raupp-Pereira, J. A. Labrincha, and D. Hotza, "Life cycle assessment of the production of cement: A Brazilian case study," Journal of Cleaner Production, vol. 137, pp. 1293-1299, 2016/11/20/ 2016, doi: https://doi.org/10.1016/j.jclepro.2016.07.050.

[31] C. Salvi Malacarne, M. Rubens Cardoso da Silva, S. Danieli, V. Gonçalves Maciel, and A. Paula Kirchheim, "Environmental and technical assessment to support sustainable strategies for limestone calcined clay cement production in Brazil," Construction and Building Materials, vol. 310, 2021, doi: 10.1016/j.conbuildmat.2021.125261.

[32] S. Sánchez Berriel et al., "Assessing the environmental and economic potential of Limestone Calcined Clay Cement in Cuba," Journal of Cleaner Production, vol. 124, pp. 361-369, 2016, doi: 10.1016/j.jclepro.2016.02.125.

[33] Y. C. Díaz et al., "Limestone calcined clay cement as a low-carbon solution to meet expanding cement demand in emerging economies," (in English), Development Engineering, vol. 2, 2017, doi: 10.1016/j.deveng.2017.06.001.

[34] M. M. H. Khan, J. Havukainen, and M. Horttanainen, "Impact of utilising solid recovered fuel on the global warming potential of cement production and waste management system: A life cycle assessment approach," Waste Manag Res, vol. 39, no. 4, pp. 561-572, Apr 2021, doi: 10.1177/0734242X20978277.

[35] S. P. Holt and N. D. Berge, "Life-cycle assessment of using liquid hazardous waste as an alternative energy source during Portland cement manufacturing: A United States case study," Journal of Cleaner Production, vol. 195, pp. 1057-1068, 2018/09/10/ 2018, doi: https://doi.org/10.1016/j.jclepro.2018.05.214.

[36] C. Valderrama, R. Granados, J. L. Cortina, C. M. Gasol, M. Guillem, and A. Josa, "Implementation of best available techniques in cement manufacturing: a life-cycle assessment study," Journal of Cleaner Production, vol. 25, pp. 60-67, 2012, doi: 10.1016/j.jclepro.2011.11.055.

[37] M. Georgiopoulou and G. Lyberatos, "Life cycle assessment of the use of alternative fuels in cement kilns: A case study," J Environ Manage, vol. 216, pp. 224-234, Jun 15 2018, doi: 10.1016/j.jenvman.2017.07.017.

[38] S. Suárez Silgado, L. Calderón Valdiviezo, S. Gassó Domingo, and X. Roca, "Multi-criteria decision analysis to assess the environmental and economic performance of using recycled gypsum cement and recycled aggregate to produce concrete: The case of Catalonia (Spain)," Resources, Conservation and Recycling, vol. 133, pp. 120-131, 2018/06/01/ 2018, doi: https://doi.org/10.1016/j.resconrec.2017.11.023.

[39] H. Hafez, D. Kassim, R. Kurda, R. V. Silva, and J. de Brito, "Assessing the sustainability potential of alkali-activated concrete from electric arc furnace slag using the ECO_2 framework," (in English), Construction and Building Materials, vol. 281, 2021, doi: 10.1016/j.conbuildmat.2021.122559.

[40] D. Burchart-Korol, "Life cycle assessment of steel production in Poland: a case study," Journal of Cleaner Production, Article vol. 54, pp. 235-243, 2013, doi: 10.1016/j.jclepro.2013.04.031.

[41] P. A. Renzulli, B. Notarnicola, G. Tassielli, G. Arcese, and R. Di Capua, "Life cycle assessment of steel produced in an Italian integrated steel mill," (in English), Sustainability (Switzerland), Article vol. 8, no. 8, 2016, Art no. 719, doi: 10.3390/su8080719.

[42] A. M. Iosif, F. Hanrot, J. P. Birat, and D. Ablitzer, "Physicochemical modelling of the classical steelmaking route for life cycle inventory analysis," (in English), International Journal of Life Cycle Assessment, Article vol. 15, no. 3, pp. 304-310, 2010, doi: 10.1007/s11367-010-0160-y.

[43] V. Strezov, X. Zhou, and T. J. Evans, "Life cycle impact assessment of metal production industries in Australia," (in English), Scientific Reports, Article vol. 11, no. 1, 2021, Art no. 10116, doi: 10.1038/s41598-021-89567-9.

[44] T. Kuramochi, "Assessment of midterm CO_2 emissions reduction potential in the iron and steel industry: a case of Japan," Journal of Cleaner Production, vol. 132, pp. 81-97, 2016, doi: 10.1016/j.jclepro.2015.02.055.

[45] A. Özdemir, Z. Günkaya, A. Özkan, O. Ersen, M. Bilgiç, and M. Banar, "Lifecycle assessment of steel rebar production with induction melting furnace: Case study in Turkey," (in English), Journal of Hazardous, Toxic, and Radioactive Waste, Article vol. 22, no. 2, 2018, Art no. 04017027, doi: 10.1061/(ASCE)HZ.2153-5515.0000385.

[46] C. O. Vadenbo, M. E. Boesch, and S. Hellweg, "Life Cycle Assessment Model for the Use of Alternative Resources in Ironmaking," (in English), JOURNAL OF INDUSTRIAL ECOLOGY, vol. 17, no. 3, pp. 363-374, JUN 2013, doi: 10.1111/j.1530-9290.2012.00543.x.

[47] G. M. Olmez, F. B. Dilek, T. Karanfil, and U. Yetis, "The environmental impacts of iron and steel industry: A life cycle assessment study," (in English), Journal of Cleaner Production, Article vol. 130, pp. 195-201, 2016, doi: 10.1016/j.jclepro.2015.09.139.

[48] M. Haupt, C. Vadenbo, C. Zeltner, and S. Hellweg, "Influence of Input-Scrap Quality on the Environmental Impact of Secondary Steel Production," Journal of Industrial Ecology, vol. 21, no. 2, pp. 391-401, 2017, doi: 10.1111/jiec.12439.

[49] R. Tongpool, A. Jirajariyavech, C. Yuvaniyama, and T. Mungcharoen, "Analysis of steel production in Thailand: Environmental impacts and solutions," (in English), Energy, vol. 35, no. 10, pp. 4192-4200, OCT 2010, doi: 10.1016/j.energy.2010.07.003.

[50] J. Y. Hu, F. Gao, Z. H. Wang, and X. Z. Gong, "Life Cycle Assessment of Steel Production," Materials Science Forum, vol. 787, pp. 102-105, 2014, doi: 10.4028/www.scientific.net/MSF.787.102.

[51] S. M. M. Rahman, R. M. Handler, and A. L. Mayer, "Life cycle assessment of steel in the ship recycling industry in Bangladesh," (in English), Journal of Cleaner Production, Article vol. 135, pp. 963-971, 2016, doi: 10.1016/j.jclepro.2016.07.014.

[52] I. Zygomalas, E. Efthymiou, C. Baniotopoulos, and R. Blok, "A newly developed life cycle inventory (LCI) database for commonly used structural steel components," (in English), STRUCTURE AND INFRASTRUCTURE ENGINEERING, vol. 8, no. 12, pp. 1173-1181, 2012, doi: 10.1080/15732479.2010.519711.

[53] C. Walsh and P. Thornley, "The environmental impact and economic feasibility of introducing an organic Rankine cycle to recover low grade heat during the production of metallurgical coke," Journal of Cleaner Production, vol. 34, pp. 29-37, 2012, doi: 10.1016/j.jclepro.2011.12.024.

[54] J. Mitterpach, E. Hroncova, J. Ladomersky, and K. Balco, "Environmental evaluation of grey cast iron via life cycle assessment," (in English), JOURNAL OF CLEANER PRODUCTION, vol. 148, pp. 324-335, APR 1 2017, doi: 10.1016/j.jclepro.2017.02.023.

[55] Z. P. Li, X. H. Fan, G. M. Yang, J. C. Wei, Y. Sun, and M. Wang, "Life Cycle Assessment of Iron Ore Sintering Process," (in English), Journal of Iron and Steel Research International, Article vol. 22, no. 6, pp. 473-477, 2015, doi: 10.1016/S1006-706X(15)30029-7.

[56] W. Lv, Z. Sun, and Z. Su, "Life cycle energy consumption and greenhouse gas emissions of iron pelletizing process in China, a case study," Journal of Cleaner Production, Article vol. 233, pp. 1314-1321, 2019, doi: 10.1016/j.jclepro.2019.06.180.

[57] R. Jing, M. W. Yasir, J. Qian, and Z. Zhang, "Assessments of greenhouse gas (GHG) emissions from stainless steel production in China using two evaluation approaches," (in English), ENVIRONMENTAL PROGRESS & SUSTAINABLE ENERGY, vol. 38, no. 1, pp. 47-55, JAN-FEB 2019, doi: 10.1002/ep.13125.

[58] C. Ji et al., "Environmental impact assessment of galvanized sheet production: a case study in Shandong Province, China," (in English), International Journal of Life Cycle Assessment, Article vol. 25, no. 4, pp. 760-770, 2020, doi: 10.1007/s11367-020-01735-7.

[59] T. Norgate, N. Haque, M. Somerville, and S. Jahanshahi, "Biomass as a Source of Renewable Carbon for Iron and Steelmaking," (in English), ISIJ INTERNATIONAL, vol. 52, no. 8, pp. 1472-1481, 2012, doi: 10.2355/isijinternational.52.1472.

[60] Y. P. Lin, W. H. Wang, S. Y. Pan, C. C. Ho, C. J. Hou, and P. C. Chiang, "Environmental impacts and benefits of organic Rankine cycle power generation technology and wood pellet fuel exemplified by electric arc furnace steel industry," (in English), Applied Energy, vol. 183, pp. 369-379, DEC 1 2016, doi: 10.1016/j.apenergy.2016.08.183.

[61] L. Petrescu, D. A. Chisalita, C. C. Cormos, G. Manzolini, P. Cobden, and H. A. J. van Dijk, "Life Cycle Assessment of SEWGS Technology Applied to Integrated Steel Plants," (in English), SUSTAINABILITY, vol. 11, no. 7, APR 1 2019, Art no. 1825, doi: 10.3390/su11071825.

[62] D. A. Chisalita, L. Petrescu, P. Cobden, H. A. J. van Dijk, A. M. Cormos, and C. C. Cormos, "Assessing the environmental impact of an integrated steel mill with post-combustion CO_2 capture and storage using the LCA methodology," (in English), JOURNAL OF CLEANER PRODUCTION, vol. 211, pp. 1015-1025, FEB 20 2019, doi: 10.1016/j.jclepro.2018.11.256.

[63] F. Li et al., "Quantifying the energy saving potential and environmental benefit of hydrogen-based steelmaking process: Status and future prospect," (in English), APPLIED THERMAL ENGINEERING, vol. 211, JUL 5 2022, Art no. 118489, doi: 10.1016/j.applthermaleng.2022.118489.

[64] V. Niccolucci, B. Rugani, and S. Bastianoni, "Life Cycle Assessment (LCA) combined with EMergy evaluation for a better understanding of the environmental aspects associated with a crystal glass supply chain," (in English), ECOSYSTEMS AND SUSTAINABLE DEVELOPMENT VII, vol. 122, 2009, doi: 10.2495/ECO090221.

[65] R. M. Pulselli, R. Ridolfi, B. Rugani, and E. Tiezzi, "Application of life cycle assessment to the production of man-made crystal glass," (in English), INTERNATIONAL JOURNAL OF LIFE CYCLE ASSESSMENT, vol. 14, no. 5, 2009, doi: 10.1007/s11367-009-0085-5.

[66] T. Gallucci, G. Lagioia, P. Piccinno, A. Lacalamita, A. Pontrandolfo, and A. Paiano, "Environmental performance scenarios in the production of hollow glass containers for food packaging: an LCA approach," (in English), INTERNATIONAL JOURNAL OF LIFE CYCLE ASSESSMENT, vol. 26, no. 4, 2021, doi: 10.1007/s11367-020-01797-7.

[67] S. Caserini et al., "Evaluation of a new technology for carbon dioxide submarine storage in glass capsules," (in English), International Journal of Greenhouse Gas Control, vol. 60, 2017, doi: 10.1016/j.ijggc.2017.03.007.

[68] C. B. Cunha, M. Brondani, F. D. Mayer, P. P. Lopes, and R. Hoffmann, ""Low-cost small-scale distillation column: assessment of polymeric materials on its economic, chemical, mechanical, and environmental performance"," (in English), Clean Technologies and Environmental Policy, vol. 22, no. 7, 2020, doi: 10.1007/s10098-020-01897-w.

[69] B. Zegardlo and K. Druzba, "Evaluation of Traditional Container Glass Recycling Systems against Selected Environmental Impact Criteria Using the LCA Method," (in English), JOURNAL OF ECOLOGICAL ENGINEERING, vol. 22, no. 6, 2021, doi: 10.12911/22998993/137673.

[70] A. E. Schwarz, T. N. Ligthart, D. Godoi Bizarro, P. De Wild, B. Vreugdenhil, and T. van Harmelen, "Plastic recycling in a circular economy; determining environmental performance through an LCA matrix model approach," Waste Management, vol. 121, pp. 331-342, 2021/02/15/ 2021, doi: https://doi.org/10.1016/j.wasman.2020.12.020.

[71] J. Santos, A. Pham, P. Stasinopoulos, and F. Giustozzi, "Recycling waste plastics in roads: A life-cycle assessment study using primary data," Science of The Total Environment, vol. 751, p. 141842, 2021/01/10/ 2021, doi: https://doi.org/10.1016/j.scitotenv.2020.141842.

[72] K. G. Harding, J. S. Dennis, H. von Blottnitz, and S. T. L. Harrison, "Environmental analysis of plastic production processes: Comparing petroleum-based polypropylene and polyethylene with biologically-based poly-β-hydroxybutyric acid using life cycle analysis," Journal of Biotechnology, vol. 130, no. 1, pp. 57-66, 2007/05/31/ 2007, doi: https://doi.org/10.1016/j.jbiotec.2007.02.012.

[73] P. Baldowska-Witos, K. Doerffer, M. Pysz, P. Doerffer, A. Tomporowski, and M. Opielak, "Manufacturing and Recycling Impact on Environmental Life Cycle Assessment of Innovative Wind Power Plant Part 2/2," (in English), MATERIALS, vol. 14, no. 1, JAN 2021, Art no. 204, doi: 10.3390/ma14010204.

[74] R. A. Witik, F. Gaille, R. Teuscher, H. Ringwald, V. Michaud, and J.-A. E. Månson, "Economic and environmental assessment of alternative production methods for composite aircraft components," Journal of Cleaner Production, vol. 29-30, pp. 91-102, 2012/07/01/ 2012, doi: https://doi.org/10.1016/j.jclepro.2012.02.028.

[75] G. David et al., "Using life cycle assessment to quantify the environmental benefit of upcycling vine shoots as fillers in biocomposite packaging materials," The International Journal of Life Cycle Assessment, vol. 26, no. 4, pp. 738-752, 2021/04/01 2021, doi: 10.1007/s11367-020-01824-7.

[76] D. Koch and B. Mihalyi, " Assessing the Change in Environmental Impact Categories when Replacing Conventional Plastic with Bioplastic in Chosen Application Fields," Chemical Engineering Transactions, vol. 70, pp. 853-858, 2018, doi: https://doi.org/10.3303/CET1870143.

[77] R. Quirós et al., "The application of LCA to alternative methods for treating the organic fiber produced from autoclaving unsorted municipal solid waste: case study of Catalonia," Journal of Cleaner Production, vol. 107, pp. 516-528, 2015/11/16/ 2015, doi: https://doi.org/10.1016/j.jclepro.2014.04.018.

[78] M. Rybaczewska-Błażejowska and A. Mena-Nieto, "Circular economy: comparative life cycle assessment of fossil polyethylene terephthalate (PET)and its recycled and bio-based counterparts," Management and Production Engineering Review, vol. vol. 11, no. No 4, 30.12.2020 2020, doi: 10.24425/mper.2020.136126.

[79] C. Chen, C. Li, Y. Zhou, L. Sui, and X. Li, "Applying Flax FRP in an Innovative Closed-Shape Stirrup for Concrete Beams," Materials, vol. 15, no. 8, 2022, doi: 10.3390/ma15082927.

[80] ISO 21930:2017 Sustainability in buildings and civil engineering works — Core rules for environmental product declarations of construction products and services, International Organization for Standardization, British Standards Online, 2017.

[81] Product Category Rules for Construction Products, EPD International, 2019.

[82] L. Beressa and B. V. Saradhi, "Life Cycle Assessment of Ethiopian Cement Manufacturing: A Potential Improvement on the Use of Fossil Fuel in Mugher Cement Factory," Applied Environmental Research, pp. 100-112, 2021, doi: 10.35762/aer.2021.43.4.8.

[83] R. B. Meshram and S. Kumar, "Comparative life cycle assessment (LCA) of geopolymer cement manufacturing with Portland cement in Indian context," International Journal of Environmental Science and Technology, vol. 19, no. 6, pp. 4791-4802, 2021, doi: 10.1007/s13762-021-03336-9.

[84] R. Feiz, J. Ammenberg, L. Baas, M. Eklund, A. Helgstrand, and R. Marshall, "Improving the CO_2 performance of cement, part I: utilising life-cycle assessment and key performance indicators to assess development within the cement industry," (in English), JOURNAL OF CLEANER PRODUCTION, vol. 98, pp. 272-281, JUL 1 2015, doi: 10.1016/j.jclepro.2014.01.083.

[85] E. Batuecas, I. Ramón-Álvarez, S. Sánchez-Delgado, and M. Torres-Carrasco, "Carbon footprint and water use of alkali-activated and hybrid cement mortars," Journal of Cleaner Production, vol. 319, 2021, doi: 10.1016/j.jclepro.2021.128653.

[86] M. R. Irshidat, N. Al-Nuaimi, W. Ahmed, and M. Rabie, "Feasibility of recycling waste carbon black in cement mortar production: Environmental life cycle assessment and performance evaluation," Construction and Building Materials, vol. 296, 2021, doi: 10.1016/j.conbuildmat.2021.123740.

[87] N. M. Alderete, A. M. Joseph, P. Van den Heede, S. Matthys, and N. De Belie, "Effective and sustainable use of municipal solid waste incineration bottom ash in concrete regarding strength and durability," Resources, Conservation and Recycling, vol. 167, 2021, doi: 10.1016/j.resconrec.2020.105356.

[88] F. Soltanzadeh, A. E. Behbahani, E. N. B. Pereira, and C. A. Teixeira, "A life-cycle approach to integrate environmental and mechanical properties of blended cements containing seashell powder," (in English), Sustainability (Switzerland), vol. 13, no. 23, 2021, doi: 10.3390/su132313120.

[89] L. Nguyen, A. J. Moseson, Y. Farnam, and S. Spatari, "Effects of composition and transportation logistics on environmental, energy and cost metrics for the production of alternative cementitious binders," Journal of Cleaner Production, vol. 185, pp. 628-645, 2018, doi: 10.1016/j.jclepro.2018.02.247.

[90] T. E. Norgate, S. Jahanshahi, and W. J. Rankin, "Assessing the environmental impact of metal production processes," Journal of Cleaner Production, Article vol. 15, no. 8/9, pp. 838-848, 2007, doi: 10.1016/j.jclepro.2006.06.018.

[91] Y. Kahlid et al., "Oxygen enrichment combustion to reduce fossil energy consumption and emissions in hot rolling steel production," Journal of Cleaner Production, vol. 320, 2021, doi: 10.1016/j.jclepro.2021.128714.

[92] B. Bieda, "Life cycle inventory processes of the Mittal Steel Poland (MSP) S.A. in Krakow, Poland-blast furnace pig iron production-a case study," (in English), International Journal of Life Cycle Assessment, vol. 17, no. 6, pp. 787-794, JUL 2012, doi: 10.1007/s11367-012-0422-y.

[93] B. Bieda, "Life cycle inventory processes of the ArcelorMittal Poland (AMP) S.A. in Kraków, Poland-basic oxygen furnace steel production," (in English), International Journal of Life Cycle Assessment, Article vol. 17, no. 4, pp. 463-470, 2012, doi: 10.1007/s11367-011-0370-y.

[94] M. Messagie, F. Boureima, J. Mertens, J. Sanfelix, C. Macharis, and J. Van Mierlo, "The Influence of Allocation on the Carbon Footprint of Electricity Production from Waste Gas, a Case Study for Blast Furnace Gas," (in English), Energies, vol. 6, no. 3, pp. 1217-1232, MAR 2013, doi: 10.3390/en6031217.

[95] Y. Tong et al., "Life cycle water use and wastewater discharge of steel production based on material-energy-water flows: A case study in China," Journal of Cleaner Production, Article vol. 241, 2019, doi: 10.1016/j.jclepro.2019.118410.

[96] I. Tsiropoulos, A. P. C. Faaij, L. Lundquist, U. Schenker, J. F. Briois, and M. K. Patel, "Life cycle impact assessment of bio-based plastics from sugarcane ethanol," Journal of Cleaner Production, vol. 90, pp. 114-127, 2015/03/01/ 2015, doi: https://doi.org/10.1016/j.jclepro.2014.11.071.

[97] R. Vidal et al., "Environmental assessment of biodegradable multilayer film derived from carbohydrate polymers," Journal of Polymers and the Environment, vol. 15, no. 3, pp. 159-168, 2007/07/01 2007, doi: 10.1007/s10924-007-0056-5.

[98] A. Brock and I. Williams, "Life Cycle Assessment of Beverage Packaging," Detritus, vol. 13, pp. 47-61, 10/05 2020, doi: 10.31025/2611-4135/2020.14025.

[99] H. H. Khoo, R. B. H. Tan, and K. W. L. Chng, "Environmental impacts of conventional plastic and bio-based carrier bags," The International Journal of Life Cycle Assessment, vol. 15, no. 3, pp. 284-293, 2010/03/01 2010, doi: 10.1007/s11367-010-0162-9.

[100] A. Furberg, R. Arvidsson, and S. Molander, "A practice-based framework for defining functional units in comparative life cycle assessments of materials," Journal of Industrial Ecology, vol. 26, no. 3, pp. 718-730, 2022, doi: https://doi.org/10.1111/jiec.13218.

[101] F. Colangelo, A. Forcina, I. Farina, and A. Petrillo, "Life Cycle Assessment (LCA) of Different Kinds of Concrete Containing Waste for Sustainable Construction," Buildings, vol. 8, no. 5, p. 70, 2018, doi: 10.3390/buildings8050070.

[102] E. R. Teixeira, R. Mateus, A. F. Camões, L. Bragança, and F. G. Branco, "Comparative environmental life-cycle analysis of concretes using biomass and coal fly ashes as partial cement replacement material," Journal of Cleaner Production, vol. 112, pp. 2221-2230, 2016, doi: 10.1016/j.jclepro.2015.09.124.

[103] O. E. Ige, O. A. Olanrewaju, K. J. Duffy, and O. C. Collins, "Environmental Impact Analysis of Portland Cement (CEM1) Using the Midpoint Method," (in English), Energies, vol. 15, no. 7, 2022, doi: 10.3390/en15072708.

[104] M. A. Putra, K. C. Teh, J. Tan, and T. S. Y. Choong, "Sustainability assessment of Indonesian cement manufacturing via integrated life cycle assessment and analytical hierarchy process method," Environ Sci Pollut Res Int, vol. 27, no. 23, pp. 29352-29360, Aug 2020, doi: 10.1007/s11356-020-09207-z.

[105] C. Li, S. P. Cui, Z. R. Nie, X. Z. Gong, Z. H. Wang, and N. Itsubo, "The LCA of portland cement production in China," (in English), INTERNATIONAL JOURNAL OF LIFE CYCLE ASSESSMENT, vol. 20, no. 1, 2015, doi: 10.1007/s11367-014-0804-4.

[106] K. C. Gomes, M. Carvalho, D. D. Diniz, R. D. C. Abrantes, M. A. Branco, and P. R. O. de Carvalho, "Carbon emissions associated with two types of foundations: CP-II Portland cement-based composite vs. geopolymer concrete," (in English), MATERIA-RIO DE JANEIRO, vol. 24, no. 4, 2019, doi: 10.1590/S1517-707620190004.0850.

[107] K. Dahlström and P. Ekins, "Combining economic and environmental dimensions: Value chain analysis of UK iron and steel flows," (in English), Ecological Economics, Article vol. 58, no. 3, pp. 507-519, 2006, doi: 10.1016/j.ecolecon.2005.07.024.

[108] J. G. Backes, J. Suer, N. Pauliks, S. Neugebauer, and M. Traverso, "Life Cycle Assessment of an Integrated Steel Mill Using Primary Manufacturing Data: Actual Environmental Profile," (in English), SUSTAINABILITY, vol. 13, no. 6, MAR 2021, Art no. 3443, doi: 10.3390/su13063443.

[109] Z. Qi, C. Gao, H. Na, and Z. Ye, "Using forest area for carbon footprint analysis of typical steel enterprises in China," (in English), Resources, Conservation and Recycling, Article vol. 132, pp. 352-360, 2018, doi: 10.1016/j.resconrec.2017.05.016.

[110] C. Wang, X. Z. Zheng, W. J. Cai, X. Gao, and P. Berrill, "Unexpected water impacts of energy-saving measures in the iron and steel sector: Tradeoffs or synergies?," (in English), APPLIED ENERGY, vol. 205, pp. 1119-1127, NOV 1 2017, doi: 10.1016/j.apenergy.2017.08.125.

[111] Y. Gu et al., "Calculation of water footprint of the iron and steel industry: A case study in Eastern China," Journal of Cleaner Production, vol. 92, pp. 274-281, 2015, doi: 10.1016/j.jclepro.2014.12.094.

[112] H. Liu, Q. Li, R. Ding, and G. Li, "Environmental and economic impact assessment of converter steelmaking based on life cycle assessment," (in English), Int. J. Innov. Comput. Inf. Control, Article vol. 17, no. 2, pp. 499-509, 2021, doi: 10.24507/ijicic.17.02.499.

[113] S. E. Tanzer, K. Blok, and A. Ramirez, "Can bioenergy with carbon capture and storage result in carbon negative steel?," (in English), INTERNATIONAL JOURNAL OF GREENHOUSE GAS CONTROL, vol. 100, SEP 2020, Art no. 103104, doi: 10.1016/j.ijggc.2020.103104.

[114] S. Vinodh, K. Jayakrishna, and D. Joy, "Environmental impact assessment of an automotive component using eco-indicator and CML methodologies," Clean Technologies and Environmental Policy, vol. 14, no. 2, pp. 333-344, 2012/04/01 2012, doi: 10.1007/s10098-011-0405-x.

[115] Y. Liu, Y. Liu, and J. Chen, "The impact of the Chinese automotive industry: scenarios based on the national environmental goals," Journal of Cleaner Production, Article vol. 96, pp. 102-109, 2015, doi: 10.1016/j.jclepro.2014.05.015.

[116] K. Vadoudi, P. Deckers, C. Demuytere, H. Askanian, and V. Verney, "Comparing a material circularity indicator to life cycle assessment: The case of a three-layer plastic packaging," Sustainable Production and Consumption, vol. 33, pp. 820-830, 2022/09/01/ 2022, doi: https://doi.org/10.1016/j.spc.2022.08.004.

[117] I. J. Gatt and P. Refalo, "Reusability and recyclability of plastic cosmetic packaging: A life cycle assessment," Resources, Conservation & Recycling Advances, vol. 15, p. 200098, 2022/11/01/ 2022, doi: https://doi.org/10.1016/j.rcradv.2022.200098.

[118] M. Samer et al., "Environmental impact assessment of bioplastics production from agricultural crop residues," Clean Technologies and Environmental Policy, vol. 24, no. 3, pp. 815-827, 2022/04/01 2022, doi: 10.1007/s10098-021-02145-5.

[119] S. Lee et al., "Static and dynamic flow analysis of PBDEs in plastics from used and end-of-life TVs and computer monitors by life cycle in Korea," Science of The Total Environment, vol. 506-507, pp. 76-85, 2015/02/15/ 2015, doi: https://doi.org/10.1016/j.scitotenv.2014.10.116.

[120] P. Loubet, J. Couturier, R. Horta Arduin, and G. Sonnemann, "Life cycle inventory of plastics losses from seafood supply chains: Methodology and application to French fish products," Science of The Total Environment, vol. 804, p. 150117, 2022/01/15/ 2022, doi: https://doi.org/10.1016/j.scitotenv.2021.150117.

[121] A. Ahamed, A. Veksha, K. Yin, P. Weerachanchai, A. Giannis, and G. Lisak, "Environmental impact assessment of converting flexible packaging plastic waste to pyrolysis oil and multi-walled carbon nanotubes," Journal of Hazardous Materials, vol. 390, p. 121449, 2020/05/15/ 2020, doi: https://doi.org/10.1016/j.jhazmat.2019.121449.

[122] L. Tähkämö, A. Ojanperä, J. Kemppi, and I. Deviatkin, "Life cycle assessment of renewable liquid hydrocarbons, propylene, and polypropylene derived from bio-based waste and residues: Evaluation of climate change impacts and abiotic resource depletion potential," Journal of Cleaner Production, vol. 379, p. 134645, 2022/12/15/ 2022, doi: https://doi.org/10.1016/j.jclepro.2022.134645.

[123] P. A. Wäger and R. Hischier, "Life cycle assessment of post-consumer plastics production from waste electrical and electronic equipment (WEEE) treatment residues in a Central European plastics recycling

plant," Science of The Total Environment, vol. 529, pp. 158-167, 2015/10/01/ 2015, doi: https://doi.org/10.1016/j.scitotenv.2015.05.043.

[124] A. Somoza-Tornos, A. Gonzalez-Garay, C. Pozo, M. Graells, A. Espuña, and G. Guillén-Gosálbez, "Realizing the Potential High Benefits of Circular Economy in the Chemical Industry: Ethylene Monomer Recovery via Polyethylene Pyrolysis," ACS Sustainable Chemistry & Engineering, vol. 8, no. 9, pp. 3561-3572, 2020/03/09 2020, doi: 10.1021/acssuschemeng.9b04835.

[125] S. S. Muthu, Y. Li, J. Y. Hu, P. Y. Mok, and X. Ding, "Eco-Impact of Plastic and Paper Shopping Bags," Journal of Engineered Fibers and Fabrics, vol. 7, no. 1, p. 155892501200700103, 2012/03/01 2012, doi: 10.1177/155892501200700103.

[126] C. Chen, G. Habert, Y. Bouzidi, A. Jullien, and A. Ventura, "LCA allocation procedure used as an incitative method for waste recycling: An application to mineral additions in concrete," Resources, Conservation and Recycling, vol. 54, no. 12, pp. 1231-1240, 2010, doi: 10.1016/j.resconrec.2010.04.001.

[127] H. Robinson et al., "BFS: a technical report on manufacturing of BFS & material status in UK," Environmental Agency: Waste Protocols Project, 2007.

[128] EU (2008). European Union Directive 2008/98/EC of the European parliament and of the council on waste and repealing certain directives.

[129] S. H. Teh, T. Wiedmann, A. Castel, and J. de Burgh, "Hybrid life cycle assessment of greenhouse gas emissions from cement, concrete and geopolymer concrete in Australia," Journal of cleaner production, vol. 152, pp. 312-320, 2017.

[130] M. U. Hossain, C. S. Poon, I. M. C. Lo, and J. C. P. Cheng, "Comparative LCA on using waste materials in the cement industry: A Hong Kong case study," Resources, Conservation and Recycling, vol. 120, pp. 199-208, 2017, doi: 10.1016/j.resconrec.2016.12.012.

[131] Y. Li et al., "Environmental impact analysis of blast furnace slag applied to ordinary Portland cement production," Journal of Cleaner Production, vol. 120, pp. 221-230, 2016, doi: 10.1016/j.jclepro.2015.12.071.

[132] H. Hafez, R. Kurda, W. M. Cheung, and B. Nagaratnam, "Comparative life cycle assessment between imported and recovered fly ash for blended cement concrete in the UK," Journal of Cleaner Production, vol. 244, 2020, doi: 10.1016/j.jclepro.2019.118722.

[133] G. Habert, "A method for allocation according to the economic behaviour in the EU-ETS for by-products used in cement industry," (in English), International Journal of Life Cycle Assessment, vol. 18, no. 1, 2013, doi: 10.1007/s11367-012-0464-1.

[134] G. Fick, O. Mirgaux, P. Neau, and F. Patisson, "Using biomass for pig iron production: A technical, environmental and economical assessment," (in English), Waste and Biomass Valorization, Article vol. 5, no. 1, pp. 43-55, 2014, doi: 10.1007/s12649-013-9223-1.

[135] X. Ma, L. Ye, C. Qi, D. Yang, X. Shen, and J. Hong, "Life cycle assessment and water footprint evaluation of crude steel production: A case study in China," Journal of environmental management, vol. 224, pp. 10-18, 2018, doi: 10.1016/j.jenvman.2018.07.027.

[136] J. Suer, M. Traverso, and F. Ahrenhold, "Carbon footprint of scenarios towards climate-neutral steel according to ISO 14067," (in English), JOURNAL OF CLEANER PRODUCTION, vol. 318, OCT 10 2021, Art no. 128588, doi: 10.1016/j.jclepro.2021.128588.

[137] J. M. Moon, J. H. Eun, and J. S. Chung, "Allocation of process gases generated from integrated steelworks by an improved system expansion method," (in English), International Journal of Life Cycle Assessment, Article vol. 11, no. 4, pp. 247-253, 2006, doi: 10.1065/lca2004.12.197.

[138] B. Chen, J. X. Yang, and Z. Y. Ouyang, "Life cycle assessment of internal recycling options of steel slag in Chinese iron and steel industry," (in English), Journal of Iron and Steel Research International, Article vol. 18, no. 7, pp. 33-40, 2011, doi: 10.1016/S1006-706X(11)60087-3.

[139] H. Kim, S. Lee, Y. Ahn, J. Lee, and W. Won, "Sustainable Production of Bioplastics from Lignocellulosic Biomass: Technoeconomic Analysis and Life-Cycle Assessment," ACS Sustainable Chemistry & Engineering, vol. 8, no. 33, pp. 12419-12429, 2020/08/24 2020, doi: 10.1021/acssuschemeng.0c02872.

[140] E. Benetto et al., "Using atmospheric plasma to design multilayer film from polylactic acid and thermoplastic starch: a screening Life Cycle Assessment," Journal of Cleaner Production, vol. 87, pp. 953-960, 2015/01/15/ 2015, doi: https://doi.org/10.1016/j.jclepro.2014.10.056.

[141] T. Blazejewski, S. R. J. Walker, R. I. Muazu, and R. H. Rothman, "Reimagining the milk supply chain: Reusable vessels for bulk delivery," (in English), SUSTAINABLE PRODUCTION AND CONSUMPTION, vol. 27, pp. 1030-1046, JUL 2021, doi: 10.1016/j.spc.2021.02.030.

[142] H. Jeswani et al., "Life cycle environmental impacts of chemical recycling via pyrolysis of mixed plastic waste in comparison with mechanical recycling and energy recovery," Science of The Total Environment, vol. 769, p. 144483, 2021/05/15/ 2021, doi: https://doi.org/10.1016/j.scitotenv.2020.144483.

[143] S. Islam, S. G. Ponnambalam, and H. L. Lam, "Review on life cycle inventory: methods, examples and applications," Journal of Cleaner Production, vol. 136, pp. 266-278, 2016, doi: 10.1016/j.jclepro.2016.05.144.

[144] R. H. Crawford, P.-A. Bontinck, A. Stephan, T. Wiedmann, and M. Yu, "Hybrid life cycle inventory methods – A review," Journal of Cleaner Production, vol. 172, pp. 1273-1288, 2018-01-01 2018, doi: 10.1016/j.jclepro.2017.10.176.

[145] Y. Ju and K. Fujikawa, "Revealing the impact of a projected emission trading scheme on the production technology upgrading in the cement industry in China: An LCA-RCOT model," (in English), "Resources, Conservation and Recycling: X", vol. 4, 2019, doi: 10.1016/j.rcrx.2019.100019.

[146] E. J. Prosman and R. Sacchi, "New environmental supplier selection criteria for circular supply chains: Lessons from a consequential LCA study on waste recovery," (in English), Journal of Cleaner Production, vol. 172, 2016, doi: 10.1016/j.jclepro.2017.11.134.

[147] W. Chen, J. Hong, and C. Xu, "Pollutants generated by cement production in China, their impacts, and the potential for environmental improvement," Journal of Cleaner Production, vol. 103, pp. 61-69, 2015, doi: 10.1016/j.jclepro.2014.04.048.

[148] L. Ren, S. Zhou, and X. Ou, "The carbon reduction potential of hydrogen in the low carbon transition of the iron and steel industry: The case of China," Renewable and Sustainable Energy Reviews, vol. 171, 2023, doi: 10.1016/j.rser.2022.113026.

[149] I. Yi, N. Itsubo, A. Inaba, and K. Matsumoto, "Development of the interregional I/O based LCA method considering region-specifics of indirect effects in regional evaluation," (in English), INTERNATIONAL JOURNAL OF LIFE CYCLE ASSESSMENT, vol. 12, no. 6, pp. 353-364, SEP 2007, doi: 10.1065/lca2007.06.339.

[150] R. Volk et al., "Techno-economic assessment and comparison of different plastic recycling pathways: A German case study," Journal of Industrial Ecology, vol. 25, pp. 1-20, 04/28 2021, doi: 10.1111/jiec.13145.

[151] V. Kočí, "Comparisons of environmental impacts between wood and plastic transport pallets," Science of The Total Environment, vol. 686, pp. 514-528, 2019/10/10/ 2019, doi: https://doi.org/10.1016/j.scitotenv.2019.05.472.

[152] G. M. Zanghelini, E. Cherubini, R. Dias, Y. H. O. Kabe, and J. J. S. Delgado, "Comparative life cycle assessment of drinking straws in Brazil," Journal of Cleaner Production, Article vol. 276, pp. N.PAG-N.PAG, 2020, doi: 10.1016/j.jclepro.2020.123070.

[153] V. Kek and V. S, "LCA Integrated ANP Framework for Selection of Sustainable Manufacturing Processes," Environmental Modeling & Assessment, vol. 21, no. 4, pp. 507-516, 2016/08/01 2016, doi: 10.1007/s10666-015-9490-2.

[154] A. A. M. Ali, A. M. Negm, M. F. Bady, M. G. E. Ibrahim, and M. Suzuki, "Environmental impact assessment of the Egyptian cement industry based on a life-cycle assessment approach: a comparative study between

Egyptian and Swiss plants," Clean Technologies and Environmental Policy, vol. 18, no. 4, pp. 1053-1068, 2016, doi: 10.1007/s10098-016-1096-0.

[155] F. N. Stafford, A. C. Dias, L. Arroja, J. A. Labrincha, and D. Hotza, "Life cycle assessment of the production of Portland cement: a Southern Europe case study," Journal of Cleaner Production, vol. 126, pp. 159-165, 2016, doi: 10.1016/j.jclepro.2016.02.110.

[156] E. Thwe, D. Khatiwada, and A. Gasparatos, "Life cycle assessment of a cement plant in Naypyitaw, Myanmar," Cleaner Environmental Systems, vol. 2, 2021, doi: 10.1016/j.cesys.2020.100007.

[157] Z. Pavlík et al., "Energy-efficient thermal treatment of sewage sludge for its application in blended cements," (in English), Journal of Cleaner Production, vol. 112, 2016, doi: 10.1016/j.jclepro.2015.09.072.

[158] S. Ruan and C. Unluer, "Comparative life cycle assessment of reactive MgO and Portland cement production," Journal of Cleaner Production, vol. 137, pp. 258-273, 2016, doi: 10.1016/j.jclepro.2016.07.071.

[159] D. García-Gusano, D. Garraín, I. Herrera, H. Cabal, and Y. Lechón, "Life Cycle Assessment of applying CO 2 post-combustion capture to the Spanish cement production," Journal of Cleaner Production, vol. 104, pp. 328-338, 2015, doi: 10.1016/j.jclepro.2013.11.056.

[160] M. E. Boesch, A. Koehler, and S. Hellweg, "Model for Cradle-to-Gate Life Cycle Assessment of Clinker Production," (in English), ENVIRONMENTAL SCIENCE & TECHNOLOGY, vol. 43, no. 19, 2009, doi: 10.1021/es900036e.

[161] C. Moro, V. Francioso, M. Schrager, and M. Velay-Lizancos, "TiO2 nanoparticles influence on the environmental performance of natural and recycled mortars: A life cycle assessment," (in English), Environmental Impact Assessment Review, vol. 84, 2020, doi: 10.1016/j.eiar.2020.106430.

[162] K. R. Bawden, E. D. Williams, and C. W. Babbitt, "Mapping product knowledge to life cycle inventory bounds: A case study of steel manufacturing," Journal of Cleaner Production, vol. 113, pp. 557-564, 2016, doi: 10.1016/j.jclepro.2015.10.014.

[163] S. Cecchel, D. Ferrario, A. Panvini, and G. Cornacchia, "Lightweight of a cross beam for commercial vehicles: Development, testing and validation," (in English), MATERIALS & DESIGN, vol. 149, pp. 122-134, JUL 5 2018, doi: 10.1016/j.matdes.2018.04.021.

[164] T. Liang et al., "Environmental impact evaluation of an iron and steel plant in China: Normalized data and direct/indirect contribution," Journal of Cleaner Production, Article vol. 264, pp. N.PAG-N.PAG, 2020, doi: 10.1016/j.jclepro.2020.121697.

[165] M. Papo and B. Corona, "Life cycle sustainability assessment of non-beverage bottles made of recycled High Density Polyethylene," Journal of Cleaner Production, vol. 378, p. 134442, 2022/12/10/ 2022, doi: https://doi.org/10.1016/j.jclepro.2022.134442.

[166] M. Mele, G. Campana, and G. Fumelli, "Environmental impact assessment of Arburg plastic freeforming additive manufacturing," Sustainable Production and Consumption, vol. 28, pp. 405-418, 2021/10/01/ 2021, doi: https://doi.org/10.1016/j.spc.2021.06.012.

[167] J. An, F. Wu, D. Wang, and J. You, "Estimated material metabolism and life cycle greenhouse gas emission of major plastics in China: A commercial sector-scale perspective," Resources, Conservation and Recycling, vol. 180, p. 106161, 2022/05/01/ 2022, doi: https://doi.org/10.1016/j.resconrec.2022.106161.

[168] J. Chu, Y. Zhou, Y. Cai, X. Wang, C. Li, and Q. Liu, "Life-cycle greenhouse gas emissions and the associated carbon-peak strategies for PS, PVC, and ABS plastics in China," Resources, Conservation and Recycling, vol. 182, p. 106295, 2022/07/01/ 2022, doi: https://doi.org/10.1016/j.resconrec.2022.106295.

[169] L. Woods and B. R. Bakshi, "Reusable vs. disposable cups revisited: guidance in life cycle comparisons addressing scenario, model, and parameter uncertainties for the US consumer," The International Journal of Life Cycle Assessment, vol. 19, no. 4, pp. 931-940, 2014/04/01 2014, doi: 10.1007/s11367-013-0697-7.

[170] M. U. Hossain et al., "Evaluating the environmental impacts of stabilization and solidification technologies for managing hazardous wastes through life cycle assessment: A case study of Hong Kong," (in English), Environment International, vol. 145, 2020, doi: 10.1016/j.envint.2020.106139.

[171] C. Strazza, A. Del Borghi, M. Gallo, and M. Del Borghi, "Resource productivity enhancement as means for promoting cleaner production: Analysis of co-incineration in cement plants through a life cycle approach," (in English), Journal of Cleaner Production, vol. 19, no. 14, 2011, doi: 10.1016/j.jclepro.2011.05.014.

[172] D. K. Panesar, K. E. Seto, and C. J. Churchill, "Impact of the selection of functional unit on the life cycle assessment of green concrete," The International Journal of Life Cycle Assessment, vol. 22, no. 12, pp. 1969-1986, 2017, doi: 10.1007/s11367-017-1284-0.

[173] J. An, R. S. Middleton, and Y. Li, "Environmental performance analysis of cement production with CO_2 capture and storage technology in a life-cycle perspective," (in English), Sustainability (Switzerland), vol. 11, no. 9, 2019, doi: 10.3390/su11092626.

[174] X. Liang, W. Dang, G. Yang, and Y. Zhang, "Eco-efficient alkaline activated binders for manufacturing blocks and pedestrian pavers with low carbon footprint: Mechanical properties and LCA assessment [Materiales cementantes activados alcalinamente para la fabricaci n de bloques y adoquines peatonales con baja huella de carbono: Propiedades mec nicas y an lisis LCA.]," (in English), Materiales de Construccion, vol. 70, no. 340, 2020, doi: 10.3989/MC.2020.17419.

[175] T. Hottle et al., "Environmental life-cycle assessment of concrete produced in the United States," (in English), Journal of Cleaner Production, vol. 363, 2022, doi: 10.1016/j.jclepro.2022.131834.

[176] D. Song, J. Yang, B. Chen, T. Hayat, and A. Alsaedi, "Life-cycle environmental impact analysis of a typical cement production chain," (in English), Applied Energy, vol. 164, 2016, doi: 10.1016/j.apenergy.2015.09.003.

[177] L. Zhang and W. E. Mabee, "Comparative study on the life-cycle greenhouse gas emissions of the utilization of potential low carbon fuels for the cement industry," Journal of Cleaner Production, vol. 122, pp. 102-112, 2016, doi: 10.1016/j.jclepro.2016.02.019.

[178] O. Yilmaz, A. Anctil, and T. Karanfil, "LCA as a decision support tool for evaluation of best available techniques (BATs) for cleaner production of iron casting," (in English), JOURNAL OF CLEANER PRODUCTION, vol. 105, pp. 337-347, OCT 15 2015, doi: 10.1016/j.jclepro.2014.02.022.

[179] A. Avinal and P. Ergenekon, "Life cycle impacts of induction furnace technology for crude steel production: case study," Energy Sources, Part A: Recovery, Utilization and Environmental Effects, vol. 44, no. 4, pp. 9974-9987, 2022, doi: 10.1080/15567036.2022.2143946.

[180] A. C. Bourtsalas, J. Zhang, M. J. Castaldi, and N. J. Themelis, "Use of non-recycled plastics and paper as alternative fuel in cement production," (in English), Journal of Cleaner Production, vol. 181, 2018, doi: 10.1016/j.jclepro.2018.01.214.

[181] G. Lonca, P. Lesage, G. Majeau-Bettez, S. Bernard, and M. Margni, "Assessing scaling effects of circular economy strategies: A case study on plastic bottle closed-loop recycling in the USA PET market," Resources, Conservation and Recycling, vol. 162, p. 105013, 2020/11/01/ 2020, doi: https://doi.org/10.1016/j.resconrec.2020.105013.

[182] G. Bishop, D. Styles, and P. N. L. Lens, "Land-use change and valorisation of feedstock side-streams determine the climate mitigation potential of bioplastics," Resources, Conservation and Recycling, vol. 180, p. 106185, 2022/05/01/ 2022, doi: https://doi.org/10.1016/j.resconrec.2022.106185.

[183] K. Kuusela, V. Uusitalo, J. Ahola, and J. Levänen, "The transformation of plastics production from net positive greenhouse gas emissions to net negative: An environmental sustainability assessment of CO_2-based polypropylene," Journal of CO_2 Utilization, vol. 52, p. 101672, 2021/10/01/ 2021, doi: https://doi.org/10.1016/j.jcou.2021.101672.

[184] N. Singh, H. Duan, and Y. Tang, "Toxicity evaluation of E-waste plastics and potential repercussions for human health," Environment International, vol. 137, p. 105559, 2020/04/01/ 2020, doi: https://doi.org/10.1016/j.envint.2020.105559.

[185] J. L. Gálvez-Martos, R. Chaliulina, A. Elhoweris, J. Mwanda, H. A., and Y. Al-horr, "Techno-economic assessment of calcium sulfoaluminate clinker production using elemental sulfur as raw material," (in English), Journal of Cleaner Production, vol. 301, 2021, doi: 10.1016/j.jclepro.2021.126888.

[186] D. M. Petroche and A. D. Ramirez, ""The Environmental Profile of Clinker, Cement, and Concrete: A Life Cycle Perspective Study Based on Ecuadorian Data"," (in English), Buildings, vol. 12, no. 3, 2022, doi: 10.3390/buildings12030311.

[187] M. E. Boesch and S. Hellweg, "Identifying improvement potentials in cement production with life cycle assessment," (in English), Environmental Science and Technology, vol. 44, no. 23, 2010, doi: 10.1021/es100771k.

[188] S. Wu et al., "Co-preparation of calcium sulfoaluminate cement and sulfuric acid through mass utilization of industrial by-product gypsum," Journal of Cleaner Production, vol. 265, 2020, doi: 10.1016/j.jclepro.2020.121801.

[189] D. Yang, L. Fan, F. Shi, Q. Liu, and Y. Wang, "Comparative study of cement manufacturing with different strength grades using the coupled LCA and partial LCC methods—A case study in China," Resources, Conservation and Recycling, vol. 119, pp. 60-68, 2017, doi: 10.1016/j.resconrec.2016.06.017.

[190] R. Abbas, M. A. Khereby, H. Y. Ghorab, and N. Elkhoshkhany, "Preparation of geopolymer concrete using Egyptian kaolin clay and the study of its environmental effects and economic cost," (in English), Clean Technologies and Environmental Policy, vol. 22, no. 3, 2020, doi: 10.1007/s10098-020-01811-4.

[191] H. Dong, S. Ohnishi, T. Fujita, Y. Geng, M. Fuji, and L. Dong, "Achieving carbon emission reduction through industrial & urban symbiosis: A case of Kawasaki," Energy, vol. 64, pp. 277-286, 2014, doi: 10.1016/j.energy.2013.11.005.

[192] W. Chen et al., "Environmental sustainability challenges of China's steel production: Impact-oriented water, carbon and fossil energy footprints assessment," (in English), Ecological Indicators, Article vol. 136, 2022, Art no. 108660, doi: 10.1016/j.ecolind.2022.108660.

[193] R. S. Trovão, L. D. Camillo, G. A. de Silva, and L. Kulay, "Verifying the Environmental and Energy Feasibility of Potential Improvement Actions in the Steel Production Chain in Brazil," "Journal of Sustainable Development of Energy, Water and Environment Systems", vol. 10, no. 1, 2022, doi: 10.13044/j.sdewes.d9.0390.

[194] R. Nezamoleslami and S. M. Hosseinian, "An improved water footprint model of steel production concerning virtual water of personnel: The case of Iran," Journal of environmental management, vol. 260, p. 110065, 2020, doi: 10.1016/j.jenvman.2020.110065.

[195] C. Broadbent, "Steel's recyclability: demonstrating the benefits of recycling steel to achieve a circular economy," (in English), International Journal of Life Cycle Assessment, Article vol. 21, no. 11, pp. 1658-1665, 2016, doi: 10.1007/s11367-016-1081-1.

[196] S. Cecchel et al., "Lightweighting in light commercial vehicles: cradle-to-grave life cycle assessment of a safety-relevant component," (in English), INTERNATIONAL JOURNAL OF LIFE CYCLE ASSESSMENT, vol. 23, no. 10, pp. 2043-2054, OCT 2018, doi: 10.1007/s11367-017-1433-5.

[197] R. J. K. Helmes, P. Goglio, S. Salomoni, D. S. van Es, I. Vural Gursel, and L. Aramyan, "Environmental Impacts of End-of-Life Options of Biobased and Fossil-Based Polyethylene Terephthalate and High-Density Polyethylene Packaging," Sustainability, vol. 14, no. 18, doi: 10.3390/su141811550.

[198] K. M. Bataineh, "Life-Cycle Assessment of Recycling Postconsumer High-Density Polyethylene and Polyethylene Terephthalate," Advances in Civil Engineering, vol. 2020, p. 8905431, 2020/03/10 2020, doi: 10.1155/2020/8905431.

[199] P. T. Benavides, U. Lee, and O. Zarè-Mehrjerdi, "Life cycle greenhouse gas emissions and energy use of polylactic acid, bio-derived polyethylene, and fossil-derived polyethylene," Journal of Cleaner Production, vol. 277, p. 124010, 2020/12/20/ 2020, doi: https://doi.org/10.1016/j.jclepro.2020.124010.

[200] P. Billen, L. Khalifa, F. Van Gerven, S. Tavernier, and S. Spatari, "Technological application potential of polyethylene and polystyrene biodegradation by macro-organisms such as mealworms and wax moth larvae," Science of The Total Environment, vol. 735, p. 139521, 2020/09/15/ 2020, doi: https://doi.org/10.1016/j.scitotenv.2020.139521.

[201] C. Chen, X. Li, C. Li, Y. Zhou, and L. Sui, "Optimized flax FRP stirrup in reinforced concrete beam: Material property and shear performance," Composite Structures, vol. 302, p. 116219, 2022/12/15/ 2022, doi: https://doi.org/10.1016/j.compstruct.2022.116219.

[202] M. Raynolds, M. D. Checkel, and R. A. Fraser, "Application of Monte Carlo Analysis to Life Cycle Assessment," Journal of Materials & Manufacturing, vol. 108, no. 5, 1999, doi: https://doi.org/10.4271/1999-01-0011.

[203] N. W. Ayer and G. Dias, "Supplying renewable energy for Canadian cement production: Life cycle assessment of bioenergy from forest harvest residues using mobile fast pyrolysis units," (in English), Journal of Cleaner Production, vol. 175, 2018, doi: 10.1016/j.jclepro.2017.12.040.

[204] C. Chen, G. Habert, Y. Bouzidi, and A. Jullien, "Environmental impact of cement production: detail of the different processes and cement plant variability evaluation," Journal of Cleaner Production, vol. 18, no. 5, pp. 478-485, 2010.

[205] L. P. T. Güereca, Nathalia and C. R. Juárez-López, "The co-processing of municipal waste in a cement kiln in Mexico. A life-cycle assessment approach," (in English), Journal of Cleaner Production, vol. 107, 2015, doi: 10.1016/j.jclepro.2015.05.085.

[206] I. Vázquez-Rowe, K. Ziegler-Rodriguez, J. Laso, I. Quispe, R. Aldaco, and R. Kahhat, "Production of cement in Peru: Understanding carbon-related environmental impacts and their policy implications," Resources, Conservation and Recycling, vol. 142, pp. 283-292, 2019, doi: 10.1016/j.resconrec.2018.12.017.

[207] P. Wang, W. Li, and S. Kara, "Cradle-to-cradle modeling of the future steel flow in China," (in English), Resources, Conservation and Recycling, Article vol. 117, pp. 45-57, 2017, doi: 10.1016/j.resconrec.2015.07.009.

[208] E. Gul et al., "Substitution of coke with pelletized biocarbon in the European and Chinese steel industries: An LCA analysis," Applied Energy, vol. 304, 2021, doi: 10.1016/j.apenergy.2021.117644.

[209] R. Xue, S. Wang, G. Gao, D. Liu, W. Long, and R. Zhang, "Evaluation of symbiotic technology-based energy conservation and emission reduction benefits in iron and steel industry: Case study of Henan, China," Journal of Cleaner Production, Article vol. 338, pp. N.PAG-N.PAG, 2022, doi: 10.1016/j.jclepro.2022.130616.

[210] M. B. London, G. M. Lewis, and G. A. Keoleian, "Life Cycle Greenhouse Gas Implications of Multi Jet Fusion Additive Manufacturing," ACS Sustainable Chemistry & Engineering, vol. 8, no. 41, pp. 15595-15602, 2020/10/19 2020, doi: 10.1021/acssuschemeng.0c04845.

[211] G. Bishop, D. Styles, and P. N. L. Lens, "Environmental performance of bioplastic packaging on fresh food produce: A consequential life cycle assessment," Journal of Cleaner Production, vol. 317, p. 128377, 2021/10/01/ 2021, doi: https://doi.org/10.1016/j.jclepro.2021.128377.

[212] X. Zhou et al., "Life cycle assessment of polycarbonate production: Proposed optimization toward sustainability," Resources, Conservation and Recycling, vol. 189, p. 106765, 2023/02/01/ 2023, doi: https://doi.org/10.1016/j.resconrec.2022.106765.

[213] S. Samad, A. Shah, and M. C. Limbachiya, "Strength development characteristics of concrete produced with blended cement using ground granulated blast furnace slag (GGBS) under various curing conditions," Sādhanā, vol. 42, no. 7, pp. 1203-1213, 2017, doi: 10.1007/s12046-017-0667-z.

[214] J. S. J. Van Deventer, R. San Nicolas, I. Ismail, S. A. Bernal, D. G. Brice, and J. L. Provis, "Microstructure and durability of alkali-activated materials as key parameters for standardization," Journal of Sustainable Cement-Based Materials, vol. 4, no. 2, pp. 116-128, 2015, doi: 10.1080/21650373.2014.979265.

[215] A. Ventura, V.-L. Ta, T. S. Kiessé, and S. Bonnet, "Design of concrete : Setting a new basis for improving both durability and environmental performance," (in English), Journal of Industrial Ecology, vol. 25, no. 1, 2021, doi: 10.1111/jiec.13059.

[216] A. S. Gutierrez, J. J. Cabello Eras, C. A. Gaviria, J. V. Caneghem, and C. Vandecasteele, "Improved selection of the functional unit in environmental impact assessment of cement," (in English), Journal of Cleaner Production, 2017, doi: 10.1016/j.jclepro.2017.09.007.

[217] P. Van den Heede and N. De Belie, "Environmental impact and life cycle assessment (LCA) of traditional and ‘green’concretes: Literature review and theoretical calculations," Cement and Concrete Composites, vol. 34, no. 4, pp. 431-442, 2012.

[218] E. Possan, E. F. Felix, and W. A. Thomaz, "CO_2 uptake by carbonation of concrete during life cycle of building structures," Journal of Building Pathology and Rehabilitation, vol. 1, no. 1, 2016, doi: 10.1007/s41024-016-0010-9.

[219] R. Heijungs and S. Suh, The Computational Structure of Life Cycle Assessment. Dordrecht, The Netherlands: Kluwer Academic Publishers, 2002.

[220] W. Leontif, "Environmental Repercussions and the Economic Structure: An Input-Output Approach," The Review of Economics and Statistics, vol. 52, no. 3, pp. 262-271, 1970.

[221] G. Finnveden et al., "Recent developments in Life Cycle Assessment," J Environ Manage, vol. 91, no. 1, pp. 1-21, Oct 2009, doi: 10.1016/j.jenvman.2009.06.018.

[222] G. J. Treloar, "Extracting Embodied Energy Paths from Input–Output Tables: Towards an Input–Output-based Hybrid Energy Analysis Method," Economic Systems Research, vol. 9, no. 4, pp. 375-391, 1997, doi: 10.1080/09535319700000032.

[223] M. D. Wilkinson et al., "The FAIR Guiding Principles for scientific data management and stewardship," Scientific Data, vol. 3, no. 1, p. 160018, 2016/03/15 2016, doi: 10.1038/sdata.2016.18.

[224] E. McGuire, J. L. Provis, P. Duxson, and R. Crawford, "Geopolymer Concrete: Is there an alternative and viable technology in the concrete sector which reduces carbon emissions?," presented at the Concrete 2011: Building a Sustainable Future, Perth, Australia, 2011.

[225] D. Maga, C. Galafton, J. Blömer, N. Thonemann, A. Özdamar, and J. Bertling, "Methodology to address potential impacts of plastic emissions in life cycle assessment," The International Journal of Life Cycle Assessment, vol. 27, no. 3, pp. 469-491, 2022/03/01 2022, doi: 10.1007/s11367-022-02040-1.

[226] G. Finnveden, "On the limitations of life cycle assessment and environmental systems analysis tools in general," The International Journal of Life Cycle Assessment, vol. 5, no. 4, pp. 229-238, 2000/07/01 2000, doi: 10.1007/BF02979365.

[227] J. Reap, F. Roman, S. Duncan, and B. Bras, "A survey of unresolved problems in life cycle assessment," The International Journal of Life Cycle Assessment, vol. 13, no. 5, pp. 374-388, 2008/08/01 2008, doi: 10.1007/s11367-008-0009-9.

[228] D. W. Pennington et al., "Life cycle assessment Part 2: Current impact assessment practice," Environment International, vol. 30, no. 5, pp. 721-739, 2004/07/01/ 2004, doi: https://doi.org/10.1016/j.envint.2003.12.009.

[229] Q. Tan, L. Yang, F. Wei, Y. Chen, and J. Li, "Comparative life cycle assessment of polyethylene agricultural mulching film and alternative options including different end-of-life routes," Renewable and Sustainable Energy Reviews, vol. 178, p. 113239, 2023/05/01/ 2023, doi: https://doi.org/10.1016/j.rser.2023.113239.

[230] J. C. Bare, "Life cycle impact assessment research developments and needs," Clean Technologies and Environmental Policy, vol. 12, no. 4, pp. 341-351, 2010/08/01 2010, doi: 10.1007/s10098-009-0265-9.

[231] G. C. Palermo, D. A. Castelo Branco, A. C. O. Fiorini, and M. A. V. de Freitas, "Comparative life cycle assessment of three 2030 scenarios of the Brazilian cement industry," Environ Monit Assess, vol. 194, no. 3, p. 153, Feb 8 2022, doi: 10.1007/s10661-022-09822-y.

[232] L. Moretti and S. Caro, "Critical analysis of the Life Cycle Assessment of the Italian cement industry," Journal of Cleaner Production, vol. 152, pp. 198-210, 2017, doi: 10.1016/j.jclepro.2017.03.136.

[233] World Business Council for Sustainable Development, "Pathfinder Framework Guidance for the Accounting and Exchange of Product Life Cycle Emissions," 2023. [Online]. Available: https://www.carbon-transparency.com/our-approach/pathfinder-framework

5. DECARBONISATION PATHWAYS: Evaluating Realistic UK Cement and Concrete Sector Abatement Options

The second article, published in *Sustainable Production and Consumption*, aims to balance the expectations placed on low-maturity and high-maturity decarbonisation strategies to determine a realistic route in which the UK cement and concrete sector is able to achieve net-zero targets by 2050. This was achieved through a decomposition analysis using a hybrid methodological approach that integrates material flow analysis and LCA. The results indicate that it is unlikely for the UK cement and concrete sector to reach net-zero unless concrete demand is reduced by 43% through the reuse and refurbishment of existing building stock. To implement these circular economy strategies swiftly and effectively, it is essential that economic, legislative, and social barriers are addressed by altering existing economic models, establishing new policy frameworks, and creating direct public incentives. I hereby certify that I led in the data curation, formal analysis, visualisation, and writing and reviewing of the original draft. Together with my supervisor Dr. Hisham Hafez, I conceptualised the study's aims and developed the methodology. In addition, I took full responsibility of the submission process including responding to comments left by peer-reviewers.

Thousand Cuts: A Realistic Route to Decarbonise the UK Cement and Concrete Sector by 2050

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

*Corresponding author: mcsrihner1@sheffield.ac.uk
[a]School of Chemical, Materials and Biological Engineering, University of Sheffield, S1 3JD, UK
[b]School of Civil Engineering, University of Leeds, LS2 9LG, UK

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 lower clinker factor and 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 is impeded by financial and resource constraints. Accordingly, it is improbable for the sector to meet the 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.

Keywords

decarbonisation strategies, 2050 targets, CCUS, demand reduction, low carbon cement, circular economy

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

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 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.

Table 1: Excluded roadmap cement and concrete sector decarbonisation strategies

<table>
<tr><th>Decarbonisation Strategy</th><th>Description of Strategy</th><th>Rationale for Exclusion</th></tr>
<tr><td>Material Substitution</td><td>Using alternative materials such as steel and timber</td><td>This strategy has not been proven to achieve environmental or economic savings when compared to concrete (D’Amico et al., 2021).</td></tr>
<tr><td>Decarbonisation of Transportation</td><td>Reducing fossil fuel consumption by utilising more sustainable transport methods such as electric vehicles</td><td>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).</td></tr>
<tr><td>Re-carbonation</td><td>Utilising use and end-of-life concrete as a form of carbon capture</td><td rowspan="2">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.</td></tr>
<tr><td>Leveraged Thermal Mass</td><td>Reduce operational carbon emissions generated from heating and cooling</td></tr>
</table>

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 Figure 1.

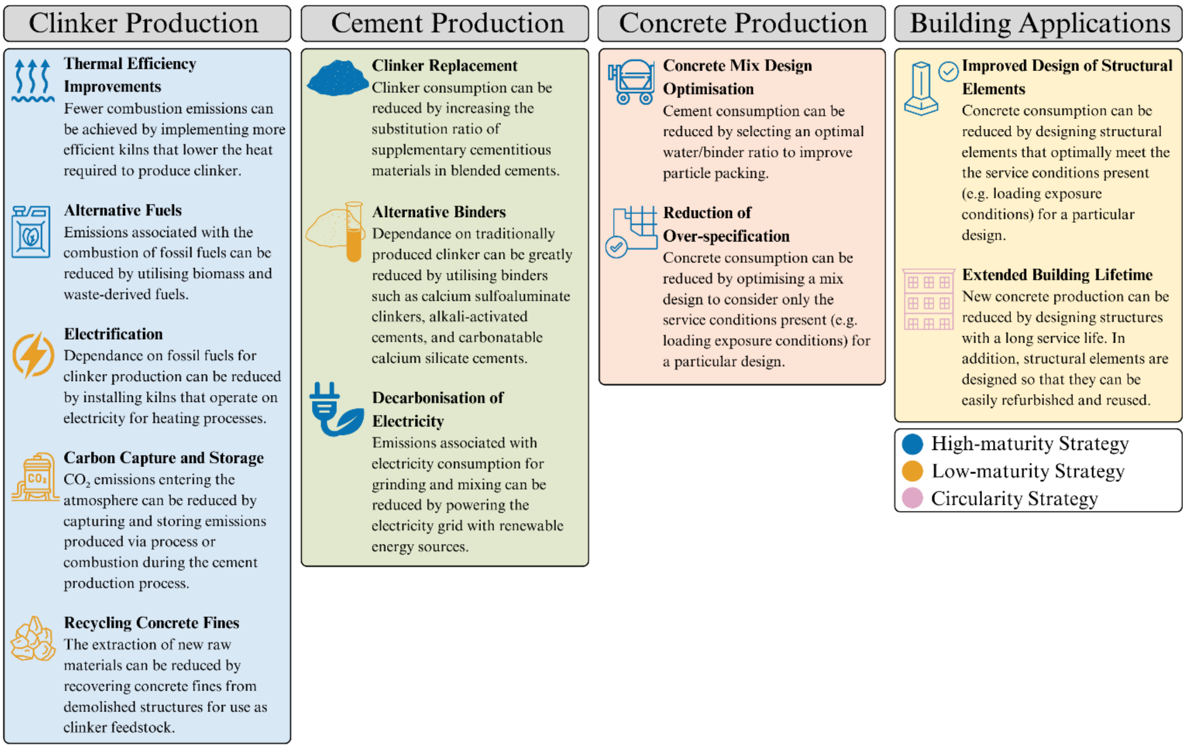


Figure 1: Cement and concrete sector decarbonisation strategies found in UK and EU roadmaps (adapted from Marsh et al. (2023))

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 the two probability combined decarbonisation potential pathways by 2050 with the net-zero pathway outlined by the UK CCC.

2. Methodology

The methodology followed for this study, along with the flow of the data, is best described in Figure 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 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.

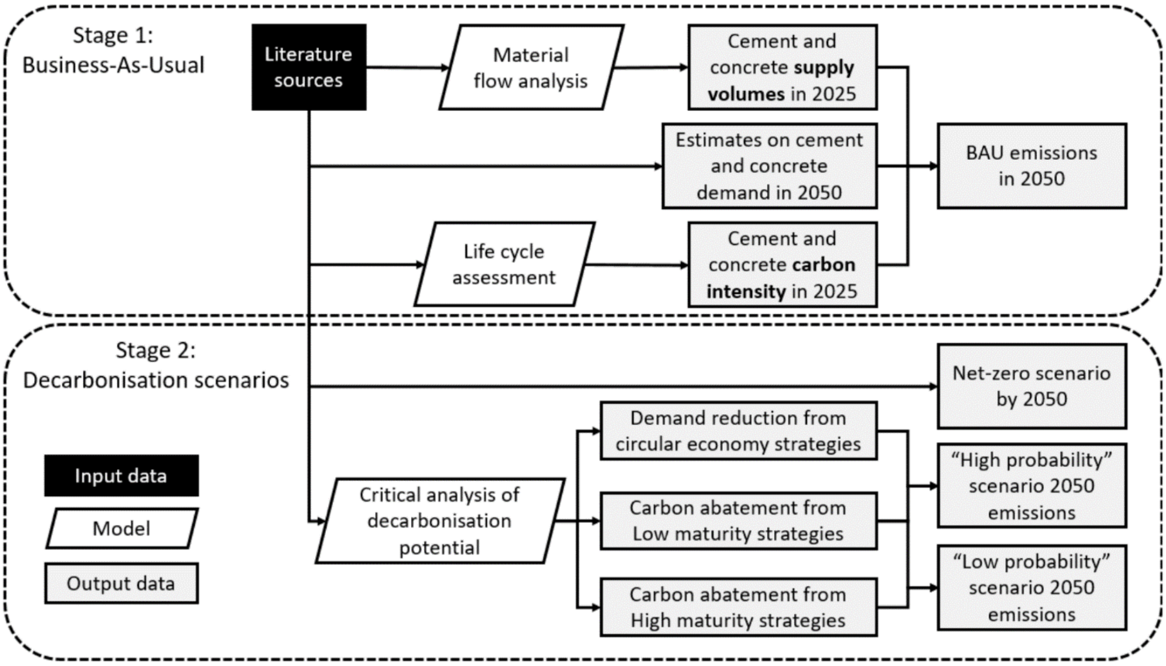


Figure 2: Methodology Flowchart

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 Equation 2.

$$\text{BAU carbon emissions } (MtCO_{2eq}/yr) = \text{Product annual consumption } (Mt/yr) * \text{Product carbon intensity } (kgCO_{2eq}/kg) \qquad \textit{Equation 2}$$

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 in infrastructure, pavements, and roads is 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 Equation 3, 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.

BAU cement and concrete sector carbon emissions in 2025 (MtCO_{2eq}/yr) = (Average concrete carbon intensity (kgCO_{2eq}/kg) * Annual concrete consumption (Mt/yr)) + (Average cement carbon intensity (kgCO_{2eq}/kg) * Annual cement consumption (Mt/yr) * Cement consumption for other uses (%))

Equation 3

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 Equation 4.

BAU cement and concrete sector carbon emissions in 2050 (MtCO_{2eq}/yr) = BAU cement and concrete sector carbon emissions in 2025 (MtCO_{2eq}/yr) * Demand for cement and concrete (%)

Equation 4

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 Equation 3. 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 product flows are measured in megatons per year. 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.

Since the clinker content in imported cement is not straightforwardly calculated nor directly reported, clinker imports were estimated by multiplying the total mass of imported cement by 0.90, reflecting an assumed high clinker content similar to CEM I (OPC). This estimate was then added to the known quantity of imported clinker. This approach provides a conservative estimate of clinker inflows, ensuring that subsequent carbon emissions calculations are not underestimated. 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 Equation 5.

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

Equation 5

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 700-1400 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.

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'. In cement and concrete LCA studies, a 'cradle-to-gate' system boundary is typically selected, which 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). Due to uncertainty regarding each product's upstream processes, the boundary for this study is limited to the processing and production stage (A3) only, resulting in the selection of a simplified 'gate-to-gate' system boundary (Rihner et al., 2025). A detailed illustration of this system boundary can be found in supplementary information (SI, Figure

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 Equations 6-8. For the impact assessment stage, embodied carbon (kgCO_2eq/kg) was the only environmental impact indicator analysed in line with the study's goal. Since the carbon intensity values were obtained from literature sources, no additional GWP characterisation factors were applied in the calculations to avoid double counting. As expressed in Equation 6, the average carbon intensity of concrete is defined by its binder and non-binder components.

Average concrete carbon intensity (kgCO_{2eq}/kg) =
Average cement carbon intensity (kgCO_{2eq}/kg) * Binder content (kg/kg) + ∑Carbon intensity of the non-binder components (kgCO_{2eq}/kg) * Non-binder content (kg/kg)

Equation 6

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 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 Equation 7.

$$\text{Average cement carbon intensity } (kgCO_{2eq}/kg) = \text{Average clinker carbon intensity } (kgCO_{2eq}/kg) * \text{Clinker factor} + \text{Average SCMs carbon intensity } (kgCO_{2eq}/kg) * (1-\text{Clinker factor}) + \text{Electric energy required for indirect processes } (MJ/kg) * \text{Carbon intensity of national electricity grid } (kgCO_{2eq}/MJ)$$

Equation 7

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 Equation 8.

$$\text{Average clinker carbon intensity } (kgCO_{2eq}/kg) = \text{Clinker process emissions } (kgCO_{2eq}/kg) + \text{Energy intensity in clinker production } (MJ/kg) * \text{Carbon intensity of fuel mix } (kgCO_{2eq}/MJ)$$

Equation 8

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.

Table 2: Summary of the inventory data used to determine cement and concrete sector carbon intensity values

Market Level	Variable	Unit	UK Value	Reference	Global Value	Reference
Clinker	Clinker process emissions	$kgCO_{2eq}/kg$	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	$kgCO_{2eq}/MJ$	0.07	Summerbell (2018)	0.09	Summerbell (2018)
	Average clinker carbon intensity	**$kgCO_{2eq}/kg$**	**Calculated**			
Cement	Clinker factor	-	0.70	MPA (2020a)	0.71	IEA (2022)
	Average SCM carbon intensity	$kgCO_{2eq}/kg$	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	$kgCO_{2eq}/kWh$	0.2	ESO (2023)	0.44	Ember (2024)
	Average cement carbon intensity	**$kgCO_{2eq}/kg$**	**Calculated**			
Concrete	Carbon intensity of the non-binder components	$kgCO_{2eq}/kg$	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	**$kgCO_{2eq}/kg$**	**Calculated**			

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 assumed 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).

Table 3: Values for current global best practice of HM strategies

Market Level	Variable	Value	Unit	Reference
Clinker	Carbon Intensity of Fuel Mix	0.058	$kgCO_{2eq}/MJ$	IEA (2018)
	Energy Intensity of Clinker Production	3.0	GJ/t clinker	Summerbell (2018)
Cement	Carbon Intensity of National Electricity Grid	0.041	$kgCO_{2eq}/kWh$	Switzerland; (Ember, 2024)
	Average SCM Carbon Intensity	0.08	$kgCO_{2eq}/kg$	MPA (2025)
	Clinker Factor	0.5	-	Scrivener et al. (2018)
Concrete	Binder Content	0.12	kg/kg kg/kg	BSI (2023)

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 (Equations 6-8).

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 Improve ments	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	
Buildings Application	Improved Design of Structural Elements	HM	More efficient structural design	Concrete material intensity in buildings

To determine the decarbonisation potential in 2050 for any LM or HM intervention, Equation 9 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.

Strategy decarbonisation potential in 2050 ($MtCO_2eq/yr$) = Product carbon intensity ($kgCO_2eq/kg$) * Strategy reduction potential beyond the current average (%) * Product annual consumption (Mt/yr)

Equation 9

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 is 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. Equation 10 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 concrete sector carbon emissions in 2050 was calculated using Equation 4.

Reduction in concrete demand (%) in 2050=
(BAU cement and concrete sector carbon emissions in 2050 ($MtCO_2eq/yr$) – total decarbonisation potential carbon savings in 2050 ($MtCO_2eq/yr$)) / BAU cement and concrete sector carbon emissions in 2050 ($MtCO_2eq/yr$)

Equation 10

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 Equation 4. For this study, the best practice concrete material intensity in buildings is conservatively projected to equal to 843 kg/m2 (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, Equation 11 was used to determine reduction potential required by LM strategies under the low probability scenario to reach net-zero.

Reduction required by LM strategies ($MtCO_2eq/yr$) =
BAU cement and concrete sector carbon emissions in 2050 ($MtCO_2eq/yr$) – Total HM decarbonisation potential ($MtCO_2eq/yr$)

Equation 11

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 Figure 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 Equation 5, 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.

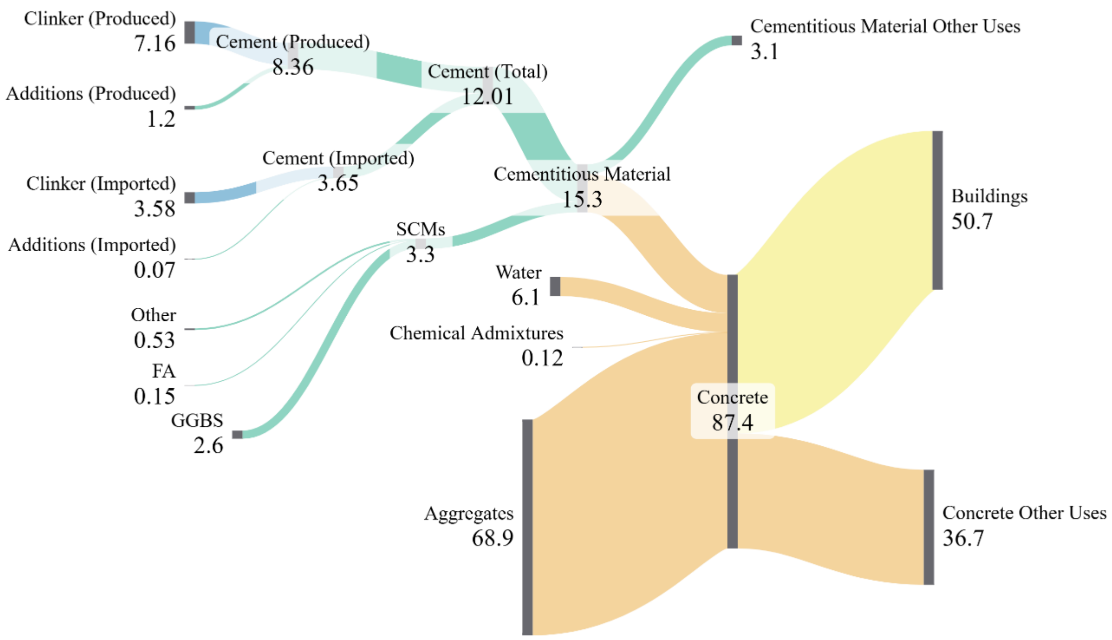


Figure 3: Material Flow Diagram of UK Cement and Concrete Consumption (Mt/yr), 2022

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 Figure 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.

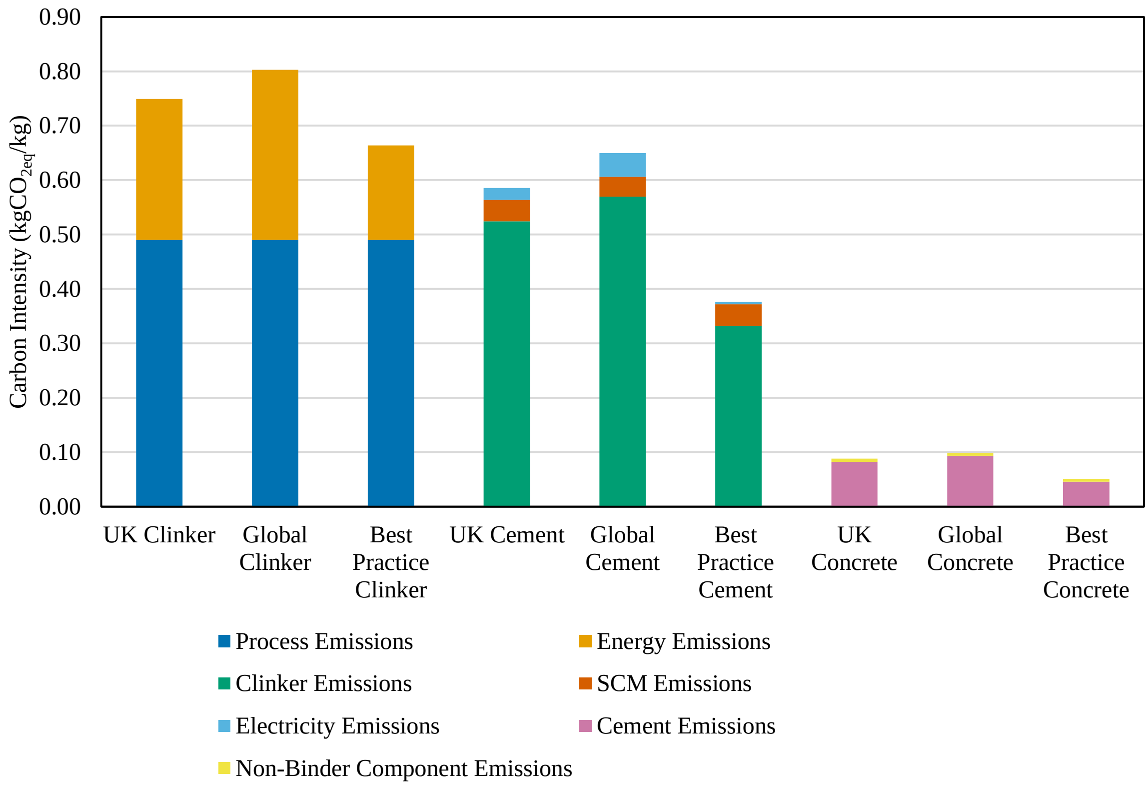


Figure 4: Carbon Intensity at each market level (UK, global, best practice)

3.2 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 Equation 9. 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 Figure 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.

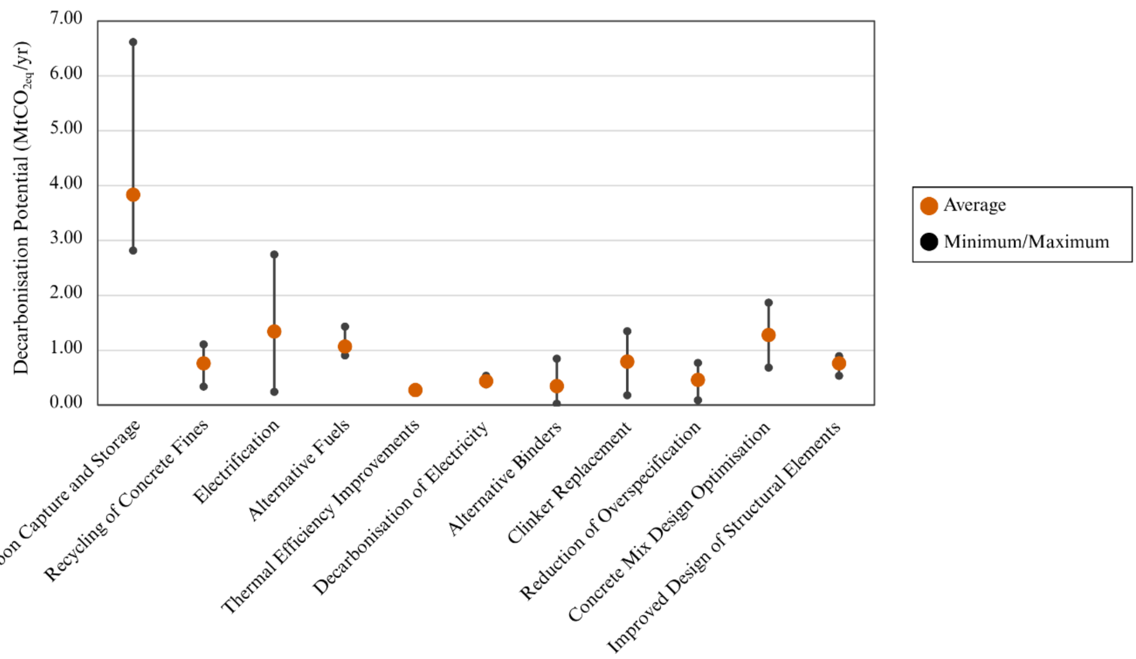


Figure 5: Decarbonisation potential in 2050 by strategy utilising values reported in five UK and EU roadmaps

While the probability of achieving the minimum decarbonisation potential of HM strategies values shown in Figure 5 is high, the lower probability implementation of the best practice for each HM strategy was also investigated. As shown in Figure 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.

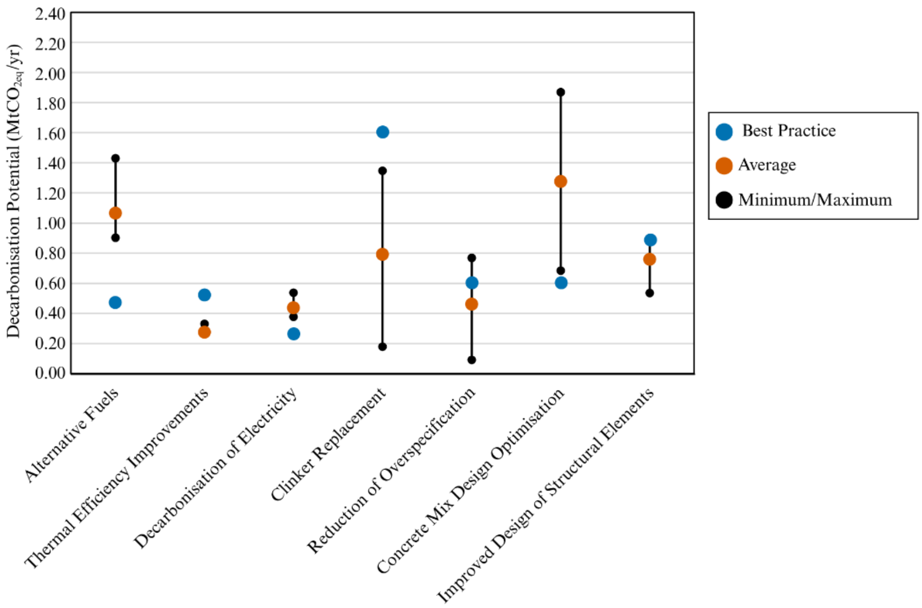


Figure 6: Decarbonisation potential in 2050 of HM strategies (roadmap values vs. best practice values)

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 (Equation 3) 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). Figure 7 illustrates the reduction in 2050 BAU emissions though the implementation of all decarbonisation strategies under both the high and low probability scenarios.

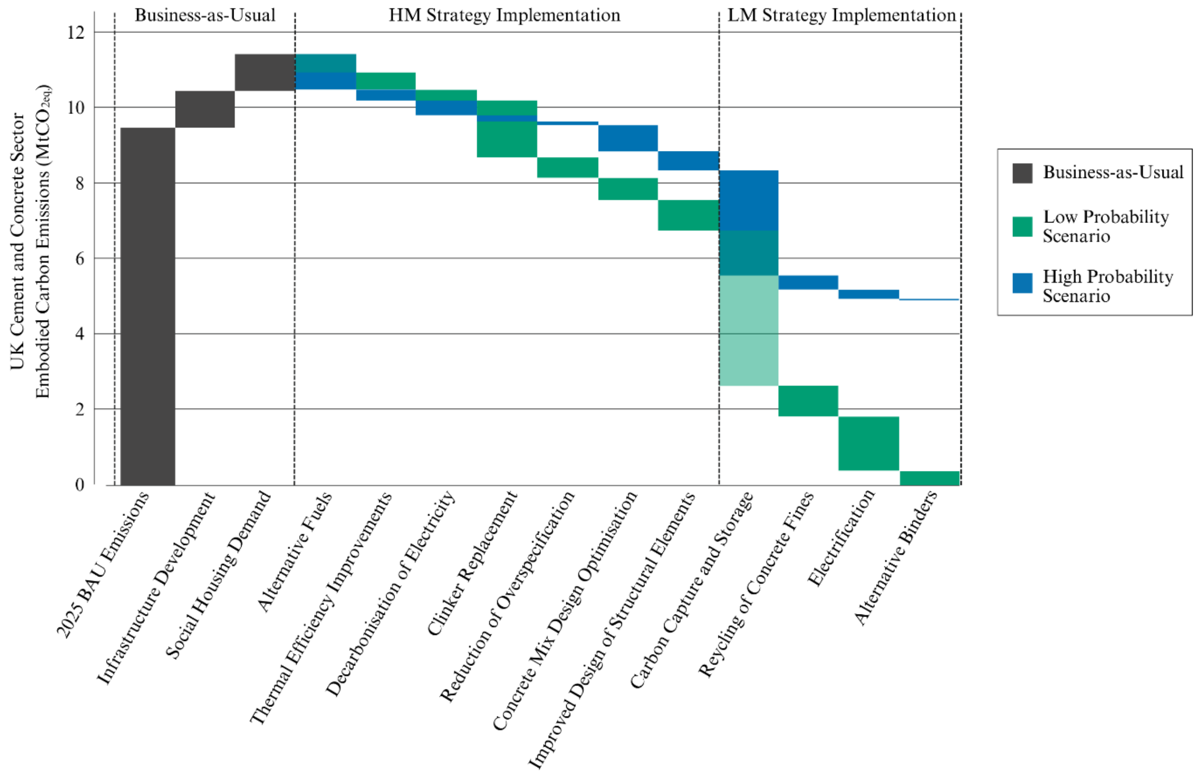


Figure 7: The business-as-usual emissions from cement and concrete consumption in the UK by 2050 and the combined decarbonisation potential values from this study's analysis of low and high probability of strategy implementation

3.2.1 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 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 Equation 10, 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).

3.2.2 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/tonne 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/tonne 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 $MtCO_2eq./yr$; 54% less than the capacity required under the low probability scenario (Hafez et al., 2024). 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.

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 et al. (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 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. Figure 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 (2020). 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.

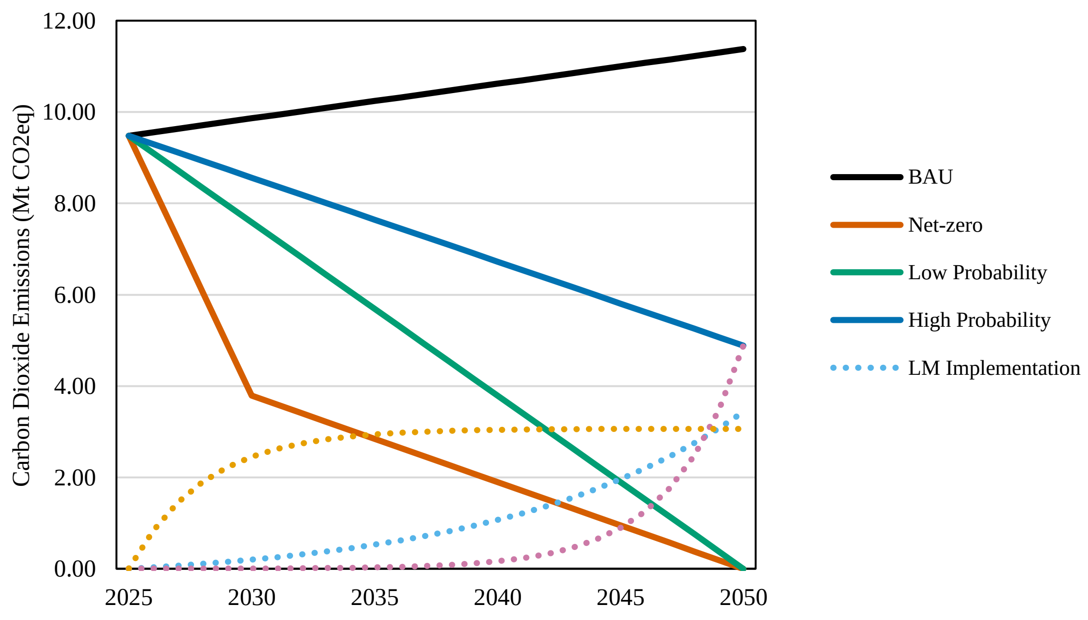


Figure 8: Cumulative Carbon Budget to 2050 Implementing Combined HM and LM Strategies

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.

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).

Declaration of Conflicting Interest

The authors declare that there is no conflict of interest.

Author Contributions

M. Rihner: Conceptualization (supporting); Data Curation (equal); Formal Analysis (lead); Methodology (supporting); Writing- Original Draft (lead); Visualization; Writing-Review & Editing (lead), H. Hafez: Project Administration; Supervision; Conceptualization (lead); Methodology (lead); Formal Analysis (supporting); Data Curation (equal); Writing- Original Draft (supporting); Writing-Review & Editing (supporting), B. Walkley: Funding Acquisition; Supervision; Writing- Review and Editing (supporting), P. Purnell: Funding Acquisition; Supervision; Writing- Review and Editing (supporting), M. Drewniok: Supervision; Writing- Review and Editing (supporting)

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. Fly Ash and Blast Furance Slag for Cement Manufacturing. In: DEPARTMENT FOR BUSINESS, E. I. S. (ed.). BEIS.

BERNSTEIN, S. & HOFFMANN, M. 2015. The Politics of Decarbonisation: 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. Resources, Conservation and Recycling, 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. Energy and Buildings, 231, 110612.

CLIMATE GROUP CONCRETEZERO 2024. Decarbonising the UK Concrete Industry: Closing the emissions gap. The Climate Change Organisation.

CURRY, K. C. 2018. Slag—Iron and Steel. In: INTERIOR, U. S. D. O. T. (ed.) 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. Journal of Cleaner Production, 279, 123487.

DAMINELI, B. L., KEMEID, F. M., AGUIAR, P. S. & JOHN, V. M. 2010. Measuring the eco-efficiency of cement use. Cement and Concrete Composites, 32, 555-562.

DENSLEY TINGLEY, D., COOPER, S. & CULLEN, J. 2017. Understanding and overcoming the barriers to structural steel reuse, a UK perspective. Journal of Cleaner Production, 148, 642-652.

DEPARTMENT FOR ENERGY SECURITY AND NET ZERO 2024. Contracts signed for UK's first carbon capture projects in Teesside. gov.uk.

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. Resources, Conservation and Recycling, 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? Journal of Cleaner Production, 464, 142709.

EBERHARDT, L., BIRGISDOTTIR, H. & BIRKVED, M. 2019. Potential of Circular Economy in Sustainable Buildings. 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. Environmental Research Letters, 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. Journal of Cleaner Production, 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 Economics, 121, 106655.

HAFEZ, H., BAJIĆ, P., AIDAROV, S., MALIJA, X., DREWNIOK, M., PURNELL, P. & TOŠIĆ, N. 2024a. Parametric study on the decarbonisation potential of structural system and concrete mix design choices for mid-rise concrete buildings. Materials and Structures, 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.

HAFEZ, H., KURDA, R., CHEUNG, W. M. & NAGARATNAM, B. 2019. A Systematic Review of the Discrepancies in Life Cycle Assessments of Green Concrete. Applied Sciences, 9, 4803.

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 [Accessed].

IEA. 2024. Switzerland [Online]. Available: https://www.iea.org/countries/switzerland [Accessed].

ISO 2006a. IInternational 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. Applied Sciences, 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. 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. Journal of Industrial Ecology, 26, 562-576.

MONTEIRO, P. J. M., MILLER, S. A. & HORVATH, A. 2017. Towards sustainable concrete. Nature Materials, 16, 698-699.

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. MPA Cement stats data collation template BH Collation F v3. In: MINERAL PRODUCTS ASSOCIATION (ed.).

MPA 2025. Fact Sheet 18 Embodied CO_2e of UK cements. In: MINERAL PRODUCTS ASSOCIATION (ed.).

NAQI, A. & JANG, J. G. 2019. Recent Progress in Green Cement Technology Utilising Low-Carbon Emission Fuels and Raw Materials: A Review. Sustainability, 11, 537.

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. Journal of Cleaner Production, 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. Journal of Industrial Ecology, 25, 359-376.

QIAN, Z., LANTSOGHT, E. O. & LUKOVIC, M. 2022. A critical review on structural behavior of alkali-activated concrete beams. 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. Renewable and Sustainable Energy Reviews, 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). Cement and Concrete Research, 114, 49-56.

SHEN, L., GAO, T., ZHAO, J., WANG, L., WANG, L., LIU, L., CHEN, F. & XUE, J. 2014. Factory-level measurements on CO_2 emission factors of cement production in China. Renewable and Sustainable Energy Reviews, 34, 337-349.

SHEN, W., CAO, L., LI, Q., ZHANG, W., WANG, G. & LI, C. 2015. Quantifying CO_2 emissions from China's cement industry. Renewable and Sustainable Energy Reviews, 50, 1004-1012.

STAMFORD, L. & AZAPAGIC, A. 2014. Life cycle sustainability assessment of UK electricity scenarios to 2070. Energy for Sustainable Development, 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. The Climate Change Act 2008 (2050 Target Amendment) Order 2019. In: DEPARTMENT FOR BUSINESS ENERGY AND INDUSTRIAL STRATEGY (ed.).

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. The International Journal of Life Cycle Assessment, 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. Industrial Decarbonisation & Energy Efficiency Roadmaps to 2050: Cement. In: DEPARTMENT OF ENERGY AND CLIMATE CHANGE AND THE DEPARTMENT FOR BUSINESS, I. A. S. (ed.).

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. Resources, Conservation and Recycling, 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. Environmental Chemistry Letters, 21, 55-80.

ZUNINO, F. 2023. A two-fold strategy towards low-carbon concrete. RILEM Technical Letters, 8, 45-58.

Supplementary Information

This supplementary information provides full details of the methods used for the analysis presented in the article Rihner et al. "Thousand cuts: a realistic route to decarbonise the UK cement and concrete sector by 2050."

S1: Benchmarking the UK cement and concrete sector consumption volumes and embodied carbon values

Table 1 outlines the values and data sources utilised to create the material flow analysis of cement and concrete consumption in the UK.

Table 1: 2022 Material Flow Analysis of Cement and Concrete Production in the UK

Material	Amount (Mt/yr)	Source
Clinker production in cement	**10.74**	
Clinker (produced)	7.16	MPA (2023)
Clinker (imported)	3.58	Estimate assuming CEM1 (90% clinker)
Total cementitious material consumption	**15.3**	
Additions	1.57	MPA (2023)
Total cement consumption	12.01	
Consumed cement produced	8.36	
Consumed cement imported	3.65	
Total SCM consumption	3.27	
GGBS	2.60	(World Steel Association, 2022; UN Comtrade, 2022)
Fly ash	0.15	UKQAA (2021)
Limestone filler	0.53	MPA (2023)
Total concrete consumption	**87.4**	**Drewniok et al. (2023)**
Cement consumption for other uses	3.1	Drewniok et al. (2023)
Cement consumption for concrete	12.2	
Aggregate consumption for concrete	68.9	-
Water consumption for concrete*	6.1	
Chemical admixtures for concrete**	0.12	
Concrete consumption for buildings	**50.7**	**Drewniok et al. (2023)**
Concrete consumption for other uses	36.7	Drewniok et al. (2023)

*Assuming 0.50 water to cement ratio

**Assuming 1% of cement content

When conducting a life cycle assessment, a system boundary must be selected. Figure 1 illustrates the 'cradle-to-gate' system boundary selected for this study which includes the manufacturing process of clinker, cement, and concrete. This study does not assess the emissions associated with raw material extraction, transportation, or a material's use.

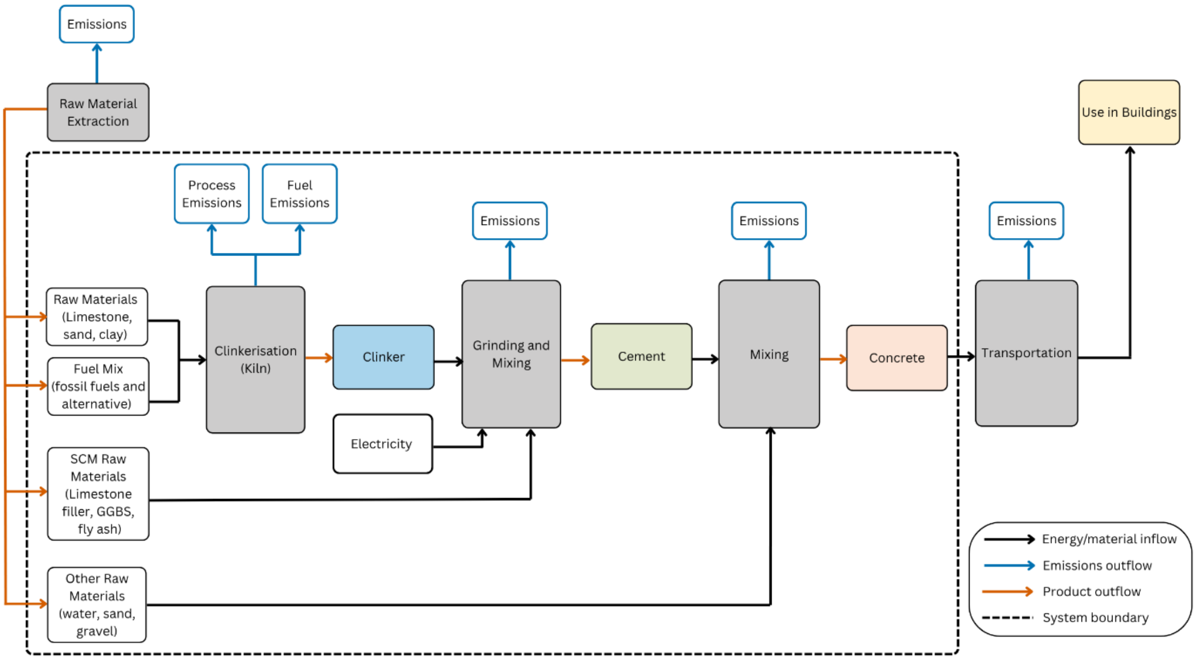


Figure 1: LCA System Boundary

To determine the percentage of concrete consumption for buildings, the concrete material intensity in buildings must be calculated. As noted in Equation 1, the UK concrete material intensity in buildings in 2022 was calculated by multiplying the percentage of new construction by the summed average material intensity for each typology. To determine the best practice concrete material intensity in buildings, the average material intensity for each typology was first multiplied by 0.80 to account for the 20% reduction in concrete use when a structure's design is optimised through either the improved design of structural elements or by considering optimised material use in a building's initial design (Drewniok et al., 2023). The material intensity and typology percentage values used for these calculations were obtained from Drewniok et al. (2023) and are reported in Table 2 alongside the calculated results.

$$Concrete\ material\ intensity\ in\ buildings \left(\frac{kg}{m^2}\right) = \sum Material\ Intensity\ (\frac{kg}{m^2}) * New\ Construction\ (\%)$$

Equation 1

Table 2: Concrete Material Intensity in UK Buildings

Typology	Average Material Intensity (kg/m^2)	Minimum Material Intensity (kg/m^2)	New Construction (%)
End Terrace	1203.5	962.8	10.1%
Mid Terrace	1013.1	810.5	10.2%
Semi-detached	1266.5	1013.2	19.1%
Detached	1332	1065.6	9.1%
Bungalow	1652.9	1322.3	1.1%
Converted Flat	457.2	365.8	4.7%
Residential (4<)	1212.2	969.7	3.7%
Residential (5-6)	716.1	572.9	0.9%
Residential (7-10)	675.4	540.3	0.2%
Residential (>10)	551.9	441.5	0.1%
Office Buildings	1361	1088.8	6.2%
Industrial Buildings	785.4	628.3	24.2%
Retail Buildings	830.4	664.3	6.9%
Other non-domestic buildings	1167.9	934.3	3.5%
UK concrete material intensity in buildings (2022)	**1054 kg/m^2**		
UK best practice concrete material intensity in buildings	**843 kg/m^2**		

Table 3 reports the values utilised to calculate the average binder content ratio for both the UK and global markets. The average binder content ratio in UK concrete was calculated by dividing the UK cement consumption for concrete by the UK's total concrete consumption. Both of these values were obtained from the material flow analysis in section S1. The UK value calculated (0.14 ratio = 380 kg/m^3) is close to that reported by the European Ready Mix Concrete Organisation 2020 report (ERMCO, 2020) which notes that 348 kg/m^3 is the average binder content used for UK ready-mix concrete. For the global benchmark, the ratio between the average annual global cement and concrete consumption was taken.

Table 3: UK and Global Average Binder Content Ratio

Market	Cement Consumption (Mt/yr)	Source	Concrete Consumption (Mt/yr)	Source	Calculated Binder Content
UK	12.23	Material Flow Analysis	87.4	Material Flow Analysis	0.14
Global	4,300	(IEA, 2022)	30,000	(Monteiro et al., 2017)	0.14

For this study, the best practice binder content was calculated by determining the lowest binder content value. This was achieved by first selecting the maximum minimum binder content for each strength class category specified in BS8500-1:2023. These values were then multiplied by the market share for each strength class noted in ERMCO (2020). To calculate the binder content ratio, the summed minimum binder content value was divided by 2400 kg/m^3; the average density of concrete. From this, the minimum binder content ratio was found to be 0.12. Table 4 provides the data used to calculate this value.

Table 4: Calculated Minimum Binder Content Ratio for Best Practice

Production by Strength Class	**Maximum Minimum Binder Content (kg/m^3)**	**Market Share**	**Maximum Minimum Binder Content * Market Share (kg/m^3)**
<C16/20	200	5%	10
C16/20-C20/25	240	17%	40.8
C25/30-C30/37	300	64%	192
>=C35/45	340	14%	47.6
		Sum	290.4

While the carbon intensity of the non-cementitious components for the UK market was obtained from Drewniok et al. (2023), the value for the global market was calculated using SimaPro and Ecoinvent 'rest-of-the-world' datasets (Wernet et al., 2016). To calculate the carbon intensity value for each component, the component's mass ratio was multiplied by its carbon intensity. For the purposes of this calculation, a typical concrete mix design was assumed. Table 5 provides a breakdown of the assumed concrete mix design, the carbon intensity values, and the summed carbon intensity value for non-cementitious components. This value was also used for the UK best practice case.

Table 5: Global Market Carbon Intensity of Non-Cementitious Components

Material	**Mass of Typical Concrete Mix Design (kg)**	**Mass Ratio**	**Carbon Intensity ($kgCO_2eq/kg$)**	**Mass Ratio*Carbon Intensity ($kgCO_2eq/kg$)**
Gravel	1100	46%	0.0102	0.0047
Sand	850	35%	0.00427	0.0015
Water	150	6%	0.00124	0.0001
Cement	300	13%	-	-
Sum	2400	100%	-	0.006

To calculate the average SCM carbon intensity, the mass ratio of each SCM was multiplied by their respective carbon intensity value and then summed. The annual global SCM consumption was assumed to be identical to the UK's consumption value. Therefore, the material use values were extracted from the material flow analysis. For this assessment, it was assumed that the remaining 0.53Mt/yr of SCMs correlates to limestone filler. Table 6 provides a summary of the data and sources utilised in addition to the calculated average global and UK SCM carbon intensity values.

Table 6: Average SCMs Carbon Intensity UK and Global Market

<table>
<tr><th colspan="6">UK Market</th></tr>
<tr><th>SCM</th><th>Material Use (Mt/yr)</th><th>Mass Ratio</th><th>Carbon Intensity (kgCO_{2eq}/kg)</th><th>Reference</th><th>Mass Ratio * Carbon Intensity (kgCO_{2eq}/kg)</th></tr>
<tr><td>GGBS</td><td>2.6</td><td>79.4%</td><td>0.155</td><td rowspan="4">MPA (2025)</td><td>0.123</td></tr>
<tr><td>Fly Ash</td><td>0.15</td><td>4.4%</td><td>0.022</td><td>0.001</td></tr>
<tr><td>Calcined Clays</td><td>0</td><td>0.0%</td><td>0.27</td><td>0</td></tr>
<tr><td>Limestone Filler</td><td>0.53</td><td>16.2%</td><td>0.044</td><td>0.007</td></tr>
<tr><td>Sum</td><td>3.27</td><td>100</td><td>-</td><td>-</td><td>0.13</td></tr>
<tr><th colspan="6">Global Market</th></tr>
<tr><th>SCM</th><th>Material Use (Mt/yr)</th><th>Mass Ratio</th><th>Carbon Intensity (kgCO_{2eq}/kg)</th><th>Reference</th><th>Mass Ratio * Carbon Intensity (kgCO_{2eq}/kg)</th></tr>
<tr><td>GGBS</td><td>2.6</td><td>79.4%</td><td>0.155</td><td>MPA (2025)</td><td>0.123</td></tr>
<tr><td>Fly Ash</td><td>0.15</td><td>4.4%</td><td>0.023*</td><td>Shi et al. (2021)</td><td>0.001</td></tr>
<tr><td>Calcined Clays</td><td>0</td><td>0.0%</td><td>0.28</td><td rowspan="2">Scrivener et al. (2018)</td><td>0</td></tr>
<tr><td>Limestone Filler</td><td>0.53</td><td>16.2%</td><td>0.008</td><td>0.001</td></tr>
<tr><td>Sum</td><td>3.27</td><td>100</td><td>-</td><td>-</td><td>0.13</td></tr>
</table>

*Data for fly ash originates from China but is assumed to represent global value

For the carbon intensity of SCMs best practice value, a homogenised cement mix in 2050 was assumed based on the best practice clinker factor value and realistic SCM substitution rates. Table 7 shows the percentage contribution of each constituent and their respective carbon intensity values. The sum of each SCM's mass ratio multiplied by its respective carbon intensity value is equal to the average best practice SCM carbon intensity in 2050.

Table 7: Average SCM Carbon Intensity for UK Best Practice

Material	Homogenised Mix (%)	SCM Mass Ratio	Carbon Intensity ($kgCO_{2eq}/kg$)	Reference	Mass Ratio*Carbon Intensity ($kgCO_{2eq}/kg$)
Gypsum	5%	0.1	0.05	Hafez et al. (2024)	0.005
Limestone Filler	15%	0.3	0.044	MPA (2025)	0.013
GGBS	15%	0.3	0.155		0.047
Calcined Clays	15%	0.3	0.048		0.014
				Sum	0.08

Table 8 summarises the current UK business-as-usual (BAU) carbon emissions. To calculate the BAU carbon emissions at each product level prior to any decarbonisation strategy implementation, the product annual consumption values obtained from the material flow analysis was multiplied by its respective average carbon intensity.

Table 8: UK BAU Carbon Emissions

Market Level	Product Annual Consumption (Mt/yr)	Carbon Intensity ($kgCO_2eq/kg$)	BAU Carbon Emissions ($MtCO_2eq/yr$)
Clinker	10.74	0.75	8.05
Cement	15.28	0.59	8.95
Concrete	87.40	0.088	7.69
Building Application	50.69	-	4.46

S2: Calculating the Cumulative Carbon Savings

Equation 2 was used to evaluate the carbon emissions savings annually for both the low and high probability scenarios where *n* is the year for which carbon savings are assessed. For the low probability scenario, the compounded high-maturity decarbonisation potential from all best practices were utilised. To determine the carbon emissions for a given year, the carbon savings (low-maturity (LM) and high-maturity (HM)) were subtracted from the BAU carbon emissions in year *n* as noted in Equation 3. For the net-zero scenario, the Climate Change Committee carbon targets in 2030, 2040, and 2050 were used. From this, a linear relationship was taken between the 2025 to 2030 values, the 2030 to 2040 values, and the 2040 to 2050 values to determine the annual net-zero scenario carbon emissions. The calculated carbon savings and carbon emissions for both the low and high probability scenarios are presented in Table 9.

$$
\begin{aligned}
&HM\ or\ LM\ Carbon\ emissions\ savings\ in\ year\ n\ (MtCO_{2eq}) \\
&\qquad = HM\ or\ LM\ carbon\ savings\ in\ year\ n-1(MtCO_{2eq}) \\
&\qquad + \frac{HM\ or\ LM\ carbon\ savings\ in\ 2050\ (MtCO_{2eq})}{25\ (yrs)}
\end{aligned}
$$

Equation 2

$$
\begin{aligned}
&Carbon\ emissions\ in\ year\ n\ (MtCO_{2eq}) \\
&\qquad = BAU\ carbon\ emissions\ in\ year\ n\ (MtCO_{2eq}) - HM\ carbon\ savings\ in\ year\ n\ (MtCO_{2eq}) \\
&\qquad - LM\ carbon\ savings\ in\ year\ n\ (MtCO_{2eq})
\end{aligned}
$$

Equation 3

Table 9: Cumulative HM and LM Carbon Savings and Emissions (2025-2050)

Year	Carbon Savings ($MtCO_2eq.$)				Carbon Emissions ($MtCO_2eq.$)			
	High Probability		Low Probability		High Probability	Low Probability	BAU	Net-Zero
	HM	LM	HM	LM				
2025	0	0	0	0	9.48	9.48	9.48	9.48
2026	0.12	0.14	0.19	0.27	9.30	9.10	9.56	8.34
2027	0.25	0.27	0.37	0.54	9.11	8.72	9.63	7.21
2028	0.37	0.41	0.56	0.81	8.93	8.34	9.71	6.07
2029	0.49	0.55	0.75	1.07	8.75	7.97	9.79	4.93
2030	0.61	0.69	0.93	1.34	8.56	7.59	9.86	3.79
2031	0.74	0.82	1.12	1.61	8.38	7.21	9.94	3.60
2032	0.86	0.96	1.31	1.88	8.20	6.83	10.01	3.41
2033	0.98	1.10	1.49	2.15	8.01	6.45	10.09	3.22
2034	1.10	1.23	1.68	2.42	7.83	6.07	10.17	3.03
2035	1.23	1.37	1.87	2.69	7.64	5.69	10.24	2.84
2036	1.35	1.51	2.05	2.95	7.46	5.31	10.32	2.66
2037	1.47	1.64	2.24	3.22	7.28	4.93	10.39	2.47
2038	1.59	1.78	2.43	3.49	7.09	4.55	10.47	2.28
2039	1.72	1.92	2.61	3.76	6.91	4.17	10.54	2.09
2040	1.84	2.06	2.80	4.03	6.73	3.79	10.62	1.90
2041	1.96	2.19	2.99	4.30	6.54	3.41	10.70	1.71
2042	2.08	2.33	3.17	4.57	6.36	3.03	10.77	1.52
2043	2.21	2.47	3.36	4.83	6.17	2.66	10.85	1.33
2044	2.33	2.60	3.55	5.10	5.99	2.28	10.92	1.14
2045	2.45	2.74	3.73	5.37	5.81	1.90	11.00	0.95
2046	2.58	2.88	3.92	5.64	5.62	1.52	11.08	0.76
2047	2.70	3.01	4.11	5.91	5.44	1.14	11.15	0.57
2048	2.82	3.15	4.29	6.18	5.26	0.76	11.23	0.38
2049	2.94	3.29	4.48	6.44	5.07	0.38	11.30	0.19
2050	3.07	3.43	4.67	6.71	4.89	0.00	11.38	0.00

References

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. Resources, Conservation and Recycling, 197, 107056.

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

HAFEZ, H., BAJIĆ, P., AIDAROV, S., MALIJA, X., DREWNIOK, M., PURNELL, P. & TOŠIĆ, N. 2024. Parametric study on the decarbonisation potential of structural system and concrete mix design choices for mid-rise concrete buildings. Materials and Structures, 57, 85.

IEA. 2022. Tracking Cement [Online]. Available: https://www.iea.org/energy-system/industry/cement [Accessed April 15th 2024].

MONTEIRO, P. J. M., MILLER, S. A. & HORVATH, A. 2017. Towards sustainable concrete. Nature Materials, 16, 698-699.

MPA 2023. MPA Cement stats data collation template BH Collation F v3. In: MINERAL PRODUCTS ASSOCIATION (ed.).

MPA 2025. Fact Sheet 18 Embodied CO_2e of UK cements. In: MINERAL PRODUCTS ASSOCIATION (ed.).

SCRIVENER, K., MARTIRENA, F. B., SHASHANK & MAITY, S. 2018. Calcined clay limestone cements (LC3). Cement and Concrete Research, 114, 49-56.

SHI, X., ZHANG, C., LIANG, Y., LUO, J., WANG, X., FENG, Y., LI, Y., WANG, Q. & ABOMOHRA, A. E.-F. 2021. Life Cycle Assessment and Impact Correlation Analysis of Fly Ash Geopolymer Concrete. Materials, 14, 7375.

UKQAA 2021. Consolidation Data 2021 Stats. In: ASSOCIATION, U. Q. A. (ed.).

UN COMTRADE 2022. 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. The International Journal of Life Cycle Assessment, 21, 1218-1230.

WORLD STEEL ASSOCIATION 2022. World Steel in Figures.

6. DECARBONISATION SYNERGIES: Exploring Cross-Sectoral Links though Blast Furnace Slag

The third article, published in *Resources, Conservation, and Recycling* examines the unintended consequences of decarbonisation in the UK steel sector on low-carbon cement production, as domestic GGBS supplies become threatened. Using material flow analysis, LCA, and economic modelling, the risks of cross-sectoral trade-offs were highlighted, demonstrating the need for integrated industrial decarbonisation policies in the UK. I hereby certify that I co-led this publication alongside Jacob W. Whittle. I was responsible for the cement sector data curation and LCA analysis which encompassed cement GGBS for use as an SCM. I also significantly contributed to the development methodology, the Excel based LCA model, visualisation of results, and writing and reviewing of the original draft. I also supported Jacob W. Whittle in the submission process including responding to comments left by peer-reviewers.

From symbiosis to scarcity: Evaluating disruption associated with decarbonisation to circular waste materials between the UK cement and steel sectors

Jacob W. Whittle (1), Madeline C.S. Rihner (2), Hisham Hafez (3), R.M. Eufrasio Espinosa (4), David I. Fletcher (1), Brant Walkley (2), Lenny S.C. Koh (4)

Affiliations

1: School of Mechanical, Aerospace, and Civil Engineering, University of Sheffield, UK, S1 3JD
2: School of Chemical, Materials, and Biological Engineering, University of Sheffield, UK, S1 4LZ
3: School of Civil Engineering, University of Leeds, UK, LS2 9LG
4: Management School, University of Sheffield, UK, S10 1FL

Corresponding Authors

Jacob W. Whittle, Department of Mechanical Engineering, University of Sheffield S1 3JD (jwwhittle1@sheffield.ac.uk)
David I. Fletcher, Department of Mechanical Engineering, University of Sheffield S1 3JD (d.i.fletcher@sheffield.ac.uk)
Lenny S.C. Koh, Management School, University of Sheffield, UK, S10 1FL (s.c.l.koh@sheffield.ac.uk)

Keywords

Steel, cement, decarbonisation, manufacturing, supply chain, net zero

Highlights

- Decarbonisation pathways will disrupt UK cement-steel industrial symbiosis
- UK steel sector is expected to reach decarbonisation targets through transition to secondary steelmaking
- Shortages in ground granulated blast furnace slag supply due to domestic and global steelmaking shifts could limit the availability of low carbon cement
- Low carbon supplementary cementitious materials threatened by reliance on imports

Glossary

BaU: business-as-usual

BF-BOF: blast furnace - basic oxygen furnace

BFS: blast furnace slag

CC: calcined clays

CCUS: carbon capture, usage and storage

CO_2: carbon dioxide

EAF: electric arc furnace

FA: fly ash

GGBS: ground granulated blast furnace slag

IEA: International Energy Agency

LC³: limestone calcined clay cement

LCA: life cycle assessment

MFA: material flow analysis

PC: Portland cement

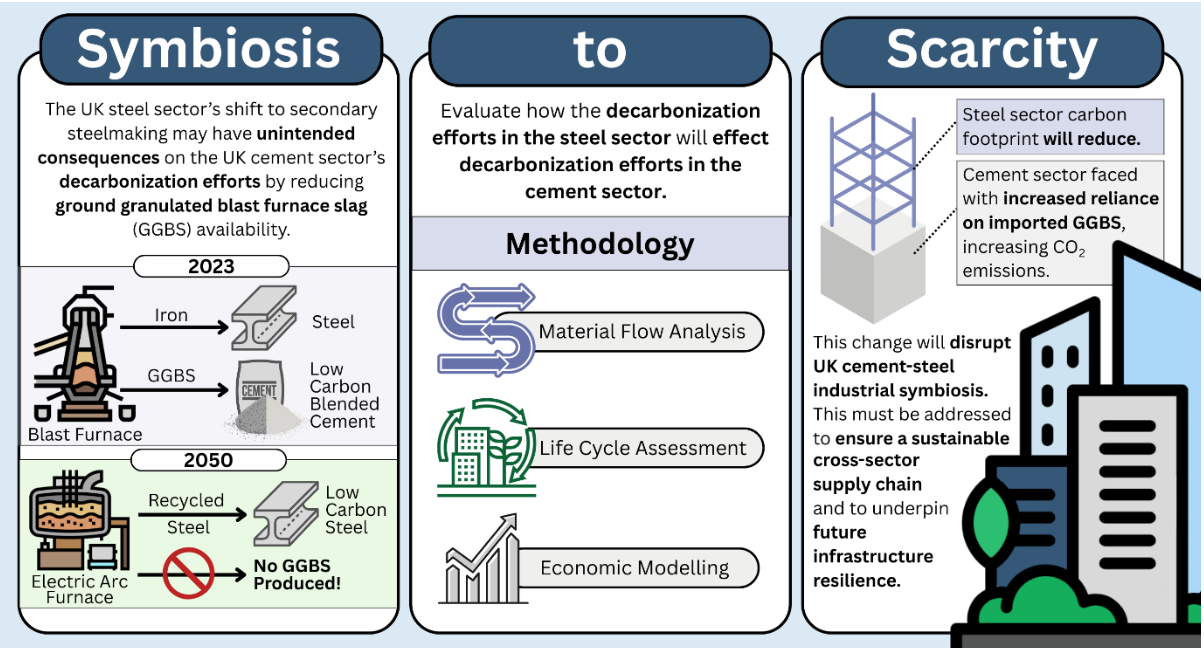
SCM: supplementary cementitious material

SDS: Sustainable Development Scenario

STEPS: Stated Policies Scenario

UK: United Kingdom

Graphical Abstract



Abstract

The UK cement and steel industries are decarbonising rapidly to meet net-zero targets. This study explores the unintended consequences of these efforts, particularly the potential disruption of industrial symbiosis between sectors. Cement production in the UK increasingly relies on ground granulated blast furnace slag (GGBS), a low carbon supplementary cementitious material (SCM). However, the shift from primary to secondary steelmaking threatens domestic GGBS supply. This research uses material flow analysis, life cycle assessment, and economic modelling to evaluate future GGBS availability, carbon intensities, and supply chain vulnerabilities. Findings indicate that although the steel sector is expected to reduce its environmental impact, this will cause the cement sector to face a potential shortfall in domestic SCMs, increasing reliance on imports through cross-

sector decoupling and stagnation of decarbonisation. Addressing these challenges is vital to ensure a sustainable cross-sector supply chain and support future UK and global infrastructure resilience.

1.Introduction

Cement and steel are fundamental to modern infrastructure, making them the two most readily consumed materials with over 4 Gt of cement [1] and nearly 2 Gt steel produced globally in 2023 [2]. However, these industries have significant environmental impacts due to their energy- and process-intensive processes [3, 4]. In 2023, both sectors generated a combined 5 Gt of direct carbon dioxide (CO_2) emissions - 15% of total anthropogenic CO_2 emissions released annually [5-7]. This has intensified pressures to decarbonise using innovative solutions and mechanisms to meet net-zero targets by 2050 [3], including an increased use of the circular economy and industrial clustering concepts [8]. These can both be underpinned by the theoretical concept and practical application of industrial symbiosis.

1.1. Industrial Symbiosis

Industrial symbiosis can be interpreted in different ways [9], but is broadly defined as the long-term engagement between different companies or industries in the physical exchange of materials, by-products, energy, or information [9-12]. Against the background of decarbonisation, this is a vital mechanism to help reduce the environmental impact of one or multiple parties. This can be achieved through converting by-product streams in one industry to form a supply chain which supplements or replaces raw and virgin materials within an industrial process typically within a separate industry [13], avoiding early disposal of otherwise useful material [14], and preserving natural resources [15]. Industrial symbiosis also has the potential to unlock further economic and social benefits beyond the parties directly involved [8, 9]. There are innumerable literature examples of industrial symbiosis at range of scales and industries including symbiosis of water sources across all industries in a single city [16], symbiosis between mushroom farmers and beer brewers [17], and synergy in regional minerals mining and production [18].

1.2. Symbiosis between cement and steel

An interesting, and long standing, global application of industrial symbiosis is between the steel and cement sectors. There are several examples of symbiosis including utilising end of life steel scrap from finished construction grade cement products as a steel scrap source [19] and the use of dusts from both industries as carbonation materials [20]. However, by far the most common is the use of ground granulated blast furnace slag (GGBS). This is created as a by-product of the steelmaking process and used within the cement manufacturing process, a symbiosis summarised in Figure 1. Initially studied for its potential as a performance enhancing material within cement, the relationship between both sectors has shifted significantly to a focus on the reduction of environmental impact within both through repurposing of this material [21-23]. However, no literature can be found which assesses the potential effects of a change in the production landscape of both sectors due to global decarbonisation efforts on this long-standing symbiotic relationship.

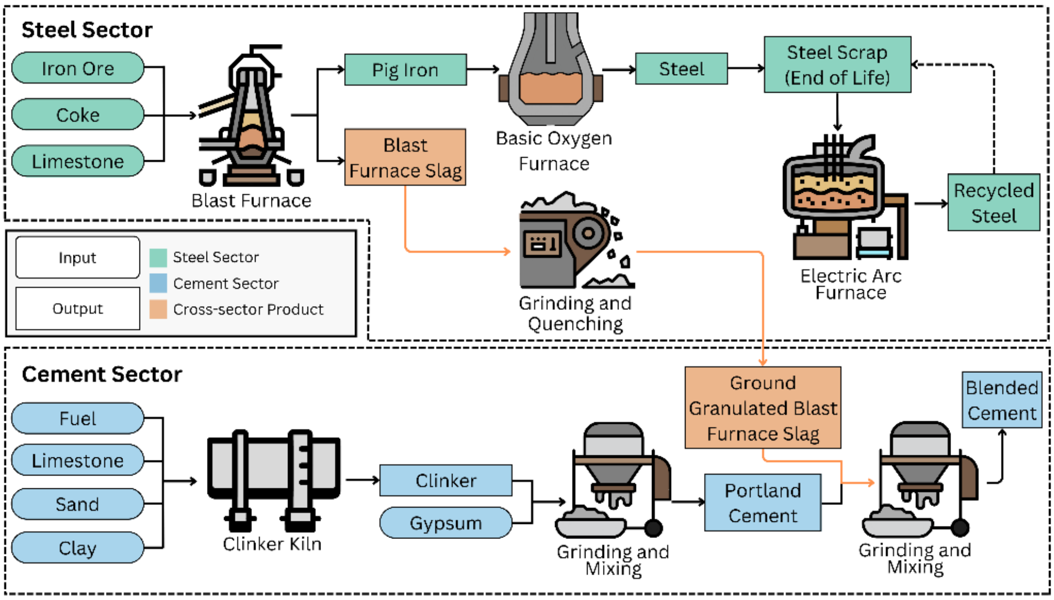


Figure 1: Cement and steel manufacturing industrial symbiosis process flowchart.

The CO_2 emissions associated with the production of cement are mostly (in excess of 90%) caused by the production of clinker, which is the main constituent of cement [24]. Clinker is a mixture of limestone and other materials that are heated within a kiln then subsequently ground into a fine material for use within cement [24], as shown in Figure 1. Globally, the clinker-to-cement ratio, also known as the clinker factor (i.e. the percentage of clinker used within a given cement mix), is approximately 0.70 [25]. Although direct circularity is possible within the cement industry [26] it is challenging [27]. Therefore, a major strategy to reduce CO_2 emissions is replacing ordinary Portland cement (PC) with cement blends that contain a greater proportion of supplementary cementitious materials (SCMs), thus reducing clinker use. These can also enhance performance [28, 29]. Commonly utilised SCMs include GGBS [28], limestone [30], calcined clays (CC) [31], and fly ash (FA) from coal combustion [28, 32]. However, SCM application varies significantly by region. Despite historically high usage and substitution rates within cement manufacture [33, 34], the United Kingdom's (UK) early transition to cleaner energy sources [35-37] has resulted in a decline of FA availability and usage as a SCM, while legacy FA recovery remains uncertain [38]. While limestone calcined clay cement (LC3) has shown technical success elsewhere [30], poor reactivity of local CC stocks [39] and the limited availability of limestone fines [40, 41] has challenged its use within the UK. As a result, GGBS is the UK's most widely used SCM [2, 41, 42]. GGBS is a fine, glassy substance produced by grinding and rapidly cooling molten blast furnace slag (BFS); a co-product of the iron smelting process within the primary blast furnace-basic oxygen furnace (BF-BOF, or BOF) steelmaking route [43] as shown in Figure 1. Consequently, UK cement decarbonisation is largely dependent on a symbiotic relationship with the steel industry. However, falling domestic steel production, over the past decade, means that the UK has begun relying heavily on GGBS imports to meet industrial demand [2, 42]. Given this, existing literature suggests limiting GGBS use to 20% in the UK for performance and material availability reasons [44]. Maintaining this rate is preferrable, but industrial shifts may result in further GGBS shortages.

The remaining major steel manufacturers in the UK, which currently utilise primary steelmaking, aim to reduce emissions by 85% by 2035 and reach net-zero by 2050 [45]. In the short to medium term this will be achieved through process decarbonisation of primary steelmaking, making use of emerging technologies including hydrogen, carbon capture, usage, and storage (CCUS), and alternative materials [45]. However, all manufacturers expect to completely transition to secondary steelmaking routes, utilising electric arc furnaces (EAF), by 2050 at the latest [46, 47]. This transition process is already underway at one of the two remaining major primary steelmaking sites [48]. Secondary steelmaking relies on implementation of a circular economy within the steel sector, with scrap material becoming the primary iron or steel source. This in turn, significantly reduces the material's environmental impact. Although there are technical [49, 50], regulatory [51], and practical [51, 52] challenges, this may occur much sooner [53]. However, these furnaces do not produce the same co-products as the BOF route. Furthermore, this transition toward cleaner steelmaking is likely to occur on a global scale, albeit at different rates [54]. As a result, the availability of steel, cement, and associated co-products will change, affecting the economic value of each material, and thus industrial decision-making [55, 56]. Changes in value could mean that GGBS is no longer economically viable or environmentally sustainable to continue importing into the UK, resulting in a major shift in the balance of global supply chains - increasing the reliance on SCM imports and potentially an increase in cement-related CO_2 emissions.

This relatively unique position of SCM use, type of steelmaking, and ambitious net-zero related targets for heavy industry make the UK a perfect case study to assess the potential unintended consequences of decarbonisation on industrial symbiosis, which has not been seen previously in literature. This paper aims to conduct a novel study into how shifts in the steel industry, both in the

UK and globally, will impact UK cement sector decarbonisation. To achieve this, several steel and cement transition scenarios will be explored, predictions will be made on material economic value, and the environmental impact of GGBS will be assessed. The findings will provide insights into the likely carbon intensity and emissions of both sectors in 2050, aiding informed decision-making and mitigate cross-sector supply chain disruption and risk. This study is structured as follows: Section 2 describes the study methodology, Section 3 presents study findings, and Section 4 discusses the results presented in Section 3, and Section 5 summarises key insights and their implications on steel-cement symbiosis.

2. Materials and Methods

2.1. Outline

To evaluate the impact of decarbonisation strategies between the synergistic steel and cement sectors, as outlined in Figure 1, the carbon emissions produced in each sector must first be assessed. The carbon emissions associated with any given product are defined by the volume of product consumed and its carbon intensity. Therefore, to determine the cement and steel sector's annual consumption volumes and embodied carbon, a material flow analysis and life cycle assessment were performed, respectively. This is a methodology combination which has been found to yield more robust and transparent results, compared to utilising these methods independently [57]. The methodology outline, in addition to the flow of data, is described in Figure 2.

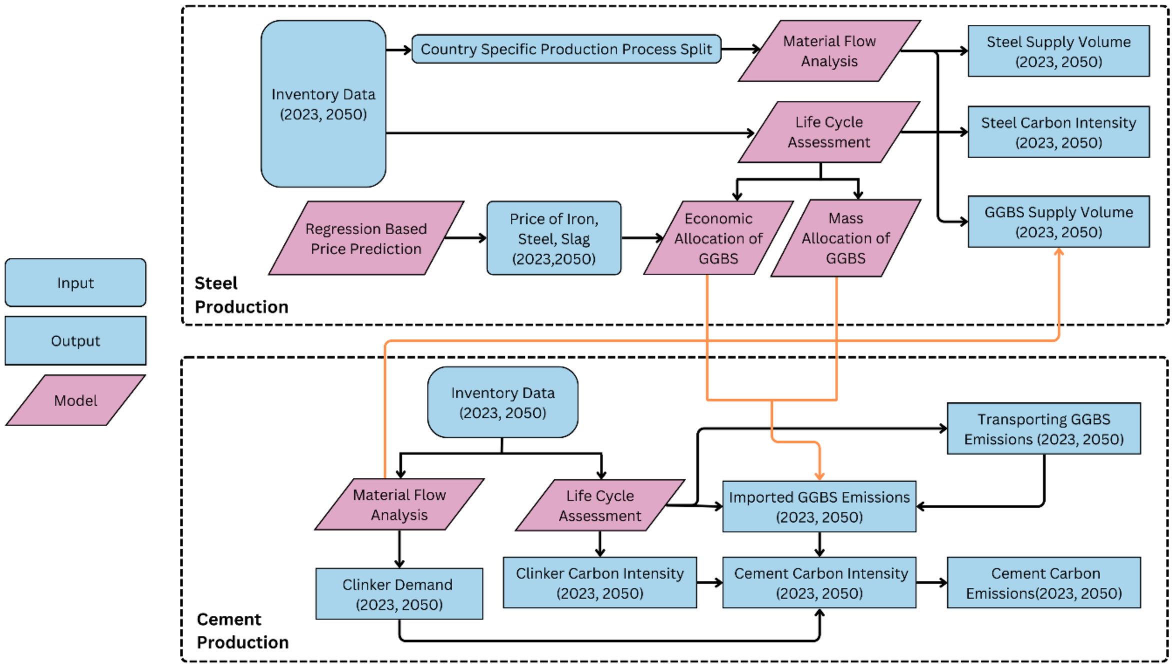


Figure 2: Methodology outline.

2.2. Scenarios

The baseline year for this study is 2023 as this is the most recent year for which complete data is available. Alongside this, several 2050 scenarios were considered which focus on key transition strategies in each sector: the shift from primary to secondary steelmaking in the steel sector and the increased use of SCMs in the cement sector. These are outlined in Table 1.

Global cementitious material demand is expected to remain constant until 2050, with production rising in the Global South but declining in the Global North [58, 59]. However, in a drive to reduce the emissions associated with cement production, the clinker factor will decrease regionally, reducing clinker demand while increasing SCM consumption. In the UK, the current clinker factor is 0.70 [41], but must drop to 0.50 to achieve net-zero targets [60]. It is also assumed that while the percentage of GGBS used will remain constant to 2050 (17% of the UK cement mix), the consumption of other SCMs will shift from FA to CC and limestone fines due to other industrial shifts.

Similarly, global steel production is expected to continue to grow by 3.5% annually to approximately 1960 Mt by 2050 [2], but steelmaking transition pathways remain unclear outside of a small number of European countries. Accordingly, the UK has been modelled as undertaking a complete shift (100%) from BOF to EAF steelmaking. This means that all steel manufactured in the UK will be produced using secondary steelmaking methods by 2050, though it is likely that this will occur much sooner. Other steel producing countries are the subject of a 25 to 75% shift in route toward EAF steelmaking to assess global sensitivity in absence of reliable transition pathway plans - particularly with respect to GGBS availability. China (90.1% BOF), India (43.6% BOF), and Japan (71.1% BOF) currently produce a combined 65% of global steel [61] and are likely to remain the largest by market share. Therefore, it is assumed that each country assessed (UK, China, India, and Japan) retains the same global production share through each scenario, but the total volume of steel produced in 2050 by BOF decreases by the percentage noted. A global average has also been assessed to understand regional disparities in production. Each country is assumed to satisfy domestic scrap demand to enable sufficient high-quality steel scrap availability [51].

Table 1: Steel and cement transition scenarios from baseline to 2050.

Country / Region	**2050 Transition Scenario**	**Cement Transition (Clinker Factor)**	**Steel Transition (EAF Shift Percentage)**
UK	Low	0.70 (Mineral Products Association, 2023)	100%
	Medium	0.60	
	High	0.50 (Rihner et al., 2025)	
China, India, Japan, Global Average	Low	0.71 (International Energy Agency, 2024)	25%
	Medium	0.63	50%
	High	0.55 (International Energy Agency, 2024, Global Cement and Concrete Association, 2024)	75%

Within the transition scenarios described, the effects of different regional and sector-specific decarbonisation strategies were also assessed, incorporating emissions reductions beyond those achieved through lower clinker factors and the transition to EAF. In both sectors, establishing current [61] and predicting future process energy intensity (GJ/tonne of product) is challenging as these values are affected by a multitude of technological, financial, and geopolitical issues. However, there are several common decarbonisation strategies, at various market readiness levels, that could contribute to the reduction of carbon emissions associated with material production. These include CCUS [64, 65], material efficiency [64], technology performance improvements (e.g. recycling concrete fines, or fitment of top-pressure recovery turbines) [64, 65], electrification [64], and use of alternative energy sources [64, 65]. In the steel sector, two decarbonisation pathways were considered based the IEA's iron and steel roadmap pathways: 'Stated Policies Scenario' (STEPS) and 'Sustainable Development Scenario' (SDS) [64]. The STEPS roadmap reflects countries' energy and climate-related commitments through 2050, while SDS outlines the major changes required to achieve the primary energy-related goals of the UN Sustainable Development Agenda (Supplementary Information (SI), section S1). For the cement sector, a recent UK cement market analysis was used to define the 2050 decarbonisation pathway [60] (SI section S4). In order to contextualise the plans of both sectors, an additional business-as-usual (BaU) scenario has been included which does not see any process decarbonisation, but does consider the decarbonisation of regional electricity, material, and fuel in addition to the transition scenarios noted in Table 1. Reductions in the carbon intensity of these indirect processes were determined based on projected values provided in industry roadmaps and relevant publications.

2.3. Material Flow Analysis

To quantify the flow and stock of material within the cement and steel sectors, and therefore aggregate sectoral consumption volumes, a material flow analysis (MFA) was performed [66]. The flow of materials is summarised in Figure 1. All data was taken from publicly available regional and international reports [2, 25, 41, 61, 62], and secondary literature sources [67]. The defined system boundary is cradle-to-gate, which encompasses the annual consumption values of clinker, iron, GGBS, other SCMs (FA, CC, and limestone fines), cement, and steel in the UK. To maintain a focus on domestic consumption, exported products were excluded. In addition, downstream flows of finished products such as concrete and reinforced concrete elements were not considered.

2.4. Life Cycle Assessment

Life cycle assessment (LCA) is a methodological framework which can provide a general perspective on the environmental impact of a product to support decision making [68, 69]. LCA is standardised by ISO14040 and ISO14044 [70, 71]. The first stage of any LCA is to define the goal. The goal of this LCA is to assess the carbon intensities associated with several products within the UK steel and cement sector including clinker, GGBS, cement, and steel. In line with this, several different scenarios were considered including different temporal (2023 and 2050), geographic (UK, China, India, Japan, global average), and transition scenarios as noted in Table 1. In an LCA, a study's scope is defined by three main components: the system boundary, functional unit, and allocation procedure [70]. The system boundary for this LCA is cradle-to-gate which includes raw material extraction (stage A1), the transportation of those raw materials to the factory (stage A2), as well as the processing and production of the product (stage A3) [72]. Since the study aims to assess the environmental impact of several products in which material function is not necessary, a mass declared unit (e.g. one kilogram) was selected for each product.

When assessing the carbon intensity of co-products, partially on a cross-sector level, special consideration must be given to the allocation procedure selected. ISO14040 defines allocation as the "partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems" [70]. Previous studies have found that within the cement and steel sector there is no clear consensus on how BFS (as well as other co-products) should be allocated within the LCA method [73, 74]. Each sector appears to favour different methods of allocation to best suit study goals which makes cross-sector comparison challenging; contradicting the core principles of LCA [72]. Therefore, in this study, both mass and economic allocation were considered when determining the carbon intensity of BFS (SI, section S5).

The core method of producing GGBS is not expected to change, and so therefore the allocation by mass between products will not change between the baseline and 2050. Within the primary steelmaking route, BFS values per country are not typically reported, so a ratio of 0.28 tonnes of BFS produced per tonne of iron was assumed for all countries [75]. The exception to this is in the UK where no iron or BFS is produced in 2050, as defined in Table 1. Allocation by economic values however will change due to fluctuations of supply and demand, and therefore price, over time. By examining import and export trade flow data, the baseline global economic values for crude iron and GGBS were found to be £0.41/kg and £0.02/kg, respectively. In the UK, the export price of iron is significantly higher at £0.86/kg. Given the low supply and high demand for GGBS in the UK, the exporting price of the material is £0.18/kg [42]. To determine the 2050 economic values for both products, the price prediction methodology outlined in Section 2.5 was used.

Inventory analysis is the second stage of an LCA which includes the evaluation and collection of data required to fulfil the study's goal. Data has been compiled from a range of secondary sources through a top-down collection method, for use across and within each sector of analysis. Each source is as spatially, temporally, and technically relevant as possible. A summary of this is outlined below, but all data, detailed calculations, and sources can be found in the associated SI. For both the steel and cement carbon intensity calculations, the electricity and fuel emission factors were regionalised where possible to most accurately model disparities in decarbonisation pathways. All data and decarbonisation pathways are taken from government (UK GHG, China CF) or literature [54] sources (SI, section S2). The carbon intensity for both BOF and EAF steelmaking (kgCO_2eq/kg) (S_{CI}) is derived from electricity (kgCO_2eq/GJ) (E_{CF}), fuel emission factors (kgCO_2eq/GJ) (F_{EF}), the fuel mixture (including electricity) (%) (F_M), and steelmaking process energy intensity (GJ/kg) (S_{PE}) as noted in Equation 1.

$$S_{CI} = S_{PE} \times ((F_M \times F_{EF}) + (F_M \times E_{CF}))$$

Equation 1

The average steelmaking process energy intensity ranges from 19.39 to 14.00 GJ/tonne of steel (SI, Section S1). The fuel mixture used in each steelmaking process is likely to shift in favour of electricity driven, and more sustainable processes. Data supporting the aggregated fuel mixture consumption for each transition scenario is taken from WorldSteel [61] or IEA [64] (SI, section S3).

The carbon intensity of cement (kgCO_2eq/kg) (Ct_{CI}) in the UK is derived from the carbon intensity of clinker (kgCO_2eq/kg) (Cl_{CI}), the cement's clinker factor (C_F), the average carbon intensity of SCMs (kgCO_2eq/kg) (SCM_{CI}), the carbon intensity of GGBS (kgCO_2eq/kg) (G_{CI}), the cement's GGBS factor (i.e. the amount of GGBS used within a given cement mix) (G_F), the carbon intensity of the transport of both GGBS (kgCO_2eq/kg) (G_T) and clinker (kgCO_2eq/kg) (Cl_T), the electric energy required for indirect processes (GJ/kg) (EC_R), and the emission factor of electricity (kgCO_2eq/GJ) (EC_{EF}). This calculation is summarised in Equation 2.

$$Ct_{CI} = Cl_T + (Cl_{CI} \times C_F) + (SCM_{CI} \times (1 - (G_F + C_F)) + (G_{CI} \times (G_F)) + G_T + (EC_R \times EC_{EF})$$

Equation 2

The carbon intensity of the clinker component consists of process emissions occurring from calcination in the cement kiln and the carbon intensity of the kiln's fuel mix (SI, section S4). The clinker factor is noted in Table 1. The average SCM carbon intensity was determined by taking the mass ratio of all three other SCM types considered (limestone, CC, and FA) in addition to gypsum and multiplying each by their respective consumption volume and carbon intensity value. All these materials were assumed to have negligible transportation distances given current material stocks. To determine carbon intensity of GGBS, the same calculation procedure was applied as was done for the other SCMs.

In addition to the emissions arising from the clinker and GGBS production process, those arising from importation must also be accounted for. In the case of clinker, this includes clinker sold as a product and cement products containing clinker (e.g., CEM1). While countries that are in close proximity to the UK such as the Netherlands, Spain, Germany and France have consistently been some of the UK's largest exporters of GGBS, a shift to relying on slag exports from China, India, and Japan is likely due to pan-European decarbonisation targets [76]. This shift has already begun in the case of Japan. Since 2019, it has become one of the largest slag exporters to the UK with the country accounting for 22% of all UK GGBS imports annually. To determine the carbon intensity of transportation for the 2023 baseline scenario, a weighted average was taken between the top four exporting countries of each product. To account for regional production process differences, carbon intensity values for clinker and GGBS production were retrieved from literature sources for each of the four exporting countries considered. The processing of BFS into GGBS was assumed to take place in the country of origin and all slag imported into the UK is GGBS (SI, section S5). Transportation distances were determined using secondary data sources [77, 78] (SI, section S6). When assessing the impact of transportation in 2050, the top four exporters of clinker and cement products were assumed to be the same from the 2023 baseline. For GGBS however, several importation scenarios for 2050 were considered and are detailed in the interpretation step. Lastly, the electric energy required for indirect processes values were taken from secondary sources [25, 79].

In line with the study scope, embodied carbon is the only impact indicator analysed at the impact stage. The interpretation of the LCA results include examining several decarbonisation pathways in both the steel and cement sector. In addition, several transportation scenarios were considered for GGBS. For the baseline scenario, a weighted average between the top four GGBS importers (comprising 77% of all slag imports to the UK) was considered which include the Netherlands, Spain, France, and Japan. Six different importation scenarios were considered for 2050: (1) the 2023 GGBS import countries and their import ratios, (2) equal import from the assumed top three steel producers (China, India, and Japan), (3) all GGBS import from China, (4) all GGBS import from India, (5) all GGBS import from Japan, and (6) a global transport average. The global average scenario considers a weighted average based on the GGBS amount of the seven exporting countries assessed.

2.5. Economic Modelling

To perform a sensitivity analysis on the impact of the allocation method, the economic value of iron and GGBS in 2050 must be predicted. A robust, time series methodology was tailored to address the inherent volatility and inconsistencies in trade flow data [80, 81]. Outliers were removed to reduce variation in the dataset. An ARIMA [82] model was used to predict future trends and cyclical patterns, and the model was implemented in Python. The prediction provided insights into the long-term price movements of slag and steel up to 2050 (SI, section S7). Although regional differences were observed, the volatility of material production and value results in wide confidence intervals regardless of source. Therefore, the same changes in economic value of material were applied to all countries of interest to enable useful comparison.

3. Results

3.1. Material Flow Analysis

The results of each MFA conducted are shown in Figure 3a through Figure 3d, where Figure 3d shows the effects of a high transition in both sectors globally. The MFA of both the low and medium transition scenarios can be found in the SI (SI, section S8), but the numerical outcome of both analyses is discussed below. Examining the UK baseline (Figure 3a), it was determined that 7.5 Mt of steel and 15.24 Mt of cementitious materials are consumed annually. Although not directly relevant to the analysis, the UK exported 2.6 Mt, imported 4.5 Mt, and domestically produced 5.6 Mt of steel. This indicates that the trade of steel is broadly driven by specialisation of UK manufacturers in different steel grades and products but also highlights that the domestic circular economy in steel is fractured. Consequently, this may be limiting the domestic availability of sufficient high-quality grades of scrap as well as increasing UK reliance on complex global supply chain routes (in a similar parallel to the cement industry as noted below). Currently 1.1 Mt of UK steel is produced via EAF, however the expected complete shift to secondary steelmaking by 2050 (Figure 3b) will increase the UK global market share of EAF steel to 0.59-0.83%, depending on global trends (Figure 3d). China, India, and Japan are expected to remain dominant producers in 2050, accounting for at least 263 Mt of global EAF steel (37.5% of production). While the total consumption of cementitious materials is expected to stay constant to 2050, the reduction in clinker factor in the UK cement sector will increase demand for SCMs from 3.23-6.86 Mt, whilst reducing clinker demand from 10.74-7.62 Mt. This will reduce reliance on existing material sources (Spain, France, Algeria, and Ireland), but at least 2.54 Mt will still need to be imported. The assumption that GGBS demand will stay as a constant proportion of SCM use in the UK (17% of cement mix design) means the demand volume falls slightly to 2.59 Mt in 2050. This is a major supply chain risk to the sector, as the global demand for GGBS (366 Mt) will outstrip supply (274 Mt to 353 Mt available, depending on the transition scenario) due to increased SCM consumption (a 62% rise) as global clinker rates drop and BOF steelmaking decreases globally as illustrated in Figure 3d. As shown in Figure 3c, the consumption of CC (2.13 Mt) and limestone filler (2.13 Mt) also rises dramatically in the UK (accounting 62% of SCMs by 2050). However, this is also true globally which will demand approximately 871 Mt of each by 2050. Although the analysis of these materials is not the core focus of this study, this will compound issues surrounding the UK's reliance on imported material. This could leave UK manufacturers vulnerable to material availability and cost, and therefore impact the sector's competitiveness and rate of decarbonisation.

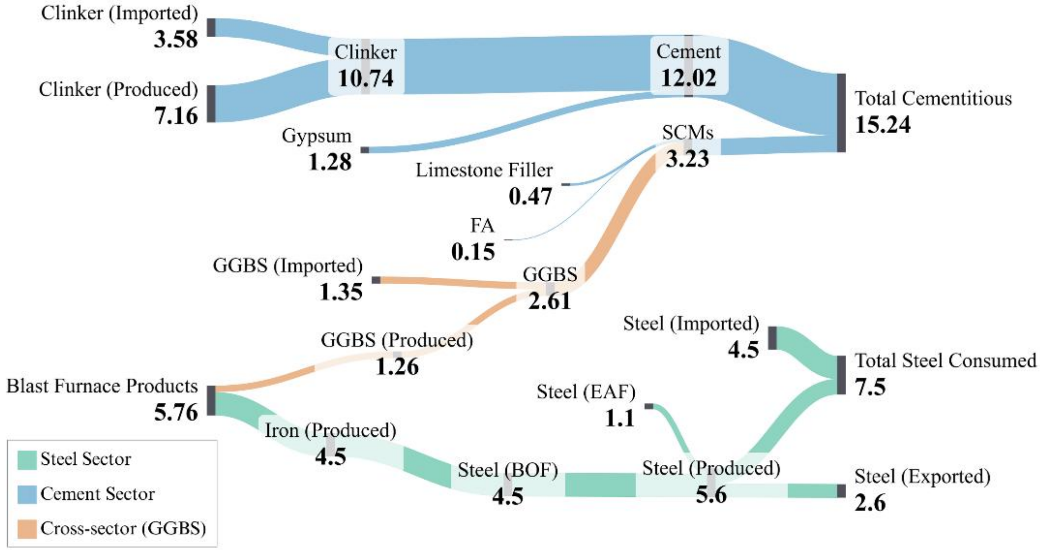


Figure 3a: UK MFA 2023, where all values are in Mt/year.

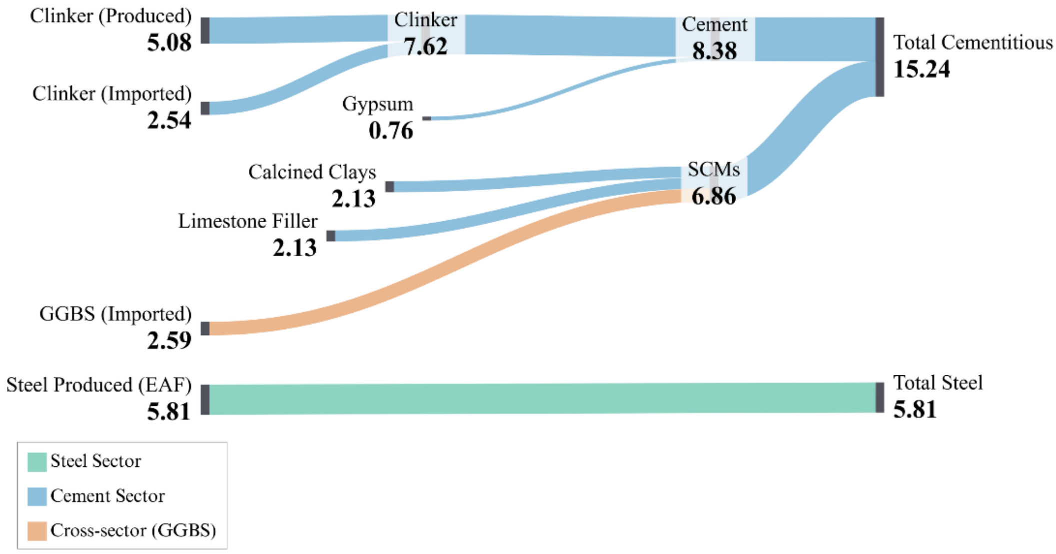


Figure 3b: UK MFA 2050, where all values are in Mt/year.

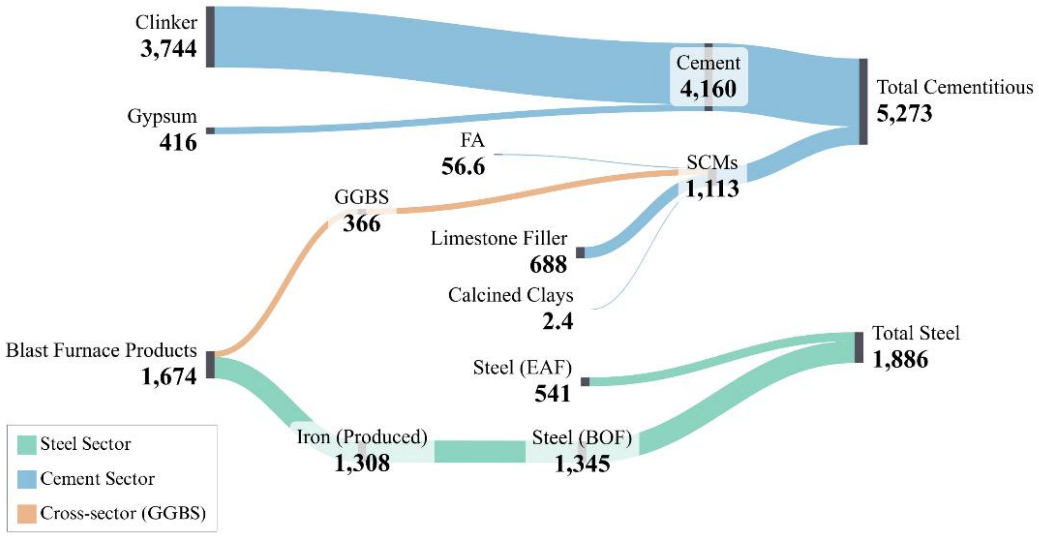


Figure 3c: Global MFA 2023, where all values are in Mt/year.

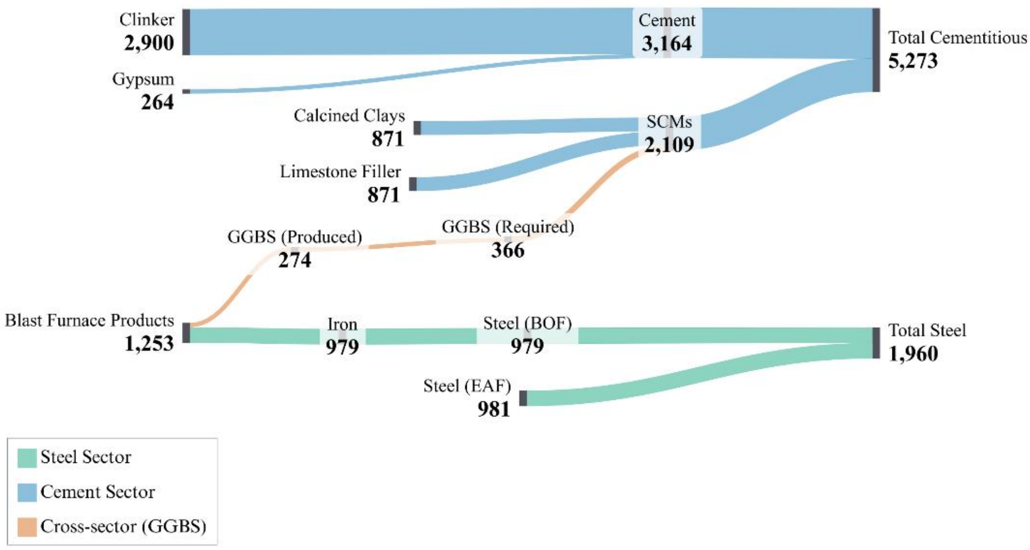


Figure 3d: Global MFA 2050 under the high scenario, where all values are in Mt/year.

3.2. Carbon Emissions

The analysis conducted indicates that the current combined global baseline carbon emissions for both the cement and steel sector is approximately 6.6 $GtCO_2eq$/yr. Whilst the cement sector value is in broad agreement with figures reported by the IEA [25], the steel sector value is approximately 20% higher [2]. As outlined, this study relies on a range of secondary data sources and a top-down data collection approach, which due to error truncation, is likely to result in an overestimation of values. However, the correlation of this combined global value gives confidence that the analysis performed is accurate enough to make predictions on the likely outcomes of the described transition scenarios and decarbonisation pathways. If both the highest rate of steel and cement production route transition is undertaken, as outlined in Table 1, alongside the implementation of all sector specific decarbonisation pathways, it is likely that the combined global carbon emissions will fall to approximately 2.9 $GtCO_2eq/yr$ by 2050, representing a joint emissions reduction of 56%. The effects of each scenario are explored on a sectoral level in Section 3.3.

The overall carbon emissions of each country or region of analysis are a function of material volume produced, split in production route, and the carbon intensity of each production route. As expected, results shown in Figure 4a indicate that carbon emissions arising from the UK steel sector will reduce by between 74% and 84%, depending on the decarbonisation pathway, due to the transition to EAF steelmaking. This represents a reduction in overall emissions to approximately 2 $MtCO_2eq$ from 10.2 $MtCO_2eq$ at the baseline. Emissions within the UK cement sector are expected to decrease between 2% and 56% compared to the baseline depending on the decarbonisation scenario; significantly less than the reduction potential expected from the steel sector. The higher range present in the expected cement sector reduction potential is attributed to the lack of certainty in decarbonisation strategy implementation. The UK steel sector's action calling for a complete shift to EAF steelmaking allows for a greater predicted emission reduction, whereas the cement sector can only estimate reductions based on minimum reported reduction values. Despite cement having a lower carbon intensity value (0.281 $kgCO_2eq/kg$) when compared to steel produced by EAF (0.312 tCO_2eq/t) under the 2050 UK SDS high cement decarbonisation scenario, the overall emissions associated with cement production (3.71 $MtCO_2eq$) are more than double that of steel production (1.81 $MtCO_2eq$) due to the predicted production volume of cement being higher than that of steel.

Examining the global market, as shown in Figure 4b, the current cement and steel sector baseline carbon emissions are 2.8 $GtCO_2eq/yr$ and 3.8 $GtCO_2eq/yr$ respectively. Also as indicated by Figure 3b, in the cement sector, these values are expected to decrease in 2050 by 25-66% depending on the cement transition scenario as well as the rate of implementation of regional decarbonisation strategies which have been modelled. The global steel sector will likely see a reduction in carbon emissions by 14-49% depending on the implementation of each decarbonisation pathway and EAF transition, and this rate of decarbonisation will have a direct impact on the cement sector as symbiosis through GGBS is present globally. This means that a reduction of overall emissions arising from the steel sector to between 3.3 to 1.9 $GtCO_2eq/yr$, by 2050, is likely. The results shown in Figure 4 are a global average, but analysis indicates that in relative terms, India is likely to experience the greatest reduction in carbon emissions due to the modelled increase in EAF steelmaking despite having the most carbon intensive sources of electricity and fuel. The world's largest steel producer, China, is also predicted to see a reduction in emissions and a small move towards EAF steelmaking. However, the country is still likely to contribute to at least half the global sector's emissions, even under the most ambitious decarbonisation targets. The production dominance of China, India, and Japan means that they contribute significantly to the global scenario.

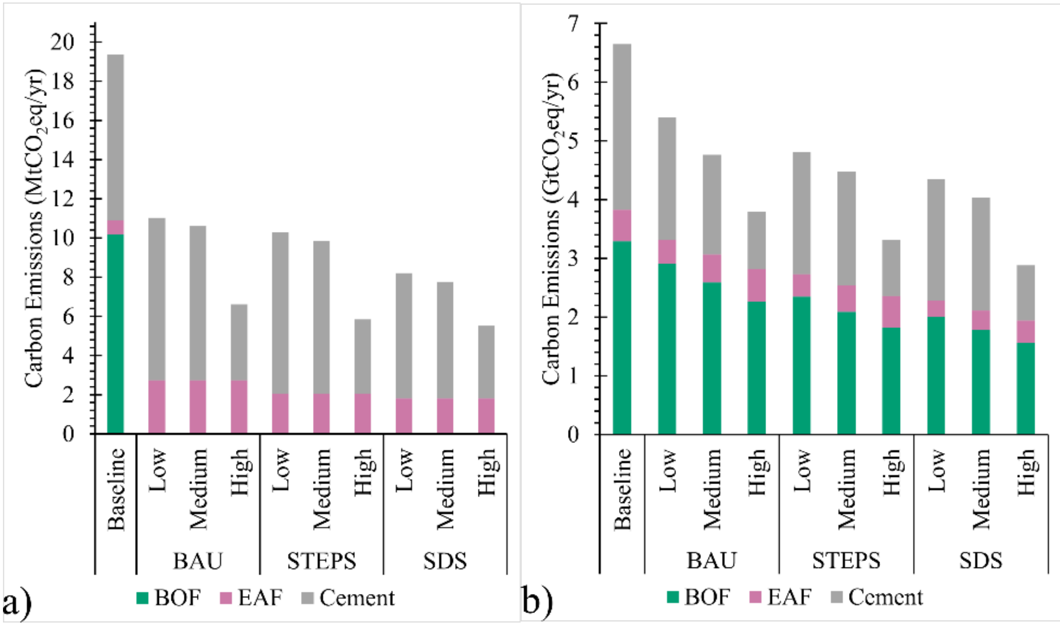


Figure 4: (a) UK, (b) global combined steel and cement sector emissions at the baseline (2023) and in 2050 under the specified scenario.

3.3. Sector Level Carbon Intensity

3.3.1. Steel

Using the LCA methodology described in Section 2.4, the carbon intensity of both steelmaking routes within each country under each decarbonisation scenario was determined as shown in Figure 5. It has been calculated that the baseline global carbon intensity of BOF and EAF steelmaking is 2.45 and 0.99 tCO_2eq/t respectively. Depending on the decarbonisation pathway selected, these values are likely to reduce to between 2.31 and 1.59 tCO_2eq/t and 0.57 and 0.39 tCO_2eq/t respectively. The analysis indicates that the UK is likely to have the lowest EAF steelmaking carbon intensity (between 0.47 and 0.31 tCO_2eq/t) primarily due to ambitious regional electricity decarbonisation targets that should result in an overall grid intensity that is at least half that of the global average. Although not directly explored here, if the UK were to retain a complementary BOF steelmaking capability, this analysis indicates that it could significantly reduce its carbon intensity from 2.28 tCO_2eq/t to between 2.23 and 1.53 tCO_2eq/t. The potential reduction in intensity is very similar to India and Japan, highlighting the minimal influence of regional decarbonisation efforts (e.g. electricity grid) in comparison to process decarbonisation through technology improvements (e.g. CCUS or hydrogen).

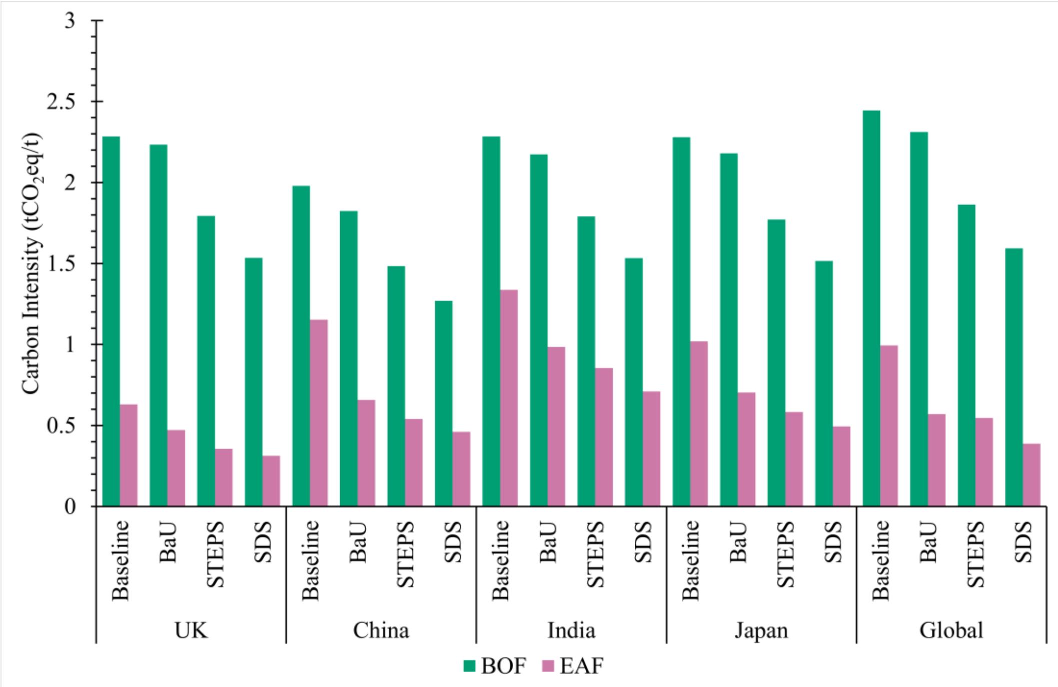


Figure 5: Carbon intensity (tCO_2eq/t) of steelmaking in each assessed country or region, under each decarbonisation scenario.

3.3.2. Cement

Ensuring the consistent use of the most representative allocation method within a system is vital to allow for effective cross-study, and cross-sector, comparison. The mass allocation of carbon intensity between iron and BFS was taken as 78.12% to 21.88%. Figure 6a illustrates the differences in proportion between economic allocation procedure by country. The economic allocation between iron and GGBS produced in the UK was calculated as 94.5% and 5.5%, respectively. Examining the values for China, India, and Japan it was found that the percentage allocation for iron and GGBS was similar between China and Japan (99.5% and 0.5%). In India the lower value of iron resulted in a higher percentage of emissions being allocated to GGBS (96.1% and 3.8%, respectively). The economic allocation of these materials is in broad agreement with existing studies [83]. Price volatility over the last decade underscores why future price estimation is necessary to accurately assess a co-product’s future environmental impact. It was determined that from 2023 to 2050, the price of iron and GGBS is expected to increase by 283.5% and 24.6%, respectively. This results in the economic allocation between iron and GGBS in the UK shifting to

97.5% and 2.5% respectively. As shown in Figure 6b, the use of economic allocation results in a significant reduction in the carbon intensity of GGBS because of the reduced allocation proportion associated with the material. The carbon intensity of GGBS at the UK baseline is 0.48 $kgCO_2eq/kg$ when mass allocation is selected. However, when economic allocation is applied, this drops to 0.15 $kgCO_2eq/kg$. In 2025, the MPA reported that the carbon intensity value (with economic allocation applied) for GGBS is equal to 0.155 $kgCO_2eq/kg$ [84]. The similarity between this study's calculated carbon intensity value and the reported MPA value supports the accuracy of the inventory data and economic values utilised. The difference in carbon intensity values present between the two allocation methods is primarily due to the economic value of GGBS. As its value is significantly lower than that of iron, the percentage allocated is also much lower. Out of the three countries assessed, GGBS from China exhibits the lowest carbon intensity with the value being 16% lower than the UK current economic baseline, despite the emissions associated with material transport. It is evident that economic allocation it is the best method to represent a complex, interlinked system because it better represents real world changes over time.

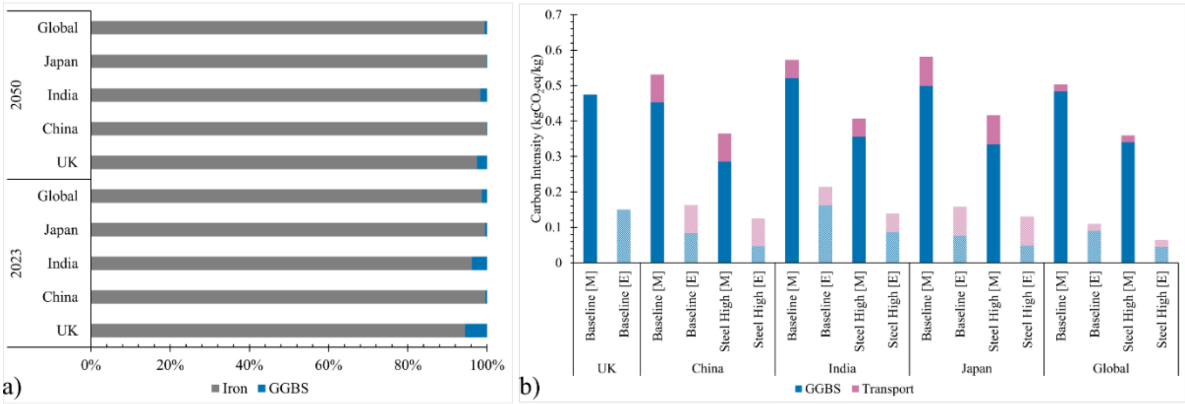


Figure 6: (a) Economic allocation percentages, (b) Impact of mass [M] and economic [E] allocation on GGBS carbon intensity.

Figure 7 illustrates the carbon intensity for the UK cement sector under the SDS steel decarbonisation pathway scenario. Multi-source scenarios consider GGBS imports from multiple countries to determine an overall carbon intensity, while single-source scenarios theoretically assess the carbon intensity of GGBS imported from a specific country. The baseline carbon intensity for cement in the UK is approximately 0.65 $kgCO_2eq/kg$ with clinker production comprising 81% of the total intensity. As expected, the overall cement carbon intensity decreases from the low to high transition scenarios, with the carbon intensity contribution from SCMs (CC, limestone, and FA) and GGBS increasing and the carbon intensity contribution of clinker decreasing. This shift is largely due to the decrease in clinker factor, with the lowest overall intensity at 0.29 $kgCO_2eq/kg$ – in which

GGBS is sourced equally from China, India, and Japan under the high cement transition scenario. Although the mass ratio between all SCMs remains constant throughout each 2050 scenario, the change in clinker factor results in an increase in SCM consumption, resulting in a greater contribution from all SCM's (including GGBS) toward overall intensity. The utilisation of sector wide decarbonisation pathways including CCUS, electrification, and the use of alternative fuels has the potential to reduce cement carbon intensity and thus emissions by over 50% in the best case. However, the route to implementation of these strategies is unclear.

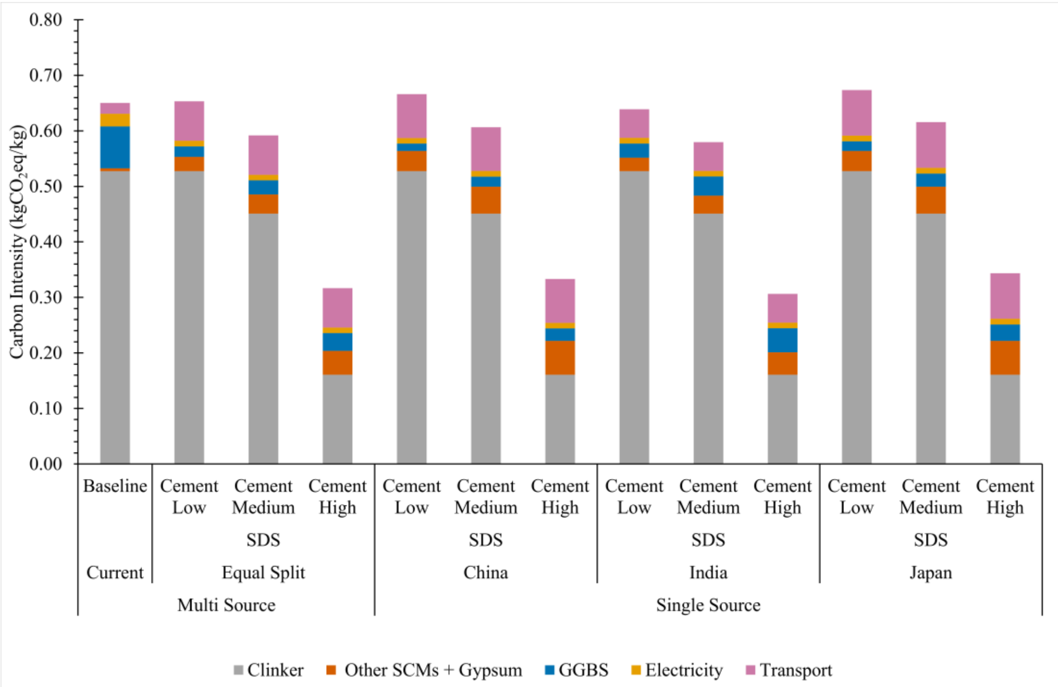


Figure 7: UK cement sector carbon intensity at the baseline and in 2050 under the best-case steel (SDS) and cement decarbonisation scenario (mass allocation).

3.4. Sensitivity Analysis

The core assumption of this study is that the UK will source GGBS in the future in an equal split between the dominant steel producers (China, India, and Japan), primarily as a method of supply chain risk management. However, it is likely that there will be periods of time where this is not the case, therefore it is important to understand the effect of sourcing from a single country; under the assumption that the material quality threshold is met. The average transportation distance between the UK and each country by sea is shown in Table 2, where the average is defined as the mean distance between the three busiest ports by material volume of each country (SI, Section S6).

Table 2: Sea transportation distance (km) between the UK and relevant countries.

Country	Average distance to the UK by sea (km)
China	22,354
India	14,639
Japan	23,309

As shown in Figure 7, the transportation distance and source can have a significant impact on the emissions associated with GGBS, and ultimately the resultant cement produced within the UK. Despite the very similar transportation distance between China and Japan (a difference of 3.9%), importing wholly from China could result in a maximum reduction in GGBS intensity of 1% compared to multi sourcing - primarily due to ambitious regional electricity decarbonisation targets despite the transport distance. Importing solely from India would result in a maximum GGBS intensity reduction of 16%, despite a greater than average steelmaking intensity. Whilst Japan's greater transportation distance (a difference of 44.6% compared to India) results in a minimum increase in GGBS intensity of 8% when compared to multi sourcing – highlighting the 'hidden' contribution of transport. This is significant as many LCA studies choose to exclude the effects of transport [72]. Although single sourcing could substantially reduce the emissions associated with GGBS, this could leave the UK cement sector vulnerable to significant supply chain, geopolitical, and transport related risks which could negate the environmental benefits.

4. Discussion

4.1. Theoretical Implications

This study has investigated the current and future carbon intensity and emissions, of key global steel producers (China, India, Japan, and the UK) and the global average, under three steelmaking route transition scenarios as well as quantifying the effects of regional and global decarbonisation efforts. These efforts are intrinsically linked to those of the global cement sector, particularly in the UK, due to the reliance on GGBS as a SCM. Therefore, this study assumed the industry will continue to use GGBS sourced from the dominant global steel producers due to a complete reduction in domestic production and consequently investigated the current and future carbon intensity and emissions associated with UK produced cement.

Our findings show that carbon emissions of steel produced in the UK will drop by up to 84% by 2050, against a global reduction of 49%; and could leave the UK responsible for just 0.09% of global steelmaking emissions. This is due to the predicted 'green' nature of the UK's electricity grid, and the use of this as primary fuel within entirely secondary steelmaking. However, the predicted volumes of steel produced are only enough to satisfy domestic demand. These reductions in emissions rely on regulatory change to ensure scrap steel supply chains can satisfy domestic demand. Otherwise scrap imports will continue, and such supply chain dependency will reduce resilience and security of UK steelmaking. China, India, and Japan will see a reduction in emissions and associated intensities but due to differences in production volumes, regional decarbonisation pathways, and the scale of EAF transition these are reduced compared to the UK decarbonisation rate. Implementation of these strategies within the global steel industry means that the UK cement sector will, consequently, also decarbonise. Emissions associated with cement production in the UK are predicted to fall by up to 56%, driven by regional and sectoral decarbonisation strategies, but also the reduction in the carbon intensity of GGBS as a result of decarbonisation efforts within the steel sector. The reduction potential is much smaller because of the emissions incurred through GGBS transportation and source. Consequently, the use of potential decarbonisation technologies in the cement sector (including CCUS and electrification) must be accelerated at a similar pace to the steel sector.

4.2. Industrial and Policy Implications

The positive efforts to reduce the environmental impact of UK steelmaking does have the potential to destabilise, disrupt, and decouple the symbiotic link between two of the UK's most important foundation industries - the products of which are vital to support continued economic growth. Although it is difficult to predict global changes with absolute certainty, it is likely that global steelmakers will accelerate steelmaking transitions in effort to decarbonise their own economies. This will ultimately reduce the global supply of GGBS further, leading to greater than modelled value increases, which could leave UK cement producers severely exposed to global supply chain failure. If the cement industry does not seek to accelerate its own pathway at the same rate as steelmaking, supported by overarching policy [85] with targets to capitalise on novel technological solutions (i.e. EAF derived low carbon cement), it will struggle to effectively decarbonise. This, compounded by the fact that there are few viable short to medium term SCM alternatives readily available in the UK, could ultimately result in a return to domestically produced PC; the adverse environmental impact of which would be higher than that studied in the baseline case. This would significantly hamper the UK's efforts to meet wider climate change mitigation targets in pursuit of net zero by 2050.

4.3. Study Limitations

This study has taken a robust approach to its analysis of cement and steel production; there are three key limitations to the study which could be tackled within a larger assessment, or as data availability changes. Firstly, the scenarios which have been modelled are relatively simplistic. This implies that the transitions toward secondary steelmaking or a lower clinker factor are discrete (i.e. only low, medium, or high) and are not currently assessed comparatively (i.e. low cement, high steel). This means that the study results are at the extreme ends of likelihood, and policy makers may benefit more from an increased number of scenarios. Secondly, although all data has been taken from a range of high-quality secondary sources which are regionalised where possible, the precision of the analysis could be enhanced by introducing additional, primary, data sources as these become available. Finally, introducing additional scenarios related to UK steelmaking which examine the effect of a reducing percentage of steel production transitioning toward secondary steelmaking would reflect the expected future trajectory of changes in domestic priorities. Therefore, the effect of this on steel-cement symbiosis could be holistically assessed.

5. Conclusion

Cement and steel are materials which are fundamental to modern infrastructure, but both have significant environmental impacts as a result of their energy- and process- intensive processes. However, both are intrinsically linked through the symbiotic use of GGBS which is primarily used as a performance enhancing, and carbon intensity reducing material within blended cements. This research has effectively characterised current and future carbon emissions, carbon intensities, and general landscape of global and UK steel production. Consequently, it has also characterised the current and future carbon emissions, carbon intensity, and general landscape of UK cement production using GGBS as a SCM - a key by-product of the BOF steelmaking route. In the work presented, it has been shown that UK steel production is projected to reduce carbon emissions by up to 84% by 2050, significantly exceeding the anticipated global reduction of 49%.

This outcome is primarily driven by the decarbonisation of the electricity grid and the adoption of secondary steelmaking processes. This reduction in carbon emissions in the steel sector will also contribute to a 56% decrease in emissions from the domestic cement industry, due to the lower carbon intensity of GGBS. However, these benefits are contingent upon the development of resilient domestic scrap steel supply chains and coordinated sectoral strategies. To achieve long-term industrial decarbonisation, parallel advancements in cement sector technologies and supportive policy interventions are imperative.

This work provides a robust methodology to analyse and effectively describe the effect of decarbonisation on the anticipated emissions of both the global and UK steel sectors, and the UK cement sector. Thus, also the effect on the symbiotic relationship between the cement and steel sectors in the UK. The enhanced understanding of the trajectory of both sectors will allow for more effective domestic planning in relation to meeting wider climate change mitigation targets. However, opportunity exists to extend this research to assess how a change in domestic steel supply chains can be supported, through technical and regulatory processes. This would ensure that high-quality steel scrap can be reused to produce further high-quality steel products to support both UK and global infrastructure. Additionally, the supply chain risk to the UK cement sector, with respect to GGBS, is clear. Therefore, a detailed supply chain analysis should be carried out to gain a better understanding of which manufacturers are producing GGBS within the regions of dominant BOF based steel production, what their decarbonisation pathways and targets are, and ultimately whether this poses a true risk to the UK supply of GGBS. Additionally, extending the methodology adopted in this study to assess other SCMs (e.g. CC) would enhance the understanding of the true emissions associated with cement production. This new understanding could also be linked to the development of more effective sustainability indexes for these materials (much like those developed for the chemical industry) [86]. There is also an opportunity to advance the attributional LCA methodology into either a consequential or dynamic model to account for fluctuations in material volumes and coefficient values (e.g. emission factors) over time, respectively. Such further research will solidify the understanding of the symbiotic relationship between two major foundation industries and ensure that regional and global decarbonisation efforts towards net-zero do not bring additional risk or result in deindustrialisation of vital segments of global economies.

Acknowledgements

The Authors would like to thank the Engineering and Physical Sciences Research Council (EPSRC) through the Advanced Metallic Systems Centre for Doctoral Training (EP/S022635/1), EPSRC and the Energy Institute at The University of Sheffield, for a DTP PhD Scholarship (EP/T517835/1), the Grantham Centre for Sustainable Futures (GCSF), and British Steel Limited.

Attribution

J.W. Whittle Conceptualization, Methodology, Software, Validation, Formal Analysis, Investigation, Data Curation, Visualization, Writing - Original Draft Preparation **M.C.S. Rihner** Conceptualization, Methodology, Software, Validation, Formal Analysis, Investigation, Data Curation, Visualization, Writing - Original Draft Preparation **H. Hafez** Conceptualization, Supervision, Writing - Review &

Editing **R.M. Eufrasio Espinosa** Formal Analysis, Writing - Review & Editing **D.I. Fletcher** Supervision, Writing - Review & Editing, Funding Acquisition **B. Walkley** Supervision, Writing - Review & Editing, Funding Acquisition **L.S.C. Koh** Supervision, Writing - Review & Editing, Funding Acquisition, Project Administration

Supplementary Information

Data supporting this publication is contained within both the Supplementary Information and the and University of Sheffield research data repository at https://doi.org/10.15131/shef.data.29224874.v1, and can be freely used under the terms of the Creative Commons Attribution (CC BY) license.

Declaration of Conflicting Interest

The Authors declare that there is no conflict of interest.

References

[1] Cembureau, "Activity Report," 2023. [Online]. Available: https://cembureau.eu/media/dnbf4xzc/activity-report-2023-for-web.pdf

[2] International Energy Agency. "Steel." https://www.iea.org/energy-system/industry/steel (accessed Nov. 8, 2024).

[3] T. Watari, A. Cabrera Serrenho, L. Gast, J. Cullen, and J. Allwood, "Feasible supply of steel and cement within a carbon budget is likely to fall short of expected global demand," Nat Commun, vol. 14, no. 1, p. 7895, Nov 30 2023, doi: 10.1038/s41467-023-43684-3.

[4] A. C. Serrenho, Z. S. Mourão, J. Norman, J. M. Cullen, and J. M. Allwood, "The influence of UK emissions reduction targets on the emissions of the global steel industry," Resour. Conserv. Recycl., vol. 107, pp. 174-184, 2016, doi: 10.1016/j.resconrec.2016.01.001.

[5] International Energy Agency. "Industry." https://www.iea.org/energy-system/industry (accessed Nov. 8, 2024).

[6] J. Kim et al., "Decarbonizing the iron and steel industry: A systematic review of sociotechnical systems, technological innovations, and policy options," Energy Research & Social Science, vol. 89, 2022, doi: 10.1016/j.erss.2022.102565.

[7] Supriya, R. Chaudhury, U. Sharma, P. C. Thapliyal, and L. P. Singh, "Low-CO2 emission strategies to achieve net zero target in cement sector," Journal of Cleaner Production, vol. 417, 2023, doi: 10.1016/j.jclepro.2023.137466.

[8] H. Cervo et al., "A Case Study of Industrial Symbiosis in the Humber Region Using the EPOS Methodology," Sustainability, vol. 11, no. 24, 2019, doi: 10.3390/su11246940.

[9] F. Boons, M. Chertow, J. Park, W. Spekkink, and H. Shi, "Industrial Symbiosis Dynamics and the Problem of Equivalence: Proposal for a Comparative Framework," Journal of Industrial Ecology, vol. 21, no. 4, pp. 938-952, 2016, doi: 10.1111/jiec.12468.

[10] G. V. Shi, J. Baldwin, S. C. L. Koh, and T. Y. Choi, "Fragmented institutional fields and their impact on manufacturing environmental practices," International Journal of Production Research, vol. 56, no. 1-2, pp. 431-446, 2017, doi: 10.1080/00207543.2017.1353712.

[11] M. A. Sellitto, F. K. Murakami, M. A. Butturi, S. Marinelli, N. Kadel Jr, and B. Rimini, "Barriers, drivers, and relationships in industrial symbiosis of a network of Brazilian manufacturing companies," Sustain. Prod. Consum., vol. 26, pp. 443-454, 2021, doi: 10.1016/j.spc.2020.09.016.

[12] M. Chertow, "Industrial symbiosis: Literature and taxonomy," Annual Review of Environment and Resources, vol. 25, pp. 313-337, 2000.

[13] D. Lyons, M. Rice, and R. Wachal, "Circuits of scrap: closed loop industrial ecosystems and the geography of US international recyclable material flows 1995–2005," The Geographical Journal, vol. 175, no. 4, pp. 286-300, 2009, doi: 10.1111/j.1475-4959.2009.00341.x.

[14] M. Hishammuddin, G. H. T. Ling, L. W. Chau, C. S. Ho, W. S. Ho, and A. M. Idris, "Circular Economy (CE): A Framework towards Sustainable Low Carbon Development in Pengerang, Johor, Malaysia," Chemical Engineering Transactions, vol. 63, 2018, doi: 10.3303/CET1863081.

[15] S. Hashimoto, T. Fujita, Y. Geng, and E. Nagasawa, "Realizing CO2 emission reduction through industrial symbiosis: A cement production case study for Kawasaki," Resour. Conserv. Recycl., vol. 54, no. 10, pp. 704-710, 2010, doi: 10.1016/j.resconrec.2009.11.013.

[16] N. B. Jacobsen, "Industrial Symbiosis in Kalundborg, Denmark: A Quantitative Assessment of Economic and Environmental Aspects," Journal of Industrial Ecology, vol. 10, no. 1-2, pp. 239-255, 2008, doi: 10.1162/108819806775545411.

[17] J. Patricio, L. Axelsson, S. Blomé, and L. Rosado, "Enabling industrial symbiosis collaborations between SMEs from a regional perspective," Journal of Cleaner Production, vol. 202, pp. 1120-1130, 2018, doi: 10.1016/j.jclepro.2018.07.230.

[18] D. van Beers, A. Bossilkov, G. Corder, and R. van Berkel, "Industrial Symbiosis in the Australian Minerals Industry: The Cases of Kwinana and Gladstone," Journal of Industrial Ecology, vol. 11, no. 1, pp. 55-72, 2008, doi: 10.1162/jiec.2007.1161.

[19] J. F. T. de Souza and S. A. Pacca, "A low carbon future for Brazilian steel and cement: A joint assessment under the circular economy perspective," Resources, Conservation & Recycling Advances, vol. 17, 2023, doi: 10.1016/j.rcradv.2023.200141.

[20] G. Biava et al., "Accelerated Direct Carbonation of Steel Slag and Cement Kiln Dust: An Industrial Symbiosis Strategy Applied in the Bergamo-Brescia Area," Materials (Basel), vol. 16, no. 11, May 29 2023, doi: 10.3390/ma16114055.

[21] T. A. Branca et al., "Reuse and Recycling of By-Products in the Steel Sector: Recent Achievements Paving the Way to Circular Economy and Industrial Symbiosis in Europe," Metals, vol. 10, no. 3, 2020, doi: 10.3390/met10030345.

[22] W. C. Jau and D. S. Tsay, "A STUDY OF THE BASIC ENGINEERING PROPERTIES OF SLAG CEMENT CONCRETE AND ITS RESISTANCE TO SEAWATER CORROSION," Cement and Concrete Research, vol. 28, 1998, doi: 10.1016/S0008-8846(98)00117-3.

[23] H. Wan, Z. Shui, and Z. Lin, "Analysis of geometric characteristics of GGBS particles and their influences on cement properties," Cement and Concrete Research, vol. 34, no. 1, pp. 133-137, 2004, doi: 10.1016/s0008-8846(03)00252-7.

[24] Cembureau, "Clinker substitution in the cement industry," 2024.

[25] International Energy Agency. "Cement." https://www.iea.org/energy-system/industry/cement (accessed Nov. 8, 2024).

[26] C. F. Dunant, S. Joseph, R. Prajapati, and J. M. Allwood, "Electric recycling of Portland cement at scale," Nature, vol. 629, no. 8014, pp. 1055-1061, May 2024, doi: 10.1038/s41586-024-07338-8.

[27] A. T. M. Marsh, A. P. M. Velenturf, and S. A. Bernal, "Circular Economy strategies for concrete: implementation and integration," Journal of Cleaner Production, vol. 362, 2022, doi: 10.1016/j.jclepro.2022.132486.

[28] B. Lothenbach, K. Scrivener, and R. D. Hooton, "Supplementary cementitious materials," Cement and Concrete Research, vol. 41, no. 12, pp. 1244-1256, 2011, doi: 10.1016/j.cemconres.2010.12.001.

[29] M. C. G. Juenger, R. Snellings, and S. A. Bernal, "Supplementary cementitious materials: New sources, characterization, and performance insights," Cement and Concrete Research, vol. 122, pp. 257-273, 2019, doi: 10.1016/j.cemconres.2019.05.008.

[30] K. Scrivener, F. Martirena, S. Bishnoi, and S. Maity, "Calcined clay limestone cements (LC3)," Cement and Concrete Research, vol. 114, pp. 49-56, 2018, doi: 10.1016/j.cemconres.2017.08.017.

[31] "Calcined Clays for Sustainable Concrete," in Proceedings of the 1st International Conference on Calcined Clays for Sustainable Concrete, Lausanne, K. Scrivener and A. Favier, Eds., 2015, vol. 10: RILEM, doi: 10.1007/978-94-017-9939-3.

[32] M. Amran et al., "Fly Ash-Based Eco-Efficient Concretes: A Comprehensive Review of the Short-Term Properties," Materials (Basel), vol. 14, no. 15, Jul 30 2021, doi: 10.3390/ma14154264.

[33] UKQAA, "UKQAA Ash Availability Report," 2016.

[34] (2017). Fly ash and blast furnace slag for cement manufacturing: BEIS research paper no.19.

[35] (2020). The Sixth Carbon Budget: The UK's path to NetZero: Presented to the Secretary of State pursuant to section 34 of the Climate Change Act 2008.

[36] S. Evans and R. Pearce. "Carbon brief: How the UK transformed its electricity supply in just a decade." https://interactive.carbonbrief.org/how-uk-transformed-electricity-supply-decade/ (accessed Oct. 31, 2024).

[37] J. Millward-Hopkins, O. Zwirner, P. Purnell, C. Velis, E. Iacovidou, and A. Brown, "Resource recovery and low carbon transitions: The hidden impacts of substituting cement with imported 'waste' materials from coal and steel production," GLOBAL ENVIRONMENTAL CHANGE-HUMAN AND POLICY DIMENSIONS, vol. 53, pp. 146-156, 2018, doi: 10.1016/j.gloenvcha.2018.09.003.

[38] H. Hafez, R. Kurda, W. M. Cheung, and B. Nagaratnam, "Comparative life cycle assessment between imported and recovered fly ash for blended cement concrete in the UK," (in English), Journal of Cleaner Production, vol. 244, 2020, doi: 10.1016/j.jclepro.2019.118722.

[39] M. Canut, S. Miller, and M. Jolnaes, "Calcined Clays for Sustainable Concrete," in Proceedings of the 3rd International Conference on Calcined Clays for Sustainable Concrete, 2020, vol. 25: RILEM.

[40] A. Radović, H. Hafez, N. Tošić, S. Marinković, and A. de la Fuente, "ECO2 framework assessment of limestone powder concrete slabs and columns," Journal of Building Engineering, vol. 57, 2022, doi: 10.1016/j.jobe.2022.104928.

[41] Mineral Products Association, "Annual cement regional sales, 2003 - 2023," 2024. [Online]. Available: https://cement.mineralproducts.org/MPACement/media/Cement/Industry-Statistics/2024/2024-07-06-Annual-Region-Channel-of-Sale.pdf

[42] United Nations. UN Comtrade. [Online]. Available: https://comtradeplus.un.org/

[43] J. Ahmad et al., "A Comprehensive Review on the Ground Granulated Blast Furnace Slag (GGBS) in Concrete Production," Sustainability, vol. 14, no. 14, 2022, doi: 10.3390/su14148783.

[44] A. Pavlovic, G. Rust, and H. Edwards, "Decarbonising Precast Concrete Manufacturing: Implementation of Low-Carbon Concretes," presented at the fib Symposium 2023, 2023.

[45] (2025). CBP 7317: UK Steel Industry: Statistics and policy. [Online] Available: https://researchbriefings.files.parliament.uk/documents/CBP-7317/CBP-7317.pdf

[46] British Steel Limited. "British Steel's £1.25-billion decarbonisation plan given major boost as permission granted for Electric Arc Furnace in Scunthorpe." (accessed Dec. 1, 2024).

[47] TATA Steel. "Green steel future." https://www.tatasteeluk.com/green-steel-future (accessed Dec. 1, 2024).

[48] H. Thomas and P. Pigott. "Traditional steelmaking in Port Talbot ends." BBC. https://www.bbc.co.uk/news/articles/c70zxjldqnxo (accessed 04 June, 2025).

[49] S. Spooner, C. Davis, and Z. Li, "Modelling the cumulative effect of scrap usage within a circular UK steel industry – residual element aggregation," Ironmaking & Steelmaking, vol. 47, no. 10, pp. 1100-1113, 2020, doi: 10.1080/03019233.2020.1805276.

[50] S. Dworak, H. Rechberger, and J. Fellner, "How will tramp elements affect future steel recycling in Europe? – A dynamic material flow model for steel in the EU-28 for the period 1910 to 2050," Resour. Conserv. Recycl., vol. 179, 2022, doi: 10.1016/j.resconrec.2021.106072.

[51] UK Steel, "Steel scrap: A strategic raw material for net zero steel," 2023.

[52] S. C. L. Koh, A. Gunasekaran, J. Morris, R. Obayi, and S. M. Ebrahimi, "Conceptualizing a circular framework of supply chain resource sustainability," International Journal of Operations & Production Management, vol. 37, no. 10, pp. 1520-1540, 2017, doi: 10.1108/ijopm-02-2016-0078.

[53] BBC News. "Traditional steelmaking in Port Talbot ends." https://www.bbc.co.uk/news/articles/c70zxjldqnxo#:~:text=The%20controversial%20move%20at%20the,the%20closure%20%E2%80%9Cindustrial%20vandalism%E2%80%9D. (accessed Dec. 1, 2024).

[54] P. Wang et al., "Regional disparities in steel production and restrictions to progress on global decarbonisation: A cross-national analysis," Renewable and Sustainable Energy Reviews, vol. 161, 2022, doi: 10.1016/j.rser.2022.112367.

[55] M. Gong, A. Simpson, L. Koh, and K. H. Tan, "Inside out: The interrelationships of sustainable performance metrics and its effect on business decision making: Theory and practice," Resour. Conserv. Recycl., vol. 128, pp. 155-166, 2018, doi: 10.1016/j.resconrec.2016.11.001.

[56] A. Genovese, A. A. Acquaye, A. Figueroa, and S. C. L. Koh, "Sustainable supply chain management and the transition towards a circular economy: Evidence and some applications," Omega, vol. 66, pp. 344-357, 2017, doi: 10.1016/j.omega.2015.05.015.

[57] R. Meglin, S. Kytzia, and G. Habert, "Regional environmental-economic assessment of building materials to promote circular economy: comparison of three Swiss cantons," Resour. Conserv. Recycl., vol. 181, 2022, doi: 10.1016/j.resconrec.2022.106247.

[58] G. F. Randolph and M. Storper, "Is urbanisation in the Global South fundamentally different? Comparative global urban analysis for the 21st century," Urban Studies, vol. 60, no. 1, pp. 3-25, 2022, doi: 10.1177/00420980211067926.

[59] World Cement Association, "WCA Long Term Forecast for Cement and Clinker Demand," 2024.

[60] M. C. S. Rihner, H. Hafez, B. Walkley, P. Purnell, and M. Drewniok, "Thousand cuts: a realistic route to decarbonise the UK cement and concrete sector by 2050," Sustain. Prod. Consum., vol. 58, 2025, doi: 10.1016/j.spc.2025.06.010.

[61] WorldSteel. "Total production of crude steel: World total 2023." https://worldsteel.org/steel-topics/statistics/annual-production-steel-data (accessed.

[62] Mineral Products Association, "Profile of the UK Mineral Products Industry," 2023. [Online]. Available: https://mineralproducts.org/MPA/media/root/Publications/2023/Profile_of_the_UK_Mineral_Products_Industry_2023.pdf

[63] Global Cement and Concrete Association. "Getting to net zero." https://gccassociation.org/concretefuture/getting-to-net-zero/ (accessed Jan. 10, 2025).

[64] International Energy Agency, "Iron and Steel Technology Roadmap," 2024. [Online]. Available: https://iea.blob.core.windows.net/assets/eb0c8ec1-3665-4959-97d0-187ceca189a8/Iron_and_Steel_Technology_Roadmap.pdf.

[65] A. Marsh, T. Dillon, and S. Bernal, "Cement and concrete decarbonisation roadmaps – a meta-analysis within the context of the United Kingdom," RILEM Technical Letters, vol. 8, pp. 94-105, 2023, doi: 10.21809/rilemtechlett.2023.163.

[66] D. Laner and H. Rechberger, "Material Flow Analysis," in Special Types of Life Cycle Assessment, M. Finkbeiner Ed., 2016, pp. 293-332.

[67] C. Lorea, F. Sanchez, and E. Torres-Morales. Green Cement Technology Tracker, Version May 2024 (05/2024). [Online]. Available: https://www.industrytransition.org/green-cement-technology-tracker/

[68] J. W. Whittle, K. Callander, M. Akure, F. Kachwala, and L. S. C. Koh, "A new high-level life cycle assessment framework for evaluating environmental performance: An aviation case study," Journal of Cleaner Production, vol. 471, no. 143440, 2024, doi: 10.1016/j.jclepro.2024.143440.

[69] T. Ibn-Mohammed, F. A. Yamoah, A. Acquaye, K. Omoteso, and S. C. L. Koh, "Enhancing life cycle product design decision-making processes: Insights from normal accident theory and the satisficing framework," Resour. Conserv. Recycl., vol. 205, 2024, doi: 10.1016/j.resconrec.2024.107523.

[70] BS EN ISO 14040, Environmental management - Life cycle assessment - Principles and framework, British Standards Institution, 2006.

[71] BS EN ISO 14044, Environmental management - Life cycle assessment - Requirements and guidelines, B. S. Institution, 2006.

[72] M. C. S. Rihner et al., "Life cycle assessment in energy-intensive industries: Cement, steel, glass, plastic," Renewable and Sustainable Energy Reviews, vol. 211, 2025, doi: 10.1016/j.rser.2024.115245.

[73] H. Dahanni, A. Ventura, L. Le Guen, M. Dauvergne, A. Orcesi, and C. Cremona, "Life cycle assessment of cement: Are existing data and models relevant to assess the cement industry's climate change mitigation strategies? A literature review," Construction and Building Materials, vol. 411, 2024, doi: 10.1016/j.conbuildmat.2023.134415.

[74] J. Suer, M. Traverso, and N. Jäger, "Review of Life Cycle Assessments for Steel and Environmental Analysis of Future Steel Production Scenarios," Sustainability, vol. 14, no. 21, 2022, doi: 10.3390/su142114131.

[75] U. S. G. Survery., "Minerals Yearbook: Slag - Iron and Steel," 2018. [Online]. Available: https://pubs.usgs.gov/myb/vol1/2018/myb1-2018-iron-steel-slag.pdf

[76] J. Somers, "Technologies to decarbonise the EU steel industry," Publications Office of the European Union, Luxembourg, 2022.

[77] International Monetary Fund. Port monitor. [Online]. Available: https://portwatch.imf.org/pages/port-monitor

[78] Sea distances. [Online]. Available: http://ports.com/sea-route/

[79] D. L. Summerbell, "Environmental Performance Improvement in the Cement Industry," 2018.

[80] R. M. Eufrasio Espinosa and S. C. Lenny Koh, "Forecasting the ecological footprint of G20 countries in the next 30 years," Sci Rep, vol. 14, no. 1, p. 8298, Apr 9 2024, doi: 10.1038/s41598-024-57994-z.

[81] S. J. Taylor and B. Letham, "Forecasting at Scale," The American Statistician, vol. 72, no. 1, pp. 37-45, 2018, doi: 10.1080/00031305.2017.1380080.

[82] G. Melard and J. M. Pasteels, "Automatic ARIMA modeling including interventions, using time series expert software," International Journal of Forecasting, vol. 16, 2000, doi: 10.1016/S0169-2070(00)00067-4.

[83] C. Chen, G. Habert, Y. Bouzidi, A. Jullien, and A. Ventura, "LCA allocation procedure used as an incitative method for waste recycling: An application to mineral additions in concrete," Resour. Conserv. Recycl., vol. 54, no. 12, pp. 1231-1240, 2010, doi: 10.1016/j.resconrec.2010.04.001.

[84] MPA Cement, "Factsheet 18: Embodied CO2e fo UK cements," 2025.

[85] Mineral Products Association, "Decarbonising UK Concrete and Cement," 2023.

[86] L. Smith, T. Ibn-Mohammed, I. M. Reaney, and S. C. L. Koh, "A Chemical Element Sustainability Index," Resour. Conserv. Recycl., vol. 166, 2021, doi: 10.1016/j.resconrec.2020.105317.

Supplementary Information

This supplementary information file provides full details of the methods and data used for the analysis presented in the article Whittle et al. “From symbiosis to scarcity: Evaluating disruption associated with decarbonisation to circular waste materials between the UK cement and steel sectors”. The following sections describe the methods and data for:

S1: Steelmaking Process Intensity

S2: Electricity and Fuel Emission Factors

S3: Fuel Mix

S4. Carbon Intensity of Clinker

S5. Allocation Procedure for blast furnace by-products

S6. Transportation

S7. Economic Modelling

S8. Material Flow Analysis

Additional information, data, and calculations supporting this publication is contained within the University of Sheffield research data repository at https://doi.org/10.15131/shef.data.29224874.v1, and can be freely used under the terms of the Creative Commons Attribution (CC BY) license.

S1: Steelmaking Process Intensity

The baseline energy intensity (GJ/tonne of steel) for each steel production process was derived from values provided by WorldSteel [1]. Two future process intensity values were examined through the application of different decarbonisation pathways. The first is the ‘Stated Policies Scenario’ (STEPS) which takes into account countries’ energy and climate related commitments through to 2050. The second is the ‘Sustainable Development Scenario’ (SDS) which sets out the major changes required to reach the main energy-related goals of the UN Sustainable Development Agenda [2]. In order to contextualise these, an additional ‘business-as-usual’ (BaU) 2050 scenario has been included which does not see any process decarbonisation, but does take into account general electricity, fuel, and material decarbonisation. The process energy intensity under each scenario is shown in Table 1.

Table 1: Energy intensity (GJ / tonne of steel production) of BOF/EAF steelmaking

Process Route	Baseline	BaU	STEPS	SDS
BOF	23.00	23.00	18.98	16.61
EAF	10.20	10.20	8.42	7.37
Average	19.39	19.39	16.00	14.00

S2: Electricity and Fuel Emission Factors

As outlined by Wang, et al. [3], there are a number of regional disparities in emission decarbonisation pathways with the majority focussed on GHG emissions reduction through a range of cross-sector strategies. However, in each, significant focus is placed on the reduction in the carbon intensity of electricity between the time of pathway publication and 2030. Most countries have a target to reach net zero by 2050, however to enable the inclusion of electricity as an input variable in this study, the conservative assumption that countries will reach their respective 2030 targets by 2050 is used. Therefore, it is assumed that each region's general decarbonisation plan will result in the same reduction in electricity carbon intensity (e.g. a 30% reduction in peak (2025) emissions by 2030). To determine the carbon intensity and emission factors of other fuels used in this study, a range of regionalised sources have been used including the UK Government GHG Conversion Factors [4] and China Products Carbon Footprint Factors Database [5]. Where the dataset could not be regionalised (e.g. biomass in each country), the UK value has been used as a verifiable, conservative estimate. It has also been assumed that the emission factor intensity of each fuel (except electricity) will not reduce between the baseline scenario and 2050 because there are a number of wide ranging factors which could affect each of these transitions, and these are not the primary focus of this study. These are outlined in Table 2.

Table 2: Emission factors for relevant electricity and fuel for Steel and Cement Production (kgCO2eq/GJ)

Country	Electricity	Ref.	Coke	Ref.	Fuel Oil	Ref.	Natural Gas	Ref.	Biomass	Ref.
2023										
UK	56.93	**[4]**	104.65	[4]	79.23	[4]	56.29	[4]	3.14	[4]
China	161.67	[6]	81.43	[5]	74.03	[5]	60.00	[5]	3.14	
India	198.06	[6]	93.61	[7]	77.40	[8]	56.10	[8]	3.14	
Japan	134.72	[6]	98.30	[9]	77.40	[9]	56.10	[9]	3.14	
Global	127.78	[10]	107.00	[11]	74.10	[11]	56.10	[11]	3.14	
2050										
UK	25.62	[3]	104.65	[4]	79.23	[4]	56.29	[4]	3.14	[4]
China	113.17	[3]	81.43	[5]	74.03	[5]	60.00	[5]	3.14	
India	128.74	[3]	93.61	[7]	77.40	[8]	56.10	[8]	3.14	
Japan	72.75	[3]	98.30	[9]	77.40	[9]	56.10	[9]	3.14	
Global	44.72	[3]	107.00	[11]	74.10	[11]	56.10	[11]	3.14	

S3. Fuel Mix

As described by the IEA, the overall demand for material will increase under the policies currently described by steel producing countries. However, as noted in Table 2 the average energy intensity of steelmaking will drop in 2050. This suggests that the fuel mixture of each process will also shift in favour of electricity driven and more sustainable processes. This drop is also reflective of the overall shift to EAF production routes which is reflected in the transition scenarios. The aggregated fuel mixture for each production route, under each scenario is shown in Table 3.

Table 3: Fuel mix of each steelmaking route under the baseline, BAU, STEPS, and SDS scenario

Steelmaking Route	Fuel/Material	Baseline	Ref.	BAU	STEPS	SDS	Ref.
BOF	Coke	89.0%	WSA [12]	89.0%	86.0%	84.0%	IEA [2]
	Oil	0.6%		0.6%	0.6%	0.4%	
	Natural Gas	3.0%		3.0%	3.0%	3.0%	
	Biofuel	0.4%		0.4%	1.4%	3.6%	
	Electricity	7.0%		7.0%	9.0%	9.0%	
EAF	Coke	11.0%		11.0%	8.9%	8.0%	
	Oil	0.6%		0.6%	0.6%	0.6%	
	Natural Gas	38.0%		38.0%	31.0%	35.0%	
	Biofuel	0.4%		0.4%	1.0%	3.0%	
	Electricity	50.0%		50.0%	58.5%	53.4%	

S4. Carbon Intensity of Clinker

The carbon intensity of the clinker component CI_{Cl} consists of process emissions occurring from the breakdown of calcium carbonate into carbon dioxide and calcium oxide in the kiln (CI_{Pe}) and the carbon intensity of the kiln's fuel mix (e.g. the product of the carbon intensity of the fuel mix (FM_{Ci}) and the energy intensity of clinker production (CI_{Ei}). This calculation is demonstrated in Equation 1, and the specific values selected for both the UK and global scenarios are noted in Table 4.

$$CI_{cl}\ (kgCO_2eq/kg) = CI_{Pe}(kgCO_2eq/kg)\ +\ CI_{Ei}\ (MJ/kg)\ *\ FM_{CI}\ (kgCO_2eq/MJ)$$

Equation 1

Table 4: Clinker Carbon Intensity Inventory Data

Variable	Unit	United Kingdom		Global	
Clinker process emissions	$kgCO_2eq/kg$	0.49	Shen, et al. [13]	0.49	Shen, et al. [13]
Energy intensity of clinker production	GJ/t clinker	3.7	IEA [14]	3.55	IEA [15]
Carbon intensity of fuel mix	$kgCO_2eq/MJ$	0.07	Summerbell [16]	0.09	Summerbell [16]

S5. Allocation Procedure for blast furnace by-products

When a process produces more than one product, the environmental impact must be divided between each product. In the case of the blast furnace process there are two outputs of value: crude iron which is the main product and BFS which is the by-product. The two allocation procedures considered for this study are by mass and economic values. The mass allocation coefficient value (C_m) was calculated using Equation 2 where m_{BFS} and $m_{Crude\ Iron}$ are the mass values in tonne of BFS and of crude iron, respectively.

$$C_m = \frac{m_{BFS}}{m_{Crude\ Iron} + m_{BFS}}$$

Equation 2

The blast furnace process is responsible for approximately 87% of all energy consumption within the BOF production route [17]. This value is unlikely to change within the overall process, however the overall energy consumption of this route will drop. Therefore, allocation by mass was determined using the known mass of iron and BFS produced within each country as noted in the MFA. To calculate the economic allocation coefficient value (C_e), Equation 3 was used where the 2023 economic values in pound sterling (£) of BFS and crude iron are noted as ev_{BFS} and $ev_{CrudeIron}$, respectively. For the purpose of this study, the economic value for GGBS was assumed to be equal to that of BFS.

$$C_e = \frac{m_{BFS} * ev_{BFS}}{(m_{Crude\ Iron} * ev_{Crude\ Iron}) + (m_{BFS} * ev_{BFS})}$$

Equation 3

The export values of iron and slag to the UK were analysed for China, India, and Japan. In addition, a global average export value was also determined. The UK export price was also evaluated. These values were obtained from the UN Comtrade trade database [18]. In order to utilise BFS as a SCM in blended cement mixes, it must undergo grinding and granulation to become GGBS. Input data for this process was obtained from Ecoinvent 3.8. The fuel and electricity emission factors were regionalised using each country's respective fuel mix and were assumed to be the same as the values utilised for the steel sector's calculations as noted in Table 2. While the HS tariff code for granulated blast furnace slag (HS Code 2618) does not distinguish if a slag has been ground or not, it was assumed all slag imported to the UK is GGBS. To determine the 2050 economic values for both products, the price prediction methodology outlined in section 2.5 was used. Table 5 notes the 2023 and 2050 economic values for both iron and GGBS for each respective country and the emission allocation percentage for each product.

Table 5: Economic Allocation of Iron and BFS (GGBS)

Material	Country	£/kg		% Change	% Allocation	
		2023	2050		2023	2050
Iron	UK	0.86	2.44	283.54%	94.48	97.49
	China	0.52	1.46		99.56	99.80
	India	0.38	1.08		96.16	98.29
	Japan	0.43	1.21		99.51	99.78
	Global	0.41	1.16		98.63	99.39
GGBS	UK	0.18	0.22	24.63%	5.52	2.51
	China	0.01	0.01		0.44	0.20
	India	0.01	0.02		3.81	1.71
	Japan	0.01	0.01		0.49	0.22
	Global	0.02	0.03		1.37	0.61

S6. Transportation

To calculate the transportation distances, the top three ports by import value were selected for the UK and the top three ports by export value were selected for each exporting country assessed [19]. The distance from each exporting country's start port to each UK end port was then determined [20]. The county average travel distance was taken as the average distance between all three UK end ports and each exporting country's top three ports. The carbon intensity of the sea freight for a bulk carrier cargo ship was taken as 0.0035 kgCO2e/t.km [4]. Table 6 summarises the average carbon intensity of material transportation by country. To determine an average global transportation carbon intensity for GGBS imports, a weighted average was taken based on the amount of GGBS imported from seven GGBS exporting countries (China, India, Japan, Spain, France, Ireland, and the Netherlands) in 2023. This value was found to be 0.0197 kgCO2eq/kg.

Table 6: Average Carbon Intensity of Material Transportation by Country

Country	Average distance UK to Country by sea (km)	kgCO2e / t	kgCO2e / kg
China	22354	78.91	0.0789
India	14639	51.67	0.0517
Japan	23309	82.28	0.0823
Spain	3365	11.88	0.0119
France	607	2.14	0.0021
Algeria	3671	12.96	0.0130
Ireland	822	2.90	0.0029
Netherlands	704	2.48	0.0025

S7. Economic Modelling

S7.1 Time Series Model

For this time series analysis, we implemented a straightforward yet robust methodology [21, 22] tailored to address the inherent volatility and inconsistencies in the data. We began by pre-processing the dataset to enhance its suitability for forecasting. Given the significant price fluctuations in both pig iron and slag prices, we identified and removed outliers that could distort the predictive models. Specifically, we observed that outliers accounted for 3.47% of slag export prices, 12.50% in slag import prices, 12.85% in pig iron import prices, and a notably higher 18.40% in pig iron export prices. Removing these outliers helped to stabilize the dataset and improve model accuracy.

For the forecasting stage, we evaluated both ARIMA [23] and Prophet [22] models in Python [24], selecting the ARIMA model based on its superior performance in capturing the trends and cyclical patterns of our data. ARIMA is particularly suited for time series with strong autocorrelation, as it models data points based on their previous values. This made it a good fit for our dataset, which displayed clear patterns and seasonality in pig iron and slag prices. Prophet, while effective in some scenarios, did not perform as well in this case due to the unique volatility of the data. By choosing ARIMA, we achieved better accuracy and forecast reliability, offering more actionable insights into long-term price movements up to 2050.

S7.2 Historical Data (2000-2023)

Figure 1 shows iron and slag exportation and importation prices. Slag export prices are relatively volatile, with frequent spikes, while import prices show a gradual increase with occasional peaks.

This suggests that slag prices may be influenced by shorter-term supply-demand factors. Overall, the historical data indicates that slag prices experience frequent, potentially local disruptions. On the other hand, pig iron export and import price plots show distinct trends compared to slag. Pig iron export prices exhibit significant volatility, with frequent and sharp spikes throughout the observed period, indicating that exports could be highly sensitive to global demand and supply shocks. Peaks in export prices may correlate with global disruptions in raw material availability or shifts in industrial production. Conversely, pig iron import prices show a generally lower and more stable trend, with occasional fluctuations. This stability might indicate a reliance on consistent international suppliers or less frequent market disruptions affecting imports. The relatively lower volatility in import prices compared to export prices suggests that domestic industries depend on stable pricing to ensure predictable production costs.

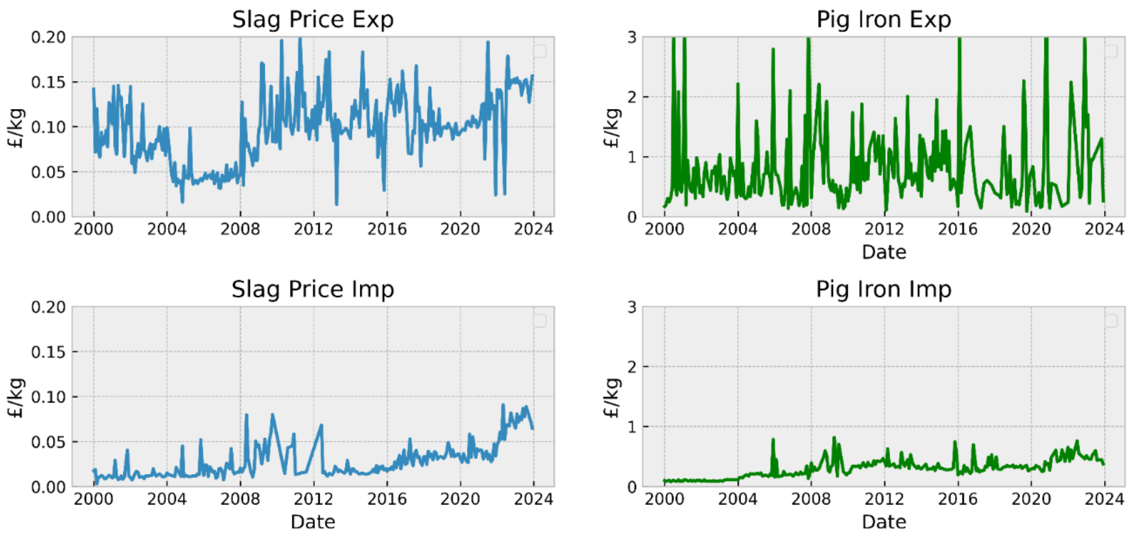


Figure 1: Historical Slag and Pig Iron Import and Export Prices, 2000-2023 (£/kg)

Figure 2 shows the monthly price distribution from 2000–2023 for slag and pig iron exports and imports. Slag exports and imports exhibit a high monthly variability, with larger interquartile ranges and more outliers, especially in the spring and summer months (April–July). This seasonal fluctuation could indicate periods of heightened demand or supply disruptions, which could be driven by industry-specific factors or market demand cycles. Conversely, pig iron exports show a higher degree of monthly variability compared to slag. The interquartile ranges are wide, and the presence of numerous outliers indicates significant price fluctuations, possibly influenced by volatile global market conditions or supply chain disruptions. Pig iron imports, on the other hand, display more stability, with narrower interquartile ranges and fewer outliers. This stability suggests consistent sourcing and supply dynamics, minimizing monthly price swings. However, minor variability is observed, particularly in summer months, which could be attributed to seasonal production cycles or shifts in demand. Overall, pig iron prices exhibit the strongest contrast between exports and imports, with exports being more volatile and imports remaining relatively stable. This highlights distinct supply and demand dynamics for pig iron compared slag across international markets.

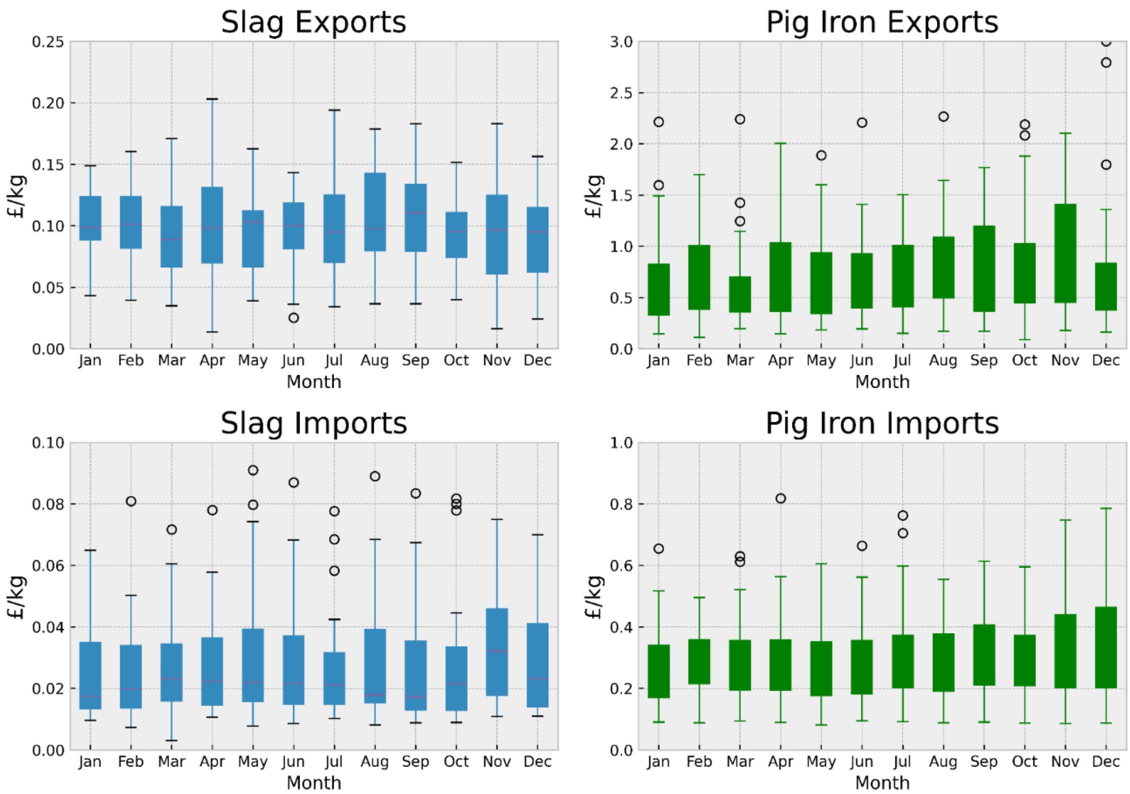


Figure 2: Historical Slag and Pig Iron Import and Export Prices, Monthly Distribution 2000-2023 (£/kg)

S7.3 Model Evaluation

Figure 3 evaluates the performance of ARIMA models for forecasting historical data and prices from 2023 to 2050.

a) **Slag Exports:** The model has a MAPE of 25.40%. However, the relatively low MSE (0.0009) and RMSE (0.0312) imply the model performs well for overall trend detection, though struggles with variability and sharp price changes.

b) **Slag Imports:** This model exhibits the highest MAPE of 30.41%, reflecting substantial variability in slag import prices. Despite the low MSE (0.0003) and RMSE (0.0229), the high relative error suggests challenges in predicting localized price spikes or rapid changes in import conditions.

c) **Pig Iron Exports**: The model demonstrates significant variability, with the highest error rates—MAPE of 77.40% and RMSE of 0.7409. This indicates difficulty in capturing the highly erratic price behaviour of pig iron exports.

d) **Pig Iron Imports**: Despite a relatively stable MAPE of 18.56% and low RMSE of 0.0111, the model shows increasing uncertainty in later years, reflecting the interplay of market forces and import dynamics.

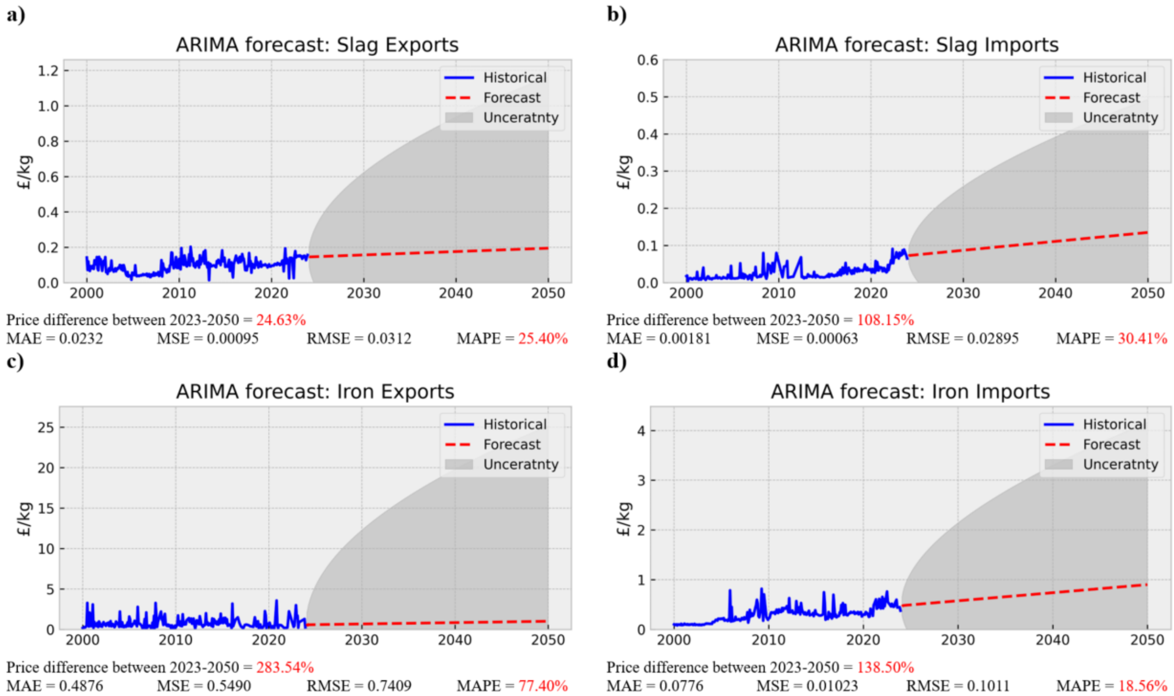


Figure 3: Import and Export Price Forecasting for Iron and Slag to 2050 (£/kg)

Our model evaluation highlights the ARIMA model's performance in forecasting price trends for slag and pig iron exports and imports. Slag price models show higher relative errors (MAPE ~25–30%), reflecting difficulties in predicting their more variable and localized fluctuations. Pig iron price models exhibit the greatest variability, particularly for exports (MAPE ~77%), indicating challenges in modelling highly erratic price behaviours. Overall, the models are effective for general trend predictions but struggle with short-term price spikes and high-volatility scenarios, emphasizing the need for caution when interpreting results.

S7.4 Conclusions

The forecasted price trends provide insights into the projected growth of slag and pig iron prices over the next few decades. Slag export prices are predicted to increase by 24.63%, reflecting relative stability. The trend is flat, with wide uncertainty bands indicating possible volatility driven by localized supply and demand dynamics. Slag imports however present a substantial increase of

108.15% is forecasted for slag import prices, the highest among all categories. This suggests potential supply constraints or rising demand, with broader uncertainty indicating challenges in accurately forecasting these price dynamics.

Pig iron exports prices are forecasted to increase dramatically by 283.54%, reflecting extreme volatility and the influence of global market conditions on exports. The steep rise and high uncertainty indicate significant unpredictability in future export dynamics. Pig iron imports prices are expected to grow by 138.50%, showing a more stable upward trend than exports but with considerable uncertainty by 2050, likely due to shifting import dependencies and market forces. All forecasts predict long-term price increases for slag and pig iron, reflecting growing demand and inflation. Both materials seem to exhibit volatility however, presenting uncertainty which grows significantly toward 2050, emphasizing the potential influence of unforeseen market disruptions and economic shifts.

S8: Material Flow Analysis

The results of each MFA conducted, not shown in the main manuscript, are shown in Figure 4 and Figure 5. These are the results of the global MFA under both the low and high transition scenarios.

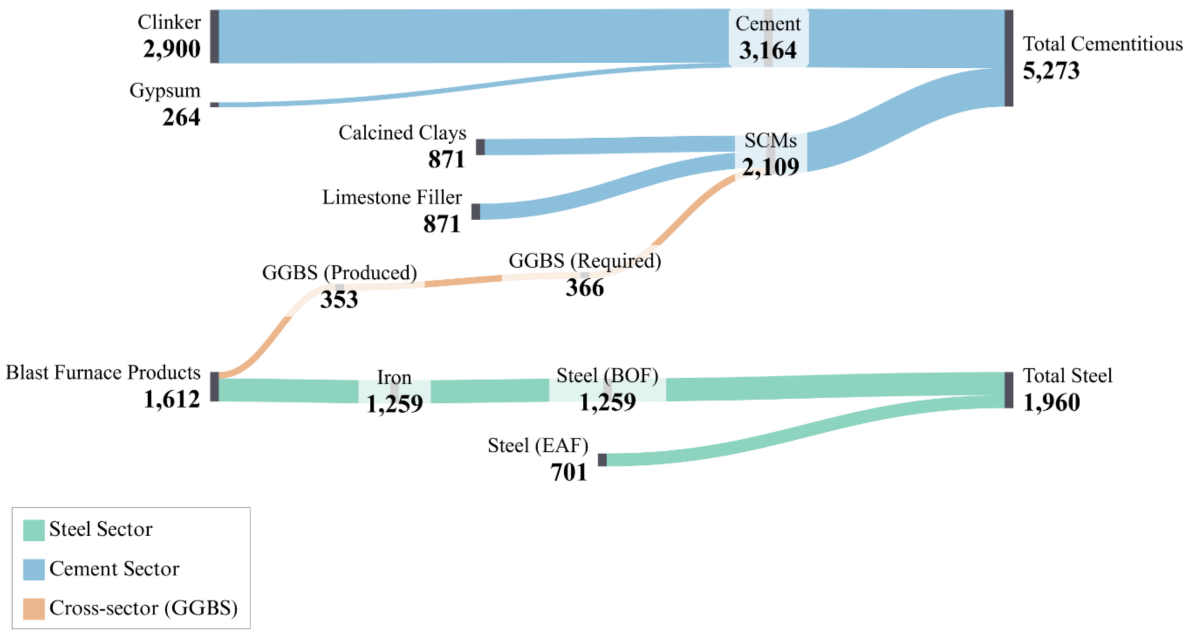


Figure 4: Global MFA 2050 under Low scenario, where all values are in Mt/year.

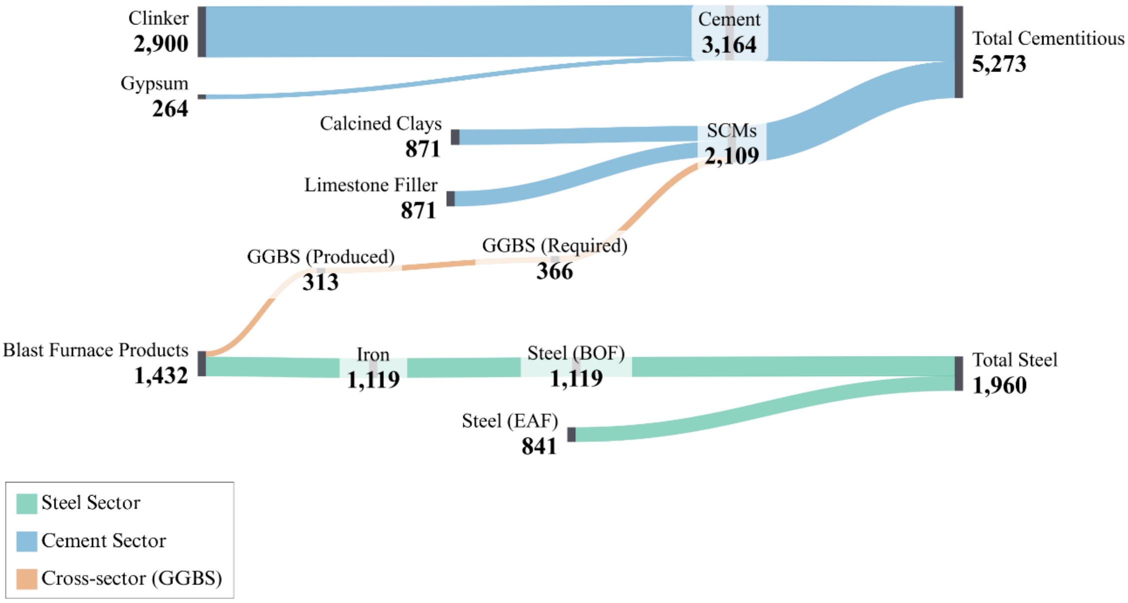


Figure 5: Global MFA 2050 under Medium scenario, where all values are in Mt/year.

References

[1] WSA. *World Steel in Figures*. [Online]. Available: https://worldsteel.org/steel-topics/statistics/world-steel-in-figures/
[2] IEA, "Iron and Steel Technology Roadmap Towards more sustainable steelmaking," 2024.
[3] P. Wang *et al.*, "Regional disparities in steel production and restrictions to progress on global decarbonisation: A cross-national analysis," *Renewable and Sustainable Energy Reviews,* vol. 161, p. 112367, 2022/06/01/ 2022, doi: https://doi.org/10.1016/j.rser.2022.112367.
[4] (2024). *Greenhouse gas reporting: conversion factors 2024*. [Online] Available: https://www.gov.uk/government/publications/greenhouse-gas-reporting-conversion-factors-2024
[5] CAEP. "Emission Factor: Coke." China Products Carbon Footprint Factors Database. https://www.climatiq.io/data/emission-factor/236eb158-41d8-456d-b0cb-febb9b844470 (accessed).
[6] Ember and Energy Institute. *Carbon intensity of electricity generation, 1990 to 2023*. [Online]. Available: https://ourworldindata.org/grapher/carbon-intensity-electricity?tab=table
[7] TERI, "INDIA'S GHG INVENTORY PROCESS: An analysis of sectoral emission profile, data sources, institutional framework and key challenges," MacArthur Foundation, The Energy and Resources Institute, , 2018.
[8] R. Mohan Rachel, N. Dharmala, M. Ramakrishnan, P. Kumar, and A. Bose, "Methodology Note National & State Level Greenhouse Gas Estimates 2005 to 2015: September 2019 Energy Sector," STEP, 2019.
[9] IPCC, "2006 IPCC Guidelines for National Greenhouse Gas Inventories," 2006.
[10] IEA. "Electricity." https://www.iea.org/energy-system/electricity (accessed 2024).
[11] D. Vidovic, L. Zore, J. A. Moya, and A. Marmier, *Greenhouse gas emission intensities of the steel, fertilisers, aluminium and cement industries in the EU and its main trading partners*. Publications Office of the European Union, 2023.
[12] WSA, "Fact sheet: Energy use in the steel industry," 2021.
[13] W. Shen, L. Cao, Q. Li, W. Zhang, G. Wang, and C. Li, "Quantifying CO2 emissions from China's cement industry," *Renewable and Sustainable Energy Reviews,* vol. 50, pp. 1004-1012, 2015/10/01/ 2015, doi: https://doi.org/10.1016/j.rser.2015.05.031.
[14] IEA. "Thermal specific energy consumption per tonne of clinker in selected countries and regions." https://www.iea.org/data-and-statistics/charts/thermal-specific-energy-consumption-per-tonne-of-clinker-in-selected-countries-and-regions-2018 (accessed).

[15] IEA, "Global thermal energy intensity of clinker production by fuel in the Net Zero Scenario, 2010-2030," 2023. [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.
[16] D. L. Summerbell, "Environmental Performance Improvement in the Cement Industry," University of Cambridge, 2018.
[17] L. D. D. Harvey, "A bottom-up assessment of recent (2016–20) energy use by the global iron and steel industry constrained to match a top-down (International Energy Agency) assessment," *Energy,* vol. 293, p. 130675, 2024/04/15/ 2024, doi: https://doi.org/10.1016/j.energy.2024.130675.
[18] UN Comtrade. [Online] Available: https://comtrade.un.org/
[19] IMF PortWatch. "Port Monitor." https://portwatch.imf.org/pages/port-monitor (accessed.
[20] Ports.com. "Sea route & distance." http://ports.com/sea-route/ (accessed October 15th, 2024.
[21] R. M. Eufrasio Espinosa and S. C. Lenny Koh, "Forecasting the ecological footprint of G20 countries in the next 30 years," *Scientific Reports,* vol. 14, no. 1, p. 8298, 2024/04/09 2024, doi: 10.1038/s41598-024-57994-z.
[22] S. J. Taylor and B. Letham, "Forecasting at Scale," *The American Statistician,* vol. 72, no. 1, pp. 37-45, 2018/01/02 2018, doi: 10.1080/00031305.2017.1380080.
[23]G. Mélard and J. M. Pasteels, "Automatic ARIMA modeling including interventions, using time series expert software," *International Journal of Forecasting,* vol. 16, no. 4, pp. 497-508, 2000/10/01/ 2000, doi: https://doi.org/10.1016/S0169-2070(00)00067-4.
[24] L. Menculini *et al.*, "Comparing Prophet and Deep Learning to ARIMA in Forecasting Wholesale Food Prices," *Forecasting*, vol. 3, no. 3, pp. 644-662doi: 10.3390/forecast3030040.

7. Conclusions and Future Work

7.1 Conclusions

As the 2050 net-zero carbon target approaches, the UK cement and concrete sector, one of the country's most energy-intensive industries, faces increasing pressure to rapidly decarbonise. This pressure has resulted in the inconsistent application of methodological approaches, overly optimistic assumptions, and a general lack of transparency across academic studies, industrial reports, and decarbonisation roadmaps. Therefore, this thesis aimed to investigate how the UK cement and concrete sector can realistically achieve net-zero by 2050 by assessing the implementation of standardised LCA practices, identifying viable decarbonisation pathways by considering economic, social, and political barriers, and analysing the wider cross-sector impacts of decarbonisation strategy implementation.

To determine how the application of LCA methodology can be improved to provide a transparent, robust, and consistent representation of direct sectoral emissions across different energy-intensive industries to support more effective decision-making, a systematic literature review was conducted. By using a unique search string and removing off-topic, non-compliant, and non-accessible articles, 256 articles (109 of which were cement LCA studies) were retrieved for analysis to assess how each study implemented the standardised LCA methodology specified by ISO 14040 and ISO 14044. The results found that declared units were favoured over functional units despite the importance of considering performance in comparative studies. 'Cradle-to-gate' was the most commonly selected system boundary type due to the nature of LCAs for specific products. As a result, downstream flows, including a material's use and end-of-life impacts, were often not assessed. The allocation procedure selected for the same by-product varied across different sectors with the cement sector favouring economic allocation and the steel sector favouring mass allocation. Most LCAs conducted relied heavily on secondary data due to difficultly obtaining primary data from industry. While this is the case, data quality checks on secondary data were often not carried out. Less than half the studies conducted a sensitivity or uncertainty analysis despite being a key component of LCA interpretation. To ensure the effectiveness of LCA as a decision making framework, it is recommended to use proper functional units when making comparative assertions, ensure cross-sector consistency when selecting an allocation procedure for by-products, and utilise a 'cradle-to-cradle' system boundary where possible. It is also recommended to implement objective and transparent data quality checks, ensure greater uniformity across impact assessment methodologies and categories, and encourage the use of quantitative interpretation methods as specified in the standardised framework. By implementing these recommendations, LCA application across energy-intensive industries can facilitate more informed decision-making in the pursuit of net-zero targets.

To evaluate net-zero pathways in the UK cement and concrete sector with a high degree of certainty, accounting for varying levels of market readiness and technological maturity, decarbonisation strategies outlined in UK and EU roadmaps were assessed under two probability scenarios. The high probability scenario assumed that the minimum reduction potential reported for each decarbonisation strategy across all assessed roadmaps are achieved, and any remaining emissions must be met through the reduction of concrete demand. The low probably scenario assumed that the best practice reduction potential for high-maturity strategies are achieved and no reduction in demand is required. Any remaining emissions would be abated through the implementation of low-maturity strategies. It was determined that in 2025, the UK cement and concrete market emitted approximately 9.5 $MtCO_2eq$. Given the predicted 20% increase in cement and concrete consumption by 2050, 11.4 $MtCO_2eq$ must be abated to achieve net-zero targets. Under the high probability scenario, it is evident that 4.9 $MtCO_2eq$ remains unabated after the implementation of all high-maturity and low-maturity decarbonisation strategies. To close this emissions gap, the demand for cement and concrete must be reduced. Current economic, social, and legislative barriers however limit the UK's ability to implement circular strategies effectively. For the UK to achieve net-zero emissions by 2050 under the low probability scenario, 59% of all emissions must be abated through the implementation of low-maturity strategies, which is economically and technically challenging. Therefore, the results indicate that it is unlikely for the UK cement and concrete sector to reach net-zero unless concrete demand is reduced by 43% through the reuse and refurbishment of existing building stock. Given the time-critical nature of climate change mitigation, it is recommended to implement low-maturity strategies (e.g., CCS, electrification of the kiln) immediately and to begin shifting towards optimised best practice cases for high-maturity strategies (e.g., decarbonising the electricity grid, improving the thermal efficiency of the kiln).

To examine how the UK steel sector's shift to electric arc furnace production may affect decarbonisation efforts in the UK cement and concrete sector, given their existing cross-sectoral dependency, the future availability of GGBS, sectoral carbon intensities, and supply chain vulnerabilities were assessed. Due to the UK steel sector's decarbonisation efforts, specifically their transition from primary to secondary steelmaking, the sector is projected to reduce its carbon emissions by up to 84% by 2050. While the global transition to secondary steelmaking is not as straightforward, a 14-49% reduction in the sector's global carbon emissions is expected depending the rate of transition. This shift, both globally and domestically, will leave the global GGBS supply chain vulnerable as the demand for GGBS will outstrip supply by as much as 28% assuming GGBS demand remains constant from 2025 to 2050. Over the same 25-year period, the UK cement sector is predicted to reduce its carbon emissions by up to 56%, however this reduction is subject to the sector's ability and willingness implement strategies quickly and effectively. The price of blast furnace slag and iron is predicted to increase by 25% and 284%, respectively, resulting in GGBS having a lower carbon intensity, as iron carries a larger percentage of the environmental burden when allocation is based on economic value. While this is the case, the reliance on imports to meet demand may impact the material's overall carbon intensity depending on the exporting country steel sector's rate of decarbonisation and distance to the UK. To ensure industrial decarbonisation is achieved by 2050, the UK and global cement sector must implement decarbonisation strategies such as carbon capture and storage and alternative SCMs at the same rate as the steel sector to reduce cross-sector decoupling.

From this, it can be concluded that in order for UK cement and concrete sector to feasibility achieve net-zero carbon emissions by 2050, several key approaches and decarbonisation strategies must be implemented. Advancing circular approaches is critical, therefore efforts should be directed towards reducing material demand across the sector's entire supply chain. When reporting baseline and predicted carbon emissions, data and methodological transparency is needed. Lastly, greater symbiosis between different industrial sectors, and industry and academia, is essential to facilitate informed and sustainable decision-making within the UK's built environment. Despite the comprehensive approach taken in this thesis, several limitations should be acknowledged. Since the implementation of decarbonisation strategies is influenced by economic, social, and legislative barriers, the scenarios and projections explored have been simplified and may not capture all potential pathways, particularly as new technologies emerge. Furthermore, the LCAs conducted in this thesis relied primarily on secondary data, which, while temporally and geographically relevant where possible, exhibits some degree of uncertainty due to variability across data sources. Future work should seek to reduce these uncertainties through greater scenario variability and the use of primary data to validate findings.

7.2 Future Work

While this thesis focused on assessing cement and concrete decarbonisation on a product level, future work should aim to assess the wider environmental impacts of these products on a building scale. By conducting a building-level LCA, specific performance based functional units can be selected which would allow the product's system boundary to be expanded to include use and end-of-life impacts. Using a dynamic LCA model would allow for even greater sectoral analysis as changes in decarbonisation strategy implementation and current material and building stocks can be assessed over time. By combining this approach with a techno-economic assessment, decarbonisation strategy implementation can be further optimised by assessing economic viability quantitatively. Lastly, assessing decarbonisation must extend to beyond the cement and concrete by implementing a hybrid life cycle assessment which combines process level assessment with economic input-output data to assess the environmental impacts and potential decarbonisation pathways for the entire building and construction sector.

8. REFERENCES

ABBAS, R., ABO-EL-ENEIN, S. A. & EZZAT, E.-S. 2010. Propiedades y durabilidad del cemento con adición de metacaolín: mortero y hormigón. *Materiales de Construcción,* 60**,** 33-49.

ALBERICI, S., DE BEER, J. & VAN DER HOORN, I. S., MAARTEN 2017. Fly Ash and Blast Furance Slag for Cement Manufacturing. *In:* DEPARTMENT FOR BUSINESS, E. I. S. (ed.). BEIS.

AMRAN, M., FEDIUK, R., MURALI, G., AVUDAIAPPAN, S., OZBAKKALOGLU, T., VATIN, N., KARELINA, M., KLYUEV, S. & GHOLAMPOUR, A. 2021. Fly Ash-Based Eco-Efficient Concretes: A Comprehensive Review of the Short-Term Properties. *Materials,* 14**,** 4264.

ANTUNES, M., SANTOS, R. L., PEREIRA, J., ROCHA, P., HORTA, R. B. & COLAÇO, R. 2022. Alternative Clinker Technologies for Reducing Carbon Emissions in Cement Industry: A Critical Review. *Materials,* 15**,** 209.

ARIAS, B., ALONSO, M. & ABANADES, C. 2017. CO2 Capture by Calcium Looping at Relevant Conditions for Cement Plants: Experimental Testing in a 30 kWth Pilot Plant. *Industrial & Engineering Chemistry Research,* 56**,** 2634-2640.

ASLANI, A., HACHEM-VERMETTE, C. & ZAHEDI, R. 2023. Environmental impact assessment and potentials of material efficiency using by-products and waste materials. *Construction and Building Materials,* 378**,** 131197.

ASPDIN, J. 1824. *Artificial Stone*. United Kingdom patent application.

ASTUTININGSIH, S., SURA, W. & ZAKIYUDDIN, A. 2018. Comparison of The Compressive Strength and The Microstructure of Metakaolin Metastar and Metakaolin Bangka as Additive in Ordinary Portland Cement. *EDP Science,* 67.

AWADH, O. 2017. Sustainability and green building rating systems: LEED, BREEAM, GSAS and Estidama critical analysis. *Journal of Building Engineering,* 11**,** 25-29.

BACKES, J. G., SUER, J., PAULIKS, N., NEUGEBAUER, S. & TRAVERSO, M. 2021. Life Cycle Assessment of an Integrated Steel Mill Using Primary Manufacturing Data: Actual Environmental Profile. *Sustainability,* 13**,** 3443.

BAENA-MORENO, F. M., RODRÍGUEZ-GALÁN, M., VEGA, F., ALONSO-FARIÑAS, B., VILCHES ARENAS, L. F. & NAVARRETE, B. 2019. Carbon capture and utilization technologies: a literature review and recent advances. *Energy Sources, Part A: Recovery, Utilization, and Environmental Effects,* 41**,** 1403-1433.

BAKHAREV, T., SANJAYAN, J. G. & CHENG, Y. B. 2002. Sulfate attack on alkali-activated slag concrete. *Cement and Concrete Research,* 32**,** 211-216.

BBC. 2017. *UK government spent £100m on cancelled carbon capture project* [Online]. Available: https://www.bbc.co.uk/news/uk-scotland-scotland-business-38687835 [Accessed].

BEIS 2020. Historical coal data: coal production, availability and consumption. *In:* DEPARTMENT FOR BUSINESS, E. I. S. (ed.).

BGS 2005. Cement Raw Materials: Mineral Profile. British Geological Survey.

BGS 2006. Mineral Planning Factsheet: Gypsum. *In:* GOVENMENT, D. F. C. A. L. (ed.). British Geological Survey: Natrual Environment Research Council.

BGS 2009. Kaolin. *Mineral Planning Factsheet.* British Geological Survey.

BGS 2014. Mineral Planning Factsheet: Cement. *In:* GOVENMENT, D. F. C. A. L. (ed.). British Geological Survey.

BIDE, T., BROWN, T. J., IDOINE, N. E. & MANKELOW, J. M. 2018. United Kingdom Minerals Yearbook 2018. British Geological Survey.

BIDE, T., EVANS, E., IDOINE, N. E. & J, M. 2024. United Kingdom Minerals Yearbook 2024. British Geological Survey: Decarbonisation and Resource Management Programme.

BIERNACKI, J. J., BULLARD, J. W., SANT, G., BROWN, K., FREDRIK, JONES, S., LEY, T., LIVINGSTON, R., NICOLEAU, L., OLEK, J., SANCHEZ, F., SHAHSAVARI, R., STUTZMAN, P. E., SOBOLEV, K. & PRATER, T. 2017. Cements in the 21 st century: Challenges, perspectives, and opportunities. *Journal of the American Ceramic Society,* 100**,** 2746-2773.

BLEZARD, R. G. 2004. The History of Calcareous Cement. *In:* HEWLETT, P. C. (ed.) *Lea's Chemistry of Cement and Concrete Fourth Edition.* Elsevier Science & Technology Books.

BRIMACHOMBE, L. & PARSONS, S. 2022. An Introduction to Life Cycle Assessment Course. IOM3 London.

BRITISH STEEL. 2024. *British Steel granted permission to build Electric Arc Furnace on Teesside as part of £1.25-billion decarbonisation plan* [Online]. Available: https://britishsteel.co.uk/news/british-steel-granted-permission-to-build-electric-arc-furnace-on-teesside-as-part-of-125-billion-decarbonisation-plan/ [Accessed].

BROWN, T. J., IDOINE, N. E., WRIGHTON, C. E., RAYCRAFT, E. R., HOBBS, S. F., SHAW, R. A., EVERETT, P., DEADY, E. A. & KRESSE, C. 2019. World Mineral Production 2015-19. *British Geological Survey.*

BS EN 197-1 2019. BSEN 197-1:2011 Cement Part 1: Composition, specifications and conformity criteria for common cements. *BRITISH STANDARD.*

BULLE, C., MARGNI, M., PATOUILLARD, L., BOULAY, A.-M., BOURGAULT, G., DE BRUILLE, V., CAO, V., HAUSCHILD, M., HENDERSON, A., HUMBERT, S., KASHEF-HAGHIGHI, S., KOUNINA, A., LAURENT, A., LEVASSEUR, A., LIARD, G., ROSENBAUM, R. K., ROY, P.-O., SHAKED, S., FANTKE, P. & JOLLIET, O. 2019. IMPACT World+: a globally regionalized life cycle impact assessment method. *The International Journal of Life Cycle Assessment,* 24**,** 1653-1674.

BYE, G. 2011. *Portland Cement Third Edition,* London, ICE Publishing.

CEICDATA.COM. 2023. Available: https://www.ceicdata.com/en/japan/port-cargo-throughput-by-commodity-port-of-osaka?page=2 [Accessed].

CHEN, C., HABERT, G., BOUZIDI, Y. & JULLIEN, A. 2010. Environmental impact of cement production: detail of the different processes and cement plant variability evaluation. *Journal of Cleaner Production,* 18**,** 478-485.

CHRISTODOULIDES, P., ARESTI, L., PANAYIOTOU, G. P., TASSOU, S. & FLORIDES, G. A. 2022. Adoption of Waste Heat Recovery Technologies: Reviewing the Relevant Barriers and Recommendations on How to Overcome Them. *Operations Research Forum,* 3**,** 3.

CLIMATE CHANGE COMMITTEE 2020. The Sixth Carbon Budget: The UK's Path to Net Zero.

COLANGELO, F., FORCINA, A., FARINA, I. & PETRILLO, A. 2018. Life Cycle Assessment (LCA) of Different Kinds of Concrete Containing Waste for Sustainable Construction. *Buildings,* 8**,** 70.

COMTRADE, U. 2025. United Nations.

CURRY, K. C. 2018. Slag—Iron and Steel. *In:* INTERIOR, U. S. D. O. T. (ed.) *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. *Journal of Cleaner Production,* 279**,** 123487.

DANACI, D. & BUI, M. 2022. Maps of the size and distribution of CO2 emission sources in the UK (2019). Zenodo.

DE BRABANDERE, L., GRIGORJEV, V., VAN DEN HEEDE, P., NACHTERGAELE, H., DEGEZELLE, K. & DE BELIE, N. 2025. Using Fines from Recycled High-Quality Concrete as a Substitute for Cement. *Sustainability,* 17**,** 1506.

DE LENA, E., SPINELLI, M., MARTÍNEZ, I., GATTI, M., SCACCABAROZZI, R., CINTI, G. & ROMANO, M. C. 2017. Process integration study of tail-end Ca-Looping process for CO2 capture in cement plants. *International Journal of Greenhouse Gas Control,* 67**,** 71-92.

DECC 2014. RHI Evidence Report: Direct Application of Renewable Heat. Department of Energy and Climate Change.

DHANDAPANI, Y., MARSH, A. T. M., RAHMON, S., KANAVARIS, F., PAPAKOSTA, A. & BERNAL, S. A. 2023. Suitability of excavated London clay as a supplementary cementitious material: mineralogy and reactivity. *Materials and Structures,* 56**,** 174.

DHANYA, B. S., SANTHANAM, M., GETTU, R. & PILLAI, R. G. 2018. Performance evaluation of concretes having different supplementary cementitious material dosages belonging to different strength ranges. *Construction and Building Materials,* 187**,** 984-995.
DING, Y., DAI, J.-G. & SHI, C.-J. 2016. Mechanical properties of alkali-activated concrete: A state-of-the-art review. *Construction and Building Materials,* 127**,** 68-79.
DUXSON, P., PROVIS, J. L., LUKEY, G. C. & VAN DEVENTER, J. S. J. 2007. The role of inorganic polymer technology in the development of 'green concrete'. *Cement and Concrete Research,* 37**,** 1590-1597.
EN 15804 2012. Sustainability of construction works — Environmental product declarations — Core rules for the product category of construction products. European Standard.
EU 2008. European Union Directive 2008/98/EC of the European parliament and of the council on waste and repealing certain directives. European Union.
EUROPEAN COMMISSION 2021. Environmental Footprint Methods: Product Environmental Footprint (PEF) Guide, version 3.1.: Joint Research Centre.
EUROPEAN COMMISSION JRC 2025. Environmental and Socio-Economic Impacts of the Circular Economy Transition in the EU Steel Sector: Analysing steel material flows with life cycle-based and macroeconomic assessment models. Joint Research Centre: EU Science Hub.
FARIA, D. G., CARVALHO, M. M. O., NETO, M. R. V., DE PAULA, E. C., CARDOSO, M. & VAKKILAINEN, E. K. 2022. Integrating oxy-fuel combustion and power-to-gas in the cement industry: A process modeling and simulation study. *International Journal of Greenhouse Gas Control,* 114**,** 103602.
FEIZ, R., AMMENBERG, J., BAAS, L., EKLUND, M., HELGSTRAND, A. & MARSHALL, R. 2015. Improving the CO2 performance of cement, part I: utilizing life-cycle assessment and key performance indicators to assess development within the cement industry. *Journal of Cleaner Production,* 98**,** 272-281.
FERNANDEZ, R., MARTIRENA, F. & SCRIVENER, K. L. 2011. The origin of the pozzolanic activity of calcined clay minerals: A comparison between kaolinite, illite and montmorillonite. *Cement and concrete research,* 41**,** 113-122.
GARCÍA-LUNA, S., ORTIZ, C., CARRO, A., CHACARTEGUI, R. & PÉREZ-MAQUEDA, L. A. 2022. Oxygen production routes assessment for oxy-fuel combustion. *Energy,* 254**,** 124303.
GELOWITZ, M. D. C. & MCARTHUR, J. J. 2016. Investigating the Effect of Environmental Product Declaration Adoption in LEED® on the Construction Industry: A Case Study. *Procedia Engineering,* 145**,** 58-65.
GERBELOVÁ, H., VAN DER SPEK, M. & SCHAKEL, W. 2017. Feasibility Assessment of CO2 Capture Retrofitted to an Existing Cement Plant: Post-combustion vs. Oxy-fuel Combustion Technology. *Energy Procedia,* 114**,** 6141-6149.
GLAVIND, M. 2009. Sustainability of cement, concrete and cement replacement materials in construction. *In:* KHATIB, J. M. (ed.) *Sustainability of Construction Materials.*
GLOBAL CEMENT AND CONCRETE ASSOCIATION. 2024. *Getting to net zero* [Online]. Available: https://gccassociation.org/concretefuture/getting-to-net-zero/ [Accessed Jan. 10 2025].
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.
HABERT, G. 2013. A method for allocation according to the economic behaviour in the EU-ETS for by-products used in cement industry. *The International Journal of Life Cycle Assessment,* 18**,** 113-126.
HAMMOND, G. P. 2022. The UK industrial decarbonisation strategy revisited. *Proceedings of the Institution of Civil Engineers - Energy,* 175**,** 30-44.
HARRISON, T. A., JONES, M. R., NEWLANDS, M. D., KANDASAMI, S. & KHANNA, G. 2012. Experience of using the prTS 12390-12 accelerated carbonation test to assess the relative performance of concrete. *Magazine of Concrete Research,* 64**,** 737-747.

HEIDELBERG MATERIALS 2025a. Heidelberg Materials UK – UK Average Regen GGBS. International EPD System.

HEIDELBERG MATERIALS. 2025b. *Major breakthrough for plasma-heated cement kiln in Sweden* [Online]. Available: https://www.heidelbergmaterials-northerneurope.com/en/major-breakthrough-for-plasma-heated-cement-kiln-in-sweden [Accessed].

HELM, D. 2023. Net zero electricity: the UK 2035 target. *Oxford Review of Economic Policy,* 39**,** 779-795.

HIGGINS, D. D. 2009. Portland cement replacement: within-mixer or factory-blend? Mineral Products Association.

HUNTZINGER, D. N. & EATMON, T. D. 2009. A life-cycle assessment of Portland cement manufacturing: comparing the traditional process with alternative technologies. *Journal of cleaner production,* 17**,** 668-675.

IDOINE, N. E., BIDE, T., BROWN, T. J. & RAYCRAFT, E. R. 2015. United Kingdom Minerals Yearbook 2015. British Geological Survey.

IDOINE, N. E., RAYCRAFT, E. R., HOBBS, S. F., EVERETT, P., EVANS, E. J., MILLS, A. J., SHAW, I. R., WATKINS, I. & SHAW, R. A. 2025. World Mineral Production 2019-23. British Geological Survey

IEA 2021. Net Zero by 2050 -A Roadmap for the Global Energy Sector. International Energy Agency: International Energy Agency.

IEA. 2022. *Cement* [Online]. Paris. Available: https://www.iea.org/reports/cement [Accessed].

IEA. 2024. *United Kingdom: Electricity* [Online]. Available: https://www.iea.org/countries/united-kingdom/electricity#where-does-united-kingdom-get-its-electricity [Accessed].

INTERNATIONAL ENERGY AGENCY. 2024. *Cement* [Online]. Available: https://www.iea.org/energy-system/industry/cement [Accessed Nov. 8 2024].

IPCC. 2022. *Intergovernmental Panel on Climate Change* [Online]. Available: https://www.ipcc.ch/ [Accessed].

ISLAM, M., ISLAM, S., RAHMAN, A. & DAS, A. 2011. Strength Behavior of Mortar Using Slag as Partial Replacement of Cement.

ISO 14025 2006. Environmental labels and declarations — Type III environmental declarations — Principles and procedures. International Standards Organisation

ISO 14040 2006. International Organization for Standardization (ISO) 14040, Environmental management - Life cycle assessment - Principles and framework. British Standards Online.

ISO 14044 2006. International Organization for Standardization (ISO) 14044, Environmental management - Life cycle assessment - Requirements and guidelines. British Standards Online.

ISO 21930 2017. Sustainability in buildings and civil engineering works — Core rules for environmental product declarations of construction products and services. International Standards Organisation.

JACKSON, P. J. 2004. Portland Cement: Classification and Manufacture. *In:* HEWLETT, P. C. (ed.) *Lea's Chemistry of Cement and Concrete Fourth Edition.* Elsevier Science & Technology Books.

JOLLIET, O., SAADE-SBEIH, M., SHAKED, S., JOLLIET, A. & CRETTAZ, P. 2015. Environmental Life Cycle Assessment.

JOZEPA, I. 2025. 7317: UK Steel Industry: Statistics and policy House of Commons Library: UK GOV.

KARRI, S. K., RAO, G. R. & RAJU, P. M. 2015. Strength and durability studies on GGBS concrete. *SSRG International Journal of Civil Engineering (SSRG-IJCE),* 2**,** 34-41.

KARS, F., OZCAN, G., GULBANDILAR, E. & KOCAK, Y. 2024. Prediction of Compressive Strength and Design Parameters of C30/37, C35/45 and C40/50 Concrete Classes by ANN. *Journal of Civil Engineering and Urbanism,* 14**,** 356-367.

KOKULU, N., ÖZGÜNLER, S. A., SEZGIN, İ. V. & ÇAKMAK, A. 2025. A BIBLIOMETRIC ANALYSIS OF THE LITERATURE ON ENVIRONMENTAL PRODUCT DECLARATION(S) FROM 2000 TO 2024. *Journal of Green Building,* 20**,** 197-222.

KOSMATKA, S. H., KERKHOOF, B. & PANARESE, W. C. 2002. *Design and Control of Concrete Mixtures,* Portland Cement Association.

LAFARGE 2021. Lafarge Portland Cement (CEM 52.5N) International EPD System.

LAVAGNA, L. & NISTICÒ, R. 2023. An Insight into the Chemistry of Cement—A Review. *Applied Sciences,* 13**,** 203.

LEHNE, J. & PRESTON, F. 2018. Making Concrete Change: Innovation in Low-Carbon Cement and Concrete. *Chatham House.* The Royal Institute of International Affairs.

LEWIS, R., SEAR, L., WAINWRIGHT, P. & RYLE, R. 2003. Cementitous Additions. *In:* NEWMAN, J. & CHOO, B. S. (eds.) *Advanced Concrete Technology: Constituent Materials.* Elsevier.

LI, N. & UNLUER, C. 2025. Towards a net zero built environment: decarbonisation of the UK cement industry. *npj Materials Sustainability,* 3**,** 10.

LI, S., FENG, Y., YANG, J. & CERNY, R. 2021. Expansion Mechanism and Properties of Magnesium Oxide Expansive Hydraulic Cement for Engineering Applications. *Advances in Materials Science and Engineering,* 2021**,** 1-9.

LIU, X., JIN, K., LI, X. & YANG, R. 2023. Low-carbon cement manufacturing enabled by electrified calcium looping and thermal energy storage. *International Journal of Greenhouse Gas Control,* 129**,** 103986.

LIU, Z., WANG, Z., YUAN, M. Z. & YU, H. B. 2015. Thermal efficiency modelling of the cement clinker manufacturing process. *Journal of the Energy Institute,* 88**,** 76-86.

LUCAS, A. 1934. *Ancient Egyptian materials & industries*, E. Arnold & Co.

MANNING, D. A. C., TANGTINTHAI, N. & HEIDRICH, O. 2019. Evaluation of raw material extraction, processivng, construction and disposal of cement and concrete products: datasets and calculations. *Data in Brief*.

MARENCO-PORTO, C. A., FIERRO, J. J., NIETO-LONDOÑO, C., LOPERA, L., ESCUDERO-ATEHORTUA, A., GIRALDO, M. & JOUHARA, H. 2023. Potential savings in the cement industry using waste heat recovery technologies. *Energy,* 279**,** 127810.

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.

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. *Proceedings of the 3rd International Conference on the Durability of Concrete Structures, ICDCS 2012.*

MCGUIRE, E., PROVIS, J. L., DUXSON, P. & CRAWFORD, R. 2011. Geopolymer Concrete: Is there an alternative and viable technology in the concrete sector which reduces carbon emissions? *Concrete 2011: Building a Sustainable Future.* Perth, Australia.

MINERAL PRODUCTS ASSOCIATION 2023. Profile of the UK Mineral Products Industry.

MOHAMED, A. K., MOUTZOURI, P., BERRUYER, P., WALDER, B. J., SIRAMANONT, J., HARRIS, M., NEGRONI, M., GALMARINI, S. C., PARKER, S. C., SCRIVENER, K. L., EMSLEY, L. & BOWEN, P. 2020. The Atomic-Level Structure of Cementitious Calcium Aluminate Silicate Hydrate. *Journal of the American Chemical Society,* 142**,** 11060-11071.

MONTEIRO, P. J. M., MILLER, S. A. & HORVATH, A. 2017. Towards sustainable concrete. *Nature Materials,* 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 2020. UK Concrete and Cement Industry Roadmap to Beyond Net Zero.

MPA 2023. Annual Cementitious.

MPA 2025. Written evidence submitted by Mineral Products Association (BIO0004). *In:* UK PARLIAMENT COMMITTEES (ed.).

MSINJILI, N. S., GLUTH, G. J. G., STURM, P., VOGLER, N. & KÜHNE, H.-C. 2019. Comparison of calcined illitic clays (brick clays) and low-grade kaolinitic clays as supplementary cementitious materials. *Materials and Structures,* 52**,** 94.

OGE, M., OZKAN, D., CELIK, M. B., SABRI GOK, M. & CAHIT KARAOGLANLI, A. 2019. An Overview of Utilization of Blast Furnace and Steelmaking Slag in Various Applications. *Materials Today: Proceedings,* 11**,** 516-525.

OGIRIGBO, O. 2016. *Influence of Slag Composition and Temperature on the Hydration and Performance of Slag Blends in Chloride Environments.*

OKSRI-NELFIA, L., MAHIEUX, P. Y., AMIRI, O., TURCRY, P. & LUX, J. 2016. Reuse of recycled crushed concrete fines as mineral addition in cementitious materials. *Materials and Structures,* 49**,** 3239-3251.

OLSSON, J. A., MILLER, S. A. & KNEIFEL, J. D. 2024. A review of current practice for life cycle assessment of cement and concrete. *Resources, Conservation and Recycling,* 206**,** 107619.

ONER, A. & AKYUZ, S. 2007. An experimental study on optimum usage of GGBS for the compressive strength of concrete. *Cement and concrete composites,* 29**,** 505-514.

ONS 2024. UK greenhouse gas emissions: Final figures 2023 Department for Energy Security and Net Zero: Office for National Statistics.

OSBORNE, G. J. 1991. Determination of the sulphate resistance of blastfurnace slag cements using small-scale accelerated methods of test. *Advances in Cement Research,* 2.

OUELLET-PLAMONDON, C. & HABERT, G. 2015. Life cycle assessment (LCA) of alkali-activated cements and concretes. *Handbook of alkali-activated cements, mortars and concretes.* Elsevier.

OUR WORLD IN DATA. 2023. *Coal, oil, gas, cement: how much does each contribute to CO2 emissions?* [Online]. Available: https://ourworldindata.org/co2/country/united-kingdom [Accessed 2025].

OVERMANN, S., VOLLPRACHT, A. & MATSCHEI, T. 2024. Reactivity of Calcined Clays as SCM—A Review. *Materials,* 17**,** 312.

OZCAN, D. C., AHN, H. & BRANDANI, S. 2013. Process integration of a Ca-looping carbon capture process in a cement plant. *International Journal of Greenhouse Gas Control,* 19**,** 530-540.

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. *Journal of Industrial Ecology,* 25**,** 359-376.

PANESAR, D. K., SETO, K. E. & CHURCHILL, C. J. 2017. Impact of the selection of functional unit on the life cycle assessment of green concrete. *The International Journal of Life Cycle Assessment,* 22**,** 1969-1986.

PCR 2019:14 2025. Product Category Rules for Construction Products Version 2.0.1.

PIEMONTI, A., CONFORTI, A., COMINOLI, L., SORLINI, S., LUCIANO, A. & PLIZZARI, G. 2021. Use of Iron and Steel Slags in Concrete: State of the Art and Future Perspectives. *Sustainability,* 13.

POGGIANTI, B., PALAZZOLO, R. & MOLINER, C. 2024. Rotary kiln simulation for energy recovery: The precalciner cement kiln case. *Thermal Science and Engineering Progress,* 54**,** 102806.

POSSAN, E., FELIX, E. F. & THOMAZ, W. A. 2016. CO2 uptake by carbonation of concrete during life cycle of building structures. *Journal of Building Pathology and Rehabilitation,* 1.

PROVIS, J. L. 2009. Activating solution chemistry for geopolymers. *Geopolymers: Structures, Processing, Properties and Industrial Applications.*

PROVIS, J. L. 2018. Alkali-activated materials. *Cement and Concrete Research,* 114**,** 40-48.

PROVIS, J. L. & BERNAL, S. A. 2014. Geopolymers and Related Alkali-Activated Materials. *Annual Review of Materials Research,* 44**,** 299-327.

QUEVEDO PARRA, S. & ROMANO, M. C. 2023. Decarbonization of cement production by electrification. *Journal of Cleaner Production,* 425**,** 138913.

RAFFETTI, E., TRECCANI, M. & DONATO, F. 2019. Cement plant emissions and health effects in the general population: a systematic review. *Chemosphere,* 218**,** 211-222.

RAJCZAKOWSKA, M., KOTHARI, A., BUASIRI, T. & CWIRZEN, A. 2025. Recycled and mechanically activated concrete fines as a complete substitute for Portland cement − Feasibility and life cycle assessment. *Case Studies in Construction Materials,* 22**,** e04798.

RATHNARAJAN, S., VADDEY, N. P., PILLAI, R. G., GETTU, R. & SANTHANAM, M. Modelling carbonation rates in concretes with similar strength and with and without slag. Conference: ICACMS, 2017.

REEVES, G. M., SIMS, I. & CRIPPS, J. 2006. *Clay materials used in construction*, Geological Society of London.

REHSI, S. S. 1983. Magnesium Oxide in Portland Cement. *Advances in Cement Technology***,** 467-483.

RIHNER, M. C. S., HAFEZ, H., WALKLEY, B., PURNELL, P. & DREWNIOK, M. 2025. Thousand cuts: a realistic route to decarbonise the UK cement and concrete sector by 2050. *Sustainable Production and Consumption,* 58.

ROBINSON, H., PAYNE, L., HIGGINS, D. D., MACLEOD, A., LAIDLAW, S., STEER, M., O'NEIL, D., GOODLAD, N., O'SULLIVAN, A., MCCALLAN, C., FAVAGER, R., BROOM, M., GHAZIREH, N., HARRIS, J., BARRITT, J., CLAYTON, S. & MOSELEY, S. 2007. BFS: a technical report on manufacturing of BFS & material status in UK. Environmental Agency: Waste Protocols Project.

SABERIAN, M., ZHANG, J., GAJANAYAKE, A., LI, J., ZHANG, G. & BOROUJENI, M. 2022. Life cycle assessment (LCA) of concrete containing waste materials: comparative studies. *Handbook of Sustainable Concrete and Industrial Waste Management.* Elsevier.

SCHINDLER, V., KOHLHASE, M., HOTTENROTH, H. & TIETZE, I. 2025. Closing Data Gaps with Emission Factors: A General Method Applied to EPDs in Brightway. *Preprints.* Preprints.

SCRIVENER, K., MARTIRENA, F. B., SHASHANK & MAITY, S. 2018. Calcined clay limestone cements (LC3). *Cement and Concrete Research,* 114**,** 49-56.

SEA-DISTANCES. 2023. Available: https://sea-distances.org/ [Accessed].

SHI, X., ZHANG, C., LIANG, Y., LUO, J., WANG, X., FENG, Y., LI, Y., WANG, Q. & ABOMOHRA, A. E.-F. 2021. Life Cycle Assessment and Impact Correlation Analysis of Fly Ash Geopolymer Concrete. *Materials,* 14**,** 7375.

SHIN, S., KIM, G., KIM, J., KIM, H. & PARK, S. 2025. Assessment of the Applicability of Waste Concrete Fine Powder as a Raw Material for Cement Clinker. *Recycling,* 10**,** 17.

SHOURIDEH, M. & YASSERI, S. 2024. Uncertainties and Barriers to CCS Acceptance & Implementation. Vol. 8**,** 18-30.

TEIXEIRA, E. R., MATEUS, R., CAMOES, A. F., BRAGANÇA, L. & BRANCO, F. G. 2016. Comparative environmental life-cycle analysis of concretes using biomass and coal fly ashes as partial cement replacement material. *Journal of Cleaner Production,* 112**,** 2221-2230.

TONIOLO, S., MAZZI, A., SIMONETTO, M., ZULIANI, F. & SCIPIONI, A. 2019. Mapping diffusion of Environmental Product Declarations released by European program operators. *Sustainable Production and Consumption,* 17**,** 85-94.

UK GOV 2021. Plans unveiled to decarbonise UK power system by 2035.

UK GOV 2024. UK carbon capture, usage and storage (CCUS). *In:* ZERO, D. F. E. S. A. N. (ed.).

UK GOVERNMENT 2025. Digest of UK Energy Statistics (DUKES) 2025 – Chapter 2: Coal. *In:* DEPARTMENT FOR BUSINESS, E. I. S. (ed.).

UK HOUSE OF PARLIAMENT 2019. The Climate Change Act 2008 (2050 Target Amendment) Order 2019. *In:* DEPARTMENT FOR BUSINESS ENERGY AND INDUSTRIAL STRATEGY (ed.).

UMER, M., AHMAD, J. & MUKHTAR, H. 2024. Innovative valorization of biomass waste-derived sodium silicate for geopolymer concrete synthesis: Sustainability assessment and circular economy potential. *Journal of Cleaner Production,* 452**,** 142181.

UNFCCC. 2016. *The Paris Agreement* [Online]. Available: https://unfccc.int/process-and-meetings/the-paris-agreement [Accessed].

VAN OSS, H. 2005. Background Facts and Issues Concerning Cement and Cement Data. US Geological Survey.

VOLAITY, S. S., AYLAS-PAREDES, B. K., HAN, T., HUANG, J., SRIDHAR, S., SANT, G., KUMAR, A. & NEITHALATH, N. 2025. Towards decarbonization of cement industry: a critical review of electrification technologies for sustainable cement production. *npj Materials Sustainability,* 3**,** 23.

WANG, X. & SONG, C. 2020. Carbon Capture From Flue Gas and the Atmosphere: A Perspective. *Frontiers in Energy Research,* Volume 8 - 2020.

WERNET, G., BAUER, C., STEUBING, B., REINHARD, J., MORENO-RUIZ, E. & WEIDEMA, B. 2016. The ecoinvent database version 3 (part I): overview and methodology. *The International Journal of Life Cycle Assessment,* 21**,** 1218-1230.

WORLD STEEL ASSOCIATION 2014. A methodology to determine the LCI of steel industry co-products.

WSA 2025. World Steel in Figures. *In:* ASSOCIATION, W. S. (ed.).

WU, K., CHEN, B. & YAO, W. 2001. Study of the influence of aggregate size distribution on mechanical properties of concrete by acoustic emission technique. *Cement and Concrete Research,* 31**,** 919-923.

XUE, B., YU, Y., CHEN, J., LUO, X. & WANG, M. 2017. A comparative study of MEA and DEA for post-combustion CO2 capture with different process configurations. *International Journal of Coal Science & Technology,* 4**,** 15-24.

YANG, J., ZHANG, T., GUO, L., ZHI, S. & HAN, J. 2023. Effect of Sodium Silicate on the Hydration of Alkali-Activated Copper-Nickel Slag Materials. *Metals,* 13**,** 596.

YOON, H.-J., ATSBHA, T. A., YOON, T., SHIN, D., AN, J., ZAREI, M., CHERIF, A., SUH, S. & LEE, C.-J. 2025. Sustainable and feasible carbon capture and utilization pathways towards net-zero. *Renewable and Sustainable Energy Reviews,* 211**,** 115331.

YOUNSI, A., TURCRY, P., AÏT-MOKHTAR, A. & STAQUET, S. 2013. Accelerated carbonation of concrete with high content of mineral additions: effect of interactions between hydration and drying. *Cement and concrete research,* 43**,** 25-33.

ZAJAC, M., SKOCEK, J., BEN HAHA, M. & DEJA, J. 2022. CO2 Mineralization Methods in Cement and Concrete Industry. *Energies* [Online], 15.

ZENG, D., SHCHERBINA, V. & LI, J. 2023. Thermal efficiency analysis of the rotary kiln based on the wear of the lining. *International Journal of Applied Mechanics and Engineering,* 28**,** 125-138.

ZHANG, H. 2011. *Cement*, Woodhead Publishing.

ZHANG, J., ZHANG, L., XU, B. & YUAN, J. 2023. Influences of Carbonated Recycled Concrete Fines on Cement Hydration. *Buildings,* 13**,** 926.

ZHANG, T., ZHANG, M., JIN, L., XU, M. & LI, J. 2024. Advancing carbon capture in hard-to-abate industries: technology, cost, and policy insights. *Clean Technologies and Environmental Policy,* 26**,** 2077-2094.