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Life cycle assessment of both cement and sand-replaced biochar concrete

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ABSTRACT

Biochar holds significant potential for reducing carbon emissions in the concrete industry. However, despite increasing interest in biochar-concrete, there is a lack of Life Cycle Assessment (LCA) studies to elucidate the mechanisms by which biochar reduces carbon emissions of concrete across various stages, such as raw material acquisition, transportation, manufacturing, etc. This study compares the environmental impacts of using biochar derived from wood and fruit waste as partial replacements for cement and sand in concrete. The research applies LCA methodology, in line with ISO 14040 and 14044, to evaluate the environmental impacts and resource use of biochar-concrete across its cradle-to-gate life cycle. Cement exhibits high carbon emissions of 0.63 kg CO_2-eq per kg, of which 80 % originates from the calcination whereas crushed sand demonstrates very low carbon emissions. On the other hand, both wood and fruit biochar exhibit negative carbon emissions due to the combined effects of co-product energy recovery and carbon sequestration. Using wood biochar to replace 20 vol.% of cement reduces carbon emissions of concrete by 42 %. Replacing 60 vol.% of sand with fruit biochar reduces concrete carbon emissions by 167 %, indicating a net-negative carbon footprint and demonstrating its effectiveness as a carbon sink. Moreover, fruit biochar shows a greater potential for reducing the carbon footprint of concrete and exhibits higher sensitivity to intensity analysis due to the larger volume replacement it offers for sand. The findings indicate that key parameters, such as biomass moisture content and biochar yield rate significantly influence the decarbonization potential, whereas applied average transportation distance contributes minimally to overall emissions.

1. Introduction

Climate change remains one of the most pressing global challenges, driven primarily by excessive anthropogenic carbon emissions (Wang et al., 2024a). One of the most significant contributors is the construction sector, which is responsible for substantial energy consumption and greenhouse gas (GHG) emissions. In 2018, the sector accounted for 34 % of global energy use and 37 % of energy- and process-related carbon emissions, with approximately 11 % originating from the production of construction materials, such as steel, cement, and glass (IEA, 2024). With the intensifying urgency of climate action and the global shift toward sustainable development, demand for environmentally responsible construction materials has grown rapidly (Luo et al., 2023). Given that cement production is responsible for roughly 8 % of global carbon emissions, it represents one of the most critical materials for targeted strategies to reduce environmental impacts and GHG emissions (Knight et al., 2023; Meyer, 2009). Concrete is composed of cement, fine and coarse aggregates, water, and various chemical admixtures (Luo et al., 2023). The resulting composite is both durable and versatile, making it one of the most widely used materials in construction (Luo et al., 2022).

The production of cement, the primary binding agent in concrete, comprises several main stages, including raw material preparation,

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clinker production, and cement grinding. The manufacturing process is highly energy-intensive, with substantial carbon emissions arising from the calcination of limestone, fossil fuel combustion in cement kilns, and energy use in raw material and cement grinding (Shan et al., 2019; van Deventer et al., 2021). Cement production remains highly carbon intensive, with emissions typically ranging between 0.5–0.9 ton of CO_2 per ton of cement, depending on the production process and regional practices (Das et al., 2023; Ige et al., 2024). In addition to the significant carbon emissions released from cement production, growing demand for cement has further intensified the extraction of aggregates, particularly limestone, a key raw material. Overexploitation of such natural resources risks accelerating resource depletion, leading to permanent ecological deterioration such as biodiversity loss, vegetation degradation, ecosystem destruction, river damage, dust contamination and broader environmental impact including global warming and climate change (Blankendaal et al., 2014; Mohamad et al., 2022). Therefore, there is a need for effective decarbonization strategies, and optimised concrete mix designs.

Besides cement, aggregates constitute approximately 65–80 vol.% of a concrete mix (Bhoopathy and Subramanian, 2022; Wang et al., 2024b). Among these, sand has become the most intensively extracted material in the 21st century. In 2015, global material extraction reached 89 gigatons, with non-metallic minerals accounting for more than half of this total (Krausmann et al., 2018). Aggregates alone represented 80 % of non-metallic mineral consumption (Miatto et al., 2017), making sand the most extracted resource worldwide (Torres et al., 2017). This high demand has caused severe geomorphological changes, depletion of existing resources, reduced natural replenishment of sand, ecological damage, and environmental pollution, ultimately leading to shortages at both local and global scales (Bendixen et al., 2019). These environmental and resource challenges have increased interest in the construction sector in developing and adopting sustainable alternatives to natural sand for concrete production.

Biochar is a co-product formed during biomass pyrolysis, which is the thermal decomposition of organic matter (often waste) such as rice straw, wood waste, food waste, sludge, and manure under limited oxygen (or inert conditions) at temperatures typically between 350 and 900 °C (Mohanty et al., 2024). This process stabilizes carbon, allowing it to be stored for extended periods (Bridgwater, 2003; Cuthbertson et al., 2019). It is characterized by high carbon content, a large specific surface area, and a porous structure. The composition of biochar typically contains 44 % to 95 % carbon, 0 % to 45 % oxygen, and 1 % to 9 % hydrogen, depending on the feedstock and processing techniques (Mohanty et al., 2024). Different biomass sources vary in moisture content, carbon content, mineral-ash content, and chemical composition, all of which influence the properties of the final product (Barbhuiya et al., 2024). In general, higher pyrolysis temperatures and longer residence times increase carbon and ash content while reducing the volatile matter (Mohanty et al., 2024). Owing to its distinctive physicochemical properties, biochar has a wide range of applications, including carbon sequestration (Lin et al., 2026), soil amendment, and high value applications such as polymer composite development (Zhang et al., 2026), energy storage and conversion (Li et al., 2026), activated carbon, supercapacitors and food/feed supplement (Barbhuiya et al., 2024; Mensah et al., 2021). Biochar in concrete can act as a filler and provide internal curing, which can reduce shrinkage and enhance durability (Chen et al., 2025; Lin et al., 2026), as well as aid in the stabilization and solidification of heavy metals (Sun et al., 2026). The high porosity of biochar may also reduce pressure buildup in concrete during high-temperature exposure (Zeng et al., 2026). Incorporating biochar into concrete can potentially reduce its carbon footprint, as biochar can be made to be carbon-negative or carbon-neutral, depending on conditions and because it is derived from renewable biomass. Biochar is produced from waste, but the heat released during pyrolysis can be captured and used as an energy resource, reducing the overall energy input. Additionally, the stable carbon in biochar prevents carbon emissions by securely storing carbon that would otherwise be released into the atmosphere. It is estimated that one ton of biochar has the potential to sequester 2.0 to 2.6 ton of CO_2, contributing a highly environmental friendly practice (Azzi et al., 2019; Li et al., 2026).

Life Cycle Assessment (LCA) methodology is often used to investigate the potential environmental benefits of biochar in concrete. According to the ISO14040–14044 standards (ISO, 2006a, 2006b), LCA is a science-based methodology applied to evaluate the environmental impacts of a product, process, or service throughout its entire life cycle (Kaynak et al., 2025). It consists of four stages that are goal and scope definition, inventory analysis, impact assessment, and interpretation. The LCA system boundary of biochar concrete typically includes raw material extraction and processing, manufacturing, chemicals and reagents, transportation, and concrete mixing (Ee et al., 2025). Through LCA, environmental impacts arising from these processes, such as energy consumption, emissions and resource depletion, are systematically quantified and evaluated (Barbhuiya et al., 2024). A range of specialised software tools, including SimaPro, OpenLCA, and GaBi, can be employed to model and analyse the product's life cycle (Shmlls et al., 2023).

To date, international scholars have attempted to evaluate the environmental impact of biochar-concrete in comparison with conventional concrete. For example, Campos et al. (2020) examined the environmental and health impacts of concrete containing 0–20 wt.% biochar rice husk-ash as a partial cement replacement. Results demonstrated that increasing the proportion of biochar consistently reduced the environmental impact of concrete. The study by Chen et al. (2022) placed greater emphasis on incorporating biochar as an aggregate in concrete, proposing an innovative design of biochar-augmented carbon-negative concrete. The LCA confirmed that each 10 wt.% increase in biochar content significantly reduced total CO_2 emissions. Moreover, concrete blocks containing 30 wt.% biochar combined with supplementary cementitious materials achieved a carbon-negative profile, sequestering between 59 and 65 kg CO_2 per ton of product. Elsewhere, Labianca et al. (2024) not only identified reliable designs for biochar-augmented cementitious products but also explored their general applications through comprehensive assessments including technical, environmental, and economic aspects. LCA results showed that higher biochar dosages reduced the Glocal Warming Potential (GWP), with the greatest reduction reached 720 kg CO_2 per ton, representing a 200 % decrease relative to the reference mix. A recent study by Ee et al. (2025) demonstrated that incorporating biochar into concrete, particularly in combination with fly ash, offers substantial environmental benefits. Replacing 5 wt.% of cement with biochar and 35 wt.% with fly ash resulted in a 23 % reduction in GWP of concrete.

Existing research has demonstrated that biochar can significantly reduce the environmental impacts of concrete production, either through its use in cementitious composites or as a substitute for fine aggregates/sand. However, most studies have focused on simple substitution strategies in isolation, without comparing the manufacturing processes of biochar, cement, and sand. This study addresses this gap by providing a direct and systematic comparison of the decarbonization potential and underlying reasons for biochar being used as a cement and sand (i.e., fine aggregate) replacement in concrete, using a region-specific LCA model for Sweden. This study employs a cradle-to-gate LCA to analyze the decarbonization potential of incorporating biochar into concrete. Data were sourced from open literature, industrial reports, and our own mechanical strength test results. First, the carbon emissions from the production of biochar powder, biochar aggregate, cement, and crushed sand (from raw material collection to product) were analyzed separately. These were then compared to identify why biochar can contribute to concrete decarbonization. Then, the decarbonization potential was quantified using our experimental concrete mix design.

2. Research significance

In Sweden, the construction sector contributes approximately 20 % of the country's total annual carbon emissions (Karlsson et al., 2020). Of cement-related emissions in 2021, about 40 % stemmed from fossil fuel combustion for kiln heating, while roughly 60 % resulted from the thermal decomposition of limestone, an unavoidable process in cement production (Cementa and Sverige, 2018; Driessen and Grönlund, 2024). Although decarbonizing the construction sector is urgent and biochar shows promise in concrete applications, research on its decarbonization effects remains limited in Sweden. Existing reports (Jacqueline et al., 2022; Azzi, 2021) outline the potential of biochar in the concrete Swedish industry but provide no detailed emission analyses. Likewise, a case study in Uppsala city in Sweden investigated biochar as a cement substitute but lacked data on material manufacturing processes, making it unclear whether its decarbonization potential derives from reduced energy demand, fuel consumption, or other parameters. Thus, a comprehensive assessment comparing the environmental impacts of biochar, cement, and sand is needed to identify the key drivers of emission reduction, guide effective decarbonization strategies in Sweden, and establish a replicable framework for future LCA studies in international contexts.

3. Methods

3.1. Description of the LCA model

3.1.1. Goal and scope definition

The overarching objectives of this study are threefold: (a) to develop a comprehensive life cycle inventory that reflects the current state of cement, sand, and biochar production in Sweden, encompassing raw material use, energy flows, and transportation; (b) to conduct a systematic analysis comparing biochar with cement and sand in order to identify pathways for minimizing carbon emissions within the Swedish concrete industry; and (c) to evaluate the potential advantages of integrating biochar into concrete production at a national scale.

3.1.2. System boundary definition

This study applies a cradle-to-gate system boundary to assess the environmental impacts of biochar-concrete production. The system boundary accounts for environmental impacts from raw material extraction through to the point where the product exits the factory (Vieira et al., 2016). The study defines system boundaries that encompass the production of wood-waste biochar (hereafter termed as wood biochar (Wang et al., 2025b), olive fruit-pit biochar (hereafter termed as fruit biochar (Wang et al., 2025c), cement, and sand. The system boundaries include raw material supply, transportation, product manufacturing, and disposal, following a similar pattern to that used in other research (Ma et al., 2025). Machinery use is excluded from the assessment during manufacturing process. Fig. 1 illustrates the cradle-to-gate system boundary applied in this study.

3.1.3. Materials and scenarios

The primary objective of this study is to evaluate the environmental benefits of substituting cement and sand with biochar in concrete, with the broader aim of further reducing emissions through biochar integration. The mass based functional units are employed. Specifically, 1 kg of each material, including cement, sand, wood biochar and fruit biochar, is analyzed for comparative purposes, alongside 1 m^3 of concrete as the reference unit for the final mix. To ensure a robust comparison between environmental benefits and concrete performance, the mix design derived from previous experimental results (Section 3.2) was adopted. The concrete mixtures incorporated different biochar replacement ratios: 5 vol.%, 10 vol.%, and 20 vol. % substitution of cement with wood biochar, and 30 vol.%, 60 vol.%, and 100 vol.% substitution of sand with fruit biochar. In total, two distinct biochar production pathways were considered. Wood biochar used for cement replacement, originated from logging residues, whereas fruit biochar used as a sand substitute, was sourced from waste olive fruit pits. The biomass supply chain encompassed feedstock collection, transportation, and subsequent chipping and drying at the pyrolysis facility.

The transportation-related emissions represent a significant component of LCA. This study selected ten largest Swedish cities (i.e., Stockholm, Gothenburg, Mälmö, Uppsala, Västerås, Örebro, Linköping, Helsingborg, Jönköping and Lund) as target locations for both the transportation of raw materials and the utilization of concrete within the country. According to Ma et al. (2025), transport distances for raw materials within the same urban area typically range from 20 to 70 km,

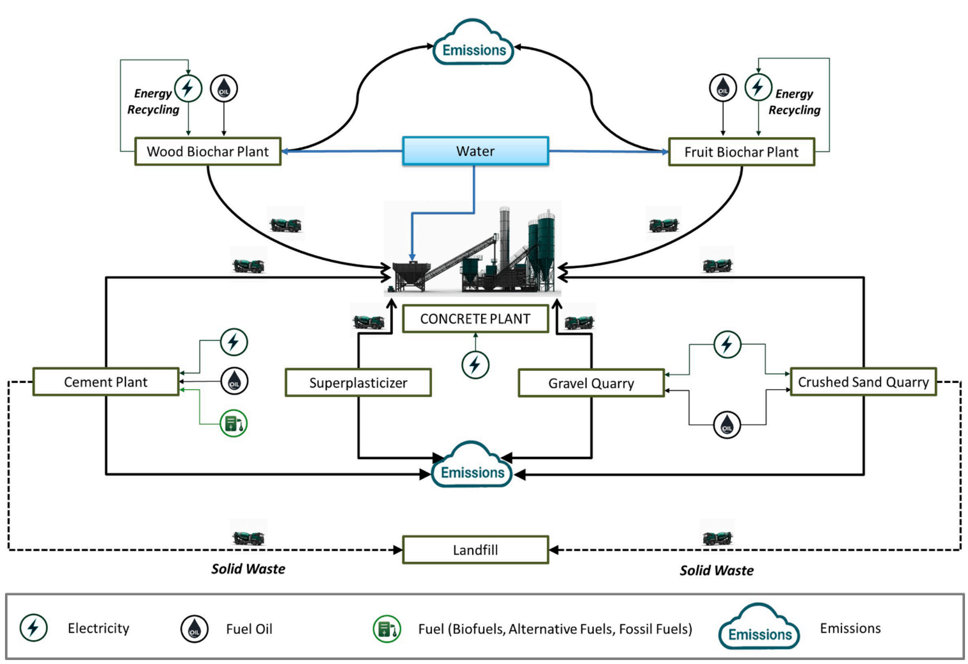

Fig. 1. Cradle-to-gate system boundaries for using biochar in concrete (Ma et al., 2025).

while other studies (Azzi et al., 2022; Huang et al., 2022) have standardized this distance at 50 km. In this study, a default transport distance of 50 km is assumed for raw materials sourced from local suppliers within the same city. For suppliers located in other cities, transport distances are derived from the actual intercity distances obtained via Google Maps, assuming lorry transport as the primary mode. The distance between biochar manufacturers to concrete mixing plants in Sweden's ten largest cities was calculated and presented in Table S1 in Appendix A. The average of these distances is used in the LCA modeling.

Heidelberg Materials Cement Sverige AB is the sole cement producer in Sweden, operating two manufacturing plants located in Skövde and Slite due to the availability of local raw materials, such as limestone, gypsum, and shale. A transport distance of 50 km for these materials was assumed, reflecting typical local sourcing practices and proximity to the production sites. The raw materials are usually situated in the vicinity of the cement production facility. The transport distances from Skövde and Slite plants to the ten largest target cities were calculated and are presented in Table S2 in Appendix A, with the average of these distances used in this study. For sand and gravel, a distance of 200 km was assumed, based on a Swedish case study (Azzi et al., 2022). A summary of all transport distances used in the analysis is presented in Table 1.

3.1.4. Life cycle impact assessment (LCIA)

Life Cycle Impact Assessment (LCIA) is a core phase of LCA. LCIA translates the inventory of emissions and resource use (from the LCA inventory phase) into environmental impact categories, such as global warming, acidification, eutrophication, and toxicity (Guinée et al., 2002). It is guided by ISO 14040 (ISO, 2006a) and ISO 14044 (ISO, 2006b), which typically includes mandatory steps (classification and characterization) and optional steps (normalization, grouping, weighting). By applying characterization factors, LCIA pinpoints where emissions and resource extractions may contribute to environmental burdens, thereby supporting the identification of trade-offs and opportunities for improvement within the system under study. In this research, the LCIA was performed using the CML-IA baseline method (version 3.09), developed at Leiden University's Institute of Environmental Sciences (CML, 2016). The CML framework is widely recognised in LCA practice for its midpoint-oriented indicators, which provide category-specific results without aggregating them into a single weighted score, thus preserving transparency in interpreting environmental trade-offs (Jolliet et al., 2016). This study primarily examines the impact of biochar on the GWP, which is quantified using the 100-year Global Warming Potential (GWP100). Other impact categories, including abiotic depletion, ozone layer depletion, human toxicity, terrestrial ecotoxicity, photochemical oxidation, and acidification, are also considered to provide a comprehensive view.

LCA was conducted using SimaPro 9.6.0.1 software with Ecoinvent 3.10 database. The modeling platform facilitated a transparent representation of processes across the system boundaries, while the database supplied consistent, process-level information on material flows, energy use, and emissions. This approach will allow for a robust and comprehensive evaluation of potential environmental impacts.

Table 1
Transport mode, distances, and moisture content during transport, assumed in the foreground model.

Activity	Distance (km)	Source	Mode
Biomass supply to pyrolysis plant (except mobile unit)			
Logging residues	150	Azzi et al.(Azzi et al., 2022)	lorry 16–32 t
Fruit pits	150	Azzi et al. (Azzi et al., 2022)	lorry 16–32 t
Biochar supply from pyrolysis plant to point of use (e.g. cement factory, soil factory)			
Wood biochar	86.4	Table S1 in Appendix A	lorry 16–32 t
Fruit biochar	86.4	Table S1 in Appendix A	lorry 16–32 t
Material supply to point of use or processing plant			
Sand, gravel within Sweden	200	Azzi et al.(Azzi et al., 2022)	lorry >32 t
Limestone, shale	50	Assumption	lorry >32 t
Portland cement	70 242.7	Table S2 in Appendix A	ship lorry >32 t
Intermediate product transport between processing plants			
Applies to all materials between two processes, if there is no specific transport assumption	5	Assumption	lorry 16–32 t
Transport to final disposal, landfill or incineration			
Solid waste	50	Assumption	lorry 16–32 t

3.2. Data collection

3.2.1. Life cycle inventory (LCI) modeling of cement

Cement is a fundamental constituent of concrete and a major source of global carbon emissions. As seen in Fig. 2, the production and supply stages comprise limestone quarrying, raw material preparation, transportation, calcination, clinker production and grinding, as well as the management of co-products and emissions from decomposition. Emissions associated with cement production can be broadly categorized into process-related and fuel-related sources. The latter primarily arise from electricity consumption for crushing and grinding, as well as fuel consumption during calcination and transportation. In the Swedish cement industry, the average thermal energy intensity is approximately 3.6 MJ per kilogram of clinker (Mossie et al., 2021). Based on 2020 data for Sweden (Mossie et al., 2021), the fuel mix employed in the cement industry consisted of 38 % fossil fuels, 39 % alternative fuels (AF), and 23 % biofuels. At Heidelberg Materials Cement Sverige AB, the primary cement manufacturer in Sweden, the clinker production relies on all three categories of fuels. In this study, hard coal is selected as the representative fossil fuel. The term AF denotes a heterogeneous mixture of biomass and waste streams, including scrap tires, municipal solid waste, residues from meat production, woody biomass, sewage sludge, textiles, paper, and agricultural by-products. For analytical simplification, municipal solid waste is adopted as the representative AF, as it typically embodies a composite of these materials. According to Global CemFuels (Global CemFuels, 2023), the most commonly used biofuels in cement production include palm kernel shells, rice husks, coconut shells, sugarcane bagasse, and sawdust. In this study, these biofuels are assumed to be used in combination. The corresponding moisture contents and calorific values for biofuels are drawn from generalized data, as summarized in the supplementary Table S3 in Appendix A. In this study, waste wood chips are considered the representative biofuel in the raw material supply chain, with moisture content and calorific value taken as the average values in Table S3 in Appendix A. For hard coal, the gross calorific value (dry ash-free basis) ranges from 33.42 to 36.58 MJ/kg, with moisture contents between 1.3 % and 5.9 % and ash contents ranging from 18.7 % to 42.9 % (Balaeva et al., 2018). In this study, the gross calorific value (dry ash-free basis) is assumed to be 35 MJ/kg, with a moisture content of 3.6 % and an ash content of 30.8 %. Based on these assumptions, the gross calorific value of hard coal, as received, is calculated to be 22.96 MJ/kg. Table 2 provides details of the fuel parameters used in this study.

For the cement used in this study (CEM II/A-LL 42.5 R), the composition is specified in SS-EN 197-1 (SS-EN 197-1, 2011), having clinker (80–94 %), limestone (LL) (6–20 %), and gypsum (typically 3–5 %). Accordingly, this study assumes a composition of 85 % clinker, 10 % limestone filler, and 5 % gypsum. Previous research (Manning et al., 2019) indicates that the production of 1 kg of clinker requires 1.2378 kg of limestone and 0.4162 kg of shale, while generating 0.0671 kg of solid waste. Furthermore, another study (Gao et al., 2016) reports that the raw material mixture prior to calcination contains 2.82 %

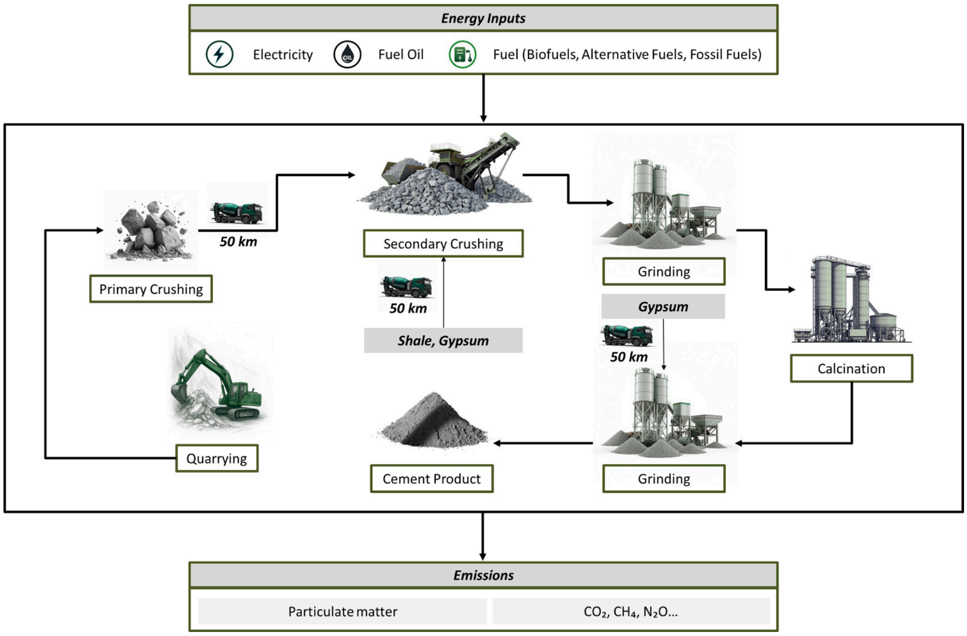

Fig. 2. Cradle-to-gate system boundaries for cement.

Table 2
Parameters of fuels used in cement production.

Materials	Percentage (%) (Mossie et al., 2021)	Moisture (%)	Calorific value (MJ/kg)
Fossil fuel (hard coal) (Mohamad et al., 2013)	38	3.6000	22.9600
AF (Municipal solid waste) (Tom et al., 2016)	39	61.2500	9
Combined biofuel (i.e., palm kernel shells, rice husks, coconut shells, sugarcane bagasse and sawdust) from Appendix Table S3.	23	20.8880	16.4624

moisture, whereas the finished cement retains 0.48 %. On this basis, the raw materials are assumed to have a moisture content of 2.82 %, and the final cement product is assumed to contain 0.48 % moisture. Consequently, the quantities of raw materials and fuels required to produce 1 kg of cement are determined using Equations (1–7) in Appendix A.

Process-related emissions in cement production primarily arise from the decomposition of carbonates in limestone. These emissions can be quantified by considering the chemical reactions occurring in the kiln: $CaCO_3 \rightarrow CaO$ (Clinker) + CO_2 and $MgCO_3 \rightarrow MgO + CO_2$. In the case of CEM II/A-LL 42.5 R used in this study (Wang et al., 2025b), the MgO content is relatively low (1.1–1.3 %). Consequently, the contribution of $MgCO_3$ in the limestone is neglected, and only the decomposition of $CaCO_3$ is considered. Previous research (Abiodun et al., 2022), indicates that the decomposition of 1 kg of limestone yields 0.56 kg of lime and releases 0.44 kg of CO_2. Accordingly, the process-related emissions associated with the production of 1 kg of CEM II/A-LL 42.5 R are calculated using Eq. (7) in Appendix A. With respect to fuel-related emissions, the electricity to produce cement is 131 kWh/t (Mossie et al., 2021). Another study indicated that cement production requires 83–110 kWh of electrical energy per ton, primarily for grinding (Hosten and Fidan, 2012), while crushing and grinding together account for approximately 70 % of this total consumption. On this basis, assuming 115 kWh per ton of cement, the electricity demand for producing 1 kg of cement is estimated at 0.0684 kWh, representing the emissions associated with crushing and grinding processes. Finally, the complete inventory of energy and material inputs required to produce 1 kg of CEM II/A-LL 42.5 R is summarized in Table 3.

3.2.2. *Life cycle inventory (LCI) creation and analysis of biochar*

The operation process of the modeled pyrolysis plant is illustrated in Fig. 3. It starts with (i) the collection of feedstocks, which is delivered to the site and ends with (ii) the delivery of energy products and biochar, along with the management of wastes. The process includes the following sub-steps: drying, pyrolysis, grinding (only for wood biochar), combustion of co-products, and quenching of biochar. The characteristics of biochar is shown in Table 4.

Two biochar production chains are considered in this study, utilizing logging residues and waste olive pits as biomass feedstocks. The biomass supply is modeled with a production yield assumed to be 30 %, according to Meyer et al. (2011). The moisture content of collected logging residues used for wood biochar production is assumed to be 50 %, based on a Swedish case study (Azzi et al., 2022). The collected olive pit biomass used to produce fruit biochar is assumed to have a moisture content of 10 % from 9-10 % in Miranda et al. (2008). The moisture content of the wood and fruit biochar used in the laboratory was measured as 3.16 % and 5.82 %, respectively. Consequently, the material parameters are summarized in Table S4 in Appendix A.

Wood biochar must be ground to match the particle size distribution of cement. The same grinding procedure used for cement is assumed, with electricity consumption representing the grinding energy required. Cement production is assumed to require 115 kWh of electrical energy

Table 3
Materials and energy needed to produce 1 kg CEM II/A-LL 42.5 R.

Materials or energy	Unit	Amount
Limestone (for clinker and SCMs)	kg	1.1790
Shale	kg	0.3620
Gypsum	kg	0.0512
Process electricity	kWh	0.0345
Crushing and Grinding electricity	kWh	0.0805
Fossil fuel (hard coal)	kg	0.0525
AF (As used)	kg	0.2138
Biofuel (As used)	kg	0.0517
CO_2 emission, at calcination	kg	0.4737
Solid waste	kg	0.0583
Transport to site of storage	km	5

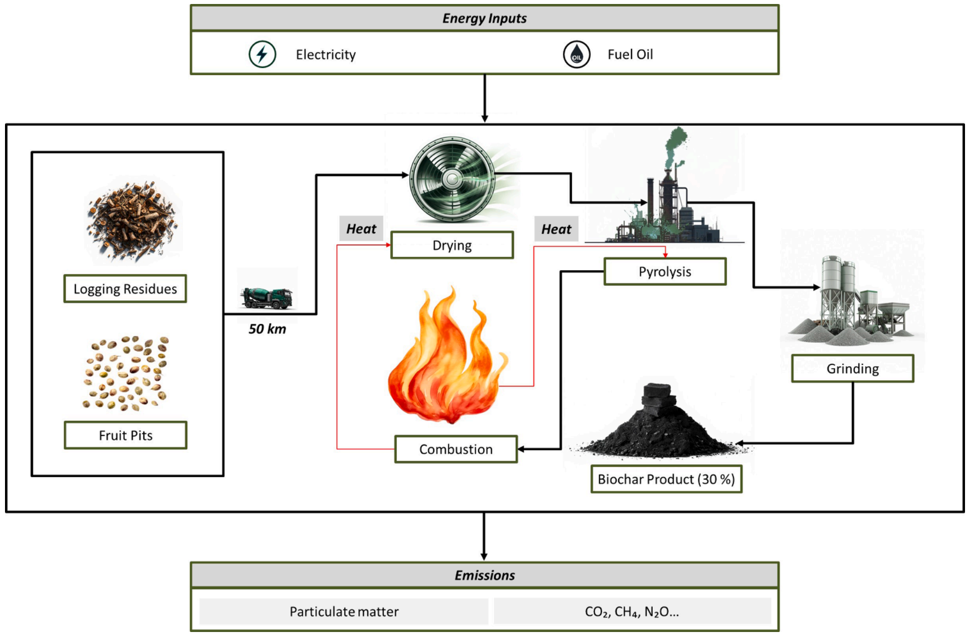

Fig. 3. Cradle-to-gate system boundaries for biochar.

Table 4
Characteristics of biochar. Pyrolysis process conditions are confidential according to NovoCarbo GmbH, the biochar supplier.

Type	True density (g/cm^3)	Hardness (GPa)	Young's Modulus (GPa)	Water absorption (tea bag test)	Particle size distribution	BET Specific surface area (m^2/g)
Wood biochar	1.9	0.28 ± 0.03	5.1 ± 0.3	2.15 g/g	10 to 140 μm with 20 μm as the dominant particle size	300, dry basis
Fruit biochar	1.58	0.22 ± 0.04	3.37 ± 0.4	1.3 g/g	2.25 to 4.75 mm with 2.95 mm as the dominant particle size	322, as received

per ton, of which approximately 40 % is attributed to grinding (Hosten and Fidan, 2012; Mossie et al., 2021). On this basis, the electricity demand for crushing and grinding 1 kg of biochar is estimated at 0.046 kWh. In contrast, fruit biochar requires no grinding since its particle size distribution is already comparable to that of sand. Similarly, biomass drying is modeled as an electricity-driven process. Previous studies (Konopka et al., 2021; Trebula P, 2005) have shown that the average specific heat consumption of drying kilns ranges between 4500 and 5600 kJ/kg of evaporated water. In this study, a value of 5000 kJ/kg (equivalent to 1.39 kWh/kg) is assumed. Table 5 provides a detailed inventory of the materials and energy input used in this study. Additional parameters, including biosphere emissions from pyrolysis, are adopted from a previous study (Azzi et al., 2022), conducted with the Pyreg 1500 syngas reactor. These emissions, detailed in Table S5 in Appendix A, are characteristic of the Pyreg 1500 system.

Table 5
Technosphere inputs and outputs for the different biochar supply chains.

Materials or energy	Unit	Logging residue	Fruit pit
Biochar (as used)	kg	1	1
Biochar produced, dry	kg	0.9684	0.9418
District electricity produced	kWh	7.3104	7.1420
Process electricity, operation	kWh	0.1928	0.1883
Drying energy required to evaporate 1 kg of water from biomass (Electricity)	kWh	1.3900	1.3900
Grinding (Electricity)	kWh	0.0460	0
Quenching water	kg	1.5424	1.5069
Start-up LPG fuel	kg	0.0026	0.0025
Biochar CO_2 sequestration, at production	kg	2.4678	3.1644
Biochar transport to storage site	km	5	5

3.2.3. Life cycle inventory (LCI) modelling of crushed sand

The modelling of crushed sand production accounts for fuel consumption, electricity use, and transportation. The production process is relatively straightforward, as illustrated in Fig. 4. Rock is first extracted and mechanically reduced to smaller blocks, after which it is conveyed to a production line for further crushing and screening. Emission parameters are adopted from previous research and the production yield is assumed to be 80 %, with the remaining 20 % classified as waste (Ma et al., 2025). Electricity consumption by process-line machinery (including the vibrating feeder, jaw crusher, cone crusher, vibrating screen, and conveyor belt) is estimated at 1.203 kWh per ton of crushed sand from supplentary file (Ma et al., 2025). Additionally, the moisture content of raw coarse aggregates after delivery typically ranges from 1-4 % by weight (MixDesignCalc), therefore, a value of 2 % is assumed. The crushed sand used in our laboratory measurements contained 0.17 % moisture. The material inputs required to produce 1 kg of crushed sand are calculated using Equations. (12) and (13) in Appendix A, and the results are presented in Table 6, together with the corresponding energy consumptions.

3.2.4. LCI modelling of concrete

According to an industrial report from Development Fund of the Swedish Construction Industry (SBUF, 2024), it is assumed that a concrete factory consumes 13 kWh of electricity and 10 kWh of heat per cubic meter of cast-in-place concrete, while a prefabrication factory consumes 70 kWh of electricity and 70 kWh of heat per cubic meter. Additionally, according to the specifications provided by the machinery manufacturer for concrete mixing factories (EPDAS, 2025), concrete mixing requires between 10 and 50 kWh per m^3. Therefore, in this study, cast-in-place concrete was chosen, and the mixing energy is assumed to

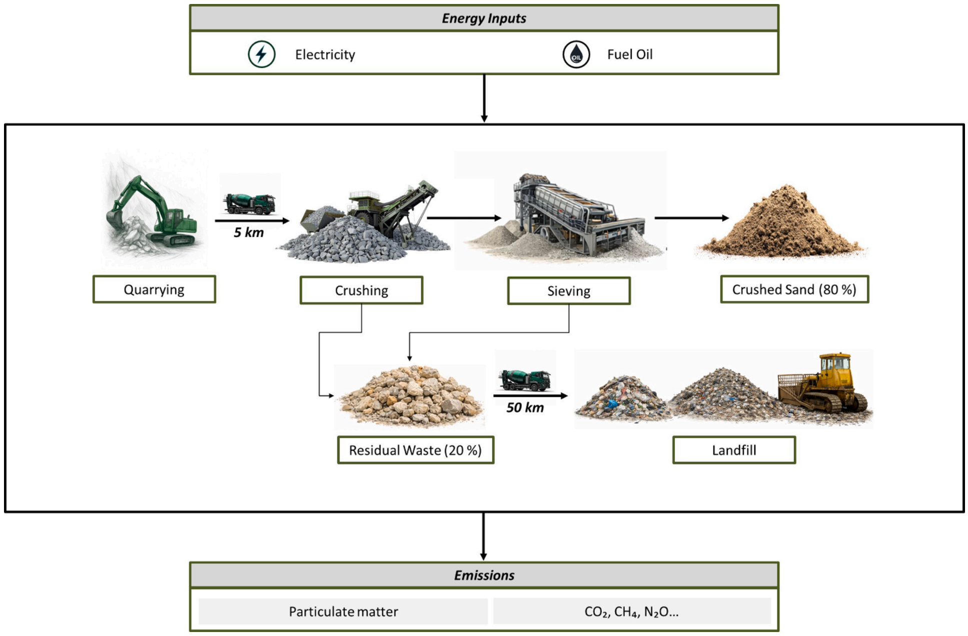

Fig. 4. Cradle-to-gate system boundaries for crushed sand.

Table 6
Materials and energy needed to produce 1 kg crushed sand.

Materials or energy	Unit	Amount
Basalt stone	kg	1.2728
Waste	kg	0.2545
Electricity	kWh	0.0012
Transport to site of storage	km	5

be 25 kWh per cubic meter of concrete. A simplified flow diagram of concrete production and delivery is provided in Fig. 5, while the concrete mix design used for the LCA is presented in Table S6 in Appendix A. This study does not explicitly model the use phase or variations in service life, as its primary focus is on evaluating the potential of biochar to decarbonize concrete materials.

3.2.5. *Other materials and supplies*

All energy inputs assumed to be consumed in Sweden (including electricity, heat, and transportation) were modeled using customized activities. Life cycle inventory (LCI) data for the production of raw materials, such as limestone, shale, water, superplasticiser, gypsum, and gravel, were obtained from the Ecoinvent database (version 3.10, cutoff system model). Electricity and diesel inputs were region-specific, taken from Swedish or European datasets within the same system model. Waste generated during the production process was assumed to be disposed of in a solid waste landfill (cutoff system model).

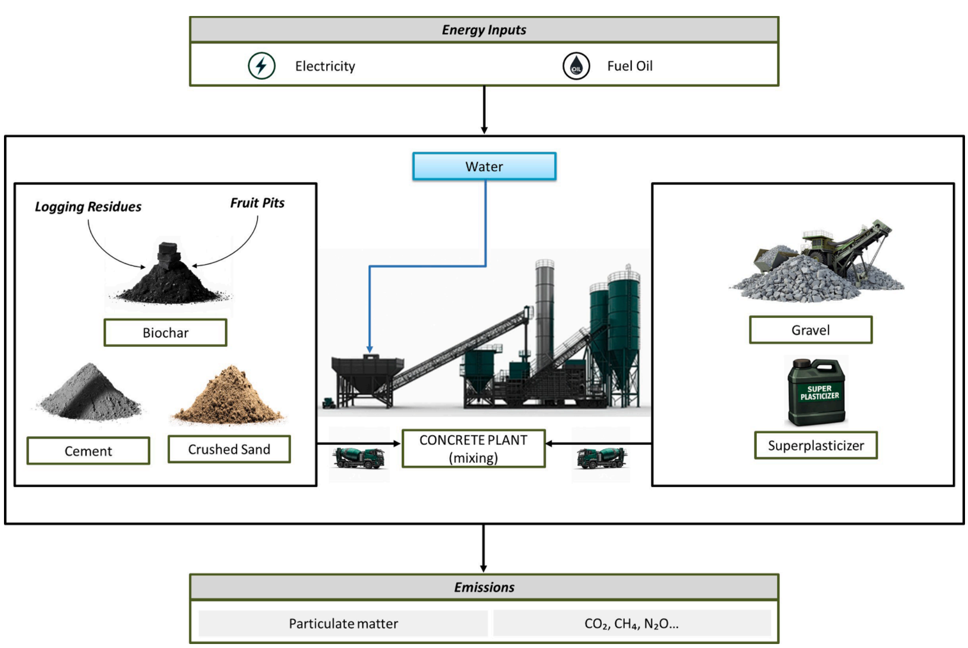

Fig. 5. Cradle-to-gate system boundaries for concrete.

4. Results

4.1. Material comparison

4.1.1. Comparison between cement and wood biochar

The details of the data sources and calculations can be found in the Appendix, and the inventory details for each material are described in Section 3.2. As shown in Fig. 6 and Table 7, cement exhibits substantially higher carbon emissions than wood biochar, primarily due to its high fuel consumption and decomposition during calcination. The carbon footprint of calcination reaches 0.56587 kg CO_2-eq per kg of cement, contributing to a total of 0.63 kg CO_2-eq per kg. In contrast, wood biochar achieves negative carbon emissions of −2.68308 kg CO_2-eq per kg, owing to energy recovery during pyrolysis and carbon sequestration. For both materials, emissions are largely associated with energy inputs. However, biochar requires more energy than cement for mechanical processing and transportation.

4.1.2. Comparison between sand and fruit biochar

As shown in Fig. 7 and Table 7, the production of crushed sand results in relatively low carbon emissions, totalling 0.0138 kg CO_2-eq per kg. Similar findings have been reported in previous studies. For e.g., the carbon footprint of natural sand production is only 0.00216 kg CO_2-eq per kg when considering solely the fuel and energy used during mechanical processing (Ma et al., 2025). Another study reported carbon emissions for natural sand ranging from 0.0078 to 0.033 kg CO_2-eq per kg, with a maximum of 0.103 kg CO_2-eq per kg (Marinković et al., 2010), which is relatively low. In contrast, fruit biochar requires substantial energy inputs for transport and mechanical processing, including drying and chipping, resulting in higher carbon emissions during production. However, due to energy recovery from co-products and the sequestration of stable carbon, fruit biochar achieves significant negative carbon emissions.

4.2. Benefits of biochar in carbon emissions of concrete

Cement serves as the primary binder in concrete, providing structural strength and durability, which results in its large volume and pivotal role in the material. As shown in Fig. 8(a), the carbon emissions of concrete are dominated by cement, accounting for approximately 80 % of the total. Although aggregates, such as gravel and sand constitute the majority of concrete by volume, over 60 % (Wang et al., 2024b), their carbon footprint remains low due to the simplicity of their production processes, which require minimal energy inputs (Ma et al., 2025). Given the high carbon intensity of cement, partial replacement by wood biochar, which exhibits negative carbon emissions, offers substantial potential for reducing the overall carbon footprint of concrete. Fig. 8(b) illustrates that the carbon emissions of plain concrete (control) are 393 kg CO_2-eq per m^3, which decrease to 228 kg CO_2-eq per m^3 when 20 vol.% of cement is replaced with wood biochar. At replacement ratios of 5 vol.%, 10 vol.%, and 20 vol.%, carbon emissions are reduced by 10.4 %, 21 %, and 42 %, respectively. These reductions are attributable to the negative carbon emissions of wood biochar, as discussed in Section 4.1, demonstrating its significant decarbonization potential in concrete. However, the mechanical properties of concrete should also be considered to assess its practical potential. According to our previous study (Wang et al., 2025b), biochar powder reduced the 56-day compressive strength by 3 %, 6 %, and 13 % at 5 vol %, 10 vol %, and 20 vol % replacement levels, respectively. In contrast, it also provided benefits in terms of shrinkage: wood biochar reduced the 28-day shrinkage of concrete by ca. 282 μm/m at both 5 vol % and 10 vol %, while the 20 vol % sample showed a greater reduction of 382 μm/m.

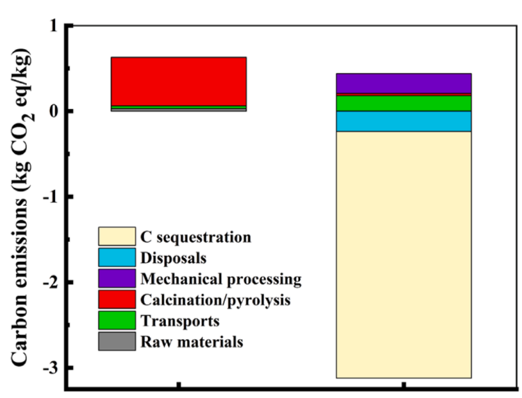

Fig. 6. Carbon emissions of cement and wood biochar (Mechanical processing includes chipping, drying, grinding for wood biochar; and grinding and crushing for cement).

Even more pronounced effects are observed when fruit biochar is used to replace sand in concrete. As shown in Fig. 8(c), replacing 30 vol.% of sand with fruit biochar reduces concrete emissions from 393 to 196 kg CO_2-eq per m^3. At 60 vol.% and 100 vol.% replacement, emissions decrease further to −1.59 and −264 kg CO_2-eq per m^3, respectively. Compared to cement replacement with wood biochar (5–20 vol %), which achieved 10.4–42 % reductions in carbon emissions, replacing sand with fruit biochar at 30 vol %, 60 vol %, and 100 vol % reduced the carbon emissions of concrete by 50.1 %, 101.4 %, and 167 %, respectively. These reductions exceeding 100 % indicate that the concrete mix achieves a net-negative carbon footprint, effectively becoming a carbon sink. Evidence can be found in the mix design presented in Table S6 in Appendix A. Using the volumetric replacement method results in a lower mass of biochar being used to replace the same volume of cement or sand in concrete, due to the lower density of biochar. For e.g., a 20 vol% cement replacement with wood biochar requires 37 kg of wood biochar to replace 100 kg of cement in concrete. This replacement reduces the amount of materials with high carbon emissions. At the same time, biochar itself has negative carbon emissions, hence, adding more biochar further decarbonizes concrete. For sand replacement, crushed sand has relatively low carbon emissions; therefore, removing it does not cause a significant reduction in emissions. However, crushed sand constitutes 29 wt.% of concrete, and replacement ratios can reach up to 100 vol.%. This allows a large amount of biochar (up to 197.74 kg) to be incorporated into concrete, thereby maximizing the decarbonization potential of the net-negative carbon emissions of fruit biochar. For the mechanical performance of concrete, as shown in our previous study (Wang et al., 2025c), fruit biochar reduced the total shrinkage of concrete; at 28 days, it decreased by 41 %, 55 %, and 65 % at 30 vol %, 60 vol %, and 100 vol % replacement levels, respectively. However, the 56-day compressive strength was also reduced by 7 %, 21 %, and 47.4 % at the same replacement levels.

4.3. Other environmental impacts

Despite the aforementioned advantages, biochar may exhibit higher impacts on certain environmental indicators compared to cement or crushed sand. As shown in Fig. 9, wood biochar causes higher levels of abiotic depletion, ozone layer depletion, human toxicity, terrestrial ecotoxicity, photochemical oxidation, and acidification than cement. These elevated impacts are largely attributable to the energy-intensive transport of wet biomass and the drying processes required prior to pyrolysis, as discussed in Section 4.1. A similar pattern is observed when comparing crushed sand and fruit biochar as fruit biochar exhibits higher potential for abiotic depletion and photochemical oxidation.

A similar pattern is observed for concrete. As shown in Fig. 10(a), replacing cement with wood biochar leads to a continuous increase in terrestrial ecotoxicity and photochemical oxidation. Consistent with previous research (Azzi et al., 2022), biochar can either increase or

Table 7
Carbon emissions of product (kg CO_2-eq per kg).

Materials	C sequestration	Disposal	Calcination/pyrolysis	Mechanical processing	Transport	Raw materials	Total
Cement	0	1.45E-4	0.56587	0.00262	0.03168	0.03019	0.63000
Wood biochar	−2.88236	−0.23800	0.02498	0.23078	0.17887	0.00265	−2.68308
Sand	0	7.25E-4	0	3.92E-5	0.00443	0.00865	0.01380
Fruit biochar	−3.16445	−0.23252	0.02458	0.06074	0.09707	0.00258	−3.21200

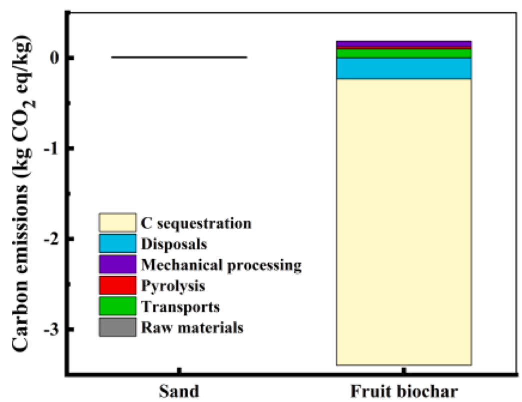

Fig. 7. Carbon emission of sand and wood biochar (Mechanical processing includes chipping, drying for fruit biochar; and crushing, sieving for crushed sand).

decrease potential impacts on resource use, human toxicity, and ecotoxicity depending on the specifics of its supply chain. However, replacing cement with wood biochar does not increase other pollutants relative to cement. Similarly, as shown in Fig. 10(b), replacing crushed sand with fruit biochar increased photochemical oxidation, while abiotic depletion decreased. As shown in Table S6 in Appendix A, replacements are based on volume to preserve cement hydration and maintain concrete strength (Wang et al., 2025b). For example, a 20 vol.% replacement requires only 37 kg of wood biochar to substitute 100 kg of cement. Similarly, fruit biochar is used to replace crushed sand on a volumetric basis due to its high porosity (Wang et al., 2025c), resulting in a lower mass of biochar needed. At higher replacement ratios, photochemical oxidation in concrete increases slightly, while other environmental impacts generally continue to decrease.

4.4. Sensitivity analysis

The properties and carbon emissions of biochar are strongly influenced by the moisture content of the feedstock. Even for the same type of feedstock, moisture content can vary depending on location and season.

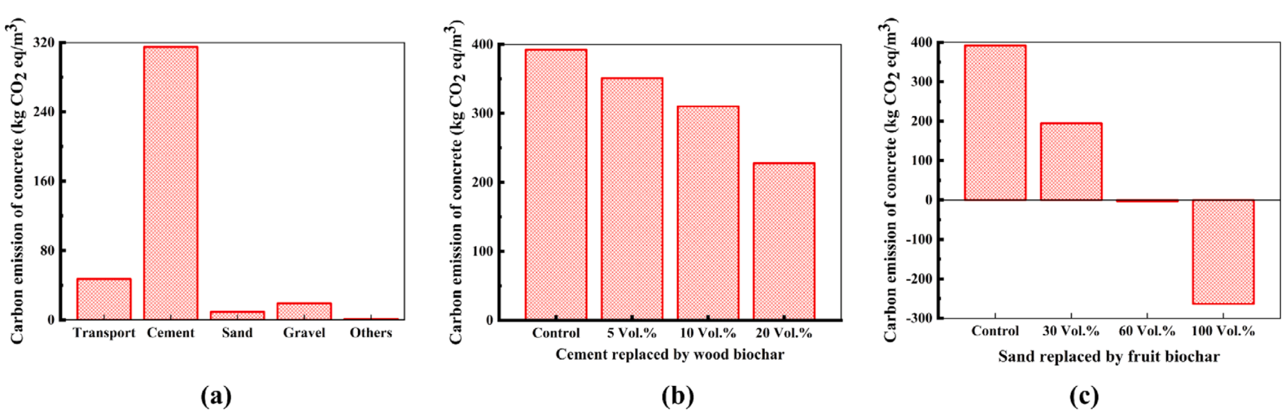

Fig. 8. Carbon emissions for concrete: (a) Plain concrete (others including mixing, water and superplasticizer); (b) Cement replacing ratio; (c) Sand replacing ratio.

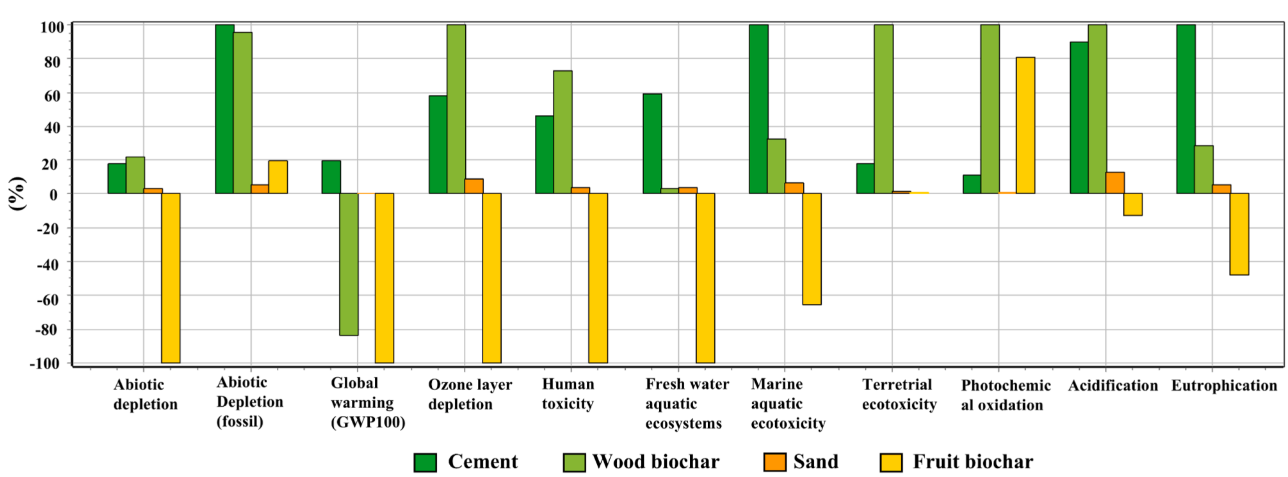

Fig. 9. Pollutants comparisons between cement, sand, wood biochar, and fruit biochar.

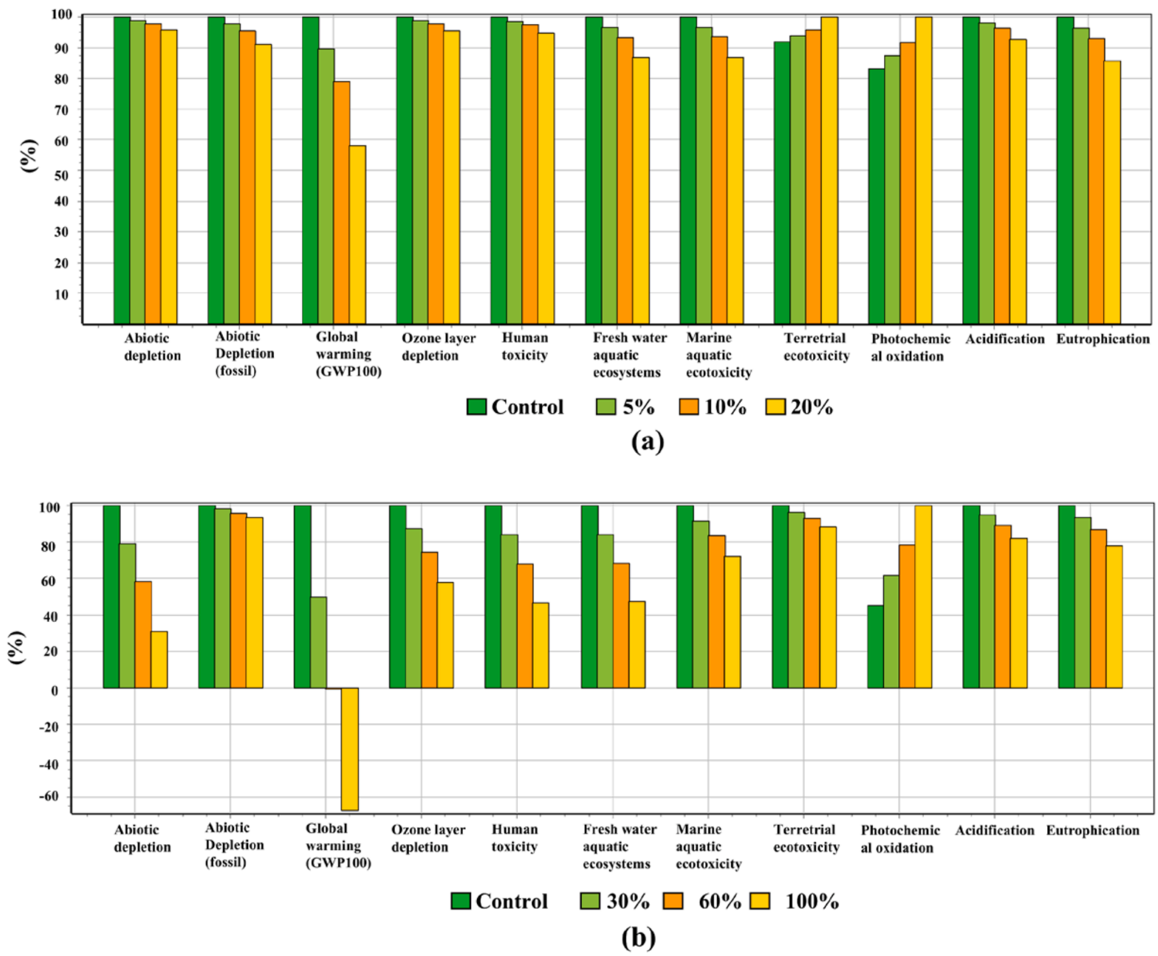

Fig. 10. Other environmental impacts of biochar concrete with a) cement and b) sand replacements.

Higher moisture levels increase the energy required for drying and elevate transportation demands. For e.g., biochar derived from Eucalyptus wood can exhibit moisture contents ranging from 20 % to 60 % when conditioned in the open air (Akossou et al., 2020), while sewage sludge and food waste can reach up to 85 wt.% (Li et al., 2022). Additionally, a review work (Manyà, 2012) have also indicated that high moisture contents (42–62 %) can improve charcoal yield under elevated pressures. Variations in moisture content, feedstock type, and pyrolysis temperature all influence biochar yield, which in turn affects its carbon emissions (Yaashikaa et al., 2020). By contrast, cement and crushed sand production are less sensitive to raw material moisture and benefit from mature production chains that ensure stable yields. Consequently, biomass moisture content and biochar yield rate are critical factors in determining the carbon footprint of biochar, and it is essential to assess their influence on life cycle emissions. Furthermore, while this study considers biochar supply across Sweden, it focuses on the ten largest cities as representative targets. The primary difference between these cities and other areas is the transportation distance for biochar. To account for variations in other regions, a distance-based intensity analysis was also conducted in this study.

4.4.1. Yield rate of biochar

As shown in Fig. 11(a), increasing the yield rate of wood biochar from 10 % to 30 % leads to a continuous decrease in carbon emissions, from −1.87 to −2.68 kg CO_2-eq per kg. This trend can be attributed to the supply and processing of raw materials: lower yield rates require more biomass, resulting in increased transport and additional energy for drying and chipping. The yield rate also has a clear impact on concrete carbon emissions; as illustrated in Fig. 11(b), a lower biochar yield rate reduces the potential carbon emissions of concrete when 20 vol.% of cement is replaced. Thus, yield rate plays a significant role in the decarbonization potential of concrete. A similar pattern is observed for fruit biochar. In Fig. 11(c), as the yield rate increases from 10 % to 30 %, the carbon emissions of fruit biochar decrease from −2.9 to −3.2 kg CO_2-eq per kg. For concrete with 60 vol.% of crushed sand replaced by fruit biochar (Fig. 11(d)), a lower yield rate results in higher carbon emissions, with negative emissions achieved only when the yield rate reaches 30 %.

Comparing Fig. 11(a) and 11(c), it is evident that yield rate has a larger impact on wood biochar than on fruit biochar. The carbon emissions of wood biochar decrease by 43.3 % as yield increases from 10 % to 30 %, whereas fruit biochar shows only a 10.7 % reduction. This difference arises from the moisture content assumptions: wood biochar is derived from logging residues with 50 % moisture, whereas fruit biochar is produced from olive pits with 10 % moisture. Higher moisture content increases transport requirements and energy consumption for drying, making moisture a key factor influencing the carbon emissions of biochar, as discussed in the following section.

4.4.2. Moisture content of biomass

As shown in Fig. 12(a), biomass moisture content has a pronounced effect on the carbon emissions of wood biochar. When moisture increased from 10 % to 50 %, carbon emissions rose from −2.93 to −2.68 kg CO_2-eq per kg. This variation also affected concrete carbon emissions; at 20 vol.% cement replacement, the emissions of concrete increased from 218.6 to 228 kg CO_2-eq per m^3 (Fig. 12(b)). A similar

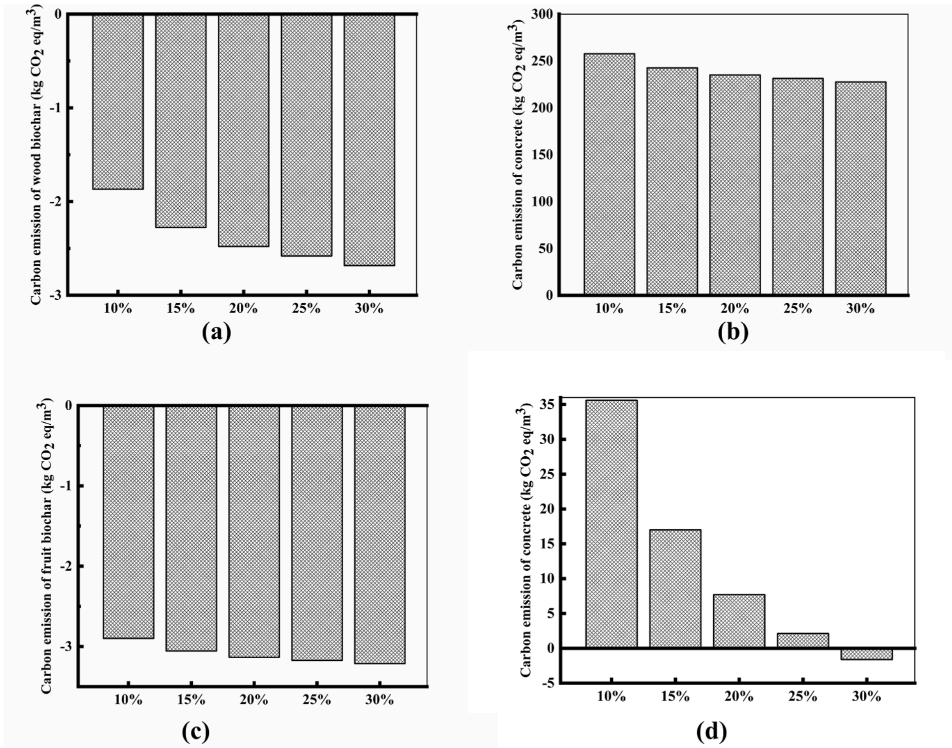

Fig. 11. The yield rate of biochar on the carbon emissions: (a) Emission of wood biochar; (b) Emission of wood biochar on concrete; (c) Emission of fruit biochar; (d) Emission of fruit biochar on concrete.

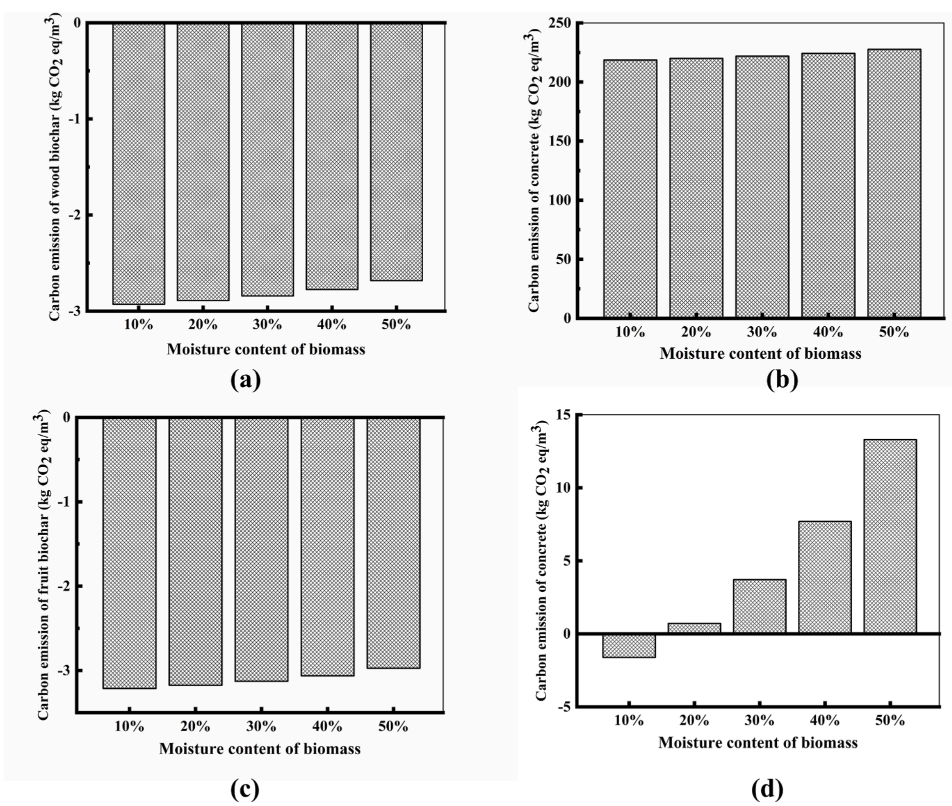

Fig. 12. The effect of biomass moisture on carbon emissions: (a) Emission of wood biochar; (b) Emission of concrete with cement replacement; (c) Emission of fruit biochar; (d) Emission of concrete with sand replacement.

trend is observed for fruit biochar. As the moisture content increased from 10 % to 50 % (Fig. 12(c)), carbon emissions of fruit biochar rose from −3.21 to −2.97 kg CO_2-eq per kg. For concrete with 60 vol.% sand replaced by fruit biochar, the carbon emissions increased from −1.61 to 13.3 kg CO_2-eq per m^3 as biomass moisture increased (Fig. 12(d)). Comparing wood and fruit biochar, the relative impact of moisture variation on carbon emissions is similar, with emissions increasing by 8.5 % for wood biochar and 7.5 % for fruit biochar as the moisture content rises from 10 % to 50 %. However, the effect on concrete is more pronounced for fruit biochar; increasing biomass moisture from 10 % to 50 % leads to a substantial rise (93 %) in concrete emissions, whereas the effect is smaller for wood biochar, with only a 4 % increase.

4.4.3. Transport distance of biochar

To assess the impact of biochar in concrete for other cities in Sweden, a transport distance intensity analysis was conducted. As shown in Fig. 13(a), increasing the biochar transport distance has a limited effect on carbon emissions of concrete. For wood biochar, a 50 % reduction in transport distance decreases emissions by only 0.159 kg CO_2-eq per m^3 of concrete, while a 300 % increase in distance raises emissions by 0.955 kg CO_2-eq per m^3. Similarly, for fruit biochar replacing 60 vol.% of sand in Fig. 13(b), a 50 % reduction in transport distance decreases concrete carbon emissions by 0.512 kg CO_2-eq per m^3, whereas a 300 % increase in distance raises emissions by 3.07 kg CO_2-eq per m^3.

5. Discussion

5.1. Modelling limitations

As demonstrated in previous experimental studies (Wang et al., 2025b; 2025c), biochar exhibits a range of properties, including porosity, water content, particle size distribution, and specific surface area. Variations in these properties influence both the replacement strategy and the mechanical performance of concrete. For e.g., wood biochar can reduce the mechanical strength of concrete after replacement (Wang et al., 2025a). Additionally, even when using the same feedstock, biomass moisture content varies, as highlighted in the intensity analysis, where higher moisture increases the carbon emissions of biochar and consequently reduces its decarbonization potential in concrete. The management of co-products and waste is also simplified in this study. Co-products from biochar pyrolysis are assumed to be used for heat generation, but emissions from this process are disregarded. In contrast, solid waste from cement and crushed sand is assumed to be landfilled in a worst-case scenario. In reality, such waste could be reused as filler in cementitious pastes (Coo and Pheeraphan, 2016), which is also not considered here.

For the analysis method, environmental impacts were assessed using the EU 25-country context and GWB 100 characterization factors. A relatively recent LCA study has shown that the choice of time horizon has no significant impact on either GWP or Global Temperature Potential (GTP) (Tisserant et al., 2022). However, in the Material Flow Analysis (MFA), biochar deployment spans several decades, with maintenance activities considered up to the year 2100. While temporal aspects are often neglected (Lausselet et al., 2021; Pauliuk et al., 2013), approaches combining time-dependent MFA, life cycle inventories, and impact assessment metrics could also be applied (Beloin-Saint-Pierre et al., 2020). Beyond moisture content, yield rate, and transport distance, reactor type and pyrolysis setup can also significantly affect biochar properties (Azzi et al., 2022), and associated environmental impacts. Different pyrolysis configurations can result in varying product distributions and compositions (Mohanty et al., 2024), which typically govern the trade-off between bioenergy generation and carbon sequestration (Azzi et al., 2019). These parameters warrant further investigation in future studies.

5.2. Results analysis

5.2.1. Biochar carbon emissions

For biochar, the primary emissions arise from mechanical processing (including drying, chipping, and grinding) and the transportation of biomass, all of which require substantial energy or fuel consumption. However, energy recovery from co-products of pyrolysis, such as bio-oil and syngas, can be utilized to generate heat or electricity (Woolf et al., 2010), significantly reducing the overall emissions associated with biochar production. Importantly, biochar is a carbon-rich material, and a substantial fraction of its carbon is stable, resistant to degradation, and remains sequestered from the atmosphere (Gamaralalage et al., 2025). This carbon sequestration enables both wood and fruit biochar to achieve negative carbon emissions, measured at −2.68 kg CO_2-eq per kg and −3.21 kg CO_2-eq per kg, respectively. The two biochar differ in their feedstock characteristics and processing assumptions. Wood biochar is produced from logging residues with a moisture content of 50 %, whereas fruit biochar is derived from olive pits with a moisture content of 10 %. The lower moisture content of fruit biochar reduces energy demands for drying and chipping as well as transportation requirements, resulting in greater benefits in terms of carbon emissions and other pollutants. Sensitivity analysis confirms that moisture content strongly influences biochar emissions. Increasing the biomass moisture content from 10 % to 50 % raises carbon emissions by 8.5 % for wood biochar and 7.5 % for fruit biochar, due to the additional energy or fuel required

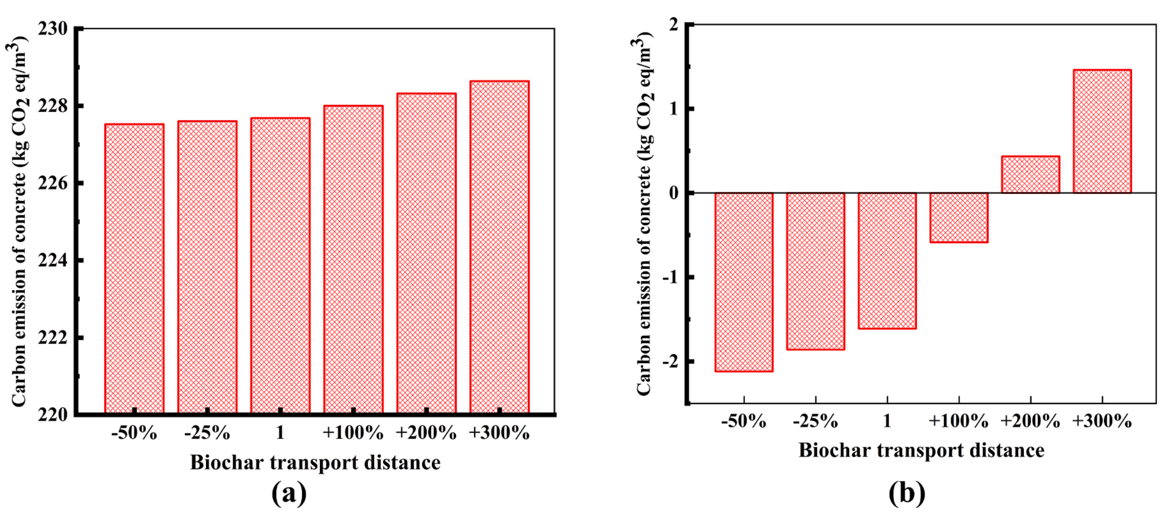

Fig. 13. The effect of transport of biochar on the carbon emissions: (a) Emission of wood biochar on concrete; (b) Emission of fruit biochar on concrete.

for drying and transport. Another important difference is that wood biochar is used to partially replace cement in concrete, and thus requires an additional grinding step to achieve a particle size distribution comparable to cement. This additional processing increases energy consumption, reducing the decarbonization potential of wood biochar in concrete.

Beyond moisture content, pyrolysis temperature, feedstock type, and reactor design also affect biochar yield (Himanshu et al., 2025). Sensitivity analysis indicates that higher yield rates substantially enhance the carbon emission reduction potential. When the yield rate increases from 10 % to 30 %, the carbon emissions of fruit and wood biochar decrease from −2.9 to −3.21 kg CO_2-eq per kg and from −1.87 to −2.68 kg CO_2-eq per kg, corresponding to reductions of 10.7 % and 43.3 %, respectively. This improvement occurs because higher yield rates reduce the amount of biomass required, thereby lowering energy inputs for drying, chipping, pyrolysis, and transportation. The minimal influence of transport distance relative to parameters, such as yield rate suggests that biochar production could be economically centralized without severely compromising its carbon reduction potential, an advantage for regional supply chains. Consequently, optimizing biomass moisture content and yield rate is critical for maximizing the decarbonization potential of biochar in industrial applications.

5.2.2. Comparison of cement and sand with biochar

Cement is a highly energy-intensive material, requiring the raw mix to be heated to approximately 1450 °C (Cement Manufacture, 2025). The calcination process alone accounts for 0.5659 kg CO_2-eq per kg of cement, contributing to a total carbon footprint of 0.63 kg CO_2-eq per kg. Emissions from calcination can be divided into two components: (i) CO_2 released from the chemical decomposition of limestone ($CaCO_3 \rightarrow CaO + CO_2$) (Cement Manufacture, 2025); and (ii) emissions from fuel combustion in the kiln. According to the Environmental Defense Fund (Fund, 2023), 85 % of total carbon emissions from cement production are attributable to calcination, of which 51 % arise from limestone decomposition and 34 % from fossil fuel combustion. In contrast, wood biochar requires more energy for biomass transport and drying than cement. This is because the biomass must be collected from forests or factories, and its high moisture content must be lowered by drying. Cement factories, by contrast, are typically located near raw material sources such as limestone and shale. Nevertheless, pyrolysis of biomass consumes relatively little energy, operating at ca. 350–900 °C (Mohanty et al., 2024), which is far lower than the 1450 °C required for cement calcination.

Moreover, biochar production requires less energy than cement production due to the recycling of co-products, particularly syngas. During pyrolysis, biomass thermo-chemically converts into bio-oil and syngas, which can be used to generate heat or electricity, thereby reducing the carbon footprint of the biochar system (Lehmann and Joseph, 2015). Most importantly, carbon sequestration renders wood biochar a negative-carbon-emission material. For instance, Li and Tasnady (2023) reported that the stable carbon fraction of biochar ranges from 48.44 to 95.17 % for pyrolysis temperatures between 300 and 600 °C. Gamaralalage et al. (2025) found that biochar derived from food biomass contains 88 % stable carbon, resulting in long-term carbon storage of approximately 1.7 t CO_2-eq per ton of biochar, with net emission reductions of 1.15–1.20 t CO_2-eq per ton. Another study (Lehmann et al., 2021) estimated that large-scale biochar application could reduce global carbon emissions by 3.4–6.3 Pg CO_2-eq, roughly half of which corresponds to CO_2 removal. These findings highlight the significant potential of biochar to offset carbon emissions associated with cement and sand in concrete production.

Crushed sand, in comparison, exhibits very low carbon emissions, with approximately 0.0138 kg CO_2-eq per kg due to its simple production process, which involves only quarrying, crushing, and sieving. Other studies report similarly low values for sand, ranging from 0.0022 to 0.0330 kg CO_2-eq per kg (Ma et al., 2025; Marinković et al., 2010). Although fruit biochar has higher emissions than crushed sand during production, primarily due to additional transport and energy required for drying, it still achieves negative emissions through energy recovery from co-products and carbon sequestration. Given the large volume of sand used in concrete and the potential for high replacement ratios, fruit biochar presents significant opportunities for decarbonizing the concrete industry.

5.2.3. Decarbonization in concrete

Concrete exhibits very high carbon emissions, approximately 393 kg CO_2-eq per m^3, with cement accounting for 80 % of these emissions. Gravel occupies the largest volume in concrete (Wang et al., 2024b), followed by sand, yet their carbon contributions are minimal due to the simplicity of their production processes. Replacing cement with wood biochar can effectively reduce the carbon footprint of concrete. Substituting 5 vol %, 10 vol %, and 20 vol % of cement by the same volume of wood biochar reduces concrete carbon emissions by 10.4 %, 21 %, and 42 %, respectively. This reduction is partly due to the lower energy requirements of wood waste pyrolysis compared to cement calcination, but it is primarily driven by the carbon sequestration of stable carbon and the energy recovery from biochar co-products.

Similarly, fruit biochar reduces concrete carbon emissions by 50.1 %, 101.4 %, and 167 % at 30 %, 60 vol %, and 100 vol % replacement of sand, respectively. These reductions exceeding 100 % indicate that the concrete mix achieves a net-negative carbon footprint, effectively becoming a carbon sink. The larger reductions compared to cement replacement by wood biochar are primarily due to the greater volume of sand in the concrete mix and the higher replacement ratios. Sand occupies more volume than cement in the mix, and the replacement ratios for fruit biochar are much higher (30–100 vol %) than those for wood biochar (5–20 vol %). Consequently, more fruit biochar can be incorporated into concrete, resulting in greater decarbonization. These trends are also reflected in the intensity analyses for moisture content, yield rate, and transport distance. Since fruit biochar is used in larger quantities, its carbon emissions are more sensitive to changes in these factors.

However, carbon emissions should not be the sole consideration when using biochar in concrete as high levels of biochar replacement may compromise the mechanical strength of the material. Our previous study (Wang et al., 2025b) found that replacing cement with wood biochar benefited the shrinkage of concrete at 28 days. Specifically, the 5 vol % and 10 vol % samples showed approximately 282 μm/m less shrinkage than the control, while the 20 vol % sample displayed an even greater reduction, with a total shrinkage decrease of 382 μm/m. Nevertheless, biochar reduced 56-day compressive strength by 3 %, 6 %, and 13 % at 5 vol%, 10 vol%, and 20 vol% replacement, respectively. Similarly, as shown in our previous study (Wang et al., 2025c), when fruit biochar replaced sand, the total shrinkage at 28 days was reduced by 41 %, 55 %, and 65 % at 30 vol %, 60 vol %, and 100 vol % replacement, respectively. However, compressive strength at 56 days was reduced by 7 %, 21 %, and 47.4 % at the same replacement levels. Therefore, although 20 vol % cement replacement with wood biochar reduced carbon emissions by 42 %, and 100 vol % sand replacement with fruit biochar achieved a net-negative carbon footprint (167 % reductions), respectively, they caused 13 % and 47.4 % reductions in compressive strength. These findings suggest that in practical applications, while the environmental benefits are clear, the application of these biochar-concrete mixes must be tailored to specific structural and non-structural applications where the reduction in mechanical strength is acceptable, highlighting the importance of a performance-based design approach.

5.2.4. Other environmental impacts

Biochar holds significant potential for decarbonization in concrete due to its energy recovery and storage of stable carbon. However, this does not imply benefits for all environmental indicators. Wood biochar exhibits higher impacts on abiotic depletion, ozone layer depletion, human toxicity, terrestrial ecotoxicity, photochemical oxidation, and acidification, primarily due to the energy-intensive transport and drying processes. Additionally, the chemical composition and decomposition of biomass during pyrolysis contribute to these impacts. For e.g., cellulose, hemicellulose, and lignin can decompose into CO, H_2, and CH_4, while nitrogen- and sulphur-containing compounds in the feedstock may form NO_x, NH_3, and SO_x, which can lead to acidification and nutrient enrichment if released (Ścierski, 2021; Xu et al., 2021). Fruit biochar similarly exhibits elevated impacts, particularly for photochemical oxidation and abiotic depletion.

Nonetheless, these effects are partially mitigated when biochar is used to replace cement or sand in concrete. For instance, replacing cement with wood biochar primarily increases terrestrial ecotoxicity and photochemical oxidation, while other pollutant categories remain largely unaffected. Similarly, fruit biochar does not significantly increase abiotic depletion in concrete. This can be attributed to the highly porous structure of biochar (Wang et al., 2025a). Environmental impacts are calculated on a mass basis (kg), but because biochar has lower density than cement or sand, replacing 1 kg of cement or sand requires <1 kg of biochar by volume. Consequently, the higher pollutant intensity of biochar per kg is partially offset in concrete, and in some cases, overall impacts may even decrease. The observed increases in terrestrial ecotoxicity and photochemical oxidation in concrete after replacement arise because these specific impacts are very high in biochar, and the reduced mass of biochar is insufficient to fully compensate. Nevertheless, when considering the use of biochar in concrete, it is essential to account for these other environmental impacts to obtain a comprehensive assessment of its sustainability.

6. Conclusions

The analysis in this study compared the environmental performance of biochar relative to cement and sand and explored its potential applications in the concrete industry, including sensitivity analyses on biomass moisture content, biochar yield rate, and transport distance. The key findings are summarized as follows:

1. Cement exhibits very high carbon emissions, with the calcination process accounting for approximately 80 % of the total, due to both limestone decomposition and fuel combustion. However, energy recovery from co-products and carbon sequestration enables wood biochar to achieve negative carbon emissions, which reduced carbon emissions of concrete by 42 % at 20 vol.% replacement of cement.
2. Sand has relatively low carbon emissions because of the simplicity of its production processes. Although fruit biochar requires additional energy for biomass transport and drying, its energy recovery and carbon sequestration reduced carbon emissions of concrete by 167 % at 100 vol.% replacement of sand, achieving a net-negative carbon footprint and confirming its role as a carbon sink.
3. Replacing sand with fruit biochar is more effective in reducing concrete carbon emissions than replacing cement with wood biochar, due to the larger volume of sand and higher replacement ratios. However, the compromised mechanical strength of concrete has limited the replacement ratio.
4. Variations in biomass moisture and biochar yield rate have a significant effect on the carbon emissions of biochar and its decarbonization potential in concrete. In contrast, transport distance has a limited effect.
5. Biochar may generate higher levels of other pollutants compared to cement or sand. However, these impacts can be partially mitigated in concrete through volumetric replacement.

In summary, the results of this current study make a significant contribution to existing knowledge by demonstrating the potential decarbonization pathways for the concrete industry through the incorporation of biochar, and by providing a robust analytical framework to support future LCA studies of biochar-concrete across different countries and regions.

Future research

Given the nature of LCA studies, results from different regions are expected to vary due to differences in inventory data for raw materials that reflect local market conditions. These variations may include differences in material composition, transport distances, and regional energy mixes and use during production. However, studies in other countries and regions could apply the analytical framework presented in this current research to evaluate the potential of biochar to decarbonize concrete, develop strategies tailored to local market conditions, and enable meaningful comparison of results across contexts. Future research could focus on the decarbonization of concrete using biochar in other countries to provide strategies for local markets. The intensity analysis could include more factors related to pyrolysis conditions, such as temperature and retention time. Moreover, the decarbonization effect of biochar on the whole structure is worthy of deeper investigation, as it could provide overall benefits for the industry.

CRediT authorship contribution statement

Dong Wang: Writing – review & editing, Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation. **Thao Thi Phuong Bui:** Writing – review & editing, Writing – original draft, Formal analysis. **Benjamin Reinke:** Writing – review & editing. **Venkata K.K. Upadhyayula:** Writing – review & editing, Visualization, Validation. **Ágoston Restás:** Writing – review & editing. **Suzanne Wilkinson:** Writing – review & editing. **Imelda Saran Piri:** Writing – review & editing, Visualization, Validation. **Sabrina Karim:** Writing – review & editing. **Oisik Das:** Writing – review & editing, Visualization, Validation, Supervision, Resources, Project administration, Methodology, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare no known competing interest.

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.rcradv.2026.200337.

Data availability

Data will be made available on request.

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