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Research article

Life cycle assessment of biochar filters for on-site wastewater treatment

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ABSTRACT

Biochar is a promising filter material for wastewater treatment. This study evaluated the environmental (climate, eutrophication, acidification) impacts of biochar filters for onsite wastewater treatment and compared them with those of a conventional sand filter. Using a parameterised life cycle assessment (LCA) approach, these three impact categories were quantified for two designs of biochar filter and a sand filter, used in normal and sensitive areas as defined by Swedish government recommendations. Different scenarios for the biochar filters, with different combinations of biochar supply chain, biochar end-of-life and energy system, were simulated and analysed. The eutrophication impact of the biochar filters was similar to that of the sand filter, while the acidification impact was generally slightly higher than that of the sand filter in sensitive areas, and lower in normal areas. The climate impact of the biochar filter varied considerably, from substantially higher to lower than that of the sand filter, depending on specific scenario. A few scenarios in which biochar filters had lower overall impacts than the sand filter were identified. In general, the biochar filters had lower environmental impacts in a renewable energy context than in a fossil fuel context. Using biochar in landscaping soil was a better end-of-life alternative than combustion. Biochar from syngas-heated pyrolysis performed considerably better than biochar from electricity-heated pyrolysis in a fossil energy context. Direct emissions to air and water from the wastewater treatment process, production of biochar and biochar end-of-life contributed most to the total impacts and variation in these for all biochar systems.

1. Introduction

In Sweden, there are nearly 700,000 onsite wastewater treatment system (OWTS) units (Hansson et al., 2019). Conventional techniques used in Sweden for these systems are a septic tank with simple or no subsequent treatment (26%), soil infiltration (30%) or a sand filter (14%) (Olshammar et al., 2015). These units contribute to eutrophication of the Baltic Sea, as the nitrogen (N) and phosphorus (P) loss from OWTS units represents around 5% and 13%, respectively, of the total estimated Swedish anthropogenic nutrient loads to the Baltic Sea (Hansson et al., 2019). The large number of OWTS units and the considerable load of pollutants to be treated means that it is vital that these units have high treatment performance. They must also use resources efficiently, thereby minimising their life cycle environmental impact. Biochar water filters are considered an interesting new technology due to their potential for efficient cleaning of wastewater, while also acting as a potential carbon sink and being derived from renewable resources (Wang et al., 2020).

The performance of OWTS is covered by recommendations from the Swedish Agency for Marine and Water Management (Swedish Agency for Marine and Water Management, 2016), which are often used as requirements by municipal authorities. The recommendations entail a 90% reduction in biochemical oxygen demand (BOD) and a 70% reduction in P under normal conditions. For areas with higher treatment requirements (where the surrounding area or water recipient is deemed sensitive), the recommendation is for a 90% reduction in P and 50% reduction in N.

Biochar is produced by thermal decomposition (pyrolysis) of organic materials at temperatures of 300–800 °C in the absence of oxygen (Steiner, 2016). The physical, chemical and structural properties of biochar vary, but in general it is characterised by large surface area (200–1000 m^2/g), low density and high porosity (Dalahmeh et al., 2016, 2017; He et al., 2016), making it an efficient adsorbent and good biofilm carrier.

Previous and ongoing research shows that biochar has good potential as a filter material in OWTS for removal of easily degradable and

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persistent organics and nutrients (Dalahmeh et al., 2016, 2017). Other studies have demonstrated good potential of biochar for adsorption of heavy metals (Babel and Kurniawan, 2004) and aromatic hydrocarbons (Mukherjee et al., 2007). Installing biochar wastewater filters also presents an opportunity to achieve carbon dioxide (CO_2) removal, which is important considering the current consensus that greenhouse gas emission reductions will have to be combined with removals in order to achieve global climate goals (IPCC, 2022).

While the reported performance of biochar filters is promising, these findings are mostly based on results obtained in laboratory experiments rather than analysis of samples from actively operating wastewater systems (Wang et al., 2020). Another aspect still to be determined is the environmental impact from a systems perspective, taking into account not just wastewater treatment performance but the entire life cycle. For example, previous research has shown that emissions to air from wastewater treatment plants can have substantial climate impacts (Delre et al., 2019; Foley et al., 2010). Since flows of carbon (C), N and P can end up in different forms in sewage sludge, air (not P) or treated water, with very different environmental implications, it is important to monitor all these system flows when conducting a comprehensive analysis (Heimersson et al., 2016). For example, N can be present as organic N in water or as nitrous oxide (N_2O) in air. In addition, while the technical performance of the treatment is important, the environmental sustainability is influenced by other factors related to production of the biochar (e.g. energy used for pyrolysis and type of feedstock), but also the post-use handling of the biochar (end-of-life).

Life cycle assessment (LCA) is an established method for assessing the environmental impacts of wastewater treatment systems and biochar systems, as it considers the environmental impacts of a system across all its components and throughout its life cycle. By modelling different processes and varying relevant inputs and outputs for these processes in LCA, it is possible to gain insights into how technical design, assumptions and data influence the total environmental impact (Zakrisson et al., 2023). Parameterised LCA is therefore a useful tool for early assessment of a technique that is not yet widely employed and for which data from practical applications are lacking.

For the specific case of biochar, Thompson et al. (2016) conducted an LCA of biochar used for removal of micropollutants compared with activated carbon. Allashimi and Aktas (2017) also compared biochar and activated carbon in an LCA, assessing both environmental and economic performance. Gongora et al. (2021) performed an assessment of onsite wastewater treatment using biochar in constructed wetlands compared with aerated treatment units. These appear to be the only published studies on LCA of biochar use in wastewater treatment applications and to our knowledge no attempt has been made previously to assess the total environmental impact of biochar filters in OWTS.

The aim of this study was to assess the environmental impact of onsite treatment of household wastewater using biochar filters and to compare the outcomes with those of a conventional filter. Taking into consideration effects of type of energy systems, different biochar supply chains and multiple end-of-life scenarios on the impacts, specific objectives were to (i) quantify the climate, eutrophication and acidification impacts of different biochar filters for onsite wastewater treatment in catchments at the normal and high level of recipient protection; (ii)

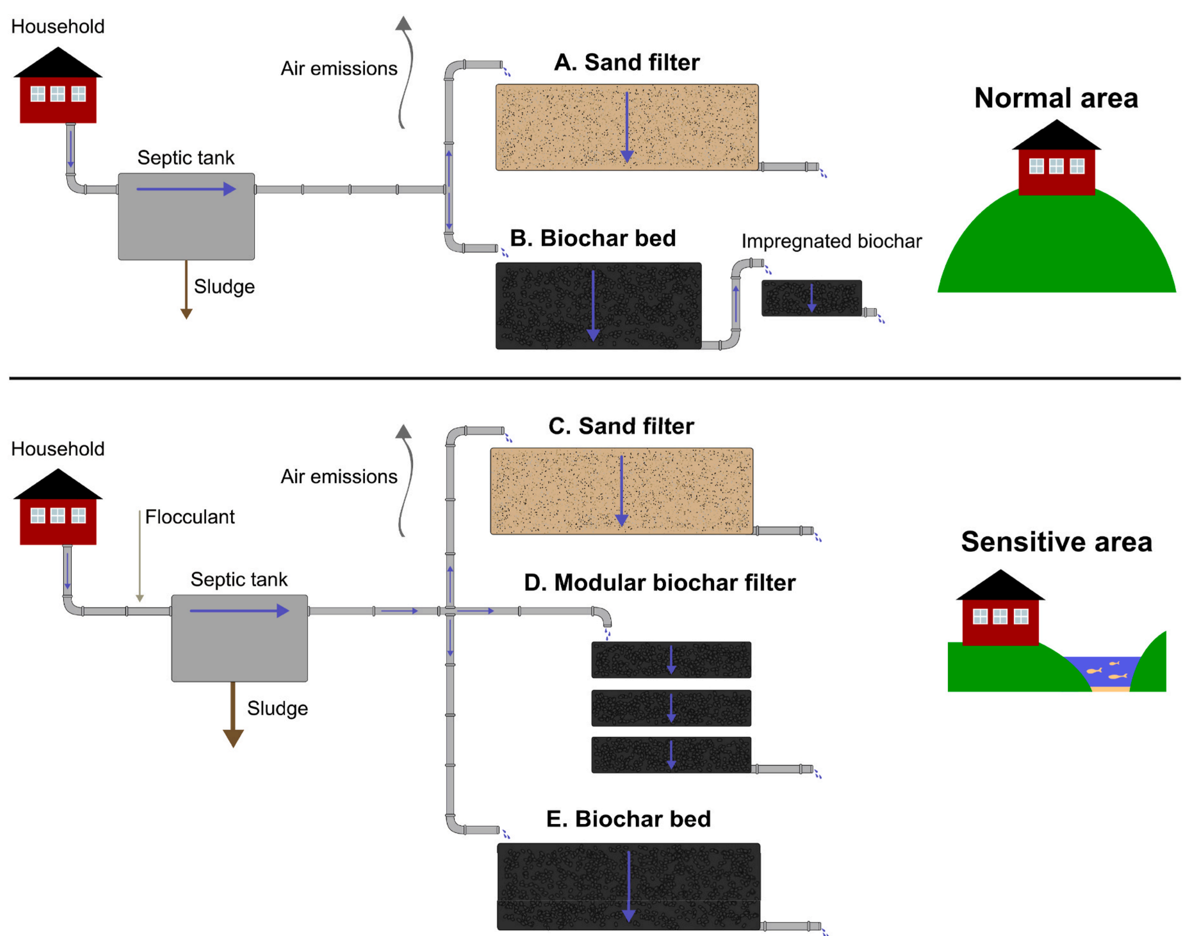

Fig. 1. Illustration of the different onsite wastewater treatment systems studied. In normal areas, wastewater was treated either in a sand filter (A) or in a biochar bed followed by a filter with impregnated biochar (B). In sensitive areas, wastewater was treated in a sand filter (C), a modular biochar filter (D) or a larger biochar bed (E).

identify and assess key uncertainties and sensitive parameters in these systems; and (iii) identify the conditions in which biochar systems are preferable to sand filters.

2. Material and methods

2.1. System overview

A one-household OWTS consisting of a plastic septic tank followed by filters, with single and several treatment steps, was modelled. Treatment of 1 m^3 of wastewater was set as the functional unit (Fig. 1). A Swedish context was assumed and the scenarios analysed covered both normal areas and areas with higher treatment requirements as defined in Swedish recommendations (Swedish Agency for Marine and Water Management, 2016), hereafter called 'normal' and 'sensitive' areas, respectively. Two different biochar filter systems were studied: an infiltration bed (hereafter 'biochar bed') constructed belowground and a compact modular biochar filter unit installed aboveground, as described by Shigei (2021). These biochar filters were compared with a conventional sand filter. The biochar bed and sand filter were designed and modelled for both normal and sensitive areas, while the modular biochar filter was only modelled for sensitive areas. For normal areas, the biochar bed was composed of the bed itself and an additional unit for P removal (impregnated biochar). For sensitive areas, both the biochar and sand filter systems included addition of flocculant to the septic tank for additional P removal.

The LCA covered the entire life cycle of all filters, i.e. production, operation and end-of-life. The OWTS was assumed to have a lifetime of 30 years and capable of treating all wastewater from a household of four people (equivalent to 178 m^3/year). The filters were dimensioned according to Swedish recommendations (Swedish Agency for Marine and Water Management, 2016) and designed for a hydraulic loading rate of 50 L m^{-2} day^{-1}. Sludge management and minor parts of the system, such as pipes and pumps, were not included in modelling in this study. This was partly because the impacts of these parts are likely to be small in relation to the total impact in the impact categories considered, at least for sand filters (Risch et al., 2021), and partly because the difference in these emissions between the systems would be negligible.

2.1.1. Technical description of filters

2.1.1.1. Filter design. The biochar bed for normal areas was assumed to be composed of an aerobic bed for organic matter degradation and nitrification (53 m^3) and a smaller filter (2 m^3) with biochar impregnated with iron chloride solution (as described by Dalahmeh et al., 2020), in order to fulfil the requirements for P removal. Since the smaller filter with impregnated biochar quickly became saturated with P (Dalahmeh et al., 2020), the impregnated biochar was replaced every second year, resulting in use of 30 m^3 of impregnated biochar over the lifetime of the filter. The biochar bed system for sensitive areas was composed of the same aerobic bed as for normal areas, followed by an anaerobic bed for denitrification, with a total volume of 65 m^3.

The modular biochar system designed for sensitive areas was composed of six biochar layers (modules) stacked on top of each other. The first three modules were designed to be aerobic, to achieve organic matter degradation and nitrification. The following two modules were anaerobic, to achieve denitrification. The last module was aerobic, to achieve bacteria purification. The filter had a total volume of 16 m^3 and was installed aboveground in plastic containers. In contrast to the other filters, no excavation was necessary to install this filter. In sensitive areas, the impregnated biochar filter was not used in combination with other filters, since high P removal would have required very frequent replacement. Instead, for all filter systems in sensitive areas, a solution of polyaluminium chloride (0.23 kg/m^3 wastewater; Carlund, 2021) was added as flocculant to the wastewater in the septic tank to ensure additional P removal. Due to its low rate of P removal (see section 2.1.2), the modular biochar filter would have had to be combined with a flocculant in a normal area and therefore it was only modelled in a sensitive area context.

The sand filter had a volume of 53 m^3 and the sand was assumed to have density of 1700 kg/m^3. In both normal and sensitive areas, the surrounding soil might or might not be suitable as a filter material, and in the latter case sand would need to be obtained and transported to the site. Both these possibilities were modelled as separate scenarios.

All components of the systems were assumed to be transported by lorry (EURO6, 16–32 tonnes, fuelled by diesel) to the location of the operation and soil was assumed to be excavated in order to make room for the system components that were installed belowground. The excavated soil was assumed to be discarded onsite, without further transportation.

2.1.1.2. Biochar feedstocks and properties. As filter media for both the biochar bed and modular biochar systems, four biomass feedstocks pyrolysed in reactors heated by syngas or electricity were modelled as described in Azzi et al. (2022), but with the model updated to Ecoinvent version 3.9.1. Six biochar supply chains were considered: wood pellets (WP) pyrolysed in a reactor heated by syngas (WP-S) or electricity (WP-E); garden waste (GW) pyrolysed in a reactor heated by syngas (GW-S) or electricity (GW-E); logging residues (LR) pyrolysed in a syngas-heated reactor; and willow chips (WC) pyrolysed in a syngas-heated reactor. Each feedstock was also modelled with a reference use, i.e. the impacts of an alternative use of the feedstock were modelled and added as a substitution effect (see section 2.3). Element content and density were different for the four different biochar feedstocks (Table 1), affecting the rest of the biochar life cycle (Fig. 2). Biochar elemental content and densities were obtained from laboratory analysis of biochars produced in commercial reactors, typically operating at a pyrolysis temperature of 500–600 °C (Azzi et al., 2022). Specifically, the density affected the mass of biochar used in filters and transportation (expressed in ton km). The element content affected all end-of-life scenarios (carbon sequestration or combustion for energy production, see section 2.1.3).

2.1.1.3. Septic tank. Data on production and installation of the plastic septic tank were obtained from the company FANN VA-teknik AB (see Table S1 in Supplementary Material (SM)). The septic tank was assumed to be transported by lorry to the site of use, where it was installed underground.

Table 1
Biochar characteristics, feedstock reference use and pyrolysis heat scenarios modelled. Data from Azzi et al. (2022). HHV = higher heating value.

Feedstock	Wood pellets (WP)	Garden waste (GW)	Logging residues (LR)	Willow chips (WC)
Properties of biochar				
Carbon (%)	93.4	69.9	91.6	81.6
Sulphur (%)	0.03	0.06	0.0	0.04
Hydrogen (%)	1.3	1.61	1.7	2.56
Oxygen (%)	2.5	6.4	2.0	9.17
Density (kg/m^3)	500	242	194	270
HHV (MJ/kg)	33.1	24.9	33.1	29.7
Ash content (%)	0.026	0.234	0.05	0.062
Reference use				
Land use	Unchanged			Fallow
Material use	Combusted for heat production		Left at location	No material produced
Pyrolysis heat				
Syngas	x	x	x	x
Electricity	x	x		

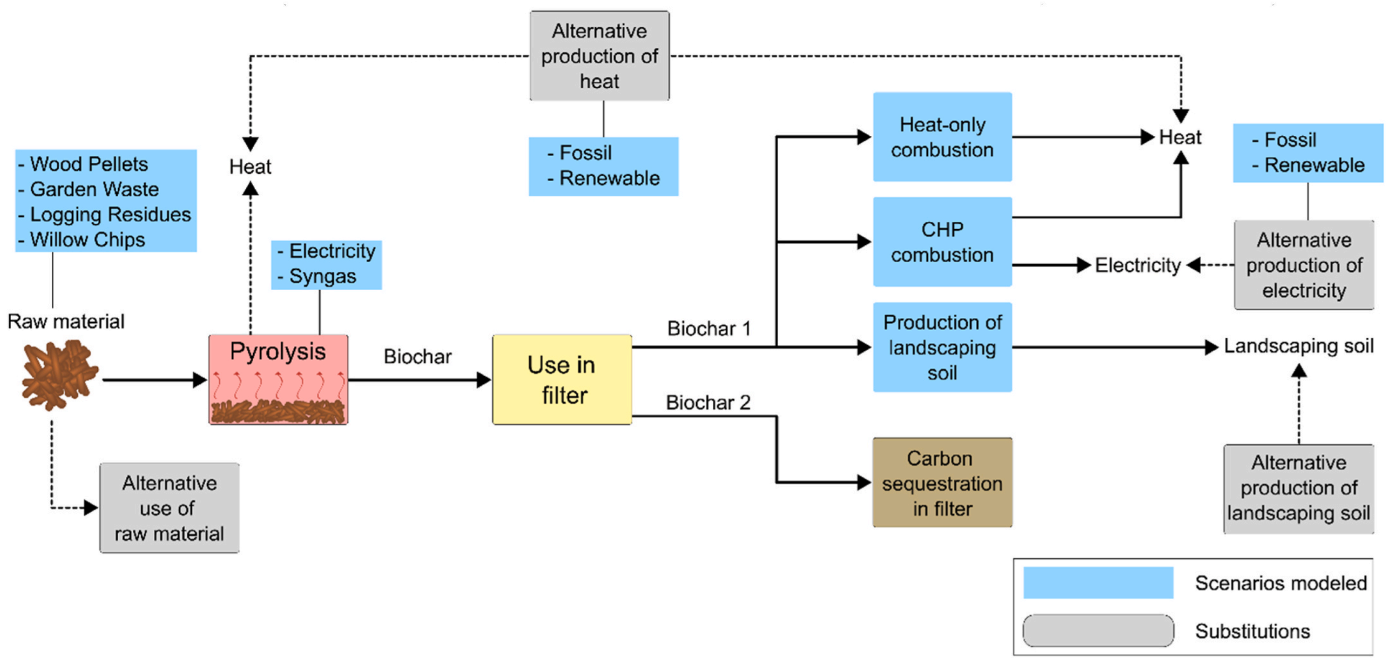

Fig. 2. Flowchart of the biochar life cycle, from production to end-of-life, showing the scenarios studied and the co-products generated. Biochar 1 = biochar in the modular filter and the impregnated biochar. Biochar 2 = biochar in the biochar bed filter.

2.1.2. Wastewater treatment

The pollutants selected for modelling were BOD_5, N and P. To ensure that all flows of pollutants were accounted for, a mass balance was set up (Fig. 3). Full details and exact quantities can be found in Table 2.

2.1.2.1. Untreated wastewater. The OWTS was assumed to be used by four people during the entirety of the operational phase and total amount of mixed wastewater from the household was assumed to be 121.7 L per person and day (Rose et al., 2015; Segerström, 2022). The concentrations of pollutants in the wastewater were assumed to be constant at 460 g BOD_5/m^3, 110 g N/m^3 and 13 g P/m^3 (Risch et al., 2021). These values were chosen in order to represent average conditions under which the filters would operate.

2.1.2.2. Pollutants removed in sludge. The percentages of pollutants (N, P, BOD_5) removed in septic tank sludge were set as constant values (Table 2), although these values differed between normal and sensitive areas due to the flocculant added in sensitive areas.

2.1.2.3. Pollutants removed in filters. The pollutants removed in the filters were modelled separately for each type of filter (sand or biochar) and for the area studied (normal or sensitive). Mass balance data from Risch et al. (2021) for a septic tank and a sand filter were used to calculate N emissions to air and water, and P emissions to water (Table 2), for both the sand and biochar filters, as there were no corresponding data available specifically for biochar filters. The impregnated biochar unit used for removing P from biochar bed effluent in normal areas was assumed to achieve 90% P removal and negligible removal of BOD_5 or N. It was also assumed that the biochar feedstock did not influence the treatment efficiency of the biochar.

2.1.2.4. Air emissions. All N removed in the filters was assumed to be converted into different gaseous N compounds (Table 2). All P removed in the filters was assumed to be absorbed in the filters, i.e. remained in the filter material. Carbon dioxide and methane (CH_4) emissions were assumed to be the same for all systems (201 g CO_2/m^3 and 94.5 g CH_4/m^3; Risch et al., 2021), both modelled with an error margin of 10%. The CO_2 formed from organic matter in wastewater was modelled as non-fossil with climate impact zero (see impact categories in section 2.2).

2.1.2.5. Pollutants in treated wastewater. With all other flows in the mass balance quantified, the amounts of pollutants left in the treated wastewater released to the surrounding environment were calculated. The composition of P and N emissions in the wastewater was modelled using the mass balance from Risch et al. (2021) (Table 2).

2.1.3. End-of-life

The biochar bed filter, the sand filter and the septic tank were assumed to be simply left in the ground after the operational phase. For the biochar in the modular biochar filter and the impregnated biochar, three different end-of-life scenarios were modelled: use as peat substitute in landscaping soil (see Table S2 in SM), combustion in a combined heat and power (CHP) plant, or combustion for heat only.

For landscaping soil, the biochar was assumed to be transported by lorry to a soil manufacturing site and mixed with other soil constituents (e.g. sand, compost), and then used for urban green areas (as modelled by Azzi et al., 2022). When used as landscaping soil or when remaining in the spent filter, the biochar carbon was assumed to constitute a long-term carbon sink (Lehmann et al., 2021). The share of biochar carbon remaining in storage for 100 years was varied between 50% and 100% of the initial biochar carbon content (see section 2.4).

For combustion of the biochar, values for heat production in the Ecoinvent 3.9.1 database (hard coal, industrial furnace 1–10 MW, in Europe without Switzerland) were used as proxy data. The values were adjusted to consider differences in element composition (C, sulphur (S), hydrogen (H), oxygen (O)) compared with coal and re-calculated specifically for each type of biochar. The adjustments involved altering the heating values, which were recalculated according to Alvarez (2006), using the element composition of the different fuels. To convert between exhaust gas concentrations and exhaust gas mass flows, total exhaust gas flow was calculated based on Alvarez (2006) and the element composition of the different fuels. For emissions, it was assumed that the burner emitted the maximum amount allowed in European Union member states (EC Commission Regulation, 2015/2193) and had the minimum thermal efficiency allowed by standard EN-303-5 as described

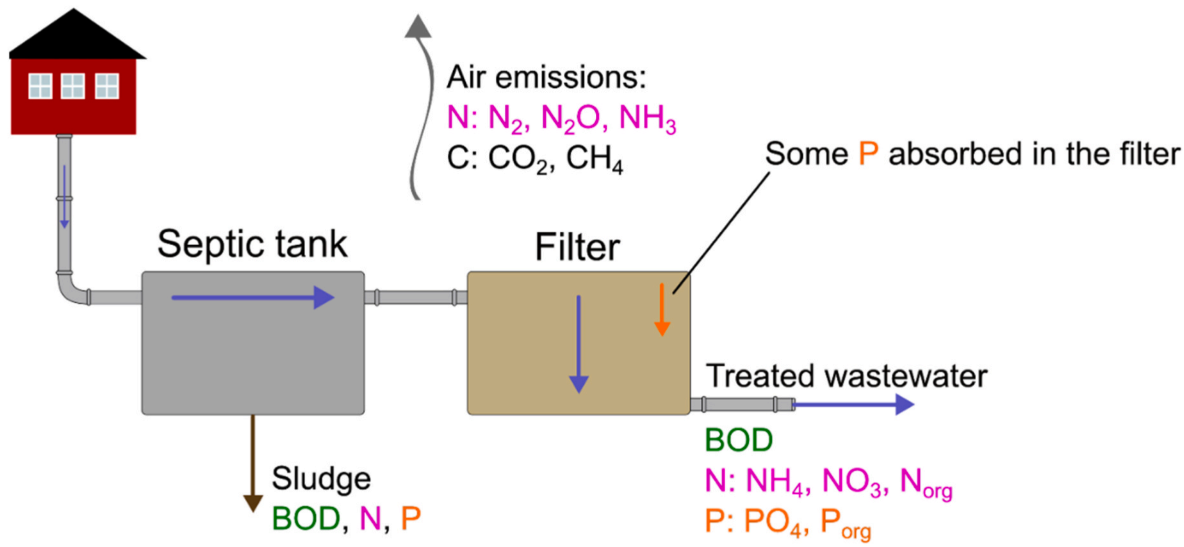

Fig. 3. Flows of biological oxygen demand (BOD), nitrogen (N) and phosphorus (P) and emissions to air of carbon dioxide (CO_2) and methane (CH_4) from the onsite wastewater treatment system.

Table 2
Mass balances for biological oxygen demand (BOD), nitrogen (N) and phosphorus (P). Pollutants removed in the septic tank ended up in the sludge removed from the system. The values and ranges shown were used in the parameterised LCA (see section 2.4).

	Biochar bed, normal area	Biochar bed, sensitive area	Modular biochar filter, sensitive area	Sand filter, normal area	Sand filter, sensitive area	Source(s)
BOD mass balance						
BOD_5 in (g/m^3)	460					Risch et al. (2021)
BOD_5 removed in septic tank (%)	8.7–50	13.7–60	13.7–60[a]	8.7–50	13.7–60	Ek et al. (2011); Johansson et al. (2005); Siegrist (2017)
BOD_5 removed in filter (% of BOD_5)	82–92	90–96	90[a]	96.7–98.4	96.7–98.4	Dalahmeh et al. (2019); Wilson et al. (2011)
BOD_5 left in wastewater	All BOD_5 not removed by the septic tank or filters					
N mass balance						
N in (g/m^3)	110					Risch et al. (2021)
N removed in septic tank (%)	5–15					Ek et al. (2011)
N removed in filter (% of remaining N)	0–22	59–83	60–70[a]	40.9–57.2	40.9–57.2	Dalahmeh et al. (2019); Wilson et al. (2011)
N air emissions	All N removed in the filters, see composition below					
N_2 air emissions (% of N removed in filter)	47.37					Risch et al. (2021)
N_2O air emission (% of N removed in filter)	5.14					Risch et al. (2021)
NH_3 air emissions (% of N removed in filter)	47.37					Risch et al. (2021)
N left in effluent	Any N not removed by the septic tank or filters, see composition below					
NH_4^+ water emissions (% of N left in water)	13.23					Risch et al. (2021)
NO_3^- water emissions (% of N left in water)	82.15					Risch et al. (2021)
N_{org} water emissions (% of N left in water)	4.63					Risch et al. (2021)
P mass balance						
P in (g/m^3)	13					Risch et al. (2021)
P removed in septic tank (%)	5–25	71–97	71–97	5–25	71–97	Ek et al. (2011); Carlund (2021)
P removed in filter (% of remaining P)	32–96	32–96	20–30[a]	43.7–93.4	43.7–93.4	Perez-Mercado et al. (2018); Wilson et al. (2011)
P removed in impregnated biochar filter (%)	90[a]	–	–	–	–	
P left in effluent	All P not removed by the septic tank or filters					
PO_4^{3-} water emissions (%)	80[a]	80[a]	80[a]	90	90	Risch et al. (2021)
P_{org} water emissions (%)	20[a]	20[a]	20[a]	10	10	Risch et al. (2021)

[a] Estimated from unpublished experimental data.

in Directive 2009/125/EC. Ash from the combustion process was assumed to be spread in agricultural fields. Further data on emissions from the combustion processes can be found in Table S3 in SM.

2.2. Impact categories

For each stage of the life cycle, inputs and outputs of materials and energy were modelled. The emissions were then summarised for the entire life cycle and classified into the chosen impact categories. The impact categories considered in the LCA were climate impact, eutrophication and acidification, respectively characterised by GWP_{100} from IPCC 2021, TRACI v2.1 and CML v4.8. The characterization factors were expressed in kilograms of equivalents; for climate impact, carbon dioxide equivalents (CO_2-eq); for eutrophication, nitrogen equivalents (N-eq); and for acidification, sulphur dioxide equivalents (SO_2-eq). Major emissions modelled for climate impact were CO_2 (1 CO_2-eq), non-fossil CH_4 (27 CO_2-eq) and N_2O (273 CO_2-eq). For eutrophication, air emissions modelled were ammonia (0.119 N-eq); water emissions modelled were ammonium (0.779 N-eq), BOD_5 (0.05 N-eq), N (0.99 N-eq). Nitrate (0.237 N-eq), P (7.29 N-eq) and phosphate (2.38 N-eq). For acidification, major emissions modelled were sulphur dioxide (1.2 SO_2-eq), ammonia (1.6 SO_2-eq), and nitrogen oxides (0.5 SO_2-eq). Climate impact and eutrophication potential are impact categories of special interest in LCA of wastewater treatment systems (Corominas et al., 2020). The climate impact is of specific interest when assessing biochar systems, since biochar is a negative emissions technology (Minx et al., 2018). Acidification is a major environmental concern for terrestrial and freshwater ecosystems in Sweden due to acid-sensitive soils and very high legacy emissions of sulphur oxides in Sweden and Western Europe (Fölster et al., 2021).

2.3. Multifunctionality

Multifunctionality was accounted for using the substitution method as defined by Heijungs et al. (2021). The biomass used for producing biochar had alternative reference uses (land application or biomass use), as modelled by Azzi et al. (2022). For the WP and GW biochars, the reference was combustion in a heat-only boiler, for LR biochar the reference was not harvested but left in the forest to decay, and for WC biochar the reference land use was fallow.

The avoided burdens from energy (heat and electricity) produced from pyrolysis and combustion (of both biochar and reference use for biochar feedstock, see Fig. 2) were modelled as either renewable or fossil energy (discrete parameters, see sections 2.4 and 2.5), referred to hereafter as the 'energy system'. Specifically, avoided electricity generated was modelled as electricity from either wind power or coal, and avoided heat was modelled as heat from wood chips or natural gas. Energy use in the systems was modelled in the same manner, i.e. if the background energy included electricity produced with wind power, energy produced and consumed were both modelled as wind power. Lastly, the landscaping soil produced with biochar was credited with the avoided burden from producing the same amount of conventional landscaping soil (as modelled by Azzi et al., 2022).

2.4. Model parameterisation and sensitivity analysis

The LCA model was parameterised, meaning that many data points were defined as a range of possible values rather than a single value. Some parameters were defined as discrete (see section 2.5) and others as continuous (e.g. treatment efficiencies in Table 2). The parameters were modelled using a uniform distribution, i.e. all parameter values had the same probability of being sampled. This in contrast to e.g. a normal distribution and was chosen since all parameter values were of equal interest. In modelling, all defined parameters were varied simultaneously and randomly across their defined ranges by computing the model multiple times (770,000 unique computations) in a global sensitivity analysis (GSA) (see Saltelli et al., 2008). All parameters and their definitions can be found in Table S4 in SM.

In LCA, sensitivity analysis is often performed by the practitioner choosing one data point they perceive as sensitive, giving it an alternative value, and then generating the results again to try and draw conclusions about the robustness of the model. By varying all potentially sensitive parameters together, in contrast to chosen data points in isolation, additional information can be obtained. For example, the risk of a sensitive data point not being highlighted because it is simply assumed not to be sensitive is minimised and interactions between several parameters can be captured. The result is potentially more informative, but also more complex and more difficult to interpret and communicate.

The environmental impacts from the life cycle were grouped, in order to perform contribution analysis with the goal of identifying parts of the life cycle that contributed most to the total impact and the variation in impacts. To identify sensitive parameters, Sobol indices were calculated for all systems and impact categories. Sobol index measures the contribution of each parameter to total variance, with the value ranging from 0 (the parameter does not contribute to the variance) to 1 (the parameter explains all the variance) (Sobol, 2001).

2.5. Scenarios

The scenarios for the biochar filters involved combinations of three discrete parameters: biochar supply chain (see section 2.1.1), background energy system (fossil or renewable, see section 2.3) and biochar end-of-life scenario (CHP combustion, heat-only combustion or use in landscaping soil, see section 2.1.3), resulting in a total of 36 different scenarios (Fig. 4). The biochar bed filters in sensitive areas did not have different end-of-life scenarios as all biochar was left in the ground after use, so for these filters 12 different scenarios were modelled. The sand filter was not affected by these discrete parameters, i.e. was only modelled for one scenario.

2.6. Software and database

The following LCA software was used: the Python-based framework for LCA *brightway2* (Mutel, 2017), its graphical user interface *activity-browser* (Steubing et al., 2020) and its algebraic extension *lca_algebraic* (Jolivet et al., 2021), which was used to compute the parameterised life cycle inventory (LCI). The Ecoinvent database, version 3.9.1 cut-off system-model (Wernet et al., 2016), was used for modelling the system. The code used to analyse the results and generate diagrams is available on Github.[1] The brightway2-model is available on request.

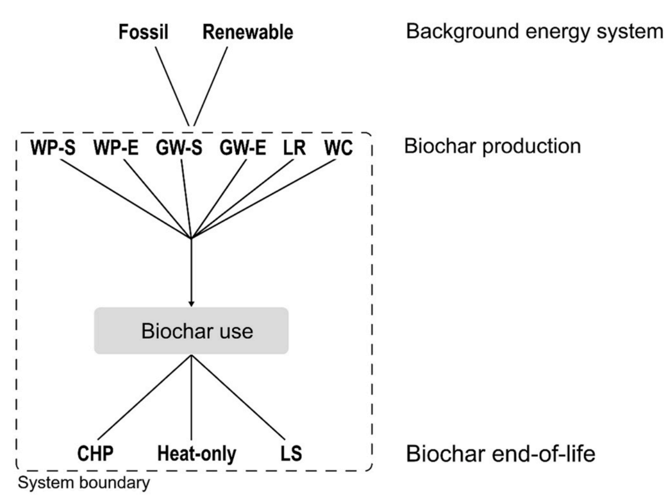

Fig. 4. Flowchart of the biochar system showing the combinations of background energy system (2), biochar supply chain (6) and end-of-life (3) resulting in 36 different scenarios. WP = wood pellets, GW = garden waste, LR = logging residues, WC = willow chips, S = syngas-heated reactor, E = electricity-heated reactor, CHP = combined heat and power plant, LS = landscaping soil.

[1] https://github.com/SLU-biochar/wwt.

3. Results

3.1. Environmental impacts and sensitive parameters

The impacts of the biochar filters varied considerably across the 36 scenarios modelled, for normal and sensitive areas and for all impact categories considered (Fig. 5). The climate impact of the biochar bed varied from −29 to 31 kg CO_2-eq/m^3 wastewater in normal areas, and from −19 to 14 kg CO_2-eq/m^3 wastewater in sensitive areas. For the modular biochar filter, the climate impact varied between −4 and 14 kg CO_2-eq/m^3 wastewater. The eutrophication impact of the biochar bed varied from 0.06 to 0.20 kg N-eq/m^3 wastewater in normal areas, and from 0.02 to 0.13 kg N-eq/m^3 wastewater in sensitive areas. The eutrophication impact of the modular biochar filter varied from 0.02 to 0.09 kg N-eq/m^3 wastewater. Lastly, the acidification impact of the biochar bed varied between −0.01 and 0.09 and 0.04–0.12 kg SO_2-eq/m^3 wastewater for the biochar bed in normal and sensitive areas, respectively. The acidification impact of the modular biochar filter varied from 0.04 to 0.08 kg SO_2-eq/m^3 wastewater. There was also variation within each scenario.

Syngas-heated pyrolysis resulted in a lower impact than electricity-heated pyrolysis for the WP and GW biochars (other biochars were only modelled in syngas-heated reactors) in a fossil energy context. This was because the internal heat in syngas-heated pyrolysis can be reused, while in electricity-heated pyrolysis the internal heat is not used and instead the environmental impact of electricity production is added to the life cycle impact. In a renewable energy context, the two pyrolysis techniques scored similarly, since electricity from wind power has very

Fig. 5. Range of climate (a, b), eutrophication (c, d) and acidification (e, f) impact for biochar and sand filters, in non-sensitive areas (a, c, e) and sensitive areas (b, d, f). The ranges plotted correspond to the minimum and maximum values, computed for all system configurations (different energy systems, biochar supply chains and biochar end-of-life) and the entire distribution of continuous parameters. Horizontal grey lines indicate the minimum and maximum values for the sand filter. BC = biochar, LS = landscaping soil, CHP = combined heat and power. Biochar supply chains: WP = wood pellets, GW = garden waste, LR = logging residues, WC = willow chips, S = syngas-heated reactor, E = electricity-heated reactor.

low environmental impact. The WP-E biochar had exceptionally high impacts in a fossil context, for all impact categories, in both normal and sensitive areas.

The biochar filters had lower impacts in general in a renewable energy system than in a fossil energy system. However, some scenarios had lower impacts in the fossil energy system, e.g. the modular biochar filter with WP-S biochar and CHP as end-of-life scenario. This was probably due to fossil energy being substituted when combusting the biochar and the WP-S biochar having relatively high higher heating value (HHV) and density, the latter resulting in a higher mass of biochar from the filter being combusted.

For all scenarios, the landscaping soil end-of-life scenario for the biochar had a lower climate impact than the combustion scenarios. This indicates that the carbon sequestration achieved by biochar is preferable to combustion of biochar for energy. The heat combustion scenario had a higher climate impact than the CHP scenario in a fossil energy context, while in a renewable context both scenarios had a similar climate impact. This was probably because the reference electricity production in the fossil energy context generated particularly high greenhouse gas emissions, while in the renewable energy system both the heat and electricity references had low impacts and therefore the difference between them was smaller.

The climate impacts varied widely for the biochar filters, with some scenarios having considerably higher impacts than for sand filters and others lower impacts. The acidification impact of the biochar filters was slightly lower than for sand filters in normal areas and slightly higher in sensitive areas, while the eutrophication impact was similar to that of the sand filter.

Pair-wise comparisons of all computations for the biochar filters and the sand filter (Table S5 in SM) revealed that all biochar filters had lower climate impacts than the sand filter when the biochar was used in landscaping soil after filter use in a renewable energy context, regardless of the biochar supply chain. This was also true in a fossil energy system, with the exception of the biochars with electricity-heated pyrolysis. Compared with the sand filter, the WC biochar was preferable climate-wise for most scenarios involving biochar bed filters, in normal and sensitive areas. This was because the willow chip feedstock, unlike the other feedstocks, did not have energy production as the reference usage and also included carbon storage in soil due to willow cultivation on fallow land (Hammar et al., 2014).

In terms of eutrophication, in all scenarios the biochar bed in normal areas generally had a higher impact than the sand filter. The best performing scenario was LR biochar in a fossil energy context with CHP as end-of-life, which had a lower eutrophication impact than the sand filter in 39.5% of the computations. In sensitive areas, eutrophication impacts for the biochar bed and modular filter were more mixed and varied between all system configurations. Thus the scenarios ranged between being preferable in 0% to up to 100% of the computations compared with the sand filter.

In terms of acidification potential, both biochar filters in sensitive areas had an overall higher impact than the sand filter in the majority of computations. The best-scoring scenario in sensitive areas was the biochar bed filter, with GW-S biochar in a fossil energy context, which had lower acidification impacts than the sand filter in 10.3% of the computations. For the biochar bed in normal areas, the WP-E biochar had a higher impact than the sand filter with heat combustion or landscaping soil as end-of-life scenario in a fossil energy context. All other scenarios had lower impacts than the sand filter for 100% or close to 100% of the computations.

The results also indicated possible trade-offs between impact categories. For example, for the biochar bed filter in normal areas and a fossil energy context, with CHP as biochar end-of-life, both the WP-S and WC biochar had lower acidification impact than the sand filter in 100% of the computations. However, the WC biochar had lower climate impact and higher eutrophication impact than the WP-S biochar.

Biochar supply chain was in general a sensitive parameter for the biochar filters, across all impact categories (Table 3). The end-of-life parameters were sensitive for some biochar filters and feedstocks. The electricity fossil/renewable parameter was sensitive for the biochar bed filter in normal and sensitive areas. The biochar stability parameter was sensitive for the climate impact for the biochar bed filter in sensitive areas. Other sensitive parameters were related to the wastewater treatment process, particularly N treatment in the biochar filters, which affected the eutrophication and acidification impacts. N and P treatment in the septic tank were sensitive parameters for the eutrophication and acidification impact of the modular biochar filter. Some parameters did not contribute to the variation in results and as such were not sensitive, e.g. transport distance. The sum of first-order Sobol indices ranged from 0.33 to 0.73 for biochar filters, indicating that interactions of multiple parameters also influenced the results. The Sobol indices are reported in full in Tables S6–S10 in SM.

3.2. Contribution analysis of best-performing biochar filters

One scenario for every biochar filter that had low impacts relative to both the sand filter and to the other scenarios was chosen for further analysis (with climate impact being prioritised as the most important impact category, since climate benefits is a main rationale for using biochar). These were: (i) Biochar bed, normal area, fossil energy context, WC biochar with CHP as end-of-life scenario; (ii) biochar bed, sensitive area, fossil energy context, GW-S biochar; and (iii) modular biochar filter, renewable energy context, WP-S biochar and use in landscaping soil as end-of-life scenario. These were further explored in contribution analysis (Fig. 6).

For all scenarios and impact categories, the emissions from wastewater treatment itself were substantial, especially in sensitive areas. In normal areas, the minimum acidification impact from wastewater treatment was close to zero. This was because of poorer N removal in the filter, which resulted in higher ammonia emissions to air, and ammonia emissions dominated the acidification impact from the wastewater treatment emissions. Production of biochar and substitution effects also had a considerable impact, especially for the climate impacts and in normal areas. Carbon sequestration was substantial for the climate impact, but showed great variation, since biochar 100-year durability varied between 50% and 100% in the model. Some groups, such as transport, had a small relative impact for all impact categories.

For the sand filters, emissions to air and water from wastewater

Table 3
Parameters with the three highest Sobol indices (marked in bold) and mean values for the biochar filters and all impact categories. BC = biochar, EOL = end-of-life, GW = garden waste biochar, ST = septic tank, WP = wood pellet biochar.

BC Bed Normal				
Parameter name	Climate	Eutrophication	Acidification	Mean
Electricity fossil/renewable	0.03	**0.09**	**0.08**	0.07
EOL-WP	**0.12**	0.02	0.02	0.05
Filter BC supply chain	**0.10**	**0.19**	**0.10**	0.13
Impregnated BC supply chain	**0.07**	**0.09**	**0.17**	0.11
N removal BC Bed	0.00	**0.11**	**0.17**	0.09
BC Bed Sensitive				
Parameter name	Climate	Eutrophication	Acidification	Mean
BC Stability	**0.10**	0.00	0.00	0.03
Electricity fossil/renewable	**0.15**	**0.19**	**0.18**	0.17
Filter BC supply chain	**0.32**	**0.37**	**0.22**	0.30
N removal BC Bed	0.00	**0.13**	**0.24**	0.12
BC Modular filter				
Parameter name	Climate	Eutrophication	Acidification	Mean
EOL-WP	**0.20**	0.04	0.04	0.09
EOL-GW	**0.03**	0.01	0.01	0.02
Filter BC supply chain	**0.07**	**0.06**	**0.10**	0.08
N removal BC filter	0.00	**0.12**	**0.29**	0.13
N removal ST	0.00	0.03	**0.15**	0.06
P removal ST	0.00	**0.44**	0.00	0.15

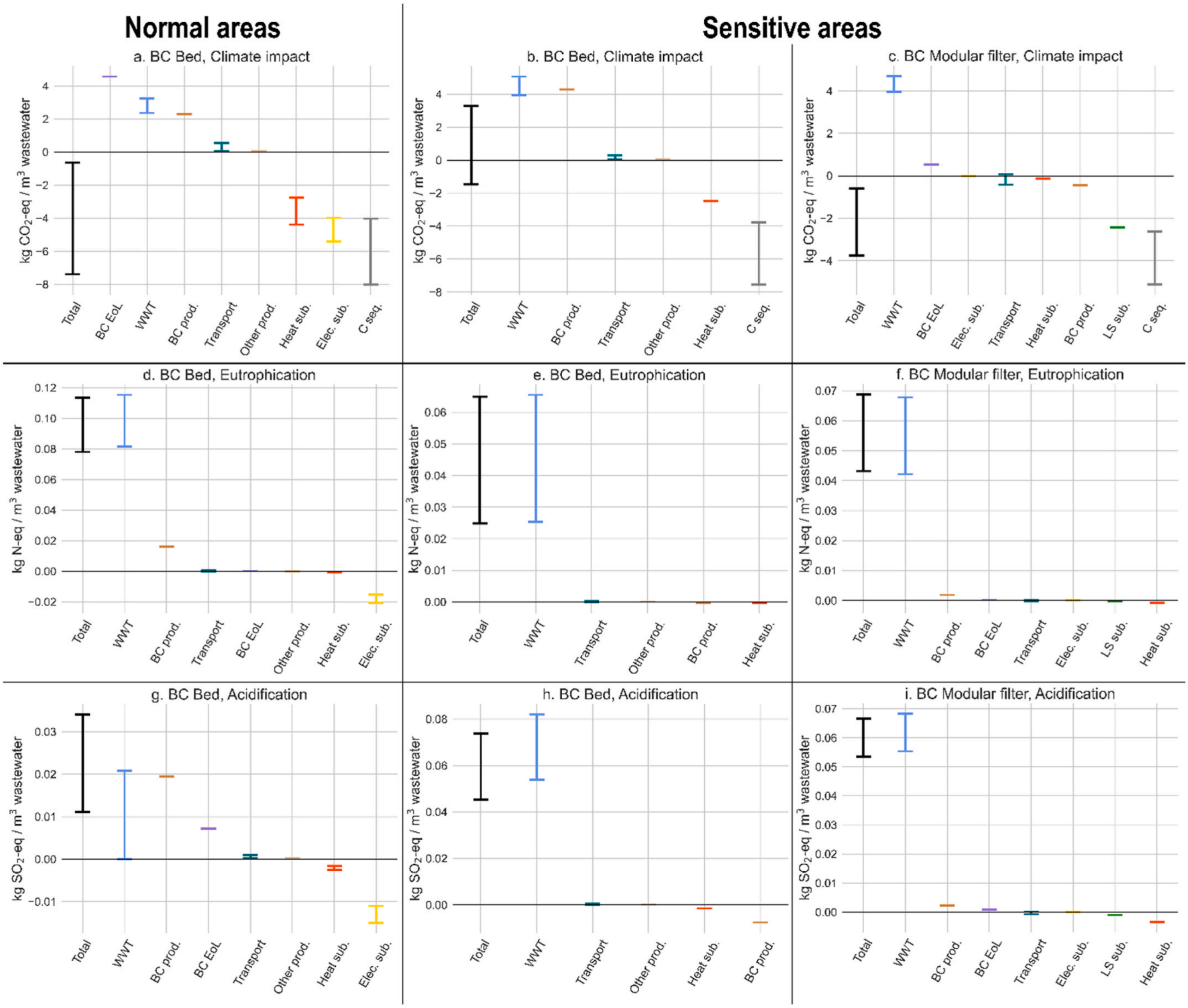

Fig. 6. Contribution of each subsystem to the total impact in three scenarios: Biochar bed, normal areas (a, d, g), biochar bed, sensitive areas (b, e, h) and modular biochar filter (c, f, i), and of three impact categories: climate impact (a–c), eutrophication (d–f) and acidification (g–i). Minimum and maximum impacts of each group and the total impact (in black) are indicated. In the contribution analysis, the impacts were divided into the following nine groups: Biochar production (including substitution effect for biochar feedstocks, processing of the feedstocks, pyrolysis and iron chloride for the impregnated biochar) wastewater treatment (emissions of pollutants to air and water), biochar end-of-life, transport, other production (production of the wastewater system excluding biochar production), electricity substitution, heat substitution, substitution of conventional landscaping soil and carbon sequestration (due to biochar being left in the filter or used in landscaping soil). BC = biochar. BC prod. = biochar production. WWT = wastewater treatment. BC EoL = biochar end-of-life. Elec. = electricity. Sub. = substitution effect. LS = landscaping soil. C seq. = carbon sequestration.

treatment accounted for most of the impacts and most of the variation in the results, while other impacts were rather insignificant. This is in line with the finding that the most sensitive parameters for sand filters (in both normal and sensitive areas) were CH_4 emissions to air (affecting the climate impact), amount of N removed in the sand filter and septic tank (affecting all impact categories) and amount of P removed in the filter and septic tank (affecting eutrophication only). Here, the sum of Sobol indices was close to 1 (0.97–1.0), meaning that the interaction between several parameters did not contribute considerably to the variation. A general contribution analysis for the sand filter and all biochar filters, i.e. not divided into the scenarios, is presented in Figs. S1–S12 in SM.

4. Discussion

4.1. Comparison of environmental impact of biochar filters and sand filters

In terms of climate impact, the biochar filters varied considerably between net negative and positive impacts, while the sand filter had a relatively low net positive climate impact. Several parts of the life cycle gave rise to this variation in results for the biochar filters. Biochar production and end-of-life, along with their substitution effects and carbon sequestration, affected the results in total and their variation. The climate impact from wastewater treatment (emissions of the greenhouse gases CH_4 and N_2O) was substantial, especially in sensitive areas, but it did not contribute considerably to the variation. This is in line with Sobol indices indicating that the most sensitive parameters were the biochar supply chain, biochar end-of-life, electricity fossil/renewable substitution and biochar stability.

For eutrophication impact the biochar filters and the sand filter were more similar, both in absolute terms and considering the variation. Biochar production and emissions from wastewater treatment contributed most to the impact of the biochar filters, while all other categories, including substitution effects, were less relevant. For the biochar bed in normal and sensitive areas, the most sensitive parameter was the biochar supply chain scenario, while for the biochar modular filter the most sensitive parameter was removal of P in the septic tank. It could be argued that as long as the national guidelines for onsite wastewater treatment are met, the eutrophication potential of the filters is acceptable in a local context, since the majority of eutrophication impacts originated from emissions from wastewater treatment itself.

The acidification impact in normal areas was lower for most biochar scenarios than for the sand filter, due to substitution effects compensating for the emissions. For some scenarios the total acidification impact was even net negative. However, for most scenarios and computations of the biochar filters in sensitive areas, the acidification impact was higher than for the sand filter. This was because of considerably higher acidification impact from biochar production compared with sand production and higher acidification impact from wastewater treatment (more ammonia emitted into air due to higher removal of N from the wastewater). The additional impacts were not compensated for by substitution effects. This is in line with the most sensitive parameter in sensitive areas being the amount of N removed in the filters.

Overall, the environmental impacts of the sand filter were dominated by the impacts from the wastewater treatment process, for all impact categories and in both normal and sensitive areas. This means that production and end-of-life emissions were relatively low for the sand filter. Since the filters had similar impacts for the wastewater treatment processes, for the biochar filter to have lower impacts than the sand filter there are two alternatives: (i) the biochar filter must have as low production and end-of-life impacts as the sand filter, which according to our results is not the case, and (ii) the biochar filter must offset its additional emissions with benefits, i.e. carbon sequestration and substitution effects.

Three scenarios that showed the high performance relative to the sand filter were selected for further analysis. For the biochar bed in normal areas, the scenario chosen was willow chips as biochar feedstock, with CHP as end-of-life, in a fossil energy system. This scenario had lower climate impact than the sand filter in 100% of the computations, 100% for acidification and 15% for eutrophication. For the climate impact, the carbon sequestration and energy substitution effects compensated for the additional climate impacts in the life cycle compared with the sand filter. Here, the biochar in the main filter was left in the ground, sequestering carbon, and the biochar impregnated with iron chloride was combusted as a CHP plant, substituting fossil energy. The acidification impact was lower than for the sand filter and, even when adding the substantial emissions from producing iron chloride to impregnate the biochar and combusting the biochar, the biochar still had a lower acidification impact in all computations. However, for the eutrophication impact, both the sand and biochar filter had similar emissions from the wastewater treatment process, and the biochar filter had an additional substantial impact from production of the biochar (mostly from producing iron chloride).

In sensitive areas, the scenario chosen for further analysis was for the biochar bed GW-S biochar in a fossil energy context, while that for the modular biochar filter was WP-S biochar in a renewable energy context and with landscaping soil as end-of-life scenario. For climate impact, both filters had lower impacts than the sand filter in 100% of the computations. Here, the biochar bed had a net negative climate impact for some computations and the modular filter for all computations. The biochar bed had higher climate impacts from biochar production, but also more carbon sequestration. For eutrophication, the filters also performed similarly and with lower impacts than the sand filter; the biochar bed had lower impacts than the sand filter in 97% of the computations and the modular biochar filter had lower impacts than the sand filter in 93% of the computations. The biochar bed also showed more variation in eutrophication impact, since the emissions from wastewater treatment had a lower minimum value than the modular biochar filter. For acidification impact, both filters showed poorer performance than the sand filter, where the biochar bed had lower impacts in 10% of the computations and the modular biochar filter had lower impacts in 0% of the computations. Both filters had higher acidification impact than the sand filter due to more emissions from the wastewater treatment process, which were not compensated for by substitution effects.

4.2. Impact of background systems, geographical area and upscaling

It was difficult to detect patterns in the results, probably due to the complexity of the model. For some system configurations, there was also a trade-off between several impact categories (see section 3.1). Trade-offs between several impact categories have been detected in previous LCAs of wastewater treatment systems (Corominas et al., 2013). The sum of Sobol indices was well below 1 (0.33–0.73), indicating that a substantial part of the variation was attributable to interactions between multiple parameters. This differs from the sand filter, where total climate impact and its variation were dominated by emissions from the wastewater treatment process.

The energy system affected several parts of the life cycle, i.e. reference usages of the biochar feedstock, pyrolysis during biochar production and the combustion end-of-life scenarios. Biochar feedstock also affected the life cycle in multiple ways, as the biochar types differed in reference feedstock use, emissions and energy from the pyrolysis process, combustion end-of-life scenarios and carbon content affecting carbon sequestration. The biochar feedstocks were modelled based on feedstock availability in Sweden, a country with extensive forest coverage; in other countries, the amount and types of feedstocks available could be vastly different.

The energy system contributed to the variation in the results, especially for the biochar bed filters with high Sobol indices for all three impact categories (Table 3). Although the biochar filters had lower impacts in general in the renewable energy system, whether the biochar was more favourable in the fossil or renewable energy system varied with the specific scenario and impact category. For example, the biochar bed with willow chip biochar in sensitive areas had lower climate impact in the fossil energy system than in the renewable energy system. It may seem counter-intuitive for a fossil energy system to result in a lower climate impact, which was due to substitution effects where co-production of bioenergy in the biochar system displaced carbon-intensive energy. The renewable energy scenario is relevant for Sweden and other countries with a relatively low-carbon energy system, but most of the world is still heavily dependent on fossil energy.

Biochar is a carbon dioxide removal technology, although in this study combustion of biochar was modelled in alternative end-of-life scenarios where no carbon sequestration was achieved. Either way, the climate impact of producing the biochar and emissions of greenhouse gases from wastewater treatment in many scenarios outweighed the carbon sequestration and substitution effects, resulting in net emissions of greenhouse gases. This confirms that a life cycle perspective is needed to evaluate biochar systems. On the other hand, several system configurations had a total climate impact of −15 kg CO_2-eq (or lower) per m^3 of wastewater treated. Considering that there are 700,000 OWTS in Sweden (Hansson et al., 2019), and assuming that all are of similar size and conditions as modelled in this study, this would amount to 1.87 million tons of CO_2-eq in Swedish OWTS annually. To put this in perspective, Sweden emitted a net total of 45.2 million tons CO_2-eq in 2022 (Naturvårdsverket, 2023).

4.3. Expansion of environmental assessment

The only water pollutants assessed in this study were BOD, N and P, as these influence the impact categories analysed and are also the pollutants covered by Swedish recommendations. However, when assessing the performance of a wastewater treatment plant there are additional pollutants of concern, such as pharmaceuticals. Such pollutants were omitted from this study, although there is evidence that biochar can perform well in removal of some (e.g. Vrchovecká et al., 2023). Some of the pollutants relevant for assessing treatment performance could be difficult to incorporate into an LCA (see e.g. Emara et al., 2018). Ecotoxicity impact categories could be assessed to capture more of these aspects and are particularly important for wastewater treatment (Corominas et al., 2020). The impact categories for toxicity and how they are characterised in LCA are not well-established and were therefore omitted in this study. A suitable approach to evaluate the impact of toxic substances could be substance flow analysis (Papageorgiou et al., 2021).

4.4. Use of parameterisation and analysis of uncertainties

Parameterisation of the LCI made it possible to model several scenarios and ranges of values, allowing analysis of variation and uncertainty in process inputs and outputs, as well as several possible system configurations. However, some uncertainties are worth highlighting.

Emissions to air from the wastewater treatment system made up a considerable part of the impacts for both the biochar and sand filters, but the composition and magnitude of gaseous emissions involved large uncertainties, as there are few sources of data on gaseous emissions from wastewater treatment, particularly in OWTS. Moreover, very different values for gaseous emissions have been reported, some of which were not compatible with the mass balance provided by Risch et al. (2021) as there was more carbon in emissions to the air than in the incoming wastewater. These values were not used in the model. This inconsistency could be due to e.g. different systems, different contexts or large fluctuations in these emissions throughout the lifetime, and thus measured values might be considerably different from the average lifetime values, the latter being of interest for this study. In addition, emissions to air depend on e.g. concentration of pollutants, pH and outdoor temperature. Lower average temperatures, as in Sweden, would result in less microbial activity and higher solubility of gases in water, which in turn would mean lower greenhouse gas emissions.

Some of the treatment efficiency values used for the biochar filters, in particular for the modular biochar filter, were based on laboratory experiments. Treatment efficiency of the biochar filters were sensitive parameters for eutrophication and acidification impacts. There is a shortage of data on biochar filters implemented at a larger scale. When scaling up this technology, additional factors could affect the performance of the filters (e.g. outdoor temperature, variations in hydraulic rate or variations in pollutant concentration in the incoming wastewater).

It was assumed that all biochars in the model had the same treatment performance, since there were no other data to rely on. However, the properties of biochar are strongly influenced by pyrolysis conditions and feedstock type (Ippolito et al., 2020). and the biochar characteristics in turn affect the wastewater treatment efficiency. For example, a higher specific surface area means more surface for bacteria to settle on, as well as more adsorption sites, potentially resulting in higher removal of BOD, N and P. A higher pyrolysis temperature is correlated to a higher specific surface area, but also a lower biochar yield (Qambrani et al., 2017). Therefore, increasing the pyrolysis temperature could result in higher wastewater treatment efficiency, but more feedstock could then be required to obtain the same amount of biochar. Furthermore, all feedstocks modelled were predominantly woody biomass, so it was reasonable to assume that they had similar properties, or at least not differing as much as, say, wood biochar and sewage sludge biochar. Biochar from woody biomass tend to have higher specific surface areas than biochar produced from other feedstocks (Ippolito et al., 2020), indicating wood as a suitable feedstock for biochar applied in wastewater treatment systems.

The sand and biochar systems were all assumed to have a hydraulic loading rate of 50 L m^{-2} day^{-1}, according to Swedish regulations based on sand filters (Swedish Agency for Marine and Water Management, 2016), which were applied when dimensioning the biochar filters since there are no corresponding regulations for biochar filters. It could be argued that biochar filters should be dimensioned in a different way, e.g. by making them smaller or larger due to higher or lower treatment efficiencies. It is also possible that biochar produced specifically for wastewater treatment would have different particle size and density than the biochars modelled in this study, which in practice could result in different amounts of biochar in the filters. Lastly, the dimensioning of filters could differ between countries, depending not only on local regulations, but also types and concentrations of pollutants in the wastewater, which in turn depends on e.g. water availability and diet.

To conclude, use of biochar filters for wastewater treatment is an interesting add-on for OWTS, but it is not yet a mature technology. Therefore, there are uncertainties and shortages of data, affecting the LCA results. Some of the uncertainties and possible scenarios were captured in this paper, but some remain to be assessed in future research, hopefully with more data and examples of full-scale implementations. It is clear from our results that the environmental impacts of biochar filters are considerable compared with those of conventional technologies and must be evaluated using a systems perspective. Biochar can be an environmentally sound alternative, sequestering carbon and adding to other societal functions while supplying the same wastewater treatment functions. In general, not all innovations are environmentally favourable, and LCA can be performed to make an initial assessment and rule out suboptimal system designs.

5. Conclusions

The climate, eutrophication and acidification impacts of a biochar bed and a modular biochar filter were calculated and compared with those of a sand filter for on-site wastewater treatment. A total of 36 scenarios were assessed, consisting of different biochar supply chains, biochar end-of-life and energy system. Based on our analyses, the following can be concluded:

- Compared with the sand filter, the acidification impacts of the biochar filters were lower in normal areas and higher in sensitive areas, the eutrophication impacts were similar and the climate impact differed vastly, being both considerably higher and lower in different scenarios than for the sand filter.
- The biochar filters had particularly low climate impacts when the biochar was used in landscaping soil at end-of-life, rather than for combustion.
- Willow chip biochar had an especially low climate impact, while all biochars produced in syngas-heated pyrolysis in a fossil energy system had lower impacts than biochars produced in electricity-heated pyrolysis.
- The biochar filters had in general lower impacts in a renewable energy system compared with the fossil energy system.
- Parts of the life cycle that contributed most to the magnitude and variation in the results were production of biochar, emissions to air and water from wastewater treatment, end-of-life of the biochar and substitution effects.
- In order for the biochar filter to have lower environmental impacts than the sand filter, the higher impacts from biochar production compared with sand production and end-of-life have to be compensated for by carbon storage in biochar and additional benefits.

- Specific data on biochar filters and on onsite wastewater treatment systems in general are currently lacking, demonstrating a need for more empirical research.

CRediT authorship contribution statement

Lisa Zakrisson: Writing – original draft, Visualization, Validation, Software, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Cecilia Sundberg:** Writing – review & editing, Supervision, Methodology, Funding acquisition, Conceptualization. **Gunnar Larsson:** Writing – review & editing, Supervision, Methodology, Conceptualization. **Elias S. Azzi:** Writing – review & editing, Software, Methodology, Conceptualization. **Sahar S. Dalahmeh:** Writing – review & editing, Supervision, Project administration, Methodology, Funding acquisition, Conceptualization.

Relevant support outside this work

Author Elias S. Azzi reports a relationship with Puro.Earth Oy that included consulting or advisory at the time when the research was conducted.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Sahar Dalahmeh reports financial support was provided by Swedish Research Council. Sahar Dalahmeh reports financial support was provided by Swedish Research Council Formas. Elias Azzi reports a relationship with Puro.Earth Oy that includes: employment. If there are other authors, they 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

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Data availability

Data will be made available on request.

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