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Lifecycle environmental impacts of biochar in Belgium: The influence of biochar feedstocks, production temperatures, and applications

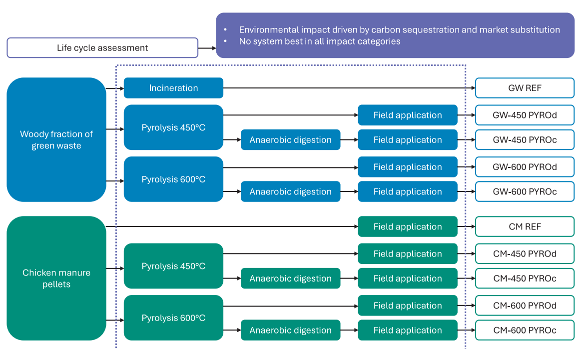
Luca Campion [a,b,*], Tobias Pape Thomsen [b], Bo Pedersen Weidema [c], Robert Malina [a], Tom Kuppens [a,d]

[a] UHasselt – Hasselt University, Centre for Environmental Sciences, Research Group Environmental Economics, Agoralaan Gebouw D, 3590 Diepenbeek, Belgium
[b] Roskilde University, Department of People and Technology, TRANSITIONS, Universitetsvej 1, 4000 Roskilde, Denmark
[c] Aalborg University, The Technical Faculty of IT and Design, Department of Planning, A.C. Meyers Vænge 15, 2450 København SV, Denmark
[d] Vrije Universiteit Brussel, Multidisciplinair Instituut Lerarenopleiding, Brussels Institute for Teacher Education, Pleinlaan 2, 1050 Brussel, Belgium

HIGHLIGHTS

- Biochar's environmental impact depends on feedstock, production, and application.
- Carbon sequestration and market substitution drive biochar's benefits.
- Cascading use of biochar may enhance its environmental performance.
- No single biochar system outperforms in all environmental impact categories.
- Life cycle assessment reveals biochar's potential for climate mitigation in Flanders.

GRAPHICAL ABSTRACT



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ABSTRACT

This study aims to support decision-making on biochar deployment in Flanders, Belgium by assessing whether diverting biomass to biochar production yields environmental benefits on a lifecycle basis, and by identifying the most favorable combinations of feedstock, production temperature, and application. Using a consequential life cycle assessment (LCA), we investigate biochars produced at 450 °C and 600 °C from the woody fraction of green waste and chicken manure. Biochar applications are direct soil amendment and cascading use, where biochar is first used to enhance anaerobic digestion. To the best of our knowledge, this is the first consequential LCA that simultaneously compares biochar's environmental impact across pyrolysis temperatures, feedstocks, and applications. For both feedstocks, no system outperforms all others in all impact categories, so the preferred option depends on the impact category. Pyrolysis, especially with cascading use, is generally preferred for global warming and fossil resource scarcity, and frequently for terrestrial acidification and marine eutrophication. Reference systems generally show lower toxicity impacts, although toxicity-related results should be interpreted with caution due to modeling simplifications. Overall, chicken manure systems outperform green waste in most categories. In the current case study, with its specific modeling assumptions and system boundaries, the

* Corresponding author at: Agoralaan gebouw D, 3590 Diepenbeek, Belgium.
E-mail addresses: luca.campion@uhasselt.be (L. Campion), tpapet@ruc.dk (T.P. Thomsen), bweidema@plan.aau.dk (B.P. Weidema), robert.malina@uhasselt.be (R. Malina), tom.kuppens@uhasselt.be (T. Kuppens).

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environmental impacts are driven more by market substitution effects and carbon sequestration than by biochar's field effects. This study shows the potential desirability of biochar production and application in specific contexts.

1. Introduction

Climate change mitigation pathways that limit global warming to 1.5 °C need both reductions of greenhouse gas (GHG) emissions and removals of already emitted GHGs. In the 2018 special report of the Intergovernmental Panel on Climate Change (IPCC), biochar is included in the list of carbon dioxide removal (CDR) technologies (Rogelj et al., 2018). Biochar is the solid product of biomass pyrolysis, which is the thermochemical conversion of organic material at elevated temperatures without oxygen and can additionally yield a gaseous and a liquid product (Bridgwater, 1994; Spokas et al., 2012). During pyrolysis, cellulose and lignin are decomposed, and aromatic carbon structures are formed in the biochar, significantly increasing carbon stability. Carbon originally captured by plants using photosynthesis can thus be stored much longer than via direct soil application, where the biomass decomposes much faster and releases the carbon as CO_2 into the atmosphere (Lehmann, 2007; Lehmann et al., 2012; Lehmann et al., 2006; Lehmann & Joseph, 2009). Therefore, biochar is considered a CDR technology. Permanence varies with pyrolysis conditions, but a sound generalization is that 75% of the carbon is still stored after 100 years (Schmidt et al., 2022).

Even though biochar is often considered a single, homogeneous product, it is variable and versatile. Its variability stems from the fact that its properties depend on the feedstock and production conditions (Lehmann & Joseph, 2009). Biochar is versatile because it can be used in many applications (Kumar et al., 2023; Schmidt & Wilson, 2014), some of which require tailored biochars. Production parameters that can be varied include residence time, heating rate, pyrolysis temperature, and feedstock (Joseph et al., 2021; Kumar et al., 2021; Li et al., 2019; Roy & Dias, 2017; Weber & Quicker, 2018). As feedstock, residual biomass can be used, so that the biochar becomes a by-product of waste management (Sikder & Joardar, 2019; Sri Shalini et al., 2020; Yargicoglu et al., 2015).

For a case study of Flanders, the northern part of Belgium, this study investigates biochar from two such residual biomass streams: the woody fraction of green waste and chicken manure pellets. These feedstocks were selected both to span a range from woody biomass to manure and for their relevance to the study's geographic scope. In line with circular economy principles, local legislation requires green waste to be composted to close carbon and nutrient cycles by applying the produced compost to agricultural soils (Oldfield et al., 2018; Vandecasteele et al., 2016). However, EU legislation promotes using biomass to produce bioenergy, affecting the availability of woody biomass and causing the woody fraction to be increasingly sieved from the green waste (Viaene et al., 2016). This fraction is studied here, as its pyrolysis could reconcile energy production and the closing of carbon and nutrient cycles (McHenry, 2009).

Following a global trend (Ritchie et al., 2023), chicken farming has been increasing in Flanders (STATBEL, 2023), involving an increased production of chicken manure. Because of its essential nutrients and organic compounds, it is often used as an organic fertilizer (Muola et al., 2021). Given the current pressure of excess nutrients on the environment in Flanders, alternative treatments of chicken manure are considered, and biochar production is one of them (Manogaran et al., 2022).

The point of departure of this study is that these residual biomass streams are a limited resource, of which biochar is only one potential use. Decisions on using these limited resources should optimize value creation through improved environmental performance. Therefore, this research aims to assess the environmental impact of diverting biomass from its current use to biochar production. The types of biochar considered here are defined by the two types of feedstock mentioned previously and two pyrolysis temperatures, namely 450 °C and 600 °C. Furthermore, biochar's environmental impact might differ depending on its application. We investigate the direct field application of biochar, arguably the most common and most studied application (Jiao et al., 2021), and a cascading use, where biochar is first used in anaerobic digestion to increase methane yield and, afterward, applied to soil, mixed in the digestate.

A consequential life cycle assessment (LCA) is used to assess the environmental impact of the different systems. Existing LCAs provide valuable insights into biochar production and use (Patel et al., 2025; Ramírez López et al., 2024). Nonetheless, this is, to our knowledge, the first consequential LCA examining the combined influence of feedstock choice, pyrolysis temperature, and application. Two additional features further distinguish this study. Regarding applications, most biochar LCAs consider only soil amendment while we compare it to biochar's use in anaerobic digestion. Many previous biochar LCAs focus on the climate impact (Matuštík et al., 2020), while we include more impact categories.

2. Material and methods

2.1. Goal and scope

This life cycle assessment (LCA) compares different biochar production-and-use systems across biomass types, pyrolysis temperatures, and biochar applications to identify the option with the lowest environmental impact. In addition, it identifies impact hotspots, drivers, and patterns. The results are intended for feedstock providers, potential biochar producers and users, and policymakers, particularly in Flanders, Belgium, but any possible generalizations are also discussed. Given its geographical scope, the LCA is conducted at the regional level. The pyrolysis plant is assumed to operate for 20 years, setting the temporal scope. The technological scope, relating to the selection of specific technologies, is described in the life cycle inventory. Because this study investigates the environmental consequences of biochar deployment, consequential LCA is used (Weidema et al., 2018). It provides the best support for comparative assessments by considering the future effects on the environment if one of the alternatives is deployed (Curran et al., 2005; Plevin et al., 2014; Weidema et al., 2018), and the approach is consistent with the ISO standards (ISO, 2006), as it eliminates the need for allocation.

Eight product systems are derived by combining two feedstocks, turned into biochar at two different temperatures and used in two different ways. Additionally, a reference system (REF) is included for each feedstock, which is inherent to the adopted consequential modeling and serves as an estimate for its environmental impact. Therefore, a total of ten product systems is studied.

The feedstocks are the woody fraction of green waste and chicken manure pellets. Green waste refers to the combination of garden waste, grass clippings, and residues from trimming and pruning. Its composition varies with the relative contributions of these different fractions, as well as collection practices, which may affect its properties and limit the generalizability of the results to other contexts. Collection happens through a combination of deliveries from garden contractors and maintenance of public parks, retrieval from container parks, and house-to-house retrieval. The woody fraction that is increasingly sieved from the green waste, due to an increasing demand for woody biomass for bioenergy purposes (Viaene et al., 2016), is the actual feedstock studied here. While only considering this woody fraction, the distinction is not made further in this paper, and the feedstock is referred to as GW.

The collected chicken manure is a mixture of bedding material, wasted feed and water, excreta, feathers, and other detritus from the chickens (Van Zwieten et al., 2013). The actual feedstock considered here consists of pellets thereof, which have been through composting, hygienization, and pelletization. This pre-treated version of the feedstock was chosen because it is already being used as a soil amendment, so its biochar is expected to be directly applicable to fields, too. Furthermore, recent experimental data on its pyrolysis exist (Lataf et al., 2022). However, pre-treatment is costly, increases emissions, and reduces the potential for energy production. Therefore, the implications of using raw chicken manure instead are discussed in Section 3.2.2. The distinction is not made further and the feedstock is referred to as CM.

The product systems and system boundaries are shown in Fig. 1. For GW, the starting point is the sieved fraction collected at waste treatment facilities in Flanders. The management of the remaining fractions is assumed to be unaffected by the treatment of the woody fraction and thus excluded from this analysis. Pellets from manure treatment companies form the starting point for CM. In the reference systems, which are in line with current practice, GW is used in a biomass power plant and CM is transported to vineyards in France, to be used as fertilizer (Schmitt et al., 2019; Topcu et al., 2022). Whereas shredding is excluded for GW because of identical requirements across systems, it is left out for CM because there is none. The AD installations using biochar are so-called pocket digesters, installed and managed by farmers to treat their livestock's manure (De Dobbelaere et al., 2015). In line with current practice, the digestate is applied to fields.

To ensure comparability of the pyrolysis systems with the reference systems, utilizing a metric ton (1 Mg) of wet biomass is the chosen functional unit (FU). This input-based FU aligns with our goal to find the best use of these scarce resources (Ahlgren et al., 2015) and is similar to other biochar LCAs (Azzi et al., 2019; Bartocci et al., 2016; Field et al., 2013; Ibarrola et al., 2012; Roberts et al., 2010). Furthermore, this FU enables a comparison of feedstocks using the impact per Mg relative to their reference use.

Finally, the following assumptions have been made in addition to the specific ones described in Section 2.2. i) The increased demand for biochar will not increase the production of either waste stream. ii) The deployment of biochar in AD only influences methane production, so the biomass used in AD is not modeled. iii) Biochar's field effects are limited to fertilizer substitution, differences in diesel consumption, and selected emissions to soil, air, and water. Effects on soil quality, irrigation, or drought resistance are excluded, although they are an integral part of biochar's carbon dioxide removal (CDR) potential. Their strong variability, context dependence, and data uncertainty hinder robust modeling. As these effects are generally expected to favor biochar use, their exclusion is consistent with the conservative approach adopted in this study. iv) All modeled fields are assumed to produce corn, the dominant crop in Flanders (Boone et al., 2016; STATBEL, 2023). The reference field is fertilized with digestate, applied to the Flemish limit of 170 kg N/ha, and supplemented by chemical fertilizer. The overall limit for active N is assumed to be 135 kg N/ha. The 60% fertilizer value means 102 kg of active N is applied with the digestate, so 33 kg of chemical N fertilizer is added.

2.2. Life cycle inventory

This section discusses the most important aspects of the life cycle inventory, while the Appendix provides more details, including the mass and carbon balances of the analyzed product systems. Background data from the consequential ecoinvent 3.6 database was used with SimaPro 9.

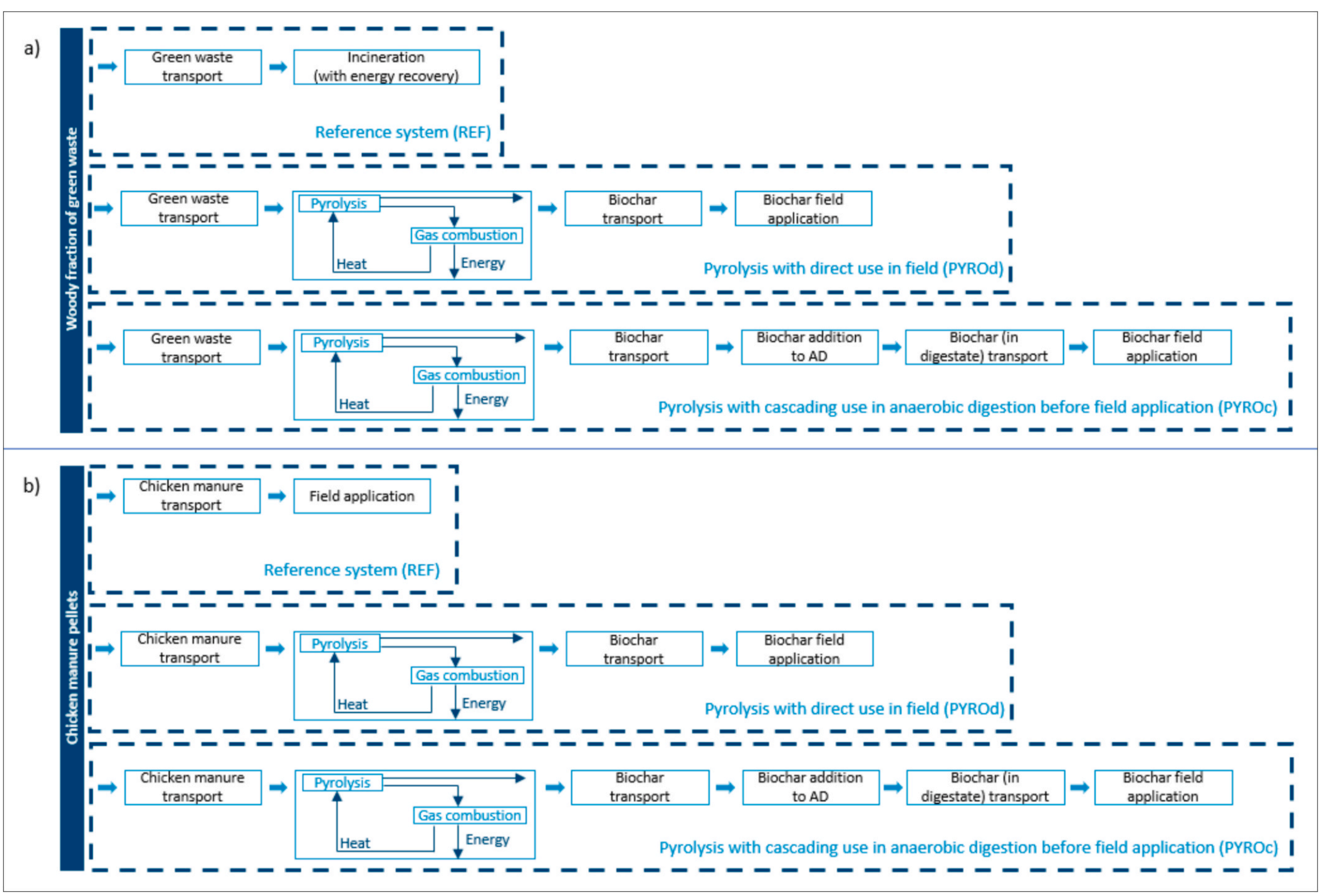

Fig. 1. Analyzed product systems and system boundaries, showing the included processes for (a) GW and (b) CM.

2.2.1. Transport

Feedstock transport distances are twice the average distance from all collection points to the relevant destination, accounting for roundtrips. The hypothetical pyrolysis plant was located centrally amongst these collection points. The location of the only biomass power plant in Flanders that incinerates GW was used for that reference. The calculations for the CM reference, involving transport to French vineyards, are detailed in the Appendix . In the PYROd systems, biochar transport distances are calculated as twice the average distance from the pyrolysis plant to the centroids of the provinces in Flanders, weighted by maize-cropped agricultural area. In the PYROc systems, biochar is first transported to AD facilities using twice the average distance between the pyrolysis plant and the AD facilities. Transports from AD facilities to fields use the same destinations as in PYROd systems. As farmers deliver manure and collect digestate independently of biochar integration, only the additional one-way transport associated with the biochar mass is considered. The importance of transport distances is investigated in the sensitivity analysis (SA).

2.2.2. Green waste reference: Incineration

Electricity production in the biomass power plant is calculated from the energy content of GW, derived from its elemental composition (Maksimuk et al., 2020) and assuming an energy efficiency of 28.5% (Tisserant et al., 2022). Three assumptions are tested in the SA: (i) no heat co-generation, (ii) substitution of ecoinvent's consequential electricity mix by the produced electricity, and (iii) valorization of ashes in Portland cement. Complete oxidation of carbon to CO_2 is assumed (Kranert et al., 2010), while emissions of N_2, NO_2, and SO_2 are based on industry data and corrected for local emission limits for NO_2 and SO_2. Plant utilities are assumed identical for incineration and pyrolysis and are omitted from all GW systems. Heavy metal (HM) pollution is modeled using a simplified representation of partitioning in thermal treatment processes. Based on Nzihou and Stanmore (2013), As, Cd, Cr, and Pb are assumed to concentrate in solid residues, while Hg volatilizes. Although differences between incineration and pyrolysis exist and depend on process conditions, these processes share similar governing mechanisms and general partitioning trends. For modeling purposes, incineration is assumed to result in no net HM emissions due to the incorporation of ash in concrete, whereas pyrolysis leads to full soil release of the retained metals. This assumption represents a deliberate and conservative simplification, consistent with the overall approach adopted in this study. However, the simplified modeling of HM emissions means that toxicity-related results should be interpreted with caution.

2.2.3. Chicken manure reference: Field application

As for biochar, a tractor requires 2.16 L diesel per Mg CM spread out (Thomsen, 2021). CM application substitutes mineral N, P, and K fertilizers based on its nutrient content, assuming fertilizer values of 100% for P and K and 20% for N. HM impacts are omitted for all CM systems, following a combination of identical and conservative modeling. This modeling, however, means that toxicity-related results should be interpreted with caution. Polycyclic aromatic hydrocarbons (PAHs) formed during pyrolysis are considered for biochar but are not present in CM (Alharbi et al., 2023). Both fossil and biogenic carbon emissions are modeled, and it is assumed that 97% of the carbon in the CM is re-emitted to air in 100 years (Thomsen, 2021). Because CM has a legislative fertilizer value of 20% for N, its application allows higher total N inputs compared to mineral fertilizer, resulting in increased N_2O emissions following IPCC Tier 1 methodology.

2.2.4. Pyrolysis plant operation

Electricity production from pyrolysis is calculated from the energy content of the syngas, with an efficiency of 28.5% (Tisserant et al., 2022), and assumed to substitute ecoinvent's consequential electricity mix. The SA includes an alternative scenario in which heat is produced instead. Although pyrolysis provides its own heat via syngas combustion, auxiliary utilities are required for process control and start-up (Haeldermans et al., 2020). However, these are assumed to be comparable for incineration and are, therefore, omitted for all GW systems. A water addition of 25% on a dry-mass basis is assumed for both GW and CM to quench the biochar upon reactor exit, resulting in a biochar moisture content of 20% after quenching (Rödger et al., 2016). All carbon not retained in the biochar is assumed to be emitted as CO_2. Emissions of NO_2 and SO_2 are based on industry measurements and corrected for local emission limits.

2.2.5. Biochar addition in anaerobic digestion

Biochar generally increases biomethane production from AD. A *meta*-analysis of 27 studies (Xiao et al., 2021) reports substantial variability in this effect but does not identify statistically significant differences between feedstock types. Similarly, no statistically significant differences were found between biochars produced below 500 °C and those produced between 500 °C and 700 °C. As these ranges include the temperatures considered in this study, the effect is not differentiated across biochar types. Following Shen et al. (2020), a baseline increase of 11.48% is assumed. The uncertainty associated with this parameter is evaluated through SA.

2.2.6. Biochar field application

Biochar is applied using a tractor consuming 2.16 L diesel per Mg biochar (Thomsen, 2021) at a rate of 10 Mg/ha, with effects assumed to persist for three years (Borchard et al., 2019; Schmidt et al., 2021; Sha et al., 2019). Six biochar field effects are considered: (i) carbon sequestration, (ii) reduced N_2O emissions, (iii) reduced NH_3 emissions, (iv) reduced NO_3 leaching, (v) reduced fertilizer demand, and (vi) the introduction of PAHs and HMs. Carbon sequestration, nutrient-related emission reductions, fertilizer substitution, and contaminant inputs are parameterized based on literature and differ by feedstock and pyrolysis temperature. HM impacts are included for GW biochars only, based on Lataf et al. (2022), while PAH impacts are considered for all biochars. Because these impacts are simplified and based on literature, toxicity-related results should be interpreted with caution. In the PYROc systems, fertilizer regulations restricting the amount of digestate applicable to soils limit the biochar application rate to 0.25 Mg/ha, which is insufficient to induce measurable changes in N_2O and NH_3 emissions or NO_3 leaching (Borchard et al., 2019; Sha et al., 2019). The other effects are assumed identical to those in the PYROd systems. Details are provided in the Appendix .

2.3. Life cycle impact assessment (LCIA)

Following literature recommendations (Bjørn et al., 2018; Chen et al., 2021), results obtained using different LCIA methods are compared. ReCiPe2016 Midpoint (Hierarchist) is used as the main method (Huijbregts et al., 2017). The SA assesses the effect of applying the Individualist perspective of ReCiPe2016, using a 20-year instead of a 100-year time horizon for Global Warming Potential (GWP), as well as the Stepwise2006 method and the European Commission's Environmental Footprint method (Fazio et al., 2018). To improve interpretation, only results for relevant impact categories are discussed. Relevance is defined as impact categories that together account for at least 95% of the total impact across all 10 product systems when the results are weighted using CE Delft's medium environmental prices (de Bruyn et al., 2023). Results for the remaining impact categories are reported in the Appendix . Biogenic and fossil carbon emissions are not distinguished for two reasons. First, treating biogenic carbon as climate-neutral can be an oversimplification, particularly due to temporal differences between carbon uptake and release (Ahlgren et al., 2015; Vance et al., 2022). Second, as this is a comparative assessment of biobased systems, assumptions that would affect all alternatives similarly are not introduced.

2.4. Sensitivity and uncertainty analysis

Parameter uncertainty is assessed using Monte Carlo simulations, varying foreground parameters according to predefined probability distributions (Bamber et al., 2020). Where available, parameter ranges are based on experimental data. However, standard uncertainty ranges had to be used for most, following the approach of Brun et al. (2002). In this approach, parameters are classified as accurately known (5% relative uncertainty), moderately inaccurate (20%), or very poorly known (50%) (Brun et al., 2002). Effects of biochar application in anaerobic

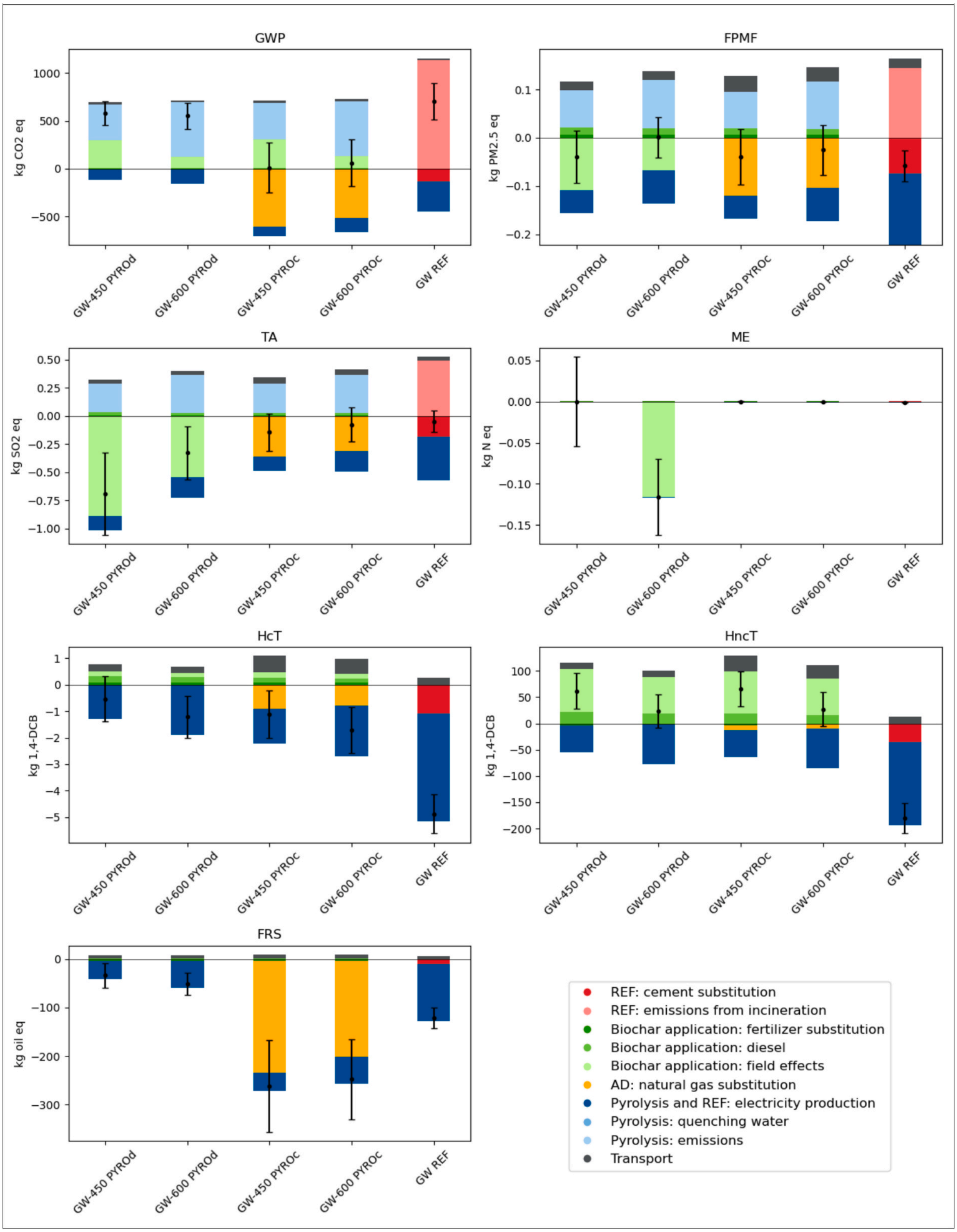

Fig. 2. GW results accounting for parameter uncertainty, including contribution analysis. Bars represent 95% confidence intervals. Abbreviations: GWP (global warming potential), FPMF (fine particulate matter formation), TA (terrestrial acidification), ME (marine eutrophication), HcT (human carcinogenic toxicity), HncT (human non-carcinogenic toxicity), FRS (fossil resource scarcity).

digestion and in the field, except for direct carbon sequestration, are classified as very poorly known. Other parameters lacking experimental data are considered moderately inaccurate. A triangular distribution is assumed for these parameters, as available data are insufficient to define more detailed probability distributions. Its definition by minimum, maximum, and most likely values provides a transparent and consistent approach (Bann et al., 2017; Kuppens et al., 2018). This choice is also consistent with the classification of uncertainty ranges, which is based on expert judgment, for which triangular distributions are commonly applied (Kuppens et al., 2018). Confidence intervals are reported alongside point estimates, as suggested by Bamber et al. (2020).

The SA is performed using scenario and contribution analyses. Contribution analysis quantifies the contribution of individual life cycle processes to overall environmental impacts (Laurent et al., 2020). Scenario analysis addresses model and scenario uncertainty (e.g., substitution choices and LCIA methods, respectively), in addition to parameter uncertainty, by varying one aspect of the system at a time (Bamber et al., 2020).

3. Results and discussion

The interpretation of the results follows the framework proposed by Laurent et al. (2020). A detailed discussion can be found in the Appendix, together with the results for all impact categories. Here, only the relevant categories are discussed, making up at least 95% of the environmental impact for all systems. These impact categories are global warming potential (GWP), fine particulate matter formation (FPMF), terrestrial acidification (TA), marine eutrophication (ME), human carcinogenic toxicity (HcT), human non-carcinogenic toxicity (HncT), and fossil resource scarcity (FRS).

3.1. Results including parameter uncertainty

3.1.1. Green waste (GW)

Across most impact categories, differences between systems are small and often not statistically significant (Fig. 2), indicating that trade-offs between energy substitution, field effects, and process emissions largely offset each other. For GWP, the PYROd systems are comparable to the reference, reflecting a balance between carbon sequestration and lower electricity production. In contrast, the PYROc systems show significantly lower impacts, driven by increased biomethane production in anaerobic digestion (AD). No significant differences are observed for FPMF. Similar patterns apply to TA, but the GW-450 PYROd system performs better than the PYROc systems and the reference. Regarding ME, the impact is driven solely by biochar's effect on nitrate leaching. Consequently, the GW-600 PYROd system shows the lowest impact due to a stronger assumed leaching reduction. For human toxicity (HcT and HncT), the reference system consistently outperforms the pyrolysis systems, primarily due to greater electricity substitution. However, these results should be interpreted with caution given the simplified modeling of contaminants. In terms of FRS, the PYROc systems perform best because of natural gas substitution with biomethane, whereas the reference outperforms the PYROd systems. Overall, the results highlight the importance of energy substitution, particularly electricity and biomethane, as key drivers of system performance. Because experimental data were missing from Lataf et al. (2022), these outcomes remain sensitive to assumptions regarding energy balances, which were approximated based on biomass and biochar composition (Maksimuk et al., 2020).

3.1.2. Chicken manure (CM)

The PYROc systems generally show the lowest impacts across categories (Fig. 3), followed by the PYROd systems and the reference, though differences are not always statistically significant. For GWP, pyrolysis systems reduce impacts relative to the reference by biochar's carbon sequestration and by enabling energy recovery. The PYROc systems tend to perform best due to additional biomethane production, although differences with the CM-600 PYROd system are not statistically significant. A similar pattern is observed for FRS, where the PYROc systems benefit from natural gas substitution, while the CM-600 PYROd system outperforms the reference through electricity generation. FPMF and TA follow comparable trends, with the PYROc systems outperforming the reference, but without statistically significant differences between PYROc and PYROd systems, and between the latter and the reference. Regarding ME, the impacts are driven by biochar's effect on nitrate leaching. The PYROd systems do not differ significantly from each other, but they do from the other systems. For HcT, no statistically significant differences are observed, reflecting substantial uncertainty across systems. In contrast, for HncT, the reference outperforms the pyrolysis systems, which do not differ significantly from each other. The reference benefits mostly from substituting nitrogen fertilizer, which was not modeled in the pyrolysis systems due to negligible plant-available nitrogen in the biochar. This assumption is especially conservative for biochar produced from a nutrient-rich feedstock such as CM, so caution is required with these results, as the modeling decisions actively favor the reference and the modeling of contaminants is simplified. Overall, the results highlight the importance of energy recovery and substitution effects, particularly biomethane and electricity, while also demonstrating the sensitivity of toxicity-related outcomes to key modeling assumptions.

3.2. Sensitivity of results to modeling choices

Table 1 summarizes the results of the scenario analysis, used to account for scenario and model uncertainty, showing whether the preferred option changes when changing assumptions. This approach is reasonable because decision-making is the main reason for this LCA. The Appendix shows all results.

3.2.1. Green waste (GW)

Regarding biochar's field effects, the results are rather insensitive to the choices made, as the preferred option rarely changes. A lower biochar application rate and longer persistence of its field effects benefit the PYROd systems, while a shorter duration of the field effects is detrimental. As expected, assuming higher carbon contents of the GW biochars changes the preferred option for GWP, because higher carbon contents lead to greater carbon sequestration. The GW-600 PYROc system becomes the preferred option because the carbon content of the biochar produced at 600 °C is assumed to be greater than that produced at 450 °C. This scenario was included because the carbon contents, based on Lataf et al. (2022), were relatively low and showed a surprising trend. The carbon content went from 51.9% at 450 °C to 46% at 600 °C, while it usually increases with increasing pyrolysis temperature (Weber & Quicker, 2018). According to Lataf et al. (2022), this inconsistency was due to the heterogeneity of the GW. To test the sensitivity of the results, this scenario included a carbon content of 66% for the biochar produced at 450 °C (Roberts et al., 2010) and 83% for the one produced at 600 °C (Tu et al., 2022).

Halving the assumed increase of biomethane production in AD worsens the results for GWP, FPMF, TA, HcT, and FRS due to the reduced substitution of natural gas. However, the preferred system only changes for FRS, where the GW-600 PYROc becomes preferred over the GW-450 PYROc system. In the base scenario, the GW-450 PYROc system achieves a larger substitution of natural gas and, therefore, incurs a larger penalty when the biomethane enhancement effect is reduced. In addition, the GW-600 PYROc system benefits more from electricity substitution, which remains unaffected in this scenario. For this same reason, the preferred option only changes for GWP and FRS when the produced biomethane is not injected into the grid but used to produce electricity. Together with FPMF and TA, these show worsened results. Producing electricity improves the results for HcT and HncT, because more of the toxic impacts associated with electricity production are avoided. The

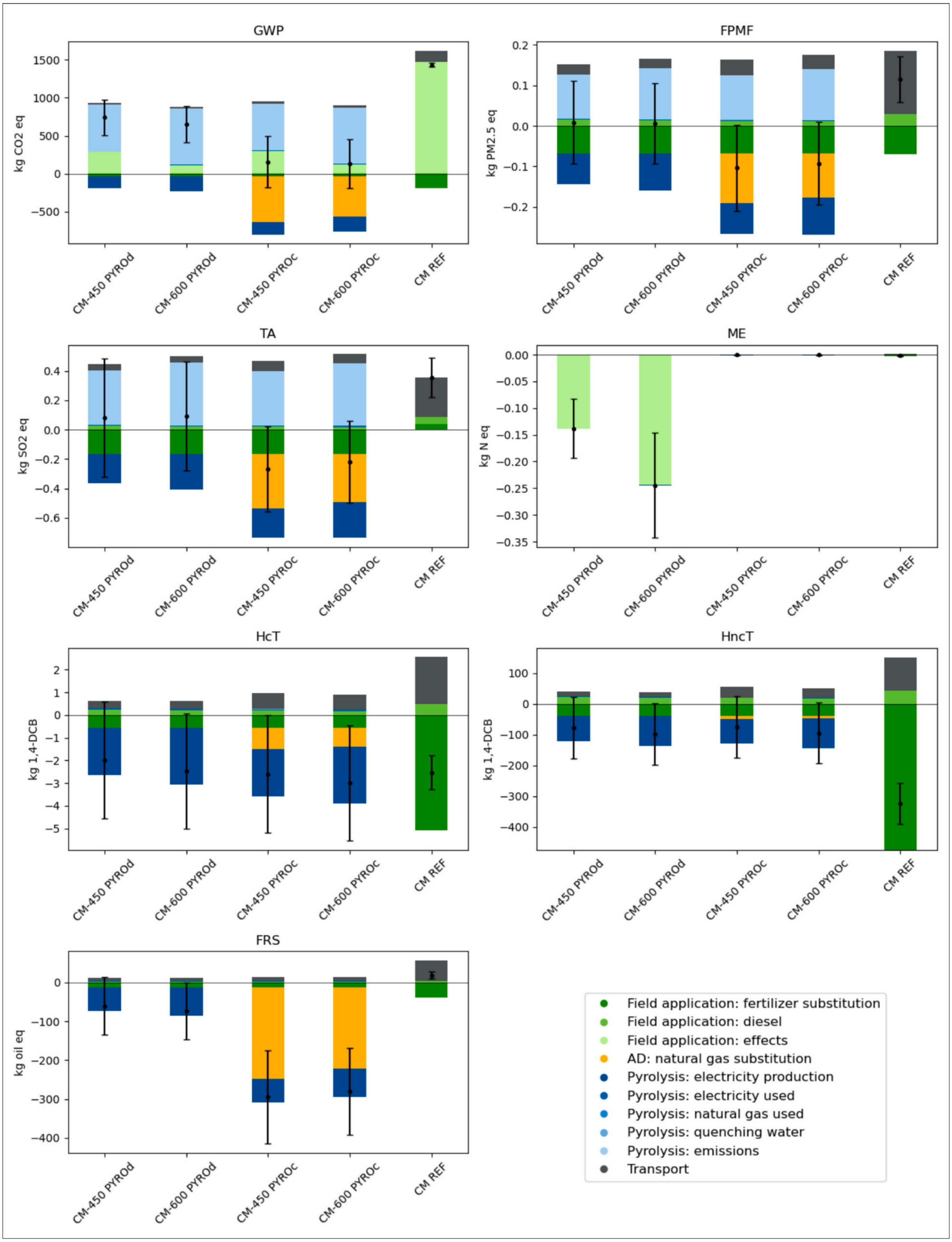

Fig. 3. CM results accounting for parameter uncertainty, including contribution analysis. Bars represent 95% confidence intervals. Abbreviations: GWP (global warming potential), FPMF (fine particulate matter formation), TA (terrestrial acidification), ME (marine eutrophication), HcT (human carcinogenic toxicity), HncT (human non-carcinogenic toxicity), FRS (fossil resource scarcity).

Table 1
Scenario analysis results showing changes in the preferred option relative to the base scenario. For GW, the second column lists the affected impact categories and the new preferred system (base scenario results: GWP: GW-450 PYROc; FPMF: REF; TA: GW-450 PYROd; ME: GW-600 PYROd; HcT: REF; HncT: REF; FRS: GW-450 PYROc). For CM, the third column does the same (base scenario results: GWP: CM-600 PYROc; FPMF: CM-450 PYROc; TA: CM-450 PYROc; ME: CM-600 PYROd; HcT: CM-600 PYROc; HncT: REF; FRS: CM-450 PYROc). Abbreviations: GWP (global warming potential), FPMF (fine particulate matter formation), TA (terrestrial acidification), ME (marine eutrophication), HcT (human carcinogenic toxicity), HncT (human non-carcinogenic toxicity), FRS (fossil resource scarcity).

Scenario	Green waste (GW) Changed preferred system	Chicken manure (CM) Changed preferred system
Biochar field application rate: 5 Mg/ha	FPMF: GW-450 PYROd	No change
Biochar field application rate: 20 Mg/ha	No change	No change
Biochar field effects persistence: 1 year	TA: GW-450 PYROc	No change
Biochar field effects persistence: 10 years	FPMF: GW-450 PYROd	No change
Biochar field effects: lower Cr content (GW-600)	No change	*Not tested for this feedstock*
Biochar field effects: higher C content (GW-450 and GW-600)	GWP: GW-600 PYROc	*Not tested for this feedstock*
Biochar in AD: biomethane production increase + 5.74%	FRS: GW-600 PYROc	No change
Biochar in AD: 50% PAH reduction	No change	No change
Biochar in AD: electricity production	GWP: GW-600 PYROc FRS: REF	FRS: CM-600 PYROc
Pyrolysis (and incineration): heat production	GWP: GW-600 PYROc FPMF: GW-450 PYROd HcT: GW-450 PYROc HncT: GW-450 PYROd FRS: REF	HcT: REF FRS: CM-600 PYROc
Pyrolysis (and incineration): marginal electricity natural gas	FPMF: GW-450 PYROd	HcT: REF
Pyrolysis (and incineration): marginal electricity wind	FPMF: GW-450 PYROd	GWP: CM-450 PYROc
GW reference: ashes in road construction	FPMF: GW-450 PYROd	*Not tested for this feedstock*
GW reference: ashes to landfill	FPMF: GW-450 PYROd	*Not tested for this feedstock*
CM reference: calcium nitrate substitution	*Not tested for this feedstock*	HcT: REF
CM reference: urea substitution	*Not tested for this feedstock*	TA: REF HncT: CM-600 PYROd
CM reference: N fertilizer value 60%	*Not tested for this feedstock*	HcT: REF
LCIA: ReCiPe2016 Midpoint (I)	No change	FPMF: CM-600 PYROc HcT: REF
LCIA: EF 3.0	FPMF: GW-450 PYROd	FPMF: CM-600 PYROc
LCIA: Stepwise2006	ME: GW-450 PYROd	ME: REF HncT: CM-600 PYROd

results for ME remain unchanged, because these solely depend on biochar's field effects.

If incineration and pyrolysis generate heat instead of electricity, the preferred option changes for all relevant impact categories, except TA and ME. The latter are characterized by an important impact from biochar's field effects, being the reduced ammonia volatilization in the case of TA and reduced nitrate leaching in the case of ME. For the impact categories where the preferred option changes, the shift is driven by the amount of energy valorized. Because the reference system produces the largest amount of energy per functional unit, changing the substituted energy from electricity to heat has the greatest effect on its environmental performance, leading to changes in the preferred system. The sensitivity of the results for these impact categories shows the importance of assumptions regarding the substituted energy carrier.

In the base scenario, electricity from pyrolysis or incineration substitutes ecoinvent's consequential electricity mix. Two scenarios instead assume marginal electricity from natural gas or wind. In both alternatives, the results for FPMF, TA, and HncT worsen, while they remain unchanged for ME. As expected, assuming a fossil marginal technology increases the GWP and FRS benefits of the systems, while the opposite is true with a non-fossil alternative. A mirrored trend is observed for HcT. Nonetheless, both alternatives only change the preferred system for FPMF. There, the worsening for the reference system is two to three times higher than the systems with pyrolysis at 450 °C and 600 °C, respectively, because the reference produces more electricity per FU.

The base scenario assumes that, with incineration, the entire GW ash content becomes fly ash used to substitute Portland cement. Two alternative scenarios assume that it is used in road construction or landfilled instead. Together, these scenarios define a range from favoring the reference to favoring pyrolysis. In both alternatives, the impacts of the reference worsen across all relevant impact categories, confirming that the base assumption is conservative. However, the preferred option only changes for FPMF, where the reference shifts from the preferred to the least desirable system, showing a high sensitivity of this impact category to assumptions regarding ash valorization. For HcT and HncT, the reference remains preferred, while pyrolysis systems already outperform it for GWP, TA, ME, and FRS. In conclusion, the results for GWP, TA, ME, and FRS are robust to this modeling decision. The same applies to HcT and HncT, though these must be interpreted with caution given the conservative and simplified way impacts were modeled.

Finally, the results were compared using different LCIA methods. Even though all relevant impact categories have a suitable alternative, the other methods use different units, so only the ranking can be compared. For GWP, TA, and FRS, it is identical across methods. While this is not true for HcT and HncT, the preferred option remains unchanged. For FPMF, the preferred option only changes with EF3, while this is the case with Stepwise2006 for ME.

3.2.2. Chicken manure (CM)

While halving the assumed increase in biomethane production considerably worsens the results for all relevant impact categories, except ME and HncT, the preferred option does not change. This insensitivity in the ranking of the CM systems indicates that it is robust to uncertainty regarding the magnitude of the biomethane enhancement effect. As the assumed PAH content of the CM-450 biochar nearly exceeds the threshold of the European Biochar Certificate (Lataf et al., 2022), the insensitivity of the results to a potential reduction shows the robustness of the conclusions for a worst-case biochar. When the biomethane is valorized by producing electricity instead of injecting it into the grid, FRS is the only impact category for which the preferred option changes. Less fossil fuels are displaced because of the energy loss when producing electricity and because the electricity mix is not made up entirely of fossil fuels.

Using the syngas to produce heat instead of electricity changes the preferred option only for HcT and FRS. For HcT, this comes from the pyrolysis systems losing their benefit from avoiding toxicity impacts associated with electricity production. For FRS, the CM-600 PYROc system benefits more from the shift from electricity to heat, because it valorizes more pyrolysis energy per FU, compared to the CM-450 PYROc system. This additional avoidance outweighs the advantage the latter derives from the greater substitution of natural gas per FU through increased biomethane production. Assuming electricity production, two scenarios replace ecoinvent's consequential electricity mix with electricity from natural gas or wind. The results change in the same way as they did for the GW systems, except for the CM reference, which does not include any energy production. However, different impact categories show a change in preferred option, as it changes for HcT with the fossil alternative and for GWP with electricity from wind.

Three modeling choices regarding the reference are investigated. The first is a change of the type of substituted nitrogen fertilizer. Regarding FPMF and TA, the results worsen when calcium nitrate is assumed but improve when this is urea. The opposite happens with HcT and HncT. Second, the nitrogen fertilizer value was changed from 20%, following French legislation, to 60%, following Flemish legislation. A higher fertilizer value means less fertilizer can be applied, reducing the total amount of nitrogen applied and the associated N_2O emissions from the field. While this alternative scenario improves the results for all relevant impact categories, except FPMF and TA, the preferred option only changes for HcT.

Third, the possible effects of using raw CM instead of pellets are discussed qualitatively. The pellets have been through composting, hygienization, and pelletization. These steps are omitted from all systems. However, it is arguably unnecessary to treat the CM before pyrolysis (Manogaran et al., 2022). In addition to reducing cost, pyrolyzing raw CM could enhance environmental benefits in two ways. First, this would increase the amount of carbon stored in the biochar, because some of the carbon is lost during composting. According to Nordahl et al. (2023), 140 kg of CO_2 and 2.82 kg of CH_4, or 79 kg CO_2 equivalents, are emitted when composting 1 Mg of CM. Assuming all CH_4 emissions and half of the CO_2 emissions are avoided with pyrolysis, the GWP could be reduced by about 150 kg of CO_2 equivalents. Second, more energy could be valorized from pyrolyzing raw CM because some of it is lost during composting. Furthermore, both pelletization and hygienization would become redundant. The latter is true because pyrolysis achieves higher temperatures than composting, so it is arguably better at eliminating contaminants. Overall, this suggests that the environmental performance of the CM biochar systems may be conservatively estimated, as the use of raw manure could provide additional benefits.

Finally, the sensitivity of the results to the chosen LCIA method was investigated. The preferred option for FPMF changes when using the Individualist version of ReCiPe2016 or the EF3 method. The former also changes the preferred option for HcT, while Stepwise2006 changes the preferred options for ME and HncT.

3.3. Comparison of results to literature

3.3.1. Green waste (GW)

To make meaningful comparisons, results from other studies would ideally have the same impact category, impact assessment method, and functional unit, as well as similar system boundaries. As most LCAs of biochar have focused on GWP (Matuštík et al., 2020), the results for this impact category are most easily compared. This is even more true because GWP is expressed in the same units (CO_2 equivalents) across multiple impact assessment methods.

The study by Azzi et al. (2022) provides the most meaningful comparison for our analysis of GW, because they assume the same reference use of an identical feedstock. With a fossil-energy background system, biochar yields a slight increase in emissions compared to the reference. Even though our results show negative emissions (−53 to −122 kg CO_2 per Mg dry GW), the results of Azzi et al. (2022) are similar to ours when considering the magnitude of negative emissions reported in other studies, going from −190 kg to −1.41 Mg CO_2 per Mg dry biomass (Field et al., 2013; Hamedani et al., 2019; Muñoz et al., 2017; Roberts et al., 2010; Thomsen, 2021; Tisserant et al., 2022). The main reasons for these vast differences are the feedstock's reference use and the associated system boundaries. More specifically, whether the reference includes energy valorization, which would be lost when producing biochar instead, or whether no alternative biomass use is considered.

However, the relatively low carbon content of the GW biochars in our study reduces their GWP benefits. While depending on many factors (Tomczyk et al., 2020), carbon contents of at least 70% are common, especially for woody feedstocks (Azzi et al., 2022; Wang & Wang, 2019). The carbon content of the GW biochars is only 46% (600 °C) and 51.9% (450 °C) (Lataf et al., 2022). Furthermore, we conservatively assumed all the ashes in the reference are used to substitute Portland cement, avoiding 130 kg CO_2 equivalents per Mg of treated GW. In other, less conservative, reference systems, there is no such avoidance.

When looking at impact categories other than GWP, comparisons are hindered by different functional units, both in studies using the same impact assessment method (Muñoz et al., 2017; Tisserant et al., 2022) and in studies using a different method (Hamedani et al., 2019). Different functional units and system configurations also contribute to hindering comparisons of the results for the PYROc systems, in addition to the difficulty due to this study's novel application pathway. There are LCAs of integrations of pyrolysis and AD, but these do not add the biochar in the AD reactor (Wang et al., 2021), focus on waste treatment through AD instead of through pyrolysis (Tian et al., 2023), or focus on biogas production (Mosleh Uddin et al., 2022). As a result, the systems in these studies are not functionally comparable to the PYROc systems considered here.

3.3.2. Chicken manure (CM)

LCAs of biochar are mostly on woody biochar, while manure-based biochars are studied much less. LCAs on biochar from pig manure (Hamedani et al., 2019), cattle manure (Struhs et al., 2020), and sewage sludge (Barry et al., 2019) use other LCIA methods and exclude the alternative biomass treatment, making meaningful comparisons difficult. Field et al. (2013) use consequential modeling for their LCA of biochar made from spent grains from breweries, providing a meaningful comparison because both modeling and feedstock are similar. The spent grains would alternatively be used as animal feed, substantially burdening the biochar due to increased corn and soy production. Additionally, drying the grains worsens the GWP impact. Field et al. (2013) found a GWP impact of −970 kg CO_2 per Mg dry biomass, differing only slightly from the CM results in this study (PYROd: −799 to −919 kg CO_2 per Mg dry CM) due to a greater assumed energy production from pyrolysis.

3.4. Comparison of green waste (GW) and chicken manure (CM)

While the previous sections examined how pyrolysis temperature and application pathway affect the LCA results for GW and CM biochars separately, this section provides another central aspect of the study by directly contrasting the feedstocks. In line with the consequential modeling approach, this is done using changes in environmental impacts relative to the respective reference systems, expressed per Mg dry biomass (Table 2). Because the functional unit, impact assessment method, and system boundaries are consistent across both feedstocks, observed differences can be attributed directly to feedstock choice, enabling a robust and internally consistent comparison.

Three groups of impact categories can be defined: (i) those where CM consistently outperforms GW (GWP, FPMF, HcT, and FRS), (ii) those where results depend on pyrolysis temperature and biochar application (TA and ME), and (iii) one where both feedstocks perform similarly (HncT). For the latter, CM shows a slight advantage when pyrolysis happens at 450 °C, whereas GW performs slightly better with pyrolysis at 600 °C. However, the differences are marginal (≤6%) and the pyrolysis systems perform worse than the reference.

CM outperforms GW across all four pyrolysis systems for GWP, FPMF, HcT, and FRS. For HcT, however, the PYROd systems with CM do not even achieve an environmental benefit relative to the reference. This reflects the importance of the avoided emissions from natural gas substitution, which are required to meet the environmental performance of the reference system. The avoided emissions from fertilizer substitution in the latter are nine times bigger than in the pyrolysis systems. The substitution of natural gas makes up for about 20% of this deficit. Furthermore, the results regarding toxicity-related impacts should be treated with caution because of simplified modeling. For GWP, FPMF, and FRS, the reason why CM is the preferred feedstock follows a similar pattern. The CM reference system performs worse than the GW

Table 2
Comparing GW and CM based on their impacts relative to their respective reference. Results are expressed per Mg dry feedstock, with the second column showing the unit for each impact category. Abbreviations: GWP (global warming potential), FPMF (fine particulate matter formation), TA (terrestrial acidification), ME (marine eutrophication), HcT (human carcinogenic toxicity), HncT (human non-carcinogenic toxicity), FRS (fossil resource scarcity). Highlighted cells in bold show the feedstock with the best environmental performance.

Impact category	Unit	Feed-stock	PYROd (450 °C)	PYROd (600 °C)	PYROc (450 °C)	PYROc (600 °C)
GWP	*kg CO2 eq*	*GW*	−146.523	−179.988	−819.224	−757.779
		CM	**−755.509**	**−853.854**	**−1399.732**	**−1422.999**
FPMF	*kg PM2.5 eq*	*GW*	0.022	0.070	0.022	0.039
		CM	**−0.117**	**−0.119**	**−0.239**	**−0.227**
TA	*kg SO2 eq*	*GW*	**−0.760**	**−0.332**	−0.113	−0.034
		CM	−0.301	−0.291	**−0.686**	**−0.631**
ME	*kg N eq*	*GW*	0.001	−0.136	0.001	0.001
		CM	**−0.150**	**−0.266**	0.002	0.001
HcT	*kg 1,4-DCB*	*GW*	5.136	4.329	4.449	3.739
		CM	**0.579**	**0.067**	**−0.074**	**−0.511**
HncT	*kg 1,4-DCB*	*GW*	286.914	**241.817**	291.360	**245.636**
		CM	**269.979**	248.065	**274.210**	251.803
FRS	*kg oil eq*	*GW*	104.248	83.803	−166.104	−148.405
		CM	**−86.884**	**−101.091**	**−344.137**	**−328.364**

reference, which benefits the CM pyrolysis systems in this relative comparison. For example, the CM reference has GWP emissions that are about double those of the GW reference. This difference arises because the GW reference includes significant avoided emissions from the substitution of Portland cement and electricity, whereas the CM reference only includes fertilizer substitution. For GWP, these avoided emissions are about 2.5 times bigger for the GW than for the CM reference. For FPMF and FRS, the transport requirements in the CM reference, which are eight times higher than in the GW reference, further contribute to this contrast. In terms of magnitude, the environmental benefit for GWP is approximately twice as large for CM as for GW in the PYROc configurations, and up to five times larger in the PYROd systems. For FPMF, GW pyrolysis systems do not provide a benefit compared to the reference. A similar pattern is observed for FRS in the PYROd systems, whereas in the PYROc configurations, the benefit for CM is roughly twice that of GW.

For TA and ME, the preferred feedstock depends on system configurations. Regarding ME, the PYROc systems show nearly identical, non-existent effects across feedstocks because these impacts are largely driven by changes in nitrate leaching. In the PYROc systems, the biochar application rate is too low, due to local fertilizer regulations, for this effect to be included in the modelling. In contrast, the PYROd systems incorporate this effect, resulting in measurable improvements relative to the reference. Both feedstocks show reduced impacts, with a greater improvement for CM, reflecting the modeling assumptions. Based on Borchard et al. (2019), CM biochar is assumed to reduce nitrate leaching more strongly than GW biochar. For biochars produced at 450 °C, the assumed reduction is 0% for GW and 12% for CM. At 600 °C, these reductions increase to 12% and 24%, respectively.

For TA, GW provides the greatest environmental benefit in direct biochar application systems, whereas CM performs better in cascading-use systems. This difference is again linked to modeling assumptions regarding biochar's field effects, specifically reductions in ammonia volatilization. Based on Sha et al. (2019), this effect was assumed to occur only for GW biochar. Furthermore, as with other field effects, the application rate in the PYROc systems is too low to produce a measurable impact. In these cascading-use systems, CM becomes the preferred feedstock for two main reasons that dominate in the absence of field effects. First, the impact of the reference system is 8.4 times higher for CM than it is for GW. In the case of GW, emissions from incineration are more than offset by avoided emissions from cement and electricity substitution, which are 9% higher than the emissions from incineration and transport. The CM reference lacks such avoided emissions, so transport emissions, which are 8 times greater than in the GW reference, are not offset. Because this comparison of feedstocks is made relative to the reference systems, this creates an advantage for CM. Second, due to its higher nutrient content, biochar derived from CM provides a greater fertilizer substitution effect across all pyrolysis systems, including the PYROc configurations. CM biochar contains approximately twice as much potassium and ten times as much phosphorus as GW biochar. Consequently, the resulting avoided emissions are about 25 times higher for CM biochar.

While no feedstock dominates across all impact categories, the results indicate that CM provides broader environmental benefits overall. However, GW shows specific advantages when minimizing TA is the priority and biochar, produced at either 450 °C or 600 °C, is directly applied in soil. Therefore, feedstock selection should depend on the targeted environmental objective.

3.5. Limitations and future research

There are several limitations to this research that future studies could address. While multiple field effects of biochar were included, others, such as impacts on soil quality, irrigation demand, and drought resistance, were excluded due to insufficient data for robust modeling. Biochar has frequently been reported to improve soil functioning, water retention, and microbial activity, although the magnitude of these effects is highly context dependent (Schmidt et al., 2021). Their exclusion may therefore lead to an underestimation of the environmental benefits associated with biochar application. Consequently, the comparative performance of the biochar systems reported in this study should be regarded as conservative. However, the magnitude of this potential underestimation remains uncertain. Future research should focus on generating field-scale, long-term data to better quantify these effects and their agronomic implications, thereby improving their representation in LCA studies. Even for one of the key field effects identified in the contribution analysis in this LCA, namely carbon sequestration, there is scope for improvement. This study applies a static modeling approach, whereas research indicates that such methods may underestimate the climate impacts of biomass soil carbon emissions by a factor of two to three (Karlsson et al., 2025). This is relevant for the CM reference, which includes such emissions.

The results are also subject to uncertainty due to the simplified representation of pyrolysis processes and the reliance on literature-based parameters. Future work could improve the robustness of such assessments by integrating kinetic and process-level models that more accurately describe biomass decomposition, product yields, energy balances, and the formation and behavior of contaminants under varying operating conditions. In particular, experimental determination of energy balances would be valuable, given the importance of energy substitution in the results. Similarly, pilot-scale experiments could provide more consistent estimates of the effects of different biochars on anaerobic digestion under varying conditions. This is important because the contribution and sensitivity analyses highlight the significant role of increased biomethane production in shaping the LCA results. Additionally, incorporating differences in infrastructure between the modeled systems could further improve the accuracy of the assessment. Such developments would enable more precise estimation of system performance and associated environmental impacts, thereby strengthening the reliability of LCA results. Finally, as no system outperforms all others across all impact categories, the application of weighting may be useful to further support decision-making.

4. Conclusion

This life cycle assessment compared valorization pathways, with and without pyrolysis, for the woody fraction of green waste and chicken manure pellets. The preferred option depends on the impact category, meaning weighing the results could be relevant to support decision making. Overall, chicken manure systems perform better across most categories. Pyrolysis is generally favored for climate change and resource use, while reference systems perform better for toxicity-related impacts. However, the latter are subject to greater uncertainty due to simplified modeling. Furthermore, conservative assumptions benefiting reference over pyrolysis systems mean results should be interpreted with caution for impact categories where pyrolysis is not preferred. Across systems, environmental performance is generally driven by market substitution effects (i.e., electricity, biomethane, and mineral fertilizers), biochar carbon sequestration, and field-level effects. Substitution effects dominate most impact categories, while field effects are generally less influential. These drivers determine when different applications are preferable. Cascading use, where biochar is first applied in anaerobic digestion, is preferable when climate change or fossil resource scarcity are the priority and the system aims to minimize system-wide impacts. In contrast, direct soil application becomes preferable when local nutrient impacts are prioritized, and with sufficiently high application rates. The performance of biochar systems also depends on the counterfactual use of the biomass, with the assumed reference systems being common treatment options. When biomass is used for bioenergy and, hence, provides substantial substitution benefits, diverting biomass to biochar requires sufficient gains in sequestration or additional substitution to be justified. For nutrient-rich feedstocks such as manure, fertilizer substitution and nitrogen-related emissions are particularly important. These findings highlight that biochar deployment should be guided by context-specific priorities, especially the relative importance of energy substitution versus local soil and nutrient management.

5. Additional information

The different sections of the Appendix provide further details (A1: life cycle inventory; A2: full results; A3: life cycle interpretation; A4: energy content calculations; A5: mass and carbon balances). The authors declare no competing interests.

6. Declaration of generative AI and AI-assisted technologies

During the preparation of this work the authors used ChatGPT to improve writing. After using this tool, the authors reviewed and edited the content as needed and take full responsibility for the content of the published article.

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.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biortech.2026.135219.

Data availability

Data will be made available on request.

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