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Assessing GHG Emission Reduction in Biomass-Derived Biochar Production via Slow Pyrolysis: A Cradle-to-gate LCA Approach

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ABSTRACT

Biochar, a potent negative emission technology (NET), sequesters carbon through biomass thermochemical conversion. Despite greenhouse gas (GHG) storage in biochar being well-understood, analyzing GHG emissions during production is challenging due to diverse carbonization methods. We utilized life cycle assessment (LCA) to address this complexity. In a literature survey, we analyzed contemporary biochar LCA studies, extracting GHG data for emission factor (EF) determination. A scenario-specific case study assesses GHG emission reduction in crop residue-derived biochar. Key factors influencing biochar emissions include feedstock type, surplus energy use, and carbonization scale. Biochar contains EFs ranging from −1.10 to 0.68 ton-CO_2e/ton-biochar. The case study demonstrates that biochar total abatements span from −625,775 to −215,712 ton-CO_2e/year (derived from 840,000 ton-crop residue/year). While overall emission reduction relies on GHG storage, carbonization emissions significantly contribute to total emission diversity. The study proposes a meticulous LCA approach for realistic biochar production, crucial for assessing carbon crediting or offsetting schemes.

1. Introduction

The imperative integration of negative emission technologies (NET) to address imminent climate change aligns with global priorities under the Paris Agreement (Lefebvre et al., 2021; Minx et al., 2018). Amid diverse NETs like afforestation and soil carbon sequestration, biochar emerges as indispensable, trapping atmospheric CO_2 and contributing significantly to NetZero emission goals (Karan et al., 2023a; Zhu et al., 2022). Renowned for carbon sequestration, biochar's climate change mitigation stems from its dense recalcitrant carbon content (Ericsson et al., 2017; Huang et al., 2022). For instance, Karan et al. (2023b) theoretical estimations propose that crop residue-derived biochar's maximum sequestration potential could be up to 3.7 GT-CO_2e/year.

One major scheme for quantifying biochar's CO_2 reduction potential is carbon crediting, an emerging strategic instrument for managing global greenhouse gas (GHG) emissions (Woolf et al., 2021) under Kyoto Protocol thresholds (Gupta, 2011; Whitman et al., 2010). This scheme requires precise accounting for GHG storage and emission. While biochar's GHG storage is well understood (Azzi et al., 2019; Uusitalo & Leino, 2019), the value adopted by the Intergovernmental Panel on Climate Change (IPCC) possibly informs storage calculations (IPCC, 2019). However, the systematic analysis of emissions during carbonization lags, owing to influential parameter complexity (Cheng et al., 2020), resulting in a lack of an emission factor reflecting production reality. For example, Japan's J-credits scheme (AG-004 method) considers both storage and emissions (Hiradate et al., 2023; J-credit, 2023), but emissions exclude those occurring during biomass open carbonization. Conversely, the National GHG Inventory Report of Japan (GIO, 2023) uses IPCC default values, i.e., the emission factor of open carbonization systems adopted from a study by Cornelissen et al. (2016), to quantify GHG emissions from biochar production, focusing solely on a single type of biochar production system.

However, the GHG emissions from biochar production vary widely, depending on the diversity of the influential factors. In accordance with the carbonization temperature and heating rate, the charring process is mainly categorized into fast pyrolysis (Kuppens et al., 2015), slow pyrolysis (Kumar et al., 2020), torrefaction (Brown, 2009), and gasification (Yaashikaa et al., 2020) where slow pyrolysis produces more biochar (Homagain et al., 2015) than other methods. Notably, about 35% of biochar is obtained from biomass slow pyrolysis (Homagain et al., 2015), while less than 10% of biochar output originates from gasification (Kambo and Dutta, 2015). In addition, mild reaction temperature (300–700°C) with a low heating rate (5–7°C/ min) (Kumar et al., 2020), plays a pivotal role in efficient energy consumption of biochar

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production from slow pyrolysis systems. The feedstock type (Rittl et al., 2018), kiln type (Cornelissen et al., 2016), temperature and other operational parameters influence the physicochemical characteristics of resultant char (Liu et al., 2023). Apparently, operational conditions govern diversity among different slow pyrolysis systems.

However, there is a knowledge gap in systematically clarifying the influence of different factors on CO_2 emissions diversity within biochar production systems (Puettmann et al., 2020), especially at the life cycle stage (Sahoo et al., 2021). Life cycle assessment (LCA) is a useful technique for disclosing GHG emissions from biochar production systems (Opatokun et al., 2017; Papageorgiou et al., 2021), encompassing processes like transportation, pre-treatment, and carbonization (Gu et al., 2018; Lugato et al., 2013). Previous experiments have determined that the remarkability of influencing parameters, such as biomass carbonization under high temperatures, would substantially reduce emissions along with operational conditions (Brassard et al., 2018). Additionally, studies by Mohammadi et al. (2017) identified the comparative GHG emission reduction potentials of open and closed carbonization systems, where closed carbonization achieved maximum carbon abatement.

Despite numerous LCA studies focusing mainly on specific locations and production methods (Dutta & Raghavan, 2014; Oldfield et al., 2018), the relationship between different biochar production systems remains unavailable. Challenges persist in providing a holistic view of emissions during biochar production based on individual studies. However, most studies only discuss differences between production methods in terms of total life-cycle CO_2 (LC-CO_2) (Cheng et al., 2020; Marazza et al., 2019; Bergman et al., 2022) and fail to provide substantial background information to realize process-specific differences. Consequently, quantifying the individual impacts of leading parameters, such as feedstock type and carbonization method, that govern differences in LC-CO_2 becomes challenging.

Detailed process-based emission data are essential to understand the real-time contribution of each biochar LCA step for precise decision-making and policy development to accomplish the expected environmental benefits from biochar (Thornley et al., 2015). This study aims to make a precise assessment of the GHG reduction effect of biochar technology by (1) developing emission factors (EFs) considering confined system boundaries reflecting cradle-to-gate biochar production to understand precise process-based carbon abatement from biochar production and determine the influence of key parameters on final emission. Subsequently, (2) a scenario-based study of biochar production and application with crop residues in Japan was conducted, realizing the effects of these process-based EFs on total GHG emission reduction.

2. Materials and methods

Initially, comprehensive EFs were determined through the extraction of process-based GHG data from a systematic literature survey of biochar LCA papers obtained from the Web of Science database. Subsequently, a scenario-based analysis was conducted to assess the impact of these EFs on the overall carbon abatement of biochar derived from crop residues in Japan.

2.1. Development of emission factors

2.1.1. Process-based GHG emission data by systematic literature review

The literature survey was executed in the Web of Science database, focusing on peer-reviewed articles spanning from 2005 to 2022, employing keywords such as ‘biochar’, ‘lifecycle assessment’, and ‘greenhouse gas’. A total of 150 papers were initially considered, as illustrated in Fig. S1. During the first screening phase, it was determined whether the primary focus of each paper centered on biochar LCA. Consequently, 60 papers were identified as not addressing biochar LCA and were excluded from further analysis. The remaining 90 papers underwent scrutiny to ascertain if the analysis primarily centered on the slow pyrolysis process and whether biochar was the principal output of the production process. This led to the exclusion of 50 papers that did not consider the slow pyrolysis process, and biochar was not identified as the primary output.

Notably, the slow pyrolysis technique, recognized for producing a more substantial percentage of biochar than bio-oil and syngas (Brown et al., 2011; Nsamba et al., 2015), guided the exclusion of papers employing alternative procedures. Subsequently, 22 papers lacking sufficient greenhouse gas emission data for each process, necessary for acquiring LC-CO_2 of biochar production systems, were excluded to ensure the analytical procedure’s robustness and mitigate disputes.

A total of 18 papers, featuring biochar as the primary product from the slow pyrolysis process, were ultimately chosen to acquire process-based GHG emission data, as detailed in Table 1. The selected papers were primarily categorized based on the carbonization method, encompassing open carbonization systems (2 papers) and closed carbonization systems (16 papers). Biomass sources were also considered, including crop residues (6 papers), wood residues (5 papers), sludge (2 papers), green waste (1 paper), and combinations of biomass sources (4 papers). Relevant datasets were extracted from these chosen papers to obtain comprehensive CO_2 emissions, encompassing LC-CO_2, and its breakdown by process.

A standardized classification method for data was established as follows. Given the primary focus of this paper on GHG emissions up to the biochar production process, estimations were conducted within the system boundaries (ISO 14040, 2006; ISO 14044, 2006). These boundaries encompassed emissions related to transportation, carbonization plant construction, pre-treatment of biomasses before carbonization, energy consumption during carbonization, and direct emissions (CH_4). Furthermore, the GHG reduction effect due to surplus energy utilization (for bio-oil and syngas) in both open and closed systems was assessed (Fig. 1). On the other hand, this study did not incorporate considerations for biogenic CO_2 related to land use. Furthermore, collection of various feedstocks was also excluded because they may be regarded as co-products of the previous cultivation or production system and their GHG emissions can vary significantly depending on the allocation method adopted.

All estimates were conducted based on the functional unit (FU) of one ton of biochar production. Sixty-two process-based LC-CO_2 data (Table 1) were acquired for subsequent analysis. It was verified that the total CO_2 from all considered processes equaled the original LC-CO_2 in each paper. Given the variation in functional units used in biochar LCA papers, such as 1 ton of biochar or 1 ton of feedstock (Sparrevik et al., 2013; Brassard et al., 2021), process-based emissions were recalculated based on available data to determine emissions for a standardized FU (1 ton of biochar).

2.1.2. Estimation of emission factors

To comprehensively account for variations in production methods and feedstock types, datasets (as illustrated in Table 1) were systematically classified into distinct groups, and corresponding EFs were computed for each group. A total of 24 groups were identified, taking into consideration the availability of data. Initially, datasets were classified based on the carbonization method, distinguishing between open (Smebye et al., 2017) and closed carbonization systems (Hammond et al., 2011). Biomass carbonization conducted using simple stoves or kilns was categorized as open carbonization systems, whereas carbonization at enclosed pyrolytic plants was classified as closed carbonization systems (Haeldermans et al., 2020; Mašek et al., 2018).

Open carbonization was subdivided into two groups based on the presence or absence of surplus energy utilization. For closed carbonization, the majority of datasets indicated the presence of a surplus energy utilization step. Consequently, further classifications were executed based on the feedstock type (wood residue, crop residue, green waste, and sludge). However, categorization according to an unspecified feedstock type was implemented under the assumption of a situation

Table 1
LC-CO_2 datasets extracted from selected literature.

No.	Selected Paper	Carbonization Method (O, C)	Scale of operation (S/M, L)	Feedstock capacity of the facility (ton-feedstock/year)	Feedstock category (A, W, G, S)	Feedstock	Carbonization temperature ($^{\circ}$C)	Surplus energy utilization	LC-CO_2 (ton-CO_2e/ton-biochar)
1	Smebye et al. (2017)	O	Unspecified	3	A, W	Woody shrub or crop residue	350	Not use	1.04
		O	Unspecified	3	A, W		350	Not use	0.82
		O	Unspecified	3	A, W		500	Not use	0.67
		O	Unspecified	3	A, W		500	Use	-0.19
2	Mohammadi et al. (2017)	O	Unspecified	3	A	Rice husk	350	Use	-0.08
		O	Unspecified	29	A		450	Not use	0.49
		C	S/M	4,670	A		650	Use	-0.22
3	Brassard et al. (2018)	C	S/M	NA	G	Switchgrass	459	Use	-1.41
		C	S/M	NA	G		591	Use	-1.33
4	Muñoz et al. (2017)	C	S/M	NA	A	Oat husk	300	Use	-0.16
		C	S/M	NA	A	Oat husk	400	Use	-0.17
		C	S/M	NA	A	Oat husk	500	Use	-0.17
		C	S/M	NA	W	Pine bark	300	Use	-0.17
		C	S/M	NA	W	Pine bark	400	Use	-0.17
		C	S/M	NA	W	Pine bark	500	Use	-0.17
5	Ibarrola et al. (2012)	C	S/M	NA	W	Wood waste	500	Use	-0.98
		C	S/M	NA	S	Sewage sludge	500	Use	-0.74
		C	S/M	NA	G	Green waste	500	Use	-1.01
6	Yang et al. (2021)	C	S/M	NA	A	Crop residue	NA	Use	-0.28
7	Leppäkoski et al. (2021)	C	S/M	NA	W	Willow woodchips	NA	Use	-0.39
8	Gievers et al. (2021)	C	S/M	NA	S	Sewage sludge	550	Use	1.10
9	Rosas et al. (2015)	C	S/M	213	A	Ripped vines' wood	550	Not use	0.12
10	Kameyama et al. (2010)	C	S/M	700	A	Sugarcane bagasse	600	Not use	0.57
11	Harsono et al. (2013)	C	S/M	4,800	A	Empty fruit bunch	400	Not use	0.22
12	Mohammadi et al. (2019)	C	S/M	4,700	S	Paper mill sludge	NA	Not use	0.10
13	Hammond et al. (2011)	C	S/M	2,000	A	Barley Straw	400	Use	-0.32
		C	S/M	2,000	A	OSR Straw	400	Use	-0.32
		C	S/M	2,000	A	Wheat Straw	400	Use	-0.32
		C	S/M	2,000	W	SaR	400	Use	-0.49
		C	S/M	2,000	W	FRC-UK	400	Use	-0.48
		C	S/M	2,000	W	SRW	400	Use	-0.48
		C	S/M	2,000	W	SRC	400	Use	-0.48
		C	S/M	2,000	W	SRF	400	Use	-0.48
		C	S/M	2,000	G	Miscanthus	400	Use	-0.48
		C	S/M	2,000	W	FRC-Canada	400	Use	-0.48
		C	L	20,000	A	Barley Straw	400	Use	-0.74
		C	L	20,000	A	OSR Straw	400	Use	-0.74
		C	L	20,000	A	Wheat Straw	400	Use	-0.74
		C	L	20,000	W	SaR	400	Use	-0.94
		C	L	20,000	W	FRC-UK	400	Use	-0.94
		C	L	20,000	W	SRW	400	Use	-0.94
		C	L	20,000	W	SRC	400	Use	-0.94
		C	L	20,000	W	SRF	400	Use	-0.94
		C	L	20,000	G	Miscanthus	400	Use	-0.94
		C	L	20,000	W	FRC-Canada	400	Use	-0.94
		C	L	100,000	A	Barley Straw	400	Use	-0.91
		C	L	100,000	A	OSR Straw	400	Use	-0.91
		C	L	100,000	A	Wheat Straw	400	Use	-0.91
		C	L	100,000	W	SaR	400	Use	-1.16
		C	L	100,000	W	FRC-UK	400	Use	-1.16
		C	L	100,000	W	SRW	400	Use	-1.16
		C	L	100,000	W	SRC	400	Use	-1.16
		C	L	100,000	W	SRF	400	Use	-1.16
		C	L	100,000	G	Miscanthus	400	Use	-1.16
		C	L	100,000	W	FRC-Canada	400	Use	-1.17
14	Hamedani et al. (2019)	C	L	19,200	W	Willow woodchips	500	Use	-0.12
15	Peters et al. (2015)	C	L	38,500	W	Poplar	450	Use	-1.05
16	Roberts et al. (2010)	C	L	70,000	A	Corn stover	450	Use	-1.03
		C	L	70,000	G	Switchgrass	450	Use	-1.28
		C	L	70,000	G	Yard waste	450	Use	-0.73
17	Ji et al. (2018)	C	L	30,000	A	Corn & wheat straw	500	Use	-0.71
18	Field et al. (2013)	C	L	NA	W	Pine bark	500	Use	-1.72

Note: O, Open carbonization; C, Closed carbonization.
S/M, Small or Medium Scale (less than 10,000 ton-feedstock/year); L, Large scale (more than10,000 ton-feedstock/year).
NA, Not available.
A, Agricultural residues; G, Green waste; S, sludge; W, Wood residues.
FRC, Forestry Residue Chips; OSR, oilseed rape; SaR, Sawmill Residues; SRC, Short Rotation Coppice; SRF, Short Rotation Forestry; SRW, Small Round Wood.

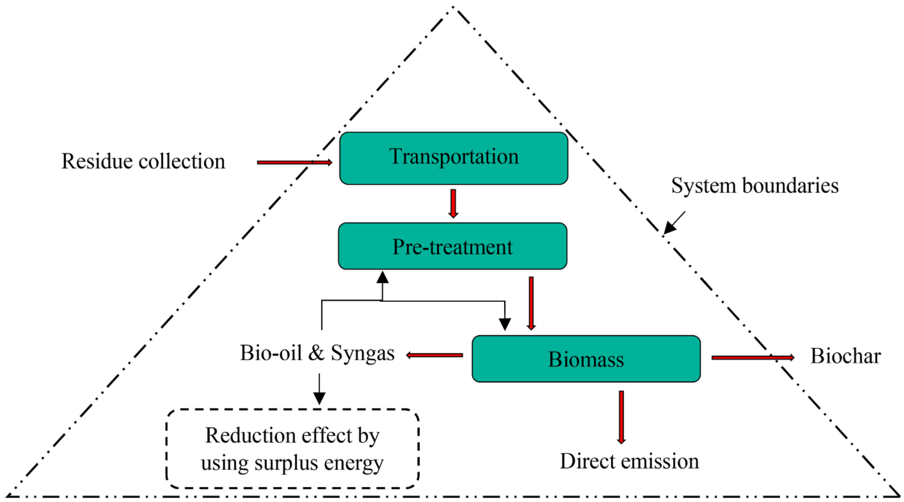

Fig. 1. System boundaries for GHG emission estimations.
Functional unit: 1 ton of biochar.

where the feedstock material is unknown. Additionally, for feedstocks with sufficient data (wood and crop residues), groupings were extended to include scale levels, while feedstocks lacking adequate data were retained as scale unspecified. The categorization of carbonization systems was contingent on the annual feedstock utilization in the pyrolytic plant. Systems processing less than 10,000 tons of feedstock annually were classified as small or medium-scale systems (S/M), while those handling more than 10,000 tons were designated as large-scale systems (L). Scale determination relied on the data availability from the selected literature (Table 1). Subsequently, assuming systems that might not facilitate surplus energy trapping and utilization, similar classification steps were followed, as mentioned above, to further refine the grouping of closed carbonization systems without surplus energy usage.

Finally, EFs were computed for various combinations of influential parameters in biochar production, encompassing carbonization method, carbonization scale, surplus energy utilization, and feedstock type. The estimation considered the most probable combinations of these parameters. The average value for each process was calculated based on the previously mentioned EF group categorization. The groups were assigned numbers (n) ranging from 1 to 24, (Table 3) reflecting the various plausible parameter combinations. Subsequently, EFs for each group were determined by summing the average values of each process (Eq. 1).

$$EF = E_{tra} + E_{pre} + E_{con} + E_{crb} + E_{dem} + (-S_{enr}), \tag{1}$$

where EF is the emission of biochar production (ton-CO_2e/ton-biochar), E_{tra} is the CO_2 emission of transportation (ton-CO_2e/ton-biochar), E_{pre} is the CO_2 emission of biomass pre-treatment process (ton-CO_2e/ton-biochar), E_{con} is the CO_2 emission of carbonization plant construction (ton-CO_2e/ton-biochar), E_{crb} is the CO_2 emission of fuel consumption in carbonization (ton-CO_2e/ton-biochar), E_{dem} is the direct emission in carbonization (ton-CO_2e/ton-biochar), and S_{enr} is the reduction effect of surplus energy utilization (ton-CO_2e/ton-biochar). Average values of the process-based emissions are used in the calculation, and the distribution ranges are obtained from uncertainty analysis (refer to Section 2.2.3 "Uncertainty analysis" for detailed information).

2.2. GHG emission reduction effects for biochar derived from crop residues

2.2.1. Scenarios of biochar production

A scenario-based analysis was conducted for biochar derived from crop residues in Japan to assess the impact of EFs on the overall carbon abatement of biochar. Given the Japanese government's intention to increase the annual crop residue utilization rate from 31% in 2019 to 45% by 2030 (MAFF, 2023), the production of biochar from crop residues emerges as an advantageous option to realize additional benefits. The assumed utilization of the increased residue (14% more in 2030) for biochar production formed the basis for estimating GHG reduction, encompassing both emissions and storage. The GHG emission of biochar production is predominantly influenced by parameters such as the carbonization system, scale, and surplus energy utilization, as revealed in the EF development. Consequently, six scenarios for biochar production were developed, considering various parameter combinations (Table S2). Additionally, a base case was included, representing a scenario where the production method is not explicitly defined.

2.2.2. Estimation of GHG emission

The total reduction in GHG emissions was calculated for each scenario in accordance with Eq. (2).

$$Total\ Emission_i = Emission_i - Storage_i, \tag{2}$$

where $Total\ Emission_i$ is the total GHG emission reduction of scenario i, $Emission_i$ is the annual emission (ton-CO_2e/year) of scenario i, and $Storage_i$ is the GHG storage of biochar (ton-CO_2e/year) of scenario i.

Subsequently, emission of annual biochar production process depending on the scenario was determined by Eq. (3)

$$Emission_i = Mb_i \cdot EF_i, \tag{3}$$

where Mb_i is the estimated annual biochar production (ton-biochar/year) of scenario i, and EF_i is the mean value of the emission factor (ton-CO_2e/ton-biochar) of scenario i.

For scenarios 1–6, emission factors (EFs) were derived from the estimated EF dataset for crop residue-derived biochar production

(Table S2). In the baseline case (Scenario 0), a default value from the IPCC report (IPCC, 2019) served as the EF.

The GHG storage was estimated by following Eq. (4).

$$Storage_i = Mb_i \cdot C_b \cdot C_{sc} \cdot 44/12, \quad (4)$$

where C_b is the organic carbon content of produced biochar (mean value, 0.57, of the herbaceous and rice husk and straw, extracted from IPCC 2019 datasets illustrated in Table 2), C_{sc} is the fraction of stable carbon content of biochar (remaining after 100 years) at medium temperature range (0.8), extracted from IPCC 2019 datasets, and the CO_2 fraction is 44⁄12.

Scenario-specific annual biochar production was calculated based on the Eq. (5)

$$Mb_i = Mcrop \cdot r \cdot m \cdot Y_i, \quad (5)$$

where *Mcrop* is the annual feedstock (crop residue) production, totaling 12,000,000 ton/year (MAFF, 2023), *r* is the crop residue utilization rate for biochar production (14% from MAFF (2023)), *m* is the moisture content of feedstocks (crop residues) (assumed as 50% in MAFF (2023)), and Y_i is the yield factor of biochar of scenario *i* (based on the literature of the current study, as illustrated in Table S1).

2.2.3. Uncertainty analysis

An uncertainty analysis, utilizing the Crystal Ball program with a Monte Carlo simulation (Cucurachi et al., 2022), was performed for both the EF values and the scenario-based case study. For EFs, probability distributions for process-specific GHG emissions were established based on data extracted from the literature. Given the diverse distributions present in the literature data for each process, a triangular distribution encompassing minimum, maximum likelihood (median), and maximum values was defined for all processes. In two cases, only one data point in a process was available; a range of ±100% of that value was assumed as its 95% confidence interval. For situations (only 2 processes) with only two available data points, a uniform distribution (minimum, maximum) was employed.

In the case study, uncertainty in the results may stem from the selection of the assessment method and the parameters incorporated in the assessment (Bamber et al., 2020). This study specifically addresses the uncertainty range of parameters involved in the EF calculation. These parameters encompass the yield rate for estimating biochar yield, organic carbon content, and stable carbon fraction of the resultant biochar used for carbon storage calculation, as well as EFs for computing GHG emissions during biochar production. The probability distributions assigned to all these parameters are detailed in Table 2.

3. Results

3.1. Process-based emission factors

The estimated EFs for the 24 groups are outlined in Table 3. These EF simulations describe the potential process-based emissions in biochar production with the most possible combinations of parameters that include carbonization method, use of surplus energy, carbonization scale, and feedstock type.

3.1.1. Emission factor by Carbonization system and scale

Approximately 62 datasets of process-based LC-CO_2 have been extracted from the selected literature and utilized in the emission estimates presented in this study (Table 1). Fig. 2a showcases the diversity in emissions or abatement across different slow pyrolysis systems categorized by the type of carbonization (open or closed systems). The results are derived from the analysis of 47 datasets containing measurable indicators of annual biochar output. Additionally, Fig. 2b provides a process-based breakdown (mean values) of emissions in open and closed carbonization systems, as well as carbonization scale (small, medium, and large). The analysis highlights emissions, rather than abatement, in open carbonization systems due to direct emissions and a lack of surplus energy trapping. Conversely, in closed carbonization systems, carbon abatement increases with the growth of biochar production, attributable to rising surplus energy production and utilization.

In Fig. 2a, six datasets have been scrutinized for the open carbonization system, revealing that the majority of these systems exhibit GHG emissions. However, instances where carbonization occurs in pyrolytic cook stoves (Mohammadi et al., 2017; Smebye et al., 2017) contribute to carbon abatement through clean energy consumption.

In particular, the annual biochar production is typically less than 10 tons due to the conventional small-scale or household-level operation of open carbonization systems with limited inputs. In contrast, closed carbonization systems typically yield negative values for LC-CO_2, indicating carbon abatement due to clean energy consumption. The observed increase in carbon abatement with rising biochar production is supported by regression analysis, revealing a negative regression fit at the 95% confidence interval when comparing GHG emissions against annual biochar production.

Table 2
Parameters used for uncertainty analysis.

	Parameters	Mean values used in calculation	Uncertainty range
Emission factor estimation	Key parameters of biochar production (transport, carbonization plant construction, pre-treatment, emissions at carbonization plant operation, direct emission, emission reduction at surplus energy production)	As per the Table 3	Given the diverse distributions present in the literature data for each process, a triangular distribution encompassing minimum, maximum likelihood (median), and maximum values was defined for all processes. In two cases, only one data point in a process was available; a range of ±100% of that value was assumed as its 95% confidence interval. For situations (only 2 processes) with only two available data points, a uniform distribution (minimum, maximum) was employed.
Case study	Emission factor (EF)	Same as above	Same as above
	Annual feedstock (crop residue) production (*Mcrop*) Crop residue utilization rate (*r*) Moisture percentage (*m*)	*Mcrop*: 12,000,000 ton-biomass/year *r*: 14% *m*: 0.5 (MAFF, 2023)	Do not consider uncertainty range. because this study focuses on difference between biochar production methods (slow pyrolysis systems)
	Organic carbon content of produced biochar (C_b)	Herbaceous: 0.65 Rice husks and rice straw: 0.49 (IPCC, 2019)	Herbaceous: ±45% Rice husks and rice straw: ±41% (IPCC, 2019)
	Stable carbon fraction of produced biochar (C_{sc})	450∼600°C (Medium temperature): 0.8 (IPCC, 2019)	450∼600°C (Medium temperature): ±11% (IPCC, 2019)
	Yield factor (Y)	Average value: 0.30	Yield factor SD: ±0.07 (Based on data extracted from this study)

Table 3
Emission factor (EF) by key influential parameters.

Carbonization method	Use of surplus energy	Feedstock type	Carbonization Scale	EF group number	EF (ton-CO_2e/ton-biochar)	Uncertainty range (CI 95%)
Open carbonization	Without surplus energy	Wood or crop residues	Unspecified	1	0.60	0.34~0.97
	With surplus energy	Wood or crop residues	Unspecified	2	0.23	-0.04~0.61
Closed Carbonization	Without surplus energy	Unspecified	Unspecified	3	0.17	0.17~1.13
			S/M	4	0.18	0.15~1.07
			L	5	0.11	0.10~0.57
		Wood residue	Unspecified	6	0.10	0.06~0.55
			S/M	7	0.07	0.04~0.15
			L	8	0.12	0.08~0.54
		Crop residue	Unspecified	9	0.15	0.08~0.48
			S/M	10	0.16	0.07~0.46
			L	11	0.10	0.09~0.23
		Green waste	Unspecified	12	0.17	0.11~0.30
		Sludge	Unspecified	13	0.68	0.27~1.09
	With surplus energy	Unspecified	Unspecified	14	-0.66	-1.39~0.44
			S/M	15	-0.31	-1.05~0.65
			L	16	-0.96	-1.61~-0.11
		Wood residue	Unspecified	17	-0.78	-1.62~-0.08
			S/M	18	-0.41	-0.88~-0.17
			L	19	-1.10	-1.71~-0.69
		Crop residue	Unspecified	20	-0.48	-0.82~0.01
			S/M	21	-0.18	-0.50~0.13
			L	22	-0.81	-0.92~-0.65
		Green waste	Unspecified	23	-1.00	-1.38~-0.48
		Sludge	Unspecified	24	0.16	-0.46~0.79

Note: EF, Emission factor; S/M, Small or Medium Scale (less than10,000 ton-feedstock/year); L, Large scale (more than10,000 ton-feedstock/year).

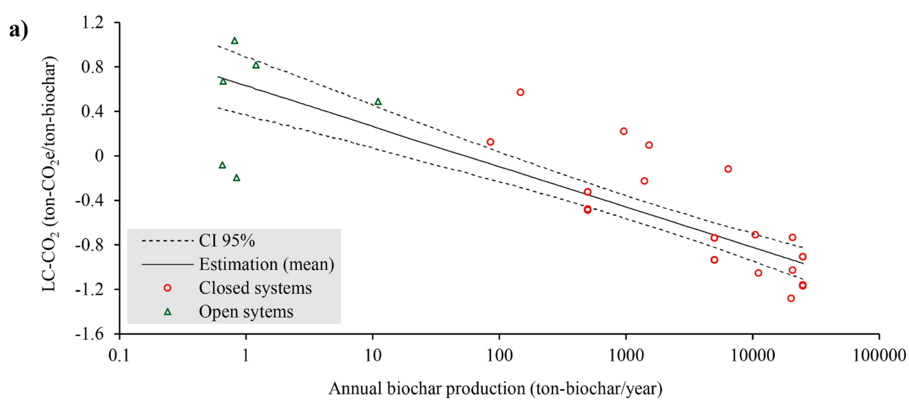

Fig. 2a. LC-CO_2 by annual biochar production.
Number of datasets for Open system (6); Closed system (43). Solid and dashed lines are the mean and 95 % confidence interval, respectively, when fitting a logarithmic function ($y = -0.156\ln(x)+0.618$, $R^2=0.66$, $P<0.001$) to the results from the linear-based method.

The impact of the carbonization system on the final EF is elucidated in Fig. 2b, considering the scale and process-based emissions of each parameter. In open carbonization systems, direct emissions, notably 0.57 ton-CO_2e/ton-biochar, play a substantial role in determining the final EF. Conversely, surplus energy released from carbonization at pyrolytic cook stoves amounts to about −0.37 ton-CO_2e/ton-biochar. Nevertheless, even with surplus energy trapping, positive EF would signify the potential negative environmental impacts of open biomass carbonization processes.

S/M closed carbonization systems exhibit notable emissions during pre-treatment (0.05 ton-CO_2e/ton-biochar), primarily arising from the sludge dehydration process. However, surplus energy production in these systems surpasses that in open carbonization systems. Consequently, carbon abatement for S/M systems is approximately −0.31 ton-CO_2e/ton-biochar. Notably, direct emission is seldom observed in confined carbonization systems due to the syngas recovery process. Large-scale systems boast the highest syngas and bio-oil recovery, approximately double that of S/M systems. Consequently, the final carbon abatement for large-scale systems stands the substantial positive environmental impact of scaling up in the closed carbonization process.

3.1.2. *Emission factor by feedstock type*

Out of the datasets extracted from the selected literature, 56 data sets have been selected from closed carbonization systems and further analyzed to understand the combined feedstock and scale effect on carbon abatement at biochar production, and the results are illustrated in Fig. 2c. However, only several feedstock types with few differences within EF are observed in the open carbonization systems. Corresponding to the results in Fig. 2c, the wood, green waste, and crop residue-derived biochar exhibit negative EF values with carbon abatement potential. However, sludge carbonization has a positive EF value due to considerable emissions during sludge drying.

As per estimations, carbonization of crop residues at a S/M system yields approximately 0.12 ton-CO_2e/ton-biochar in emissions. Conversely, at a larger scale, this emission is halved, amounting to 0.06 ton-CO_2e/ton-biochar, showcasing the discernible impact of scale on emissions. In contrast, wood residue biochar systems exhibit varied emissions during the carbonization process, with large-scale systems

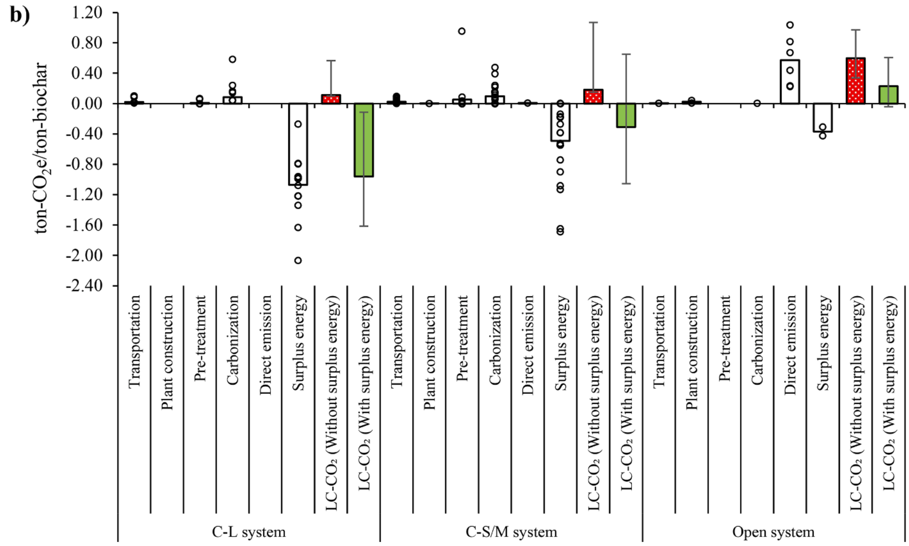

Fig. 2b. Emission factor of biochar production by annual feedstock utilization scale.
Number of datasets for Open system (6); Closed system small or medium scale (C-S/M system) (29); Closed system large scale (C-L system) (27). Median values of process-based emissions are provided along with total LC-CO_2 with or without surplus energy utilization (Uncertainty ranges are provided with a 95% confidence interval).

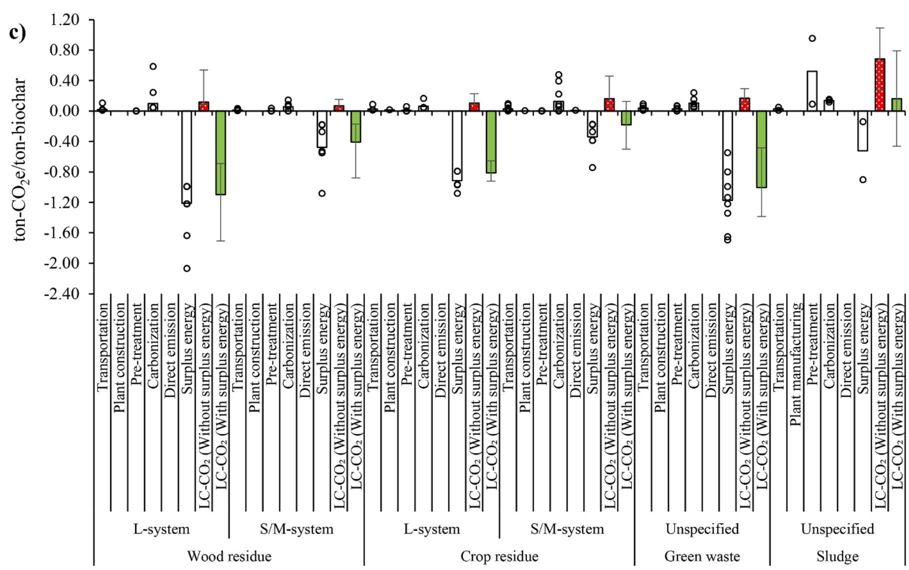

Fig. 2c. Deviation of emission factor of biochar production by feedstock types.
Datasets for Wood residue large scale system (L-system) (14); Wood residue small or medium scale system (S/M-system) (12); Crop residue large scale system (L-system) (8); Crop residue small or medium scale system (S/M-system) (11); Green waste (8); and sludge (3). Mean values of process-based emissions are provided along with total LC-CO_2 with or without surplus energy utilization (Uncertainty ranges are provided with a 95% confidence interval).

emitting 0.10 ton-CO_2e/ton-biochar and small to medium-scale systems emitting 0.05 ton-CO_2e/ton-biochar. This deviation from the typical scale effect observed in wood biomass is attributed to energy utilization, notably electricity consumption during carbonization (Peters et al., 2015). Furthermore, emissions from transportation in the majority of systems remain consistent, ranging from 0.01 to 0.04 ton-CO_2e/-ton-biochar, irrespective of biomass types. Consequently, the overall emissions during biochar production span a range of 0.07 to 0.68 ton--CO_2e/ton-biochar, excluding the utilization of surplus energy.

Conversely, the utilization of surplus energy plays a pivotal role in emission reduction, particularly concerning feedstock type and scale. Notably, wood residue exhibits a substantial reduction in emissions, amounting to −1.21 ton-CO_2e/ton-biochar for large-scale operations. In contrast, surplus energy in crop residue-derived biochar production yields a reduction of −0.92 ton-CO_2e/ton-biochar for large-scale operations albeit comparatively lower than that observed in wood or green waste-derived biochar. While sludge-derived biochar production generates substantial surplus energy (−0.52 ton-CO_2e/ton-biochar) and is reused in the feedstock drying process (Gievers et al., 2021), it results in a significant positive value in final EF. Specifically, wood, crop residue, and green waste biochar production, the final EF stands positive impact of surplus energy utilization.

3.2. *Scenario-specific GHG emissions of crop residue-derived biochar production*

The comprehensive assessment of crop residue-derived biochar production in Japan involves the calculation of total annual GHG emissions across all scenarios, as depicted in Fig. 3. Supporting information, including estimated biochar yield (Table S1) and adopted EFs for the various scenarios (Table S2), is also provided. The cumulative impact of both emissions and GHG storage contributes significantly to the overall annual biochar emission. In particular, the total annual emission is directly influenced by the utilization of surplus energy within the carbonization system. Systems incorporating surplus energy trapping and utilization exhibit greater potential for emission reduction. Consequently, the emissions resulting from biochar production exhibit a diverse range, underlining the dynamic nature of total biochar emissions. Notably, this case study demonstrates that total abatements of biochar derived from 840,000 ton-crop residue per year, i.e., 14% of annual crop residue production (dry basis) span from −215,712 to −625,775 ton-CO_2e/year based on the estimated EFs of this study.

The calculated emission value for the baseline case shows minimal deviation from that of the open system lacking surplus energy utilization. Consequently, the emission during biochar production spans a range from 57,825 to 205,632 ton-CO_2e/year in open carbonization systems. Scenarios 3 and 4 offer estimated annual emissions within closed carbonization systems without utilizing surplus energy. Results indicate limited variation in emissions during biochar production among these scenarios, owing to reduced emissions during carbonization and the absence of direct emissions in these systems. Scenarios 1 and 2 incorporate surplus energy utilization in their estimations to achieve substantial abatement effects. According to the estimations, the large-scale system (Scenario 2) with surplus energy utilization demonstrates the highest carbon abatement effect (−204,431 ton-CO_2e/year).

The baseline scenario exhibits the highest total annual emission, reaching −215,712 ton-CO_2e/year, representing the utmost emission. In contrast, the scenario with large-scale systems incorporating surplus energy utilization (Scenario 2) demonstrates the lowest total annual emission at −625,775 ton-CO_2e/year, marking an approximately threefold reduction in annual emission compared to the baseline. The estimated storage value for biochar production is consistently −421,344 ton-CO_2e/year across all scenarios, including the baseline, assuming that slow pyrolysis has occurred under medium temperature conditions for all systems. Despite the substantial contribution of GHG storage potential from biochar, the emissions during biochar production experience notable fluctuations based on whether the carbonization system is open or closed and the extent of surplus energy utilization.

4. Discussion

4.1. Contribution of process-based emission factor on precise GHG accounting of biochar

The notable diversity observed in GHG emissions among the six scenarios underscores the crucial role of process-based EF in accurate GHG accounting for biochar. The key influential parameters in carbonization, such as feedstock types, the nature of carbonization systems (open or closed), production scale, and the utilization of surplus energy, serve as primary contributors to this variability. These insights are poised to make substantial contributions to regional carbon crediting schemes, policies, and GHG inventory reports by facilitating precise EF calculations for biochar production.

Distinct surplus energy production associated with specific feedstock types becomes evident during the carbonization process, a consequence of the varying calorific values inherent in each biomass type. Wood biomass, characterized by a higher calorific value, significantly contributes to emission reduction by diminishing energy consumption from fossil fuels or electricity (Tisserant et al., 2022). Similarly, energy crops like Switchgrass and Miscanthus, mirroring the characteristics of green

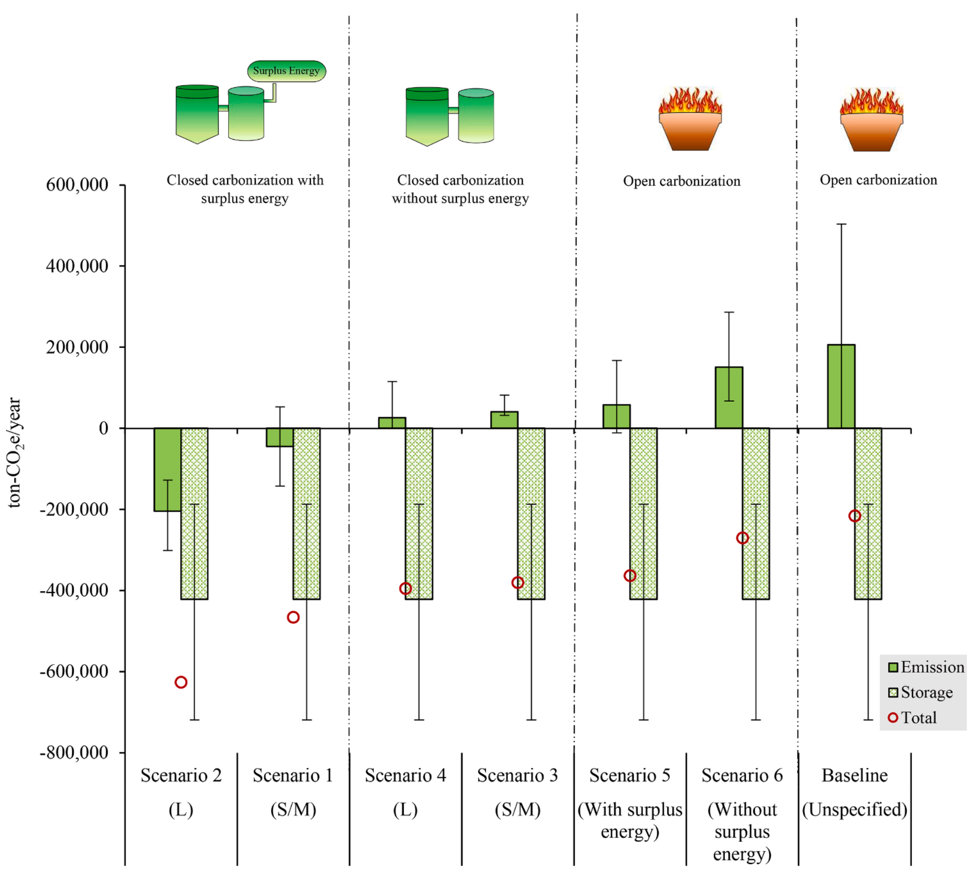

Fig. 3. CO_2 reduction effect of agricultural residue-derived biochar.
S/M, Small or Medium Scale; L, Large scale (Uncertainty ranges are provided with a 95% confidence interval).

waste, exhibit potential in thermochemical conversion systems owing to their elevated calorific values (Brassard et al., 2018). While crop residues typically yield less surplus energy, residues with high lignin content, such as husk or straw may generate more surplus energy. The necessity for an additional pre-treatment step to reduce moisture content in biomass results in relatively higher CO_2 emissions during sludge pyrolysis. Furthermore, a higher standard deviation is observed, indicating increased EF variability during biochar production due to the diverse composition of sludge.

A thorough examination of the critical influencing factors reveals that the carbonization scale plays a pivotal role in determining the final emission. It appears that the impact of temperature on emission during carbonization is potentially overshadowed by other factors, including feedstock type, biochar production scale, and the trapping and utilization of surplus energy. The present analysis does not observe a significant influence of temperature on emission during carbonization, given that slow pyrolysis is conducted within the range of 300–600°C (Fig. S3). However, biochar full LCA accounting by Cheng et al. (2020) identified that lignocellulosic biomass carbonization under higher temperatures would realize maximum energy gain and abatement.

Nevertheless, the default emission factor (EF) for biochar production outlined in IPCC (2019) introduces a higher degree of uncertainty, particularly concerning traditional carbonization methods (Cornelissen et al., 2016). Furthermore, these estimates are exclusively tailored for open carbonization systems, and the absence of kiln-specific data challenges their universal applicability in emission assessments. Even though some studies have comparable individual LCA data assortments for open carbonization systems (Mohammadi et al., 2017; Smebye et al., 2017), a generalized overview has not been accomplished yet.

It is imperative to establish a universally applicable methodology for the precise estimation of emissions from biochar production systems, encompassing not only open carbonization but also closed carbonization systems, considering production scale and other key parameters. While the present study furnishes detailed process-specific EF estimates (Table 3) with the most possible combinations, the prevailing literature as listed in Table S3 contain a limited number of combinations of EF estimates. In particular, in previous literature, only one or a few influencing factors of biochar production LCA have been evaluated, such as scale-wise comparison (Hammond et al., 2011) and temperature and feedstock biomass comparison (Muñoz et al., 2017). Nonetheless, a comparison of the consequences of pyrolysis systems with or without surplus energy consumption is yet to be outlined. Besides, the current case study aims to offer a comprehensive overview of CO_2 emissions and storage across various biochar production processes with maximum plausible combined effects.

Categorized and estimated under six scenarios of crop residue-derived biochar production, this study facilitates the selection of an optimal process based on specific requirements. Following this, the scenario-specific investigation reveals that the substantial variability in process-based emissions significantly contributes to the total emission variation within each scenario, overshadowing the well-established carbon sequestration potential of biochar, which exhibits limited variation across different scenarios. These findings underscore the pressing need for updating conventional biochar emission calculations, as the EF of the carbonization process has been inadequately considered in existing estimates. Even the carbon crediting under the J-credit scheme could benefit from enhancing their estimations by incorporating process-based EF, leading to more accurate carbon credit calculations.

4.2. GHG emission reduction nexus of biochar

This study aimed not only to identify the most favorable scenario for GHG abatement but also to delineate the GHG emission reduction potential in each scenario. The process-based EF emerges as a crucial contributor to diminishing the GHG emissions associated with biochar production. Given that the EFs elucidated in this analysis predominantly rely on process-based emission data, comprehending the highest-priority areas and taking prompt actions are paramount for achieving substantial GHG emission reductions within systems. This could entail installing the most suitable equipment or implementing technologies to enhance the efficiency of the carbonization process, coupled with significant surplus energy recovery.

Open carbonization systems typically exhibit positive emission values due to the absence of surplus energy trapping and utilization, coupled with direct emissions. Even in closed carbonization systems, surplus energy plays a pivotal role in GHG emission or abatement, followed by the contribution of carbonization scale. The establishment of open biomass carbonization systems in rural areas with limited resources is accompanied by noteworthy socio-economic benefits, along with the positive environmental impact of carbon storage in the resulting biochar (Sparrevik et al., 2015). Locally abundant feedstocks such as rice husk, straw, or wood biomass are intentionally employed in domestic open carbonization processes to capitalize on emissions reduction by bypassing pre-treatment and transportation steps, in addition to their cost-effectiveness. The common occurrence of negative environmental impact through direct emission in kilns or domestic stove pyrolysis systems stems from the use of simple, low-cost techniques without inert gas trapping. However, enhancing the energy-saving efficiency of stoves by burning syngas may mitigate these negative environmental consequences.

Pyrolytic gas trapping in closed carbonization systems yields twofold environmental benefits, encompassing carbon abatement through surplus energy utilization, as well as prevention of direct GHG emissions. Conversely, the enhancement of carbon abatement aligns with increased biochar production capacity, achieved by elevating system efficiency and efficacy through the adoption of cutting-edge technologies in sophisticated large-scale pyrolytic plants. The cumulative effect of surplus energy and scale on GHG emission reduction in biochar production becomes more pronounced in large-scale systems. Conversely, positive emissions are observed in carbonization systems without surplus energy trapping, where emissions at small or medium-scale systems surpass those at large-scale systems. Evidently, the impact of surplus energy, as the primary carbon abatement step in the charring process, outweighs other prominent factors in biomass carbonization.

While the impact of different biochar production processes on GHG reduction is evident, a comprehensive analysis of crop residue-derived biochar has yet to be fully explored. Thus, a scenario-based case study has been conducted using the IPCC default value as the base case. Total abatement values of case-study scenarios and closely related existing study outcomes were meticulously re-calculated to standardize the values into ton-CO_2e/ton-biochar, ensuring a precise assessment. The comparison of the scenarios in this study with those of existing studies (Fig. S4) not only confirms the similar trends of total emission reduction potential in each category but also provides a strong validation of our findings. The diversity within the category is primarily driven by the variety of crop biomass, including wheat straw, corn stoves, oat husk, and the cumulative effect of other operational conditions. Notably, a blend of crop and wood residues-derive biochar leads to significant variation in data of open carbonization with surplus energy use (Scenario 5). Despite the availability of closely related studies for most scenarios, the scarcity of related studies for large-scale closed carbonization systems without surplus energy trapping (scenario 4) highlights the need for extended research to obtain a comprehensive overview.

4.3. Future insights

This case study specifically focuses on the GHG emission reduction potential of crop residue-derived biochar production, considering its abundance, utilization potentials, and accountability, utilizing available statistics in Japan. The regional abundance of feedstock emerges as a crucial factor for continuous input supply, ensuring sustainable biochar production (Clarke, 2023). The current analysis may serve as an extended guideline for estimating GHG emissions in biochar production from lignin-rich biomasses like wood residue or green residue, which carry more notable environmental impacts with surplus energy production and carbon sequestration (Gu & Bergman, 2017). Nonetheless, extended studies on different feedstock carbonization processes are essential to provide precise estimates for evaluating the carbon abatement potentials of biochar. Additionally, the estimates of this study encourage the meticulous evaluation of a broader spectrum, from low-cost open carbonization systems with simple technologies to more sophisticated installations with high capital investment for mass biochar production. However, the study's scope is confined to identifying carbon abatement within production systems where biochar is the primary product (slow pyrolysis). There is still ample potential to evaluate systems with biochar as a co-product (fast pyrolysis or gasification) to assess the substitute environmental benefits of such systems. In addition, it should be noted that the results of our case study, based on EFs developed in the current study, have limitations in which the emission from the feedstock collection process is not taken into account.

While the remarkable potential of biochar production as a NET is widely acknowledged (Harsono et al., 2013), its practical applications have been limited to regions. An analysis of country and region perspectives in this study offers insights into leading countries and regions actively engaged in studies on the environmental impact assessment of biochar production and application (Fig. S2). Primarily, countries within the European Union (EU) (Azzi et al., 2021) and other developed nations have made significant academic contributions, implementing initiatives and reformative policies that endorse biochar as a prominent NET. However, some developing countries in Africa (Duku et al., 2011; Whitman et al., 2011) and Asian regions (Aberilla et al., 2019) are increasingly involved in exploring the impact of biochar as an alternative energy source, utilizing open carbonization technologies (Cornelissen et al., 2016; Mohammadi et al., 2017). The six scenarios discussed in this study further contribute to raising awareness for selecting the most appropriate and feasible biochar production process, laying a robust foundation for the formulation of sound policies in favor of the sustainable use of biochar in carbon abatement.

5. Conclusions

The diversity in key parameters within the biochar production process introduces significant uncertainty in the accurate environmental impact assessment of biochar. However, a critical evaluation of existing studies underscores the importance of carefully assessing the influence of factors governing carbon abatement potential. This study highlights that surplus energy utilization plays a pivotal role in leveraging abatement, in conjunction with the cumulative impact of the carbonization system (open or closed), scale, and feedstock type. While the carbon storage potential of biochar receives considerable attention in the carbon crediting process, the impact of emission during carbonization is inevitable in various biochar production processes. Thus, it is crucial to calculate carbonization emissions more accurately, considering all vital influential parameters. The scenario-specific case study serves as a benchmark for selecting a viable biochar production system, offering insights into its potential environmental consequences. Furthermore, it underlines the critical need for reformulating and implementing sound policy decisions to promote the use of biochar as a prominent NET.

CRediT authorship contribution statement

Gajasinghe Arachchige Ganga Kavindi: Writing – review & editing, Writing – original draft, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Longlong Tang:** Writing – review & editing, Writing – original draft, Supervision, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Yuma Sasaki:** Writing – review & editing, Methodology, Investigation, Formal analysis, Data curation.

Declaration of competing interest

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

Data availability

Data will be made available on request.

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.resconrec.2024.107900.

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