Life cycle assessment of urban uses of biochar and case study in Uppsala, Sweden

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ABSTRACT

Biochar is a material derived from biomass pyrolysis that is used in urban applications. The environmental impacts of new biochar products has however not been assessed. Here, the life cycle assessments of 5 biochar products were performed for 7 biochar supply-chains in 2 energy contexts. The biochar products (tree planting, green roofs, landscaping soil, charcrete, and biofilm carrier) were benchmarked against reference products and the oxidative use of biochar for steel production. Biochar demand was then estimated using dynamic material flow analysis for a new city-district in Uppsala, Sweden. In a decarbonised energy system and if biochar stability is high, all biochar products had a better climate performance than the reference, and most applications outperformed biomass use for decarbonising steel production. The climate benefits of using biochar ranged from -1.4 to -0.11 tonne CO_2-eq tonne^{-1} biochar in a decarbonised energy system. In other environmental impact categories, biochar products had either higher or lower impacts than the reference, depending on biochar supply-chains and materials substituted, with trade-offs between sectors and impact categories. This said, several use phase effects of biochar were not included in the assessment due to knowledge limitations. In Uppsala's new district, biochar demand was around 1700 m^3 year^{-1} during the 25 years of construction. By 2100, 23% of the biochar accumulated in landfills, raising questions for end-of-life management of biochar-containing products. Overall, in a post-fossil economy, biochar can be a carbon dioxide removal technology with benefits, but biochar applications must be designed to maximise co-benefits.

KEYWORDS

biochar; carbon dioxide removal; urban; bioeconomy; life cycle assessment; material flow analysis

DECLARATIONS

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Availability of data and material. Data behind the Figs and tables from the manuscript is to a large extent made available in Excel format as part of the supporting information. In addition, data can be supplied on request.

Code availability. Code for analysis of the modelling results is available online at https://github.com/ntropy-esa/P5_uppsala

Authors' contributions. Elias Azzi designed the study, collected the data, performed the modelling, analysed the results, and wrote the manuscript. Erik Karltun contributed to the design of the study, revised the modelling and revised the manuscript. Cecilia Sundberg contributed to the design of the study, revised the modelling, revised the manuscript, and acquired funding for the research project.

HIGHLIGHTS

- Multiple life cycle assessments of novel urban biochar applications were performed.
- Urban biochar use can have better climate impact than reference and oxidative uses.
- Low climate impact requires high biochar stability and decarbonised energy system.
- Biochar leads to shifts of environmental burden between sectors and impacts.
- In a city perspective, adequate management of future biochar waste flows is needed.

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1 Introduction

Several cities in Sweden have set ambitious climate and socio-environmental goals. Climate change goals usually include reaching neutrality between 2030 and 2050, and achieving net-negative emissions thereafter. Net-negative emissions requires removal of carbon dioxide from the atmosphere, and a major carbon dioxide removal (CDR) technologies deemed possible in Sweden is biochar production and use. Biochar, which is the carbon rich residue derived from biomass pyrolysis (Lehmann and Joseph 2009), is recognised to have a higher readiness level than other CDR technologies as it is already available at small and medium scales, while large-scale reactors are being developed (Nemet et al. 2018). Besides, various actors in Sweden (municipalities, public and private companies) have already invested in biochar production facilities.

1.1 Biochar in urban areas

The increasing interest in biochar production around urban areas is motivated, beyond CDR, by the multiple applications that biochar has in urban areas and the availability of low-grade biomass for biochar production in these areas. The most established application of biochar in Sweden is in constructed soils in urban environments. Trees planted in hard surface areas, suffering from soil compaction, have been re-planted in blends of macadam, biochar and compost since 2012 (Stockholm Stad 2020). Today, biochar-macadam structural soils are used for tree planting and rain-gardens in several Swedish cities, contributing also to storm water management. Since 2020, biochar has been added to lightweight mineral soils used in extensive green roofs (Pettersson 2020). Several soil manufacturers also now offer landscaping soil containing biochar as part of their standard catalogue (Hasselfors AB 2021).

In parallel, several new biochar applications are being developed. Adding small biochar fractions to concrete mixes is currently being tested, first for the production of carbon-neutral charcrete elements like garden tiles, tree pits or benches (Vinnova 2021). Biochar for filtering applications is also investigated (Jayakumar et al. 2021), but commercial applications are not yet common in Sweden. While biochar filters for removal of contaminants like PFAS is today excluded from commercial applications in Sweden (McCleaf 2020), the use of biochar as a support for biofilm growth and carbon oxidation (BOD removal) in water has shown promising results (Perez-Mercado et al. 2018). Other applications such as biochar-enriched asphalt, biochar-mortar, activated biochar or carbon fillers for electronics also exist but were not included in the scope of this work.

While urban applications of biochar are spreading, the potential environmental impacts of these new products have not yet been quantified in life cycle assessment (LCA) studies, nor benchmarked against current technologies. In addition, a challenge of the bioeconomy is that the climate impact of bio-based products (whether bioenergy, biomaterials, or biorefineries) is usually highly dependent on the type of biomass used, its supply-chain, the time perspective, reference land uses, as well as the modelling choices (Ahlgren et al. 2015; Brandão et al. 2021). In the case of biochar-based products, the type of biomass, the biochar properties, and the design of the biochar product may significantly influence the environmental footprint of the final product. In addition, a common critique to the use of biochar for C sequestration is that the biochar could also be used as a fuel (Peters et al. 2015) to substitute for instance the use of coal and coke in the metallurgical industry (Riva et al. 2019) for which projects exist in Sweden (Envigas AB 2020). Few LCA studies have performed comparisons between oxidative and non-oxidative uses of biochar (Peters et al. 2015), or included the benefits of the use of biochar products in terms of material substitutions (Fryda et al. 2019). Therefore, there is a need to apply LCA to the new urban uses of biochar, considering these effects, and the potential variability induced by different biochar supply-chains.

1.2 Uppsala case study: scaling-up LCA results

Some cities in Sweden are expecting population growth and therefore new districts are being planned and built. Urban expansion is tantamount to high energy and material consumption during construction, and the choice of infrastructure sets the emissions during the use phase of the district. This is the case of Uppsala, Sweden, where the municipality is planning the construction of a new city district for 57 000 inhabitants, to be built between 2025 and 2050. The ambition is to test and deploy new technologies, including biochar, with the aim of reducing the district's environmental impact. In cooperation with the municipality, the district was selected as a case study to estimate the potential for biochar C sequestration via the urban biochar applications. Working at the scale of a district allowed to place LCA results in perspective at the product level. From an industrial ecology perspective, the district scale allows to complement LCA results with material flow dynamics, which are of interest for both the management of carbon sinks and municipal development planning.

1.3 Aims and objectives

The aim of this study was to assess the environmental life cycle impacts of 5 urban applications of biochar (tree, green roof, landscaping soil, charcrete, biofilm) and one oxidative application (pig iron production). The objectives of the LCA were to: (i) evaluate the sensitivity of the LCA results of urban biochar applications with respect to biomass, pyrolysis, or reference land use assumptions, (ii) quantify the biochar demand and carbon sequestration potential of each urban biochar application, (iii) identify environmental hotspots in the life cycle of each application and its reference, and (iv) rank the environmental benefits provided by the different urban biochar applications, relative to reference technologies. The Uppsala case study was then used to convert the LCA results to the scale of an actual district in order to identify which urban biochar application had largest potential for meeting the municipal climate objectives, and highlight potential differences between LCA results at the product level and LCA results at the district level.

2 Methods

2.1 Scope definition

2.1.1 System boundaries

The system boundaries used to describe the 5 urban biochar products, pig iron production, and their references included: the production and supply of biochar, the production and supply of other materials, manufacturing of the biochar product, its use and disposal (Fig 1). The production and supply of biochar included biomass production, reference land or biomass use, biomass pyrolysis, and valorisation of the pyrolysis co-products (Fig 1). The biochar use phase and end-of-life, which are key in assessing biochar carbon sequestration and other side-effects (Azzi et al. 2021), are here part of the lifecycle of the biochar product.

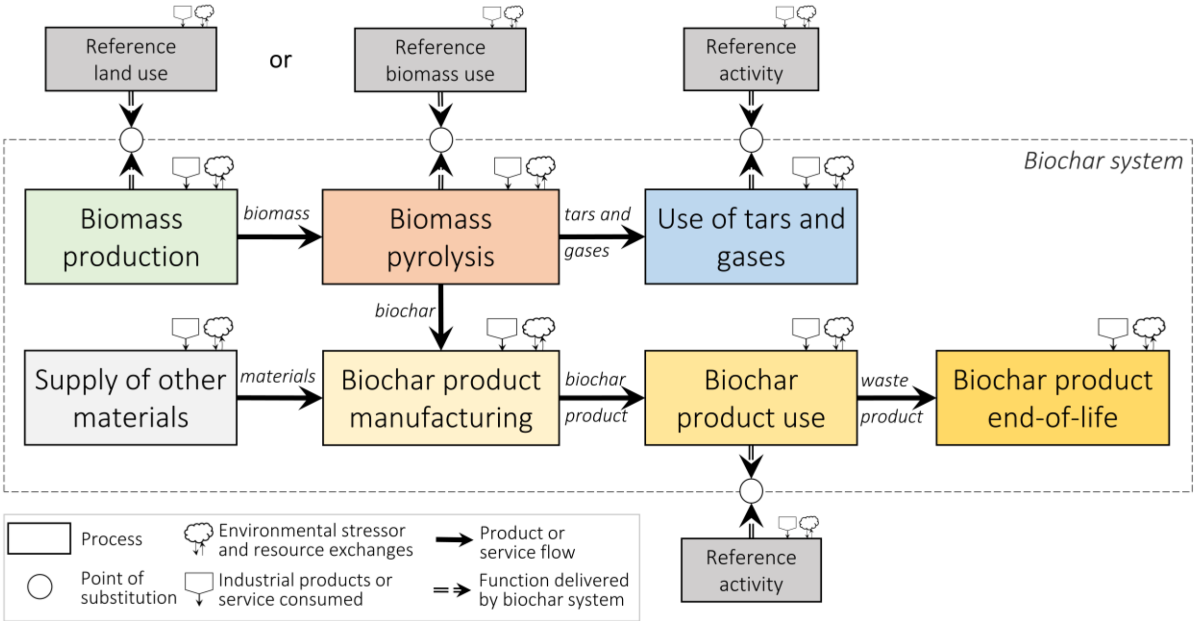

Fig 1 Generic flowchart representing lifecycle of a biochar product. The reference products (without biochar) include only supply of other materials, product manufacturing, use and disposal

Energy co-production in the biochar supply-chain or during disposal of other materials via incineration was handled by substitution. Each of the biochar products were modelled with 7 different biochar supply chains, further described in section 2.2. The impact assessment focused on climate change, characterised using GWP_{100}, but also included the 15 midpoint impact categories from the International Life Cycle Data system (ILCD) (JRC 2012) relating to resource depletion, human toxicity, and ecotoxicity.

2.1.2 Functional units

Several functional units (FU) were used to answer the research questions. For the sensitivity analysis to the type of biochar and its supply chain, the FU was 1 m^3 or 1 tonne of biochar produced (cradle-to-gate). For environmental hot-spot identification in each product-system, we used 1 unit of product (i.e. 1 tree planted, 1 m^2 year of green roof, 1 m^3 landscaping soil, 1 concrete tile of dimensions 40x40x4 cm, 1 m^3 water treated, and 1 kg pig iron). For the benchmarking of biochar applications against each other, the FU selected was: 1 tonne of biochar produced and used. For the Uppsala case study, the FU was set to the amount of final products needed to build and maintain the district over the period 2025-2100 (see 2.3).

2.2 Life cycle inventory data

2.2.1 Biochar supply chains

In total, 7 biochar supply chains were considered combining 4 biomass types and 3 pyrolysis reactors (Table 1). The 4 biomass types and their reference land or biomass use (RLBU), considered for their relevance in a Swedish context, were: (i) urban garden waste (GW), otherwise combusted for district heat production; (ii) wood pellets (WP), from residues of the wood processing industry, otherwise combusted for district heating; (iii) logging residues (LR), from tops and branches otherwise left to decay in the forest (Hammar et al. 2015; Azzi et al. 2019); and (iv) short-rotation coppice willow woodchips (WL), cultivated on previously fallow land (Hammar et al. 2014).

Supply of GW included short transportation (10 km in waste collection trucks and private cars) and chipping at the pyrolysis site. WP production was represented by the eponym ecoinvent activity, edited to reflect Swedish energy conditions, and 150 km of regional transportation to site of use were added. Cultivation and harvesting of LR and WL was modelled as in Hammar (2014, 2015). LR were sourced from the region (150 km transport), while WL was cultivated on nearby agricultural land (50 km transport). All transport assumptions are gathered in Supporting Information (SI). Biomass moisture during transport was set to 50% except for pellets where it was 10%. For LR, partial drying in the forest was neglected. Biomass was dried before pyrolysis to 10% moisture, using heat from the pyrolysis and external electricity.

Table 1. Biochar supply chains modelled, and their differences in terms of biochar carbon content, biochar bulk density, reference land or biomass use (RLBU), excess district heat co-production, electricity use during biomass drying and pyrolysis, and start-up fuel (expressed per kg of biochar). HOB: combustion in Heat-Only Boiler.

#	Biochar from a given biomass and pyrolysis reactor	Carbon content ($\% \ db$)	Bulk density ($kg \ m^{-3}, dry$)	RLBU	District heat (MJ)	Reactor electricity (kWh)	Start-up LPG (g)
WP-S	Wood pellets, syngas-heated reactor	93.4%	500	HOB	37.4	0.182	2.73
WP-E	Wood pellets, electricity-heated reactor	93.4%	500	HOB	42.2	1.75	0
GW-S	Garden waste, syngas-heated reactor	69.9%	242	HOB	27.3	0.360	2.73
GW-E	Garden waste, electricity-heated reactor	69.9%	242	HOB	32.1	1.93	0
LR-S	Logging residues, syngas-heated reactor	91.6%	194	No harvest	27.3	0.360	2.73
LR-M	Logging residues, mobile reactor	91.6%	194	No harvest	0	0	2.73
WL-S	Willow chips, syngas-heated reactor	81.6%	270	Fallow	27.3	0.360	2.73

Three pyrolysis reactors were considered: syngas-heated reactor with district heat production (i.e. representative of Pyreg and BioMaCon reactors), electricity-heated reactors with district heat production (BioGreen) and mobile syngas-heated reactors without energy recovery (Earth Systems). The reactor type was assumed to not influence biochar yield (set to 25% in all supply chains) nor the biochar properties (here, only influenced by the biomass type). The reactor type influenced other part of the LCA: heat co-production (also influenced by biomass moisture content), electricity input, start-up fuel, quenching water, and reactor manufacturing, supply and disposal (Table 1 and SI). The same direct air emissions from pyrolysis were included for all reactors, based on data from Sørmo *et al.* (2020) (SI).

Biochar carbon content and bulk density were taken from lab analyses provided by VegTech AB for WP pyrolysed in a BioMaCon reactor, by Stockholm Waste and Water AB for GW in a Pyreg reactor, Hjälmsäter Gård AB for LR in a BioMaCon reactor, and literature data for WL (Perez-Mercado et al. 2018). Two biochar stability levels were distinguished depending on the final application: 80% for all soil applications, in line with previous works (IPCC 2019), and 95% for charcrete, which is more conservative than the 100% stability recommended in the EBC C sink voluntary guidelines (Schmidt et al. 2020).

2.2.2 Other material production and supply

Apart from biochar, the biochar products and their reference applications consume other materials, such as Portland cement, gravel, sand, clay, clay granules, macadam in different size-fractions, crushed rocks, horticultural peat, pumice, compost, horticultural fleece, geotextiles, steel and concrete elements, water and fertilisers. LCA data for production of these materials were taken from the ecoinvent database (version 3.6, cut-off system-model) (Wernet et al. 2016) and, whenever relevant, direct heat and electricity inputs were edited to reflect Swedish conditions (see 2.2.3). Data for macadam production were taken from Erlandsson (2010). For peat, it was assumed that the reference land use was “no harvesting of peatland”. Compost was assumed to be produced from the treatment of park residues, and was therefore taken as burden-free. Assumptions for transport to and within Sweden are summarised in SI, for each material. Characterised impacts for production and supply of these materials, per m^3 and per tonne, are presented in SI. Mass and volume contents of materials in each studied product was calculated based on equations and material data presented in SI. Finally, all ecoinvent activities used without edition for background modelling are listed in SI.

2.2.3 Energy system assumptions

In the foreground system, all energy inputs assumed to be consumed in Sweden (electricity, heat, transportation) were replaced by custom activities. This allowed to switch between two configurations: Σ_{avg}, representing Swedish average conditions, and Σ_{fossil}, representing a fossil-based system (Table 2). Results for both energy system configurations were computed in sensitivity analysis, also expanding the representativeness of the results beyond Sweden. For Σ_{avg}, the share of biofuel in the diesel mix was set to 30% and modelled based on rapeseed biofuel from ecoinvent. For Σ_{fossil}, electricity conversion losses from high- to low- voltage were neglected.

Table 2. Set of assumptions for Swedish energy system

Energy input	Σ_{avg} – Swedish average (default)	Σ_{fossil} – Fossil
Heat	Woodchips from logging residues	Natural gas, heat-only boiler
Electricity	Swedish average mix, low-voltage	Natural gas, combined-cycle, high-voltage
Transportation	Fossil diesel, 70%; biofuel[a], 30%	Fossil diesel 100%

[a]Transportation with 100% biofuel had 40% lower climate change impact than transportation with 100% diesel.

2.2.4 Biochar applications

Inventory data for the biochar products were collected through personal communication with companies, municipal organisations, and researchers listed in SI, as well as from literature. Below, descriptions and key assumptions are provided for each biochar product and its reference.

Tree in hard-surface area

Tree in hard-surface area refers to the establishment of a new tree in a structural soil, with or without biochar, as described in Stockholm's handbook for urban greening (Stockholm Stad 2020). The product is 1 tree planted, including a 2-year establishment period, with a soil volume of 15 m^3, covering a pavement area of about 10 m^2. The manufacturing step included production of various parts (steel water inlet, geotextiles, concrete tree pit, and nursery tree), provision of macadam-biochar-soil substrate, excavation of former soil and terracing works, machinery use during installation, transportation of materials to site, and watering and fertilising during the establishment period.

The main difference between the biochar and reference tree planting techniques is the composition of the structural soil. The main layer of the reference structural soil was made of 80%bvp (bulk volume parts) macadam 32-64 mm and 20%bvp of landscaping B soil (i.e. made of 65%bvp sand and 35%bvp horticultural peat), with a bulk density of 1790 kg m^{-3} (including average moisture at delivery). In the biochar structural soil, the main layer was made of 80%bvp macadam 32-64 mm and 20%bvp biochar-compost mix in equal parts, with a final bulk density of the mixed product of 1700 kg m^{-3}.

The use phase was not explicitly modelled. In particular, potential biochar effects such as reduced fertiliser use, stormwater treatment or improved tree growth were neglected. Due to lifetime of several decades, disposal of constructed soils is unknown. Discussion with entrepreneurs revealed that constructed soil may get clogged over time, thus requiring maintenance, or complete renovation (Fridell 2020). Thus, disposal was modelled as the landfilling of the constructed soil in inert landfills after 50 years.

Extensive green roof

The modelled biochar and reference green roofs were inspired from VegTech AB's sedum mats roof, as the company started to market biochar-based sedum mats in 2020 (Pettersson 2020). These extensive green roofs have a total thickness of about 6 cm, and are made of two layers: a water-holding base layer made of synthetic material (1 cm), and a mineral soil layer (3 cm) cultivated with sedum (2 cm) during 2 years prior to installation. The sedum mats are cultivated on a bearing layer, made of a geotextile and plastic net. The manufacturing step included: production of the base layer from recycled and virgin fibres in Lithuania, open-field cultivation of sedum mats in Sweden (including machinery for sowing and harvesting, fertiliser, irrigation), production of mineral soil substrate, and transport to installation site. Other components used during installation of the roof, such as border elements in wood and steel, were excluded as well as energy use for lifting the material on the roof.

The only difference between the biochar and reference green roofs is the composition of the mineral soil. The reference mineral soil had a bulk density of 1250 kg m^{-3} (wet) and was made of was made of horticultural peat, sand, crushed rocks, scoria granulates, and clay granulates. The biochar mineral soil had a similar bulk density, and was made of horticultural peat, sand, crushed rocks, scoria granulates, clay granulates, green compost and biochar. The biochar content was set to 2.5%bvp, which is the amount used by VegTech AB in its product at the time of the study.

The lifetime of both roofs was set to 50 years. During the use-phase, application of fertiliser was assumed to take place annually for the reference roofs, and biannually for the biochar one, following the recommendations of the manufacturer. Disposal was modelled as transport to inert landfill for mineral soil, composting for organic materials, and incineration for plastics.

Landscaping soil

Several types of landscaping soil are used in urban environments, usually categorised by sieving profiles, and clay and organic matter contents. Here, only one type of landscaping soil was modelled (type A) adapted for planting of trees and bushes or grass lawn (Hasselfors AB

2020). The reference soil had a density of 1250 kg m^{-3}, and was made of sand (35%bvp), peat (35%bvp) and clay (30%bvp). In the biochar soil, peat was replaced by 20%bvp biochar and 15%bvp compost, leading to a soil mix with a bulk density around 910 kg m^{-3} (varying with biochar type). A 20% biochar volume is considered as the upper limit recommended for soil mixes (Stockholm Stad 2020). Transport to site of use was set to 50 km, but no explicit use or disposal phase was included. The soil is assumed to remain where applied.

Charcrete elements

Charcrete is made by mixing biochar, sand, gravel, cement and water in different proportions. In 2020-2021, Ecotopic AB developed several charcrete recipes that yielded a product with adequate properties for urban vegetation systems, e.g. floor tiles and tree-pit foundations (Vinnova 2021). In this study, one charcrete developed by Ecotopic AB was modelled but its recipe cannot be fully disclosed due to non-disclosure agreements. The modelled charcrete had a bulk density around 2300 kg m^{-3} (similar to normal concrete) and a biochar to cement mass ratio in the range 0.15 – 1. Reference concrete was modelled via the ecoinvent process 'unreinforced concrete production, with cement CEM II/A, geography CH', which had a bulk density of 2370 kg m^{-3}. The casted product was assumed to be 40cm x 40cm x 4cm floor tiles for urban vegetation systems. Transport to site of use and disposal to landfill were both set to 50 km. No explicit use-phase nor differences in lifetime were modelled.

Biofilm carrier

Moving bed biofilm reactors are a water purification technique that can be used e.g. for carbon oxidation (BOD removal) in drinking water treatment plants. The treatment is performed by microorganism forming a biofilm attached to carriers floating in the reactor. The reference carrier was assumed to be a K1 Anox Kaldnes carrier, made of extruded virgin high-density polyethylene (specific density 0.96 g cm^{-3}), with a 500 m^2 m^{-3} bulk specific surface area and a material weight of 145 kg m^{-3} (McQuarrie and Boltz 2011). A filling rate of 60% was assumed (McQuarrie and Boltz 2011), and a reactor size of 1100 m^3 for an annual treatment capacity of 7 million m^3 water (McCleaf 2020). For the biochar carrier alternative, the same reactor volume was assumed to be filled at 100% with raw biochar (McCleaf 2020).

Service lifetime was set to 10 years for both products (McCleaf 2020). The disposal of spent plastic carrier was incineration (heat and electricity co-generation was included), while biochar carrier were assumed to be landfilled or used in secondary soil applications. Transportation to site of use and disposal were set to 50 km.

Pig iron production

Pig iron is the intermediate product in the manufacturing of steel in a blast furnace. Production of pig iron with biochar was modelled to serve as a benchmark in the trade-off between fossil-fuel substitution and carbon sequestration in a Swedish context. Biochar was assumed to replace 100% of the fossil coke and hard coal used in the pig iron production process available in ecoinvent 3.6 with rest-of-the-world geography (Classen et al. 2009). The biochar equivalence was calculated based on both carbon and energy contents of the materials, and the most conservative outcome was selected (i.e. largest biochar requirement). The corresponding amount of fossil carbon dioxide emission was replaced by biogenic carbon. The reference pig iron production process was left unchanged. Transportation to site of use was 50 km, but no explicit use or disposal was modelled.

2.3 City-district scenario

2.3.1 Demand for final products

To upscale the LCA results from the product-level to the city-district level, the total demand for each product was estimated using 55 parameters listed in SI. The lifecycle of the new city district was divided into a construction phase (2025-2050) and a use & maintenance phase that included some renovation works (2050-2100). The construction phase usually has a high material and energy intensity but is constrained in time (Lausselet et al. 2020).

Urban trees were assumed to be planted along the 50 km of road, on both sides, with an average spacing of 10 meters. For each urban tree, a tree pit in charcrete was used. Additional decorative charcrete elements were assumed to be used, at a rate of 0.064 m^3 $tree^{-1}$ (equivalent to 10 tiles of dimensions 40x40x4 cm). Green roofs were assumed to be installed on 50% of the 837 800 m^2 of roof area, the rest being either used for PV panels or simply not usable. Landscaping soil was assumed to be used in both residential yards (i.e. green areas around the housing buildings) and public parks. Planted areas of residential yards (437 000 m^2) and public parks (184 000 m^2) were estimated from the municipality's GIS model of the future district. Soil depths and areas planted with three kinds of planting were taken from Ariluoma (2021). Annual demand for primary drinking water treatment was assumed to increase linearly with inhabitants moving in, reaching 2.9 million m^3 $year^{-1}$ (51 m^3 $pers^{-1}$ $year^{-1}$) in 2050.

Some renovation or replacement of biochar products were included. The structural soil of urban trees was assumed to have an average lifetime of 50 years (following a normal distribution with a 5-year standard deviation, $\mathcal{N}$ ~ [50, 5]). Charcrete elements were assumed to have an average lifetime of 100 years ($\mathcal{N}$ ~ [100, 10]) and therefore replaced only partly during the timeframe of the study. For maintenance of the parks and yards, landscaping soil was assumed to be used annually, at a rate of 0.001 m^3 $year^{-1}$ m^{-2} of planted area. Spent biochar products were assumed to be landfilled, maintaining the carbon sequestration function. In particular, no material cascade were assumed (e.g. spent biofilm carrier could then be used as soil amendment for constructed soil). Stock-driven material dynamics were calculated using ODYM python package for dynamic material flow analysis (Pauliuk and Heeren 2020).

2.3.2 Biochar, biomass and land requirements, and climate change mitigation potentials

From the dynamic material flow analysis, aggregated biochar demand for the two lifecycle stages of the city district were calculated, both in volume and mass. Four biochar types were used, WP-S, GW-S, LR-S, and WL-S as they have different carbon contents per volume (Table 1). Demand for WP and GW were compared to the Swedish annual production of WP and Uppsala's GW generation. For LR, land requirement was estimated via an average yield of 0.5 tonne ha^{-1} at final felling of forestry operation (Hammar et al. 2015). For WL, land requirement was derived using an average yield of 9.2 dry tonnes ha^{-1} $year^{-1}$ (Hammar et al. 2014). Finally, climate change mitigation and C sink potentials were calculated using the results from the product LCA.

3 Results

3.1 Sensitivity of climate change impact to biochar type and supply-chain

In Fig 2, the cradle-to-gate LCA of biochar production (FU = 1 tonne or 1 m^3) is used to illustrate the sensitivity of the climate change impact of biochar production to biochar type (carbon content and bulk density) and supply-chain (biomass production, reference land or biomass use, reactor type, co-product use), and to the background energy system.

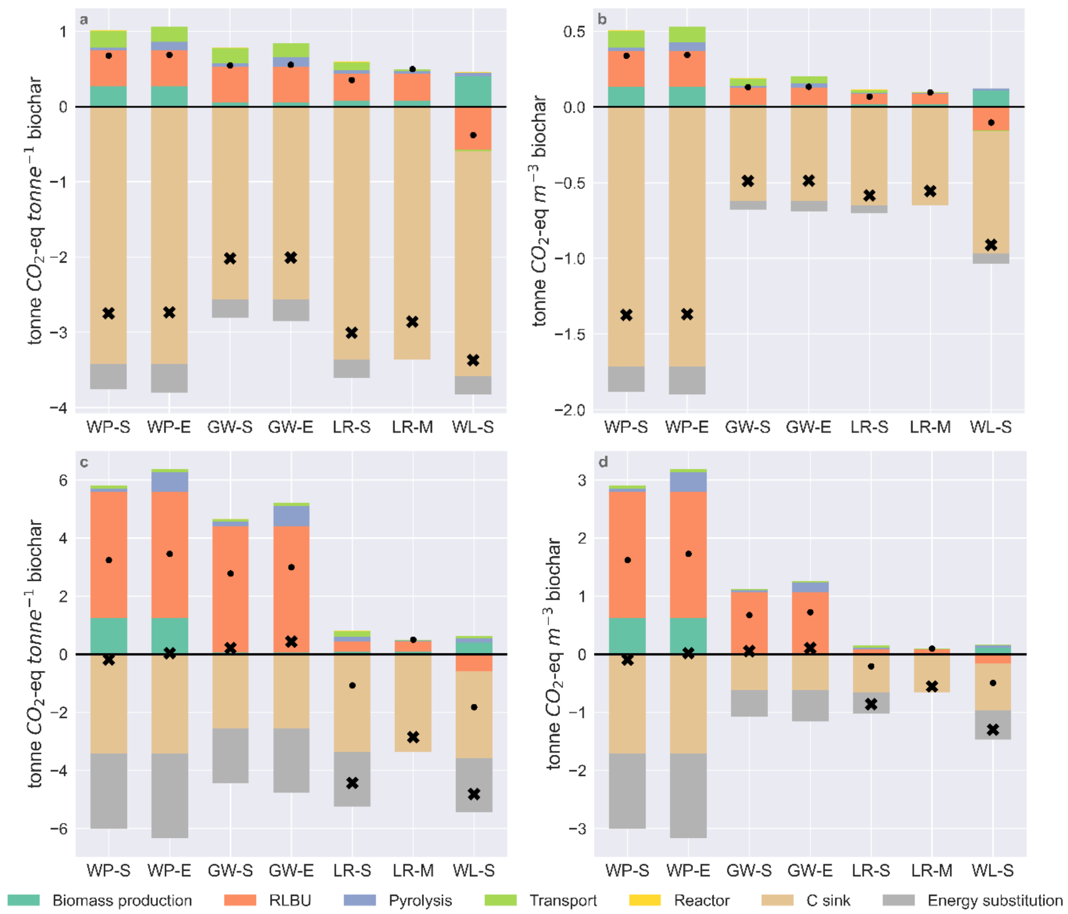

Fig 2 Climate change impact of biochar production for 7 biochar supply chains, expressed per tonne (a, c) and per cubic meter (b, d) of biochar, in two background energy systems (Σ_{avg} – a, b and Σ_{fossil} – c, d). WP: wood pellet, GW: garden waste, LR: logging residues, WL: willow woodchips; S: syngas-heated reactor, E: electricity-heated reactor, M: mobile syngas-heated reactor; RLBU = Reference land or biomass use. A cross indicates the net impact, while a dot indicates the net impact excluding the biochar C sink. *Note: different vertical scale*

In the Swedish average energy system Σ_{avg} (Fig 2a, 2b), all biochar supply chains had a net-negative climate change impact, and the initial biochar C sink was the largest contribution. Per tonne of biochar, the C sink is influenced only by the carbon content (ranging from 69.9% to 93.4%). Per cubic meter of biochar, the C sink is influenced also by the bulk density of the biochar, resulting in about 3 times more C sink for WP biochar than for the other supply chains (Fig 2b, 2d). This is important to bear in mind for applications where biochar content is set by volume. The climate impact from the supply chain excluding C sink was negative for willow biochar (-378 kg CO_2-eq $tonne^{-1}$ biochar) and positive for the other biochars, ranging from 351 to 688 kg CO_2-eq $tonne^{-1}$ biochar. The negative score for WL-S is due to increase of soil organic carbon stocks from willow cultivation (-577 kg CO_2-eq $tonne^{-1}$ biochar), which outweighs the relatively large direct emissions from fertiliser and diesel use during cultivation (+400 tonne CO_2-eq $tonne^{-1}$ biochar). Biomass production impacts are lower for the other biomass types as they require less inputs than cultivated bioenergy crops. Operation of the pyrolysis reactor (S, E, or M) led to smaller variations. Electricity-heated reactors (E) had a slightly higher impact than syngas-heated reactors (S and M), which was partly compensated by the higher amount of co-produced district heat. Net transport impact varied slightly between supply-chains. Reactor manufacturing and disposal had a negligible contribution (< 10 kg CO_2-eq $tonne^{-1}$ biochar).

In the fossil-based energy system Σ_{fossil} (Fig 2, c, d), all energy-consuming and energy producing processes had much larger contributions. Whenever the RLBU was an energy use (WP, GW), the net climate change impact was near 0 because the reference use of biomass displaced district heat from natural gas offsetting the benefits from the biochar C sink and energy co-production during pyrolysis. For LR and WL, the RLBU did not involve energy, and therefore the net impact remained negative. However, in this context it would still be preferable, from a climate change mitigation perspective, to use the biomass for bioenergy without biochar production.

3.2 Comparison of biochar urban applications

In this section, the biochar products are compared to their respective reference product, in the Swedish average energy system Σ_{avg}, to identify environmental hotspots and shifts of environmental burdens.

3.2.1 Climate change impact

All biochar products provided a reduction in climate change impact when compared to the reference technology (Fig 3), with reductions varying in from 14% to 353%. For most products, biochar from WP and WL had the lowest climate change impacts due to the high carbon content per volume of biochar for WP, and the additional soil carbon sink from willow cultivation for WL. Even when not accounting for the biochar C sink, most biochar products had a lower climate impact than the reference product (-124% to -4%). Exceptions were charcrete (+62% to +130%), because of the high cement content in the recipe modelled, and trees planted using WP biochar (+14%). Not accounting for biochar C sink is informative and can be interpreted in several ways: (i) as a worst case scenario or precautionary principle, since biochar stability is uncertain, or (ii) because the “rights” for the biochar C sink may have been traded by the biochar producer as a separate financial product. Finally, it is worth noting that biochar products do not always achieve a “net-negative” climate change impact.

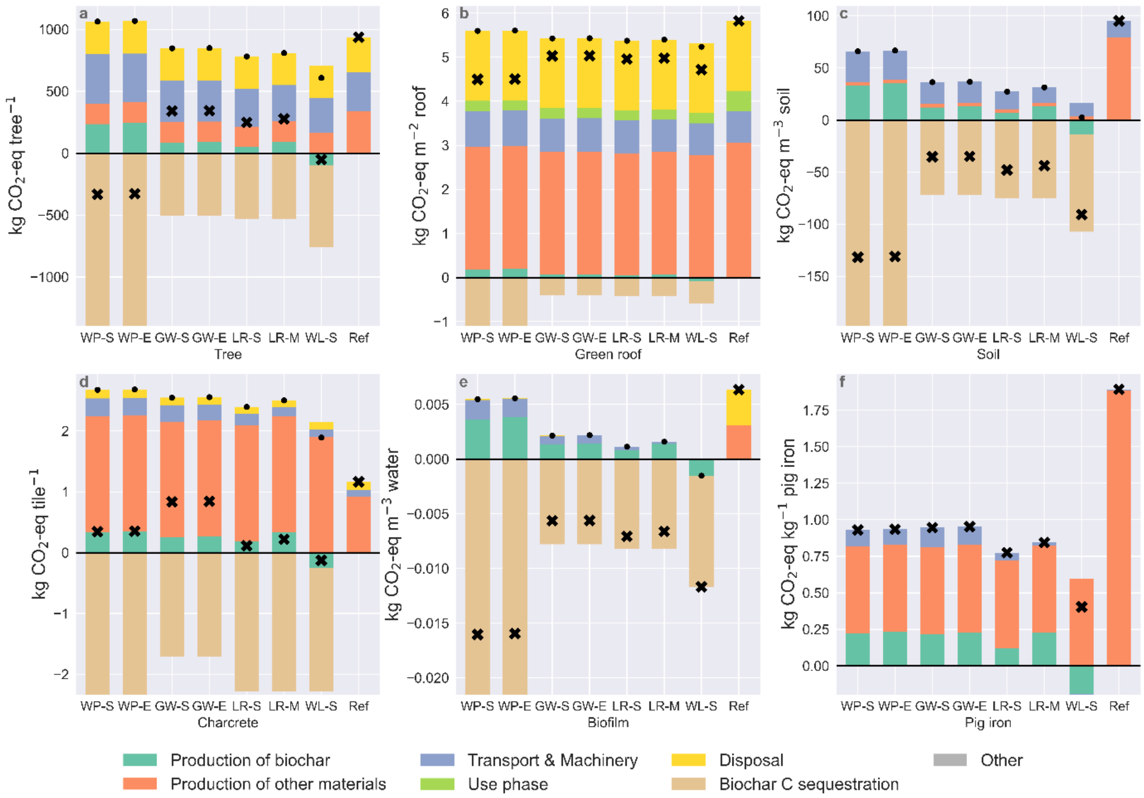

Fig 3 Climate change impact for 6 biochar products and 7 biochar supply-chains, and their respective reference products (Ref), in energy system Σ_{avg}. WP: wood pellet, GW: garden waste, LR: logging residues, WL: willow woodchips; S: syngas-heated reactor, E: electricity-heated reactor, M: mobile syngas-heated reactor. A cross indicates the net impact, while a dot indicate the net impact excluding the biochar C sink

Tree planting. Across biochar supply-chains, the transportation of materials during manufacturing and disposal and the use of machinery represented the largest positive contribution to the climate change impact. The production of biochar and other material represented between 11% and 39% of the emissions excluding C sink. Despite differences in product densities, transport related emissions varied only slightly between biochar products and reference product (18% difference between largest and smallest). **Green roof**. Across biochar supply-chains, the production of other materials contributed to about 50% of the impact, several times more than the impact from biochar production. Transportation for manufacturing and disposal also accounted for a large part of the impact (43%). Between reference and biochar products, the main difference arose from the production of other materials and the fertiliser use during the lifetime of the roof. **Landscaping soil**. Across biochar supply-chains, biochar production and transportation had the highest contributions. Production of other materials, here clay and sand, contributed little to the impact of soil production. Between reference and biochar products, the main difference came from the use of peat in the reference product. **Charcrete**. As biochar-content is set by mass, there was little difference between biochar supply-chains. The main hotspot was the production of other material, due to the high cement content in the studied charcrete formulation. **Biofilm**. For the reference product, plastic production and plastic disposal contribute equally to the climate impact. For biochar filters, the main source of impact is the biochar production. Transportation and disposal played a secondary role. **Pig iron**. Biochar products reduced the emissions of pig iron production by half when compared with hard coal and coke. Only in the case of WL, the pig iron production approaches climate neutrality thanks to the additional soil carbon sequestration from willow cultivation.

3.2.2 Resource use and toxicity impacts

Natural resource use for biochar-based products were either higher or lower than reference products, depending on the biochar supply-chain and the type of product (Fig 4), contrasting with the clearer picture given by the climate change impacts (Fig 3). No clear environmental hotspot could be identified: the contributions of biochar production, other material production, and transportation (during production or disposal) all had varying importance depending on the product and the impact category. For instance, supply of other materials for green roofs dominates the impacts except for minerals and metals were impacts are dominated by transportation (Fig 4b).

The biochar products derived from WP-E (88% of cases), WP-S (67%), GW-E (58%), and WL-S (54%) had *more often than not* increased resource use impacts than the reference products. GW-S and LR-S had more often reduced resource use impacts than the references. This can be related to the lower inputs needed to supply this biomass, and the lower electricity use to produce biochar.

Land use impacts were always highest for WL-based products, followed by WP-based products. Land use impacts were low for LR and GW because these residues were not allocated any direct land use burdens. Water impacts were always highest for WP-E based products, due to both the type of reactor and the type of biomass. GW-S and LR-S biochar production had small negative contributions to water impacts, because of the interplay between biomass production impacts, reference biomass use, and pyrolysis energy substitution.

Results for human toxicity and ecotoxicity impacts are provided in SI. Similar to resource use, no simple conclusion be drawn: biochar products could lead to either increased or reduced impacts compared to the reference products.

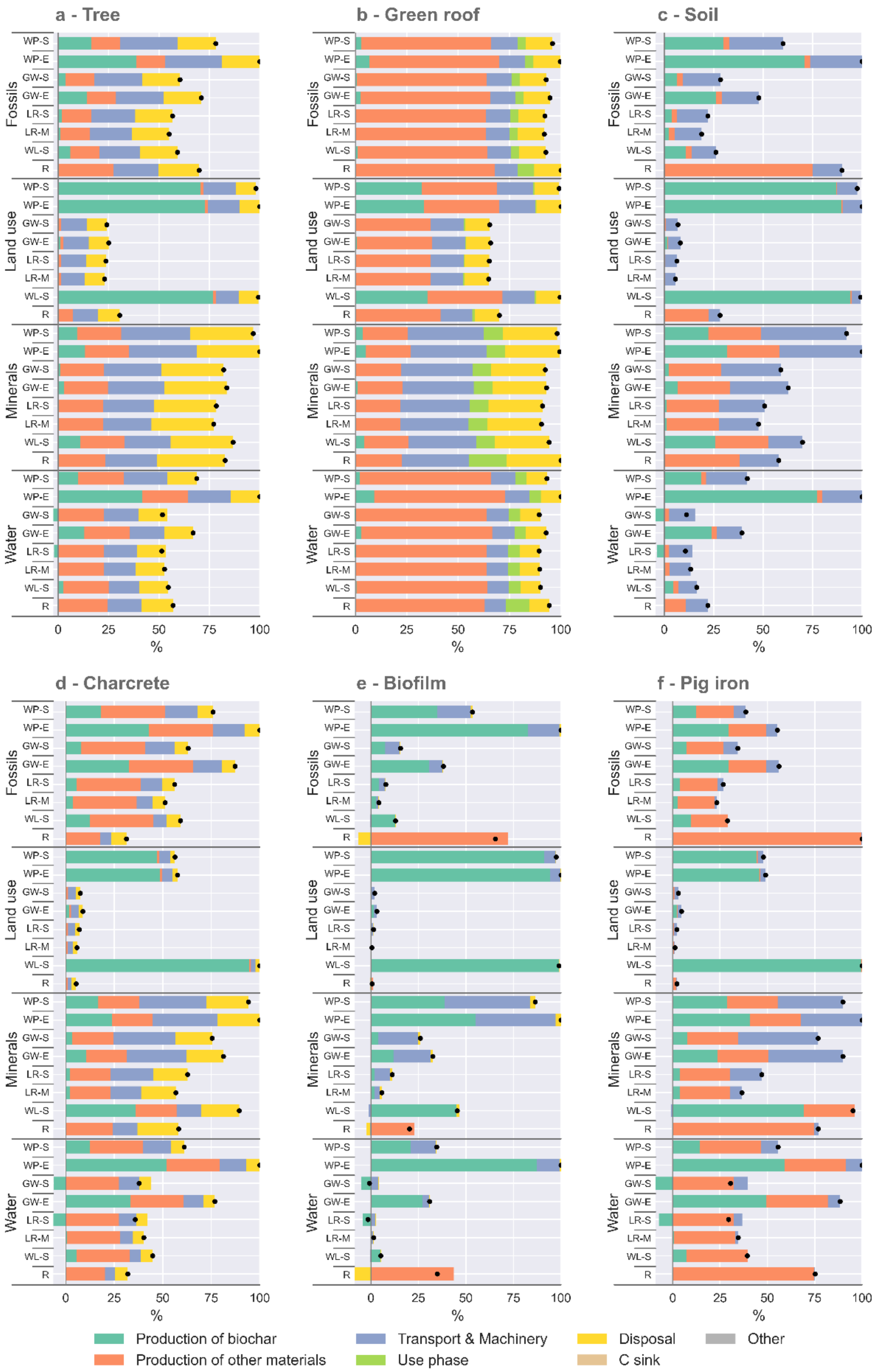

Fig 4 Resource use impacts for 6 biochar products and 7 biochar supply-chains, and the reference products (R) in energy system Σ_{avg}. The values are expressed in percentages relative to the product with highest impact in each category. WP: wood pellet, GW: garden waste, LR: logging residues, WL: willow woodchips; S: syngas-heated reactor, E: electricity-heated reactor, M: mobile syngas-heated reactor. A dot indicates the net impact

3.2.3 Climate-efficiency ranking

In the Swedish average energy system Σ_{avg}, all biochar products except charcrete achieved as much or more climate change mitigation than decarbonising pig iron production, when biochar C sink was included (Fig 5a). In addition, all biochar products led to net climate change mitigation compared with the reference when C sink is included (i.e. net score is negative in Fig 5a). When the biochar C sink is excluded, only WP-S tree planting and charcrete performed worse than the reference. Green roofs were here the best performing product due to both material substitution and fertiliser use reductions. With a mass-based FU, the low-density biochars (GW-S) had twice as much benefits from other material substitutions than high-density biochar (WP-S), in all applications with a biochar content set by volume.

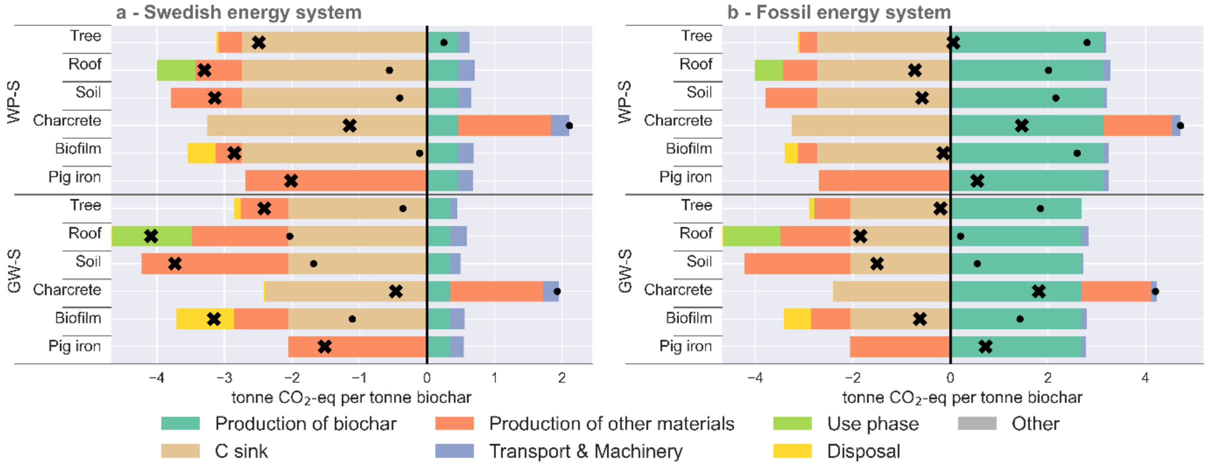

Fig 5 Climate change mitigation benefits per tonne of biochar or bio-coal produced and used in a given application, with respect to the reference technology, in energy systems Σ_{avg} (a) and Σ_{fossil} (b). WP: wood pellet, GW: garden waste, S: syngas-heated reactor

In the fossil energy system Σ_{fossil} (Fig 5b), only the biochar products which had large amounts of other materials substituted had a net climate change mitigation potential, but the potential was largely reduced or even approached zero. Even fossil fuel substitution in pig iron production was not preferable over production of district heat, substituting natural gas, for both WP-S and GW-S. An important aspect for biochar to deliver climate change mitigation are therefore that biochar stability must be high (above 80%), that the biochar product replaces products with a high climate impact, such as peat or plastics, and that heat and power supply are decarbonised.

3.3 Uppsala case study

The LCA inventory and climate change impact results were scaled to the district level using the case of Uppsala's new city district.

Biochar demand and waste flow. At the district level, biochar demand was estimated to be around 43 200 m^3 (±0.8% between biochar types) during the 25-year construction period. This corresponds to an annual average demand of 1728 m^3 $year^{-1}$. During construction, biochar was mainly consumed as landscaping soil, for parks and yards, and as constructed soils for tree planting. Green roofs, concrete elements, and water filter represented together less than 6% of the demand.

For maintenance and renovation works between 2050 and 2100, the demand was 18 300 m^3 and was concentrated in the last decades of the century (Fig 6a) with the renovation of constructed soils in hard surface areas. An equivalent volume of spent biochar material was also sent to disposal during that time (Fig 6b). Thus, at the end of the study period (2100), biochar was distributed between parks and yards (57%), other urban uses (20%) and in landfills (23%).

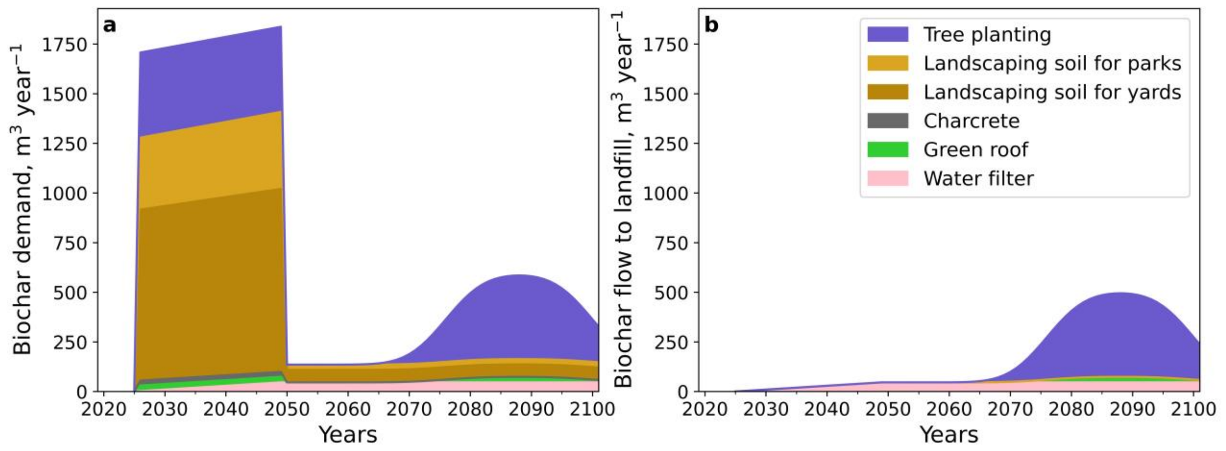

Fig 6 (a) Annual biochar demand in cubic meters per year, by application; (b) Flow of spent biochar to landfill in cubic meters, by application

Carbon sink & climate change mitigation potential. In mass units, the biochar demand ranged from to 8 500 (LR) to 21 300 (WP) tonnes, leading to a wide range of biochar C sink and climate change mitigation potential (Table 3). It can be noted that while green roofs had the largest climate change mitigation benefits at the product level (Fig 5), the effect was limited at the district level due to limited roof area compared to the size of other biochar markets (Table 3). Overall, the substitution of reference products with biochar products led to the mitigation of 4.4 (WP-S) to 30 (WL-S) 10^3 tonnes CO_2-eq, beside the biochar C sink (Table 3).

Table 3. Climate change mitigation potential for each biochar product, in Uppsala's new city district, resulting from the district's construction**.**

Biochar type	**Biomass requirement** (10^3 t)	**Biochar C sink** (10^3 t CO_2)	**Climate-change mitigation potential including biochar C sink** (Total in 10^3 t CO_2-eq and contributions in %)					
			Total	soil	tree	roof	charcrete	biofilm
WP-S	85.3	-58.6	-63.0	76%	20%	1.8%	0.54%	1.4%
GW-S	41.9	-21.6	-34.8	79%	17%	1.9%	0.40%	1.4%
LR-S	33.9	-22.9	-38.9	78%	18%	1.9%	1.1%	1.4%
WL-S	46.6	-28.0	-57.6	76%	20%	1.8%	1.2%	1.4%

Biomass and land requirement. The biochar demand was also converted to biomass requirement (Table 3), and whenever possible to land requirement. For WP, the annual demand during construction (3 400 tonnes year^{-1}) represents 0.3% of the Swedish annual WP production in 2020. For GW, the annual demand during construction (1 700 tonnes year^{-1}) is in the same range as the amount of GW collected at recycling stations in Uppsala municipality (Uppsala Vatten 2020). The LR total demand corresponds to the harvesting of residues at final felling of forestry operation over an area of more than 2 700 ha year^{-1} for 25 years. In comparison, about 200 000 ha are felled annually in Sweden (Swedish Forest Agency 2020). For WL, 203 hectares need to be cultivated for 25 years to meet the demand during construction.

4 Discussion

4.1 Modelling limitations

Biochar properties. We illustrated how different biochar bulk densities and carbon content could influence the carbon footprint of biochar products (Fig 2, Fig 3). However, the studied biochar may also differ with respect to other properties like surface area, water holding capacity or ash content. In spite of these potential differences, it was assumed that the resulting biochar products had similar performance. For soil applications (tree, roof, landscaping) in particular, it was unclear to what extent such material differences (at a constant biochar volume content) would lead to different products because several mechanisms are at play simultaneously. For water biofilm, Perez-Mercado (2018) showed that a wide variety of biochar all performed well for BOD removal. For charcrete, on-going experiments have also been performed with various biochars leading to useable products. The main point was to show the importance for biochar product manufacturers to not only disclose the volume content of biochar, but also disclose (or keep track of) other biochar supply-chain information: biomass origin, pyrolysis conditions, and biochar properties. Voluntary certificates like the European Biochar Certificate are one step in that direction, but are not complete either.

Product-design variability. For each product, a specific design was modelled even though multiple designs may exist. For instance, trees planted in urban areas may have access to root volumes in the range of 10 to 20 m^3, owing to different street dimensions and terrace depth. Also, earlier constructed soils designs included a 5 cm base layer of pure powdered biochar. Likewise, the green roof modelled had a thickness of 3 cm, but other types of green roofs exist (e.g. adapted to different roof slopes or carrying capacity). These design variations will lead to different material requirements and different contribution shares in LCA, which are of importance for city planners. In this study, however, both the reference and biochar products had the same design, leading to meaningful comparisons. The parametrised model and tools developed in the study can be re-used and adapted for different product designs.

Biochar effects. Biochar is expected to deliver beneficial side-effects during the use phase of some biochar-products (Azzi et al. 2021). The only side-effect included was reduced fertiliser use for green roof maintenance, since clear recommendations were provided by the manufacturer. For tree planting and green roofs, it is commonly mentioned that biochar can affect storm water quality and quantity. However, divergent observations have been made in Stockholm and little research is published on the subject. Trees planted in biochar constructed soils are also expected to have improved growth: Ariluoma (2021) for instance assumed that trees planted with biochar would have an improved condition (from "good" to "excellent") and a reduced mortality (from 2% to 1%), which in turn could lead to increased biogenic carbon stock in the trees. However, the long-term permanence of this carbon stock can be questioned as a 1% mortality implies that 34% of the originally planted trees will die in a 50-year timeframe (Ariluoma et al. 2021). Finally, differences in lifetime for products was not accounted either, as it is mostly unknown today.

Material flows. In the Uppsala case study, we estimated biochar demand for several applications with various degrees of confidence and constraints. Biochar demand in the district could have been higher if other product designs had been chosen or if larger markets had been assumed (e.g. green roofs covering a larger area, concrete elements used more extensively).

The Uppsala case study also allowed to estimate flows of waste biochar generated until the end of the century, a rarely discussed topic in the literature. It is worth noting that the total waste flow is in fact larger than shown in Fig 6b since biochar is mixed with other products (soil, macadam, concrete). This annual waste flow and its peak was here mainly set by the lifetime (50 years) of the constructed soils for trees. The actual lifetime of these constructed soil is today unknown leading to key uncertainty (Fridell 2020).

The biochar-material blends also affects how they can be recycled or re-used: the worst case was modelled, landfilling. Biochar-material blends could also be re-used in secondary applications, e.g. mineral soil from green roof can be re-used for new roofs (Pettersson 2020). It is worth noting here that recycling of biochar would lead to lower demand for primary biochar in the future, and thus lower total C sink generated. In addition, since biochar C sink is meant to be long-term, it was important to illustrate how biochar is dispersed in the technosphere over time. The modelling did not include biochar losses to the environment during the use phase, e.g. via water drainage and erosion in constructed soil, parks, and roofs, although such losses have been observed in agriculture (Major et al. 2010; Kätterer et al. 2019).

Climate change impact metric. In both the LCA and the MFA, climate change potential impacts were characterised using GWP_{100}, a static impact category. However, the deployment of biochar in the MFA spanned over several decades and the maintenance activities were studied up to year 2100. While it is common to neglect these temporal aspects (Pauliuk et al. 2013; Lausselet et al. 2020), other approaches combining time dependent MFA, life cycle inventories, and impact assessment metrics could have been used (Beloin-Saint-Pierre et al. 2020).

Parameter variability not investigated. In all biochar supply chains, the biochar yield was set to an average value of 25%. Different pyrolysis setups can however lead to various products distribution and composition (Woolf et al. 2014), which usually controls the trade-off between bioenergy generation and carbon sequestration (Azzi et al. 2019). Another set of parameters kept unchanged were transport distances. Distances were selected to represent generic conditions for South-Central Sweden. In this study, for several materials used, the main source of climate impact was transportation rather than their actual production (e.g. sand, clay, gravel).

4.2 Results interpretation

At the product-level. Tisserant and colleagues (2019) reviewed the climate change impact of biochar systems in 34 LCA studies (including C sink, supply-chain emissions, substitutions and soil effects) and found an average -0.9 tonne CO_2-eq tonne^{-1} biomass (range -1.5 to 0 tonne CO_2-eq tonne^{-1} biomass). Results from Fig 5 and SI (with all biochar supply-chains) converted to biomass unit were in the same range, with an average of -0.73 tonne CO_2-eq tonne^{-1} biomass (-1.4 to -0.11 tonne CO_2-eq tonne^{-1} biomass). The values on the lower end were obtained for low-density biochars used in green roofs, which was tantamount to large benefits from substitution of peat.

The most commonly assessed impact category for biochar systems is climate change, but other impact categories have been stressed to be as important to study (Tisserant and Cherubini 2019; Azzi et al. 2021; Terlouw et al. 2021). By using the ILCD impact categories, we showed that other impact categories could be either better or worse, and that the type of biomass used to produce the biochar had an important role. The results confirmed previous findings that cultivated biomass tends to have higher impacts than residual biomass. However, the amounts of residual biomass are limited and often already used (e.g. energy recovery from garden waste), and may not be enough to meet demand.

We argued that the LCA of biochar-based produced should display results both with and without C sink, in line with previous recommendations (Tisserant and Cherubini 2019; Terlouw et al. 2021). Despite research efforts (Wang et al. 2016; IPCC 2019), biochar stability is inherently uncertain, and it is to date impossible to distinguish between the stability of different biochars with high confidence. It is a precautionary principle that also without the C sink, the products designed with biochar should perform better than the reference product they intend to replace. In addition, due to the high policy and marketing attraction of CDR, many actors are interested in claiming the ownership of the biochar C sink: biochar producers, biochar-product manufacturers, biochar-product owners, but also negative-emission credit buyers.

Comparing the climate change impact of products and services in a renewable and bio-based economy is more complex than in a fossil-fuel economy because of (i) the dynamic nature of the biosphere (as opposed to the relatively static character of geological processes), and the (ii) diversity of land management options, cultivated species, and transformation pathways. In LCA terminology, this includes reference land use or alternative fate of biomass, both for the studied system, and for the reference systems used for comparisons. Here our main result (that biochar is better than a reference product, or the amount of substitution benefits) depends on assumptions on peat emissions and reference land use, biomass cultivation systems for biochar production, and green waste compost being burden free. All these terms can vary widely with the geographical and technological context.

At the district-level. We estimated an average C sink for the construction of Uppsala's new city-district of 0.87 to 2.4 10^3 tonnes CO_2 year^{-1}. This amount can be compared (i) to the territorial emissions of the whole Uppsala municipality as reported for 2020, approaching 1 10^6 tonnes CO_2-eq (Jedland 2021), (ii) to the national consumption-based GHG emissions for the expected population of the new district, around 0.5 10^6 tonnes CO_2-eq (Naturvårdsverket 2019), or (iii) to the expected emissions for the construction of the residential buildings of the new district, estimated by the municipality to be around 20 10^3 tonnes CO_2 year^{-1} (Jedland 2021). In any case, the biochar C sink was one to several orders of magnitude lower.

It can be difficult to interpret the practical implications of the climate change mitigation potentials calculated in Fig 5 and Table 3, and we want to stress two concepts relevant for the biochar industry: market mixes and market segmentation. First, market mixes refers to the fact that a product or service can be supplied by various technologies, e.g. steel can be produced in blast furnaces with hard coal, but also with bio-coal, or with hydrogen. It is unlikely that only one technology will replace hard-coal furnaces. Rather a mix of alternatives technologies will be deployed, and the total climate change impact of the future steel sector will be a weighted average of the market mix. So, the reader must bear in mind that biochar-products are not the only option to improve the environmental performance of existing products, and that the market might be shared between these options. Second, market segmentation refers to the fact that biomass and biochar may not be suitable for all applications, e.g. biochar suited for steel application may require a very specific (low) ash content, while soil applications may be less sensitive to this parameter. In other words, within the range of biochar types produced, they may not be perfectly substitutable. So, the results do not imply that all biochar should be used for the highest ranked as some biochars may not be suited for it.

5 Conclusion

We performed LCA of 5 biochar products for urban environments, compared them to reference applications, benchmarked them against an energy-use of biochar, and investigated the effect of their potential deployment in a city district. In an energy system with low-carbon heat and electricity, all biochar products had lower climate change impacts than the reference products when including biochar C sequestration. Even when excluding biochar C sequestration, most products had lower climate change impacts than the reference products due to lower use of other greenhouse gas intensive materials. However, in terms of resource use, human toxicity and ecotoxicity, biochar products could lead to either increased or decreased potential impacts depending on the biochar supply chain. Biochar produced from waste biomass tends to perform better, but waste biomass is also a limited resource. From a biomass use efficiency perspective, biochar products could provide as much or more climate change mitigation than the use of biochar in steel production, but only if biochar stability is high and if other greenhouse-gas intensive materials are substituted. At the district level, we showed that biochar deployment in the district contributes to reducing the climate footprint of the district, but the total biochar C sink is at least one order of magnitude smaller than other greenhouse gas emissions from the district.

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