Master's thesis

NTNU
Norwegian University of Science and Technology
Faculty of Engineering
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A life cycle assessment to identify benefits and trade-offs of nitrogen-enriched biochar in an agricultural context

Master's thesis in Industrial Ecology
Supervisor: Francesco Cherubini
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Abstract

Biochar is a negative emission technology (NET) that can sequester carbon in soil while providing benefits for agricultural soils. Liquid manure storage is a significant source of atmospheric emissions of ammonia (NH_3), and biochar has been shown to effectively reduce NH_3 emissions from livestock manure. In this study, the use of biochar as an adsorbent of nitrogen from pig manure, and provider of the adsorbed nitrogen to agricultural soils was investigated. A life cycle assessment (LCA) was conducted to quantify the environmental impacts of the nitrogen-enriched (N-enriched) biochar system when applied to agricultural soils. Two methods of N-enrichment were investigated, adsorption of liquid ammonium (NH_4^+) and gaseous ammonia (NH_3). To be able to identify environmental benefits and trade-offs for the system, the N-enriched biochar was compared to using non-enriched biochar, and not using biochar in agricultural soils.

The main benefits of the N-enriched biochar system are reduced terrestrial acidification and reduced particulate matter formation, due to mitigation of ammonia (NH_3) emissions from pig manure storage. N-enriched biochar using liquid adsorption and air adsorption have 120% and 170% lower impact on terrestrial acidification, respectively, compared to not using biochar in agricultural fields. The N-enriched biochar also provides a benefit for marine eutrophication, as the need for mineral nitrogen fertilizer for crop cultivation is reduced due to the recycling of nitrogen from pig manure storage. The main trade-offs are increased freshwater eutrophication, human toxicity, and climate change, depending on the N-enrichment method. For human toxicity, liquid adsorption and air adsorption have 100% and 146% higher impact compared to not applying biochar in barley fields.

The results of this study shows that there are both benefits and trade-offs related to enriching biochar with nitrogen from pig manure storage and applying it to agricultural fields compared to not using biochar, so whether to implement the system or not depends on which environmental concern is in focus. Several hotspots were identified in this study, e.g. that biochar has the ability to significantly reduce NH_3 emissions from pig manure storage. Other hotspots identified are that the benefit of carbon sequestering from biochar for climate change is almost evened out by the increased methane emissions when adding biochar to manure, and that using phosphoric acid to increase biochar's adsorptive ability is related to significant impacts across all environmental impact categories included in this analysis. These insights can be used to improve the system to perform better. As nitrogen enrichment of biochar from pig manure is a novel field of study, more research is needed to increase the knowledge on this topic.

Sammendrag

Biokull regnes som en negativ utslippsteknologi (NET), som kan binde karbon i jorda og samtidig tilføre fordeler til landbruksjord. Lagring av dyregjødsel er en betydelig kilde til utslipp av amoniakk (NH_3), og biokull har vist seg å være effektiv til å redusere NH_3-utslipp fra husdyrgjødsel. Denne studien har sett på å bruke biokull som en adsorbent av nitrogen fra grisegjødsel, og videre tilførsel av det adsorberte nitrogenet til landbruksjord. En livssyklusanalyse (LCA) ble gjennomført for å kvantifisere miljøkonsekvensene av nitrogenberiket (N-beriket) biokull i landbruksjord. To metoder for nitrogenberikelse av biokull ble undersøkt, adsorbering av flytende ammonium (NH_4^+) og NH_3 i gassform. For å kunne identifisere miljømessige fordeler og ulemper for systemet, ble N-beriket biokull sammenlignet med bruk av vanlig biokull, og ikke å tilføre biokull i landbruksjord.

De viktigste fordelene med det N-berikede biokullsystemet er redusert jordforsuring og redusert svevestøv, grunnet reduserte utslipp av NH_3 fra grisegjødsel. Biokull beriket med NH_4^+ og NH_3 har henholdsvis 120% og 170% lavere påvirkning på jordforsuring, sammenlignet med å ikke bruke biokull i landbruksjord. N-beriket biokull gir også en fordel for marin eutrofiering, ettersom behovet for mineralgjødsel blir redusert når biokullet tilfører nitrogenet som har blitt tatt opp fra grisegjødsel. De viktigeste miljømessige ulempene med N-beriket biokull er økt eutrofiering av ferskvann, menneskelig toksisitet, og global oppvarming, avhengig av hvilken metode for N-berikelse som velges. Biokull beriket av NH_4^+ og NH_3 har henholdsvis 100% og 146% høyere påvirkning på menneskelig toksisitet sammenlignet med å ikke bruke biokull i byggåkre.

Resultatene fra denne studien viser at det er både miljømessige fordeler og ulemper ved å bruke biokull beriket med nitrogen fra grisegjødsel i landbruksjord. Siden begge metodene for N-berikelse av biokull er knyttet til fordeler og ulemper, må en beslutning hvorvidt man skal implementere systemet tas på bakgrunn av hvilket miljømessig aspekt man er opptatt av. Noen nøkkelfunn fra studien er at biokull har evne til å redusere NH_3 utslipp fra grisegjødsel betydelig. Andre funn er at fordelen med karbonfangst i jorda fra biokull på klimaendringer blir nesten utjevnet av de økte metanutslippene (CH_4) som oppstår når man tilfører biokull til grisegjødsel, og at det å bruke fosforsyre til å øke adsopsjonsevnen til biokull medfører betydelig påvirkning på alle de valgte miljøindikatorene. Disse funnene kan brukes til å videre forbedre systemet til å ha en lavere miljøpåvirkning. Siden nitrogenberikelse av biokull fra grisegjødsel er et nytt forskningsfelt, er det viktig å få mer kunnskap på plass på dette området.

Preface

This master thesis concludes my Master of Science in Industrial Ecology at Norwegian University of Science and Technology (NTNU). It continues the work of my project thesis from autumn 2023, "Cascading Use of Biochar in the Agricultural Sector - Nitrogen-enriched Biochar" (Steinnes, 2023), where a system for nitrogen-enriched biochar was created and the topic of nitrogen enrichment was investigated. In this thesis, this work has been expanded, where several methods of nitrogen enrichment have been investigated, and the environmental performance of the system has been estimated by using Life Cycle Assessment (LCA). Although this master's thesis is an independent academic production, some parts are largely based on the project thesis. These parts are mainly found in the sections of introduction and methodology (Sections 2.2-2.6 and 2.8), and some of the discussion points.

I would like to thank my supervisor, Francesco Cherubini, for his valuable insights and expert advice. I am also very grateful for all the help my co-supervisor, Nariê Rinke Dias de Souza, has given me through this semester. She has put herself available to help solve problems, given thorough feedback, and motivated me along the way. I would also like to send a thank you to the researchers at NIBIO working on the AgriCascade project, who have provided advice and insight on the complex topic of biochar and nitrogen enrichment, which I am very grateful for.

Lastly, I would like to thank my classmates for contributing to making a great social and academic environment at the IndEcol master's programme. Over these two years, I have gained new perspectives on life, and together we have created memories I will carry with me in the future.

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1 Introduction

Limiting global warming to well below 2°C, in line with the Paris Agreement, requires more than just emission reductions and sustainable use of resources. The carbon dioxide (CO_2) concentration in the atmosphere has to be reduced (Field & Mach, 2017), and negative emission technologies (NET) for carbon dioxide removal (CDR) play a crucial role in achieving this goal. CDR methods intentionally remove CO_2 from the atmosphere and store it in geological, terrestrial, or ocean reservoirs (Pathak et al., 2022). These methods include among other things carbon sequestration in soil, afforestation and reforestation, direct air capture, biochar, and bioenergy with carbon capture and storage (BECCS) (Minx et al., 2018).

Biochar is produced through the pyrolysis process of burning biomass in the absence of oxygen, where carbon is stored in a stable form in the charcoal (Meiirkhanuly et al., 2020). Among different options for sequestering carbon in soil, biochar application to soil is being considered as the most promising option to sequester carbon long-term in biomass (Gupta et al., 2020). Various biomass types can be used to produce biochar, including forest and crop residues, municipal sludge, and livestock manure (Ahmad et al., 2014). Besides from sequestering carbon, biochar promotes several benefits when applied to agricultural soils. Biochar application can improve soil conditions by enhancing soil structure and aeration (Omondi et al., 2016), increase the soil's nutrient holding capacity, and mitigate nitrogen losses from soil resulting from fertilizer use (Gao et al., 2019; Q. Liu et al., 2018). A meta-analysis conducted by Q. Liu et al. (2018) found that, on average, biochar application decreases soil nitrous oxide (N_2O) emissions by 32% and nitrogen leaching by 22 to 29%, but that it can increase ammonia (NH_3) volatilization from the soil by 19%. Biochar application can lead to crop yield increase, but the effectiveness varies by region (Jeffery et al., 2017). Overall, biochar provides a solution to several environmental challenges as it sequesters carbon in the soil, improves soil health, and reduces emissions from agriculture, all without competing for resources, as existing biomass residues can be utilized.

Liquid manure storage is a significant source of atmospheric emissions of methane (CH_4), NH_3, and N_2O (IPCC, 2006). The dominant emission type from manure storage is NH_3 (Miljødirektoratet, 2020), an air pollutant that can lead to water eutrophication, soil acidification, and increased climate change through N_2O (Y. Wang et al., 2017). NH_3 also contributes significantly to the formation of secondary particulate matter ($PM_{2.5}$) aerosols (Chen et al., 2021). Various practices have been adopted to minimize NH_3 and CH_4 emissions from liquid manure storage, such as using manure covers and acidifying the manure (Fangueiro et al., 2014; Sommer et al., 2000). Biochar has been shown to effectively reduce NH_3 emissions from livestock manure (Brennan et al., 2015; Meiirkhanuly et al., 2020). Additionally, biochar has the ability to adsorb nitrogen from aqueous solutions (Jellali et al., 2022; Takaya et al., 2019). These findings suggest that biochar can have the potential to mitigate emissions from livestock manure while simultaneously adsorbing nitrogen. If the adsorbed nitrogen is released to plants after biochar is applied to the soil, this can have the potential to substitute mineral nitrogen fertilizer.

The main production method for nitrogen fertilizers is through the Haber-Bosch process, where NH_3 is synthesized from hydrogen, usually taken from CH_4, and nitrogen taken from the air (Walling & Vaneeckhaute, 2020). NH_3 is subsequentially used to produce nitrogen fertilizers, mainly in the form of urea and ammonium nitrate ($NH_4^+NO_3^-$) (Walling & Vaneeckhaute, 2020). Ammonia production for fertilizer use is related to several environmental concerns, as the Haber-Bosch process uses fossil fuels as feedstock, and is one of the largest energy consumers and greenhouse gas (GHG) emitters, responsible for 1.2% of anthropogenic CO_2 emissions (Nørskov et al., 2016; Smith et al., 2020). By recycling gaseous NH_3 or liquid ammonium (NH_4^+) from pig manure by using biochar as an adsorbent and then applying the biochar to agricultural fields, some of the mineral fertilizer production can be offset, leading to reduced fossil fuel use and energy use.

The properties of biochar vary depending on the feedstock. Wood-based biochars generally contain more carbon but fewer nutrients available for plants (Ippolito et al., 2015). Manure-based biochars show the opposite trend, with a high nutrient content but low carbon content. By using wood-based biochar and enriching it with nitrogen from manure, one can achieve both high carbon sequestration and nutrient provision for plants. Additionally, wood biochars have in general higher surface areas (Ippolito et al., 2020), which has been correlated with greater adsorptive capacity (Qambrani et al., 2017). In Norway, where large amounts of forest residues are left unused (Tisserant et al., 2022), utilizing these residues to produce biochar can contribute to a circular economy perspective.

Biochar has been extensively studied for its benefits when applied to agricultural fields (Q. Liu et al., 2018), as well as its adsorption capacity of nitrogen from aqueous solutions (Jellali et al., 2022). Some studies have looked into biochar's potential to mitigate ammonia and GHG emissions from livestock manure management (Baral et al., 2023; M. Liu et al., 2021; Meiirkhanuly et al., 2020), however, these results are varying. Interestingly, few studies have explored the multi-purpose use of biochar, which includes not only biochar production but also its role in nitrogen enrichment from manure, and its application to agricultural fields. The hypothesis for this study is that biochar can be employed in a cascading way where it is first used to sequester carbon, then to reduce NH_3 and GHG emissions while adsorbing nitrogen from manure storage, and finally be applied to agricultural soils and provide benefits such as reducing soil emissions and soil nitrogen leaching.

The general objective of this study was to explore the potential use of biochar as an adsorbent of nitrogen and a provider of the nitrogen to agricultural fields. First, biochar is used to sequester carbon, then to reduce NH_3 and GHG emissions and adsorb nitrogen from manure storage. Finally, biochar is applied to the soil where it provides the nitrogen for plant uptake and soil benefits such as reduced emissions and nutrient leaching. Based on the main objective, the research question for this study was formulated as:

What are the environmental impacts and benefits associated with the use of nitrogen-enriched biochar in an agricultural context?

To be able to answer the research question, some specific objectives have been formulated:

1. Define a system for nitrogen-enriched biochar
2. Perform a Life Cycle Assessment (LCA) of the different scenarios using the management of 1 hectare of barley field as the functional unit
3. Identify and discuss hotspots and possible improvements

2 Methodology

The methodology section is structured as follows: Section 2.1 defines the goal and scope for the LCA, and which environmental impact categories that were chosen for the analysis. Section 2.2 presents the system boundaries and the four scenarios, and Sections 2.3 to 2.8 describe the four scenarios and the relevant processes. Lastly, Section 2.9 describes the sensitivity analysis for the LCA.

2.1 LCA: Goal and Scope Definition

The goal of the analysis was to measure the environmental performance of using nitrogen-enriched (N-enriched) biochar in agricultural soils. To be able to identify the impact of the N-enriched biochar, it was compared to using non-enriched biochar, and not using biochar in the agricultural field. Environmental impacts of the different scenarios were investigated using LCA, with the scope "cradle-to-gate". The functional unit chosen for this study is 1 hectare of barley field. Background life cycle inventories were retrieved from ecoinvent v3.9 and operationalized using Brightway2.

To analyze the environmental impacts of the N-enriched biochar system, six impact categories were chosen: terrestrial acidification, particulate matter formation, marine eutrophication, freshwater eutrophication, human toxicity, and climate change. The four first impact categories were assessed by using Midpoint Hierarchist characterization factors from ReCiPe 2016 v1.03 (Huijbregts et al., 2017). To assess the toxicity of chemicals on human health, USEtox was chosen (Henderson et al., 2011), where the unit is human comparative toxic units (CTU_h), that is the cumulative cases of either cancer or non-cancer outcomes ($\text{cases/kg}_{emitted}$). For this study, the total impact is chosen which includes both carcinogenic and non-carcinogenic impacts.

For climate change, the metric GWP100 was chosen, as GHGs have a long lifetime. As the N-enriched biochar system is expected to have significant impacts on NH_3 emissions from pig manure storage, it is interesting to look at how the indirect effect NH_3 has on climate change appears in this system. Therefore, the analysis will also include how the system performs for near-term climate forcers (NTCFs), in addition to long-term GHGs. NTCFs consist of chemically and physically reactive compounds that remain in the atmosphere typically shorter than 20 years, and they can have both direct and indirect effects on climate (Szopa et al., 2021). The direct NTCFs include CH_4, ozone (O_3), short-lived halogenated compounds, and aerosols, while the NTCFs that have indirect effects on climate include nitrogen oxides (NO_x), sulfur dioxide (SO_2), carbon monoxide

(CO), non-methane volatile organic compounds (NMVOC), and NH_3 (Szopa et al., 2021). NTCFs can have either a warming or a cooling impact on the climate, and in addition, they affect precipitation and other climate-related variables (Szopa et al., 2021). The characterization factors for the GHGs are taken from IPCC 2021, while the characterization factors for NTCFs (GWP100) are taken from other sources (Table A1 in the Appendix).

Global warming potential (GWP) is the increased global atmospheric temperature due to specific air emissions (Edwards et al., 2018). Terrestrial acidification results from emitted pollutants decomposing in the atmosphere, and can lead to a reduction in soil pH, which subsequently can induce loss of plant species in the terrestrial ecosystem (Roy et al., 2014). Particulate matter formation is a common indicator for air pollution (World Health Organization (WHO), 2022), where the larger particles (PM10) can lead to allergenic responses (Arias-Pérez et al., 2020), while fine particles (PM2.5) are small enough to penetrate enter the blood system (Schulze et al., 2017) and can lead to cardiovascular and respiratory diseases, and cancer (World Health Organization (WHO), 2022). Water eutrophication refers to the increased algae growth due to excess nitrogen and phosphorus entering the waterbodies, which can result in oxygen depletion and the death of fish and other aquatic species (Y. Zhang et al., 2021). Human toxicity is an impact category that measures the impact of emissions of toxic substances on human health (Jolliet & Fantke, 2015).

2.2 Scenario Description

To be able to identify the environmental benefits and trade-offs of using N-enriched biochar in agricultural soils, the study compares the environmental impact of N-enriched biochar applied to agricultural soil with non-enriched biochar and a scenario without biochar application. Two methods of N-enrichment of biochar were investigated, resulting in four scenarios for the analysis. Figure 1 shows the four scenarios and system boundaries for the LCA. The N-enriched biochar scenario includes biochar production, N-enrichment of biochar, and application of N-enriched biochar to one hectare of barley field in Norway. As the study compares barley cultivation with or without biochar application, the functional unit considered in this analysis is the management of one hectare of barley field per year in Norway.

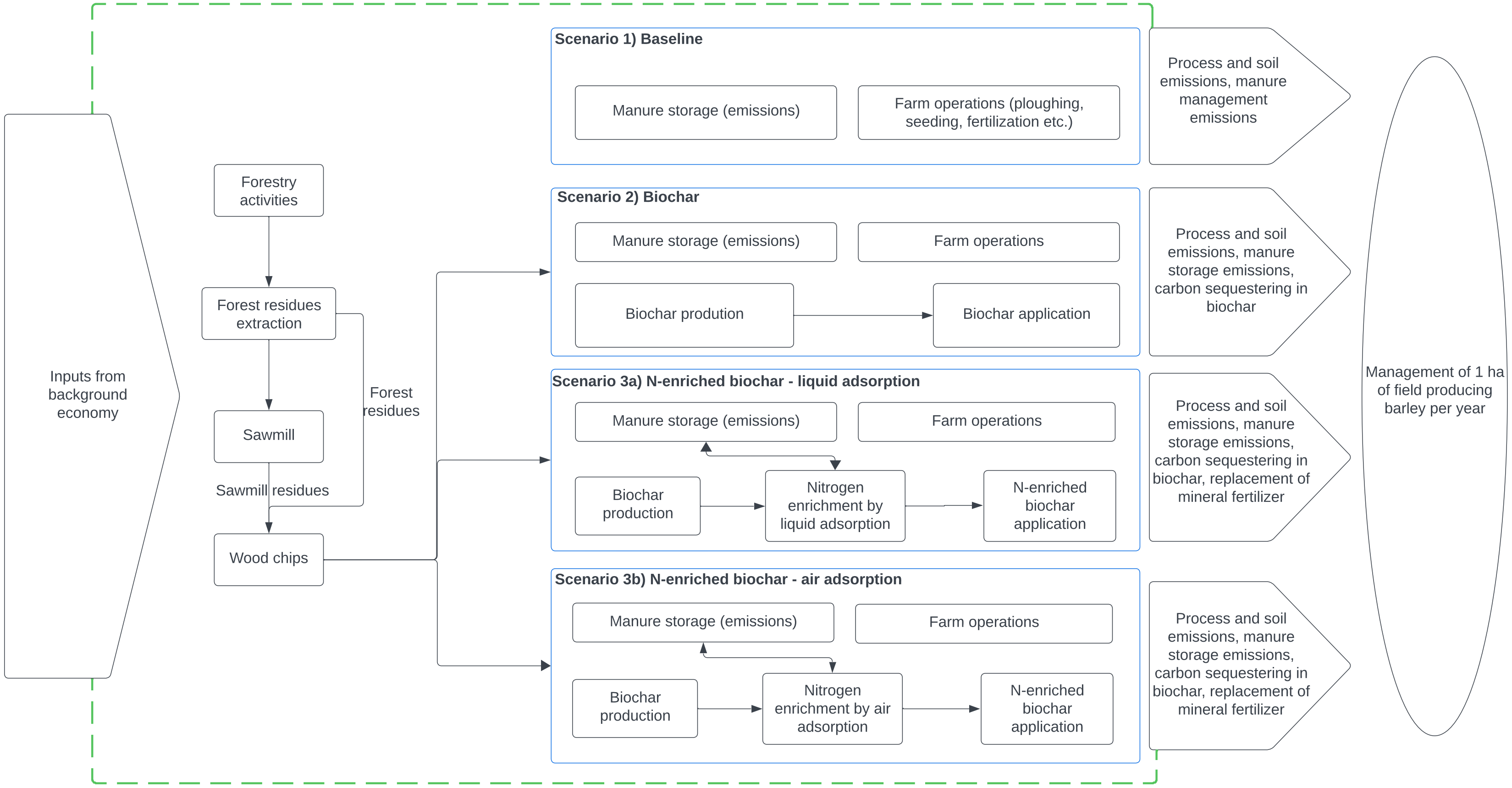


Figure 1: Scenarios: 1) Baseline, 2) Biochar, 3a) N-enriched biochar - liquid adsorption, and 3b) N-enriched biochar - air adsorption. Figure adapted from Tisserant et al. (2022) to the system for this study.

2.3 Scenario 1: Baseline

Scenario 1 includes farm operations used to manage 1 hectare of barley field across a year, such as harrowing, liming, fertilizing, and use of pesticides. In addition, the scenario includes emissions from manure storage.

2.3.1 Emissions from manure storage

For this study, pig manure was selected as the type of manure. This choice is based on that biochar addition to pig manure is well explored in literature, it has high nitrogen content, and pig manure is largely available in Norway. During storage, organic nitrogen in the manure is transformed into various nitrogen types. It can be transformed into liquid ammonium (NH_4^+ and NH_3, and subsequentially into gaseous NH_3 (Philippe et al., 2011). Further, liquid NH_4^+ can be transformed into liquid and gaseous N_2O and dinitrogen (N_2). Liquid pig manure has a high concentration of ammonium (NH_4^+), with reports of 1 280 and 2 870 mg L^{-1} (Garcia-González & Vanotti, 2015; Nunes et al., 2023).

Emissions of NH_3, CH_4, CO_2, and N_2O from manure storage used for this study are 240, 1 270, 8 000, and 2.00 mg per m^2 per hour, respectively. These are the baseline emissions for pig slurry stored in tanks based on a review study on emissions from cattle and pig slurry (Kupper et al., 2020). For CH_4 emissions, the baseline value was reported in m^3, so emissions per m^2 were calculated by filtering their dataset for pig manure emissions, pilot-scale studies, and the ones they had marked that were used for baseline emission calculations. The average value of the remaining studies after the filtering was used for this study (1 270 mg m^{-2} h^{-1}).

To quantify the yearly emissions from pig manure storage for the system in kg, the first step was to calculate the total area of manure storage needed to enrich the biochar. As the amount of biochar applied per hectare in of barley field in kg (app_{field}) was known before (explained in Section 2.4.2), the amount of biochar applied to manure in kg (app_{manure}) had to be determined (explained in Section 2.6.2) and from there estimate the total area of manure storage needed for the system in m^2 (A_{manure}), given by Equation 1.

$$A_{manure} = \frac{app_{field}}{app_{manure}} \tag{1}$$

From there, Equation 2 was used to find the emissions from manure storage for the specific gases in kg per year (E_x). This was done by taking the total area of manure storage (A_{manure}), and multiplying it by the emissions for the specific gas per hour (e_x), and the number of hours biochar was added to manure (t). The quantified parameters used in the equations for this study can be found in the appendix (Table A3).

$$E_x = \frac{A_{manure} \cdot e_x \cdot t}{10^6} \tag{2}$$

As pig manure storage is happening regardless if biochar is added or not, a manure storage facility is not included as an input in the LCA inventory (Table A7). For the same reasoning, only the changes in manure emissions due to the biochar addition compared to baseline emissions are displayed in the main results, and not the total emissions from manure storage. The CO_2 emissions from manure storage are considered biogenic as the carbon comes from biomasses, so these emissions have no impact on climate change.

2.3.2 Farm Operations

Farm operations include activities and products needed to manage 1 hectare of barley field across a year. Barley is the main cereal produced in Norway and makes up about 45% of the total grain production (SSB, 2023). The LCA inventory for the farm operations used to manage 1 hectare of barley field (Table A4) is created based on Tisserant et al. (2022). Practices included in the inventory are among other things ploughing, sowing, harrowing, fertilizing, pesticide application, and liming. Yearly fertilizer requirements per hectare of barley field are 127.5 kg nitrogen (N), 17.3 kg phosphorus (P), and 63 kg potassium (K) (Gundersen & Heldal, 2013; Kolle & Oguz-Alper, 2020), and the liming requirement per hectare is 447 kg per year (Tisserant et al., 2022).

Soil emissions due to fertilizer application for Scenario 1 are based on the Norwegian emissions inventory report (Miljødirektoratet, 2019; Tisserant et al., 2022). It is assumed that 1% of the nitrogen applied, 1% of the volatilized nitrogen, and 0.75% of the leached nitrogen is emitted as N_2O. For every kg of nitrogen applied, 5% emitted as NH_3, 0.04 kg is emitted as NO_x, and 22% is leached from the soil as nitrate (NO_3^-).

2.4 Scenario 2: Biochar

Scenario 2 includes the production of biochar, application of biochar to agricultural soils, farm operations, and emissions from manure storage.

2.4.1 Biochar Production

Biochar production can be divided into two processes, biomass collection and treatment, and the pyrolysis process. The data for both processes is based on Tisserant et al. (2022). In Norway, it is common to leave the forest residues in the forest since there is no market for utilizing low-quality wood and branches, and overall, forest residues in Norway consist of 82% spruce, 17% pine, and 1% birch (Tisserant et al., 2022). The life cycle inventory for the biomass collection includes the complete value chain, from harvesting, transport, chipping, and processing of the forest residues.

Transport of forest residues from the forest to the pyrolysis plant was estimated based on Tisserant et al. (2022), where the average transport distance of 190km was used. Spruce wood is the chosen biochar feedstock, and a temperature of 500°C was considered for the pyrolysis process. By-products from the pyrolysis process, such as bio-oil and syngas, are not included. A mass yield of 28% biochar per unit wood input (dry basis) was considered for this analysis, similarly to Tisserant et al. (2022). This ratio was used to assess the amount of wood chips needed as input per output of biochar. For the LCA inventory of biochar production (Table A5), softwood chips were used as input, as ecoinvent v3.9 does not have forest residues as an activity. As the CO_2 that is emitted while burning the wood has been taken up from the atmosphere through photosynthesis, these emissions are modeled to be biogenic and do not impact climate change.

2.4.2 Biochar Application

Transport of the biochar from the pyrolysis plant to the field was estimated based on Tisserant et al. (2022), where the average transport distance of 226 km was chosen. Their analysis considered an application rate of 2 552 kg biochar per hectare of barley field per year, so the same amount is used for this study (app_{field}). The LCA inventory for biochar application for this study (Table A6) was also created based on Tisserant et al. (2022). Biochar application includes broadcasting of biochar to agricultural soils and incorporation into the soil through harrowing. Since the spruce wood contains calcium (as $CaCO_3$) that remains in the biochar, the yearly need for liming is reduced by 145 kg per ha (Tisserant et al., 2022). In addition, a reduction in K fertilizer is obtained, as softwood biochar naturally contains 0.5% of K_2O that is available for plant uptake (Ippolito et al., 2015; Tisserant et al., 2022). The new K fertilizer need in kg ha^{-1} year^{-1} ($F_{K}new$) was calculated using Equation 3 where F_k represents the initial K fertilizer requirement in kg per year, and b_K corresponds to the amount of K_2O in softwood biochar in percentage.

$$F_K new = F_K - (app_{field} \cdot b_K) \tag{3}$$

Modeled effects on soil emissions from biochar application include N_2O, NO_x, NH_3 and nitrogen leaching, and the inventory is based on Tisserant et al. (2022), where the average values from their literature review are used. These values are 38% reduction of N_2O emissions, 5% increase in NH_3 emissions, 10% reduction in NO_x emissions, and an 8% reduction in nitrogen leaching. After biochar application, no increase in barley yield is assumed for this study, as previous studies have found that biochar has a limited effect in increasing grain yields in Norway (O'toole et al., 2018). The LCA inventory for the management of 1 hectare of barley field with modeled effects from biochar application can be found in the appendix (Table A4).

It is assumed that after 100 years, 74% of the carbon still remains in the biochar (Budai et al., 2016; Tisserant et al., 2022). Application of 2 552 kg biochar per ha of agricultural soil was found to have a carbon sequestering potential of 5 350 kg CO_2-eq. ha^{-1} $year^{-1}$ (Tisserant et al., 2022), so this value is also used for this study.

2.5 Scenarios 3a and 3b: Nitrogen-enriched Biochar

The nitrogen-enriched (N-enriched) biochar scenarios include biochar production, using biochar to adsorb nitrogen from manure storage, application of N-enriched biochar to agricultural soils, and farm operations. Production of biochar and farm operations are identical to Scenarios 1 and 2. It is assumed that the biochar is used to adsorb N from pig manure, then extracted from manure and spread onto the field within the same farm. Therefore, no extra transportation is considered for these scenarios compared to Scenario 2.

Biochar has a great capacity to adsorb nutrients due to its porous structure, and biochar can adsorb different forms of nitrogen. The literature distinguishes between the adsorption of liquid NO_3^- and NH_4^+, and gaseous NH_3. As biochar pyrolyzed at around 500°C shows no or minimal adsorption capacity of NO_3^- (M. Zhang et al., 2020), adsorption of liquid NH_4^+ and gaseous NH_3 was chosen for this study. This resulted in two subscenarios for the N-enriched biochar, where Scenario 3a includes adding biochar into the manure to adsorb liquid NH_4^+, while Scenario 3b includes attaching biochar in the air above the manure to adsorb gaseous NH_3 (see Figure A1 in the appendix).

2.6 Scenario 3a: Nitrogen-enriched Biochar Using Liquid Adsorption

2.6.1 Scenario 3a: Biochar Adsorption of Liquid Ammonium

For this study, an adsorption capacity of 20.5 mg NH_4^+-N per gram biochar from liquid pig manure was chosen. This value was based on a literature review (Table 1) where a value of 28.3 mg NH_4^+-N g^{-1} biochar was chosen, and then adjusted to 22.8 mg NH_4^+-N g^{-1} according to a re-analysis of NH_4^+ adsorption of biochar by Weldon et al. (2022). Finally, the adsorption value was adjusted down by 10% to obtain a more realistic value. A detailed explanation of the approach is presented in the following paragraphs.

Table 1: Literature review of liquid adsorption of ammonium

Biochar feedstock	Pyrolysis temperature	N concentration (in N/L)	Adsorption (in NH_4^+-N)	Source
Spruce-pine-fir	500°C	4 000 mg*	26.3 mg/g	Jassal et al. (2015)
Pinewood	550°C	40 mg	3.4 mg/g	Yang et al. (2018)
Pinewood Pinewood	550°C 550°C	40 mg 79 mg	0.38 mg/g 0.52 mg/g	Hina et al. (2015)
Spruce wood	500°C	100 mg	4.4 mg/g	Li et al. (2021)
Wood shavings	600°C	1 400 mg	42.02 mg/g	Kizito et al. (2015)
Hardwood	600°C	500 mg	114.7 mg/g	Kizito et al. (2016)
Mixed sawdust pellets	600°C	500 mg	28.3 mg/g	Kizito et al. (2016)
Oak wood Oak wood	450°C 450°C	33 mg 349 mg	7.14 mg/g 23.2 mg/g	Takaya et al. (2019)**
Oak wood Oak wood	400°C 600°C	1 000 mg 1 000 mg	129.4 mg/g 123.5 mg/g	Takaya et al. (2016)

*N concentration reported as 4 000 mg N (from $NH_4^+NO_3^-$) per L, and the study measured adsorption of NH_4^+ and NO_3^- simultaneously. **The study reported adsorption in NH_4^+, so the values in the table are transformed to NH_4^+-N.

The experimental designs of the studies chosen from the literature review (Table 1) vary in terms of biochar feedstock, pyrolysis temperature, and the N concentration in the solution to which the biochar was added. As the NH_4^+ adsorption capacity of biochar varies significantly depending on the feedstock type and the pyrolysis conditions (Jellali et al., 2022), the chosen studies from the literature should be as similar as possible to the chosen biochar for this study. Softwood, and more specifically spruce wood, is the optimal feedstock, but as most studies looking at softwood biochar have used a very low N concentration (40-100 mg N L^{-1}), these are not comparable to adsorption from pig manure, where the NH_4^+-N concentration is above 1 000 mg L^{-1}. Therefore, all wood biochar in the selected studies from the literature review has been included, with pyrolysis temperature between 400 and 600°C, excluding studies that are using N concentration at 100 mg N L^{-1} and lower. This result in five studies, and the median value of 28.3 mg NH_4^+-N per gram biochar is chosen from the literature review to prevent the result from being skewed by outliers.

In the study done by Weldon et al. (2022), biochar adsorption studies were put into critical revision, where the NH_4^+ adsorption capacity of the current literature was reanalysed. Their findings suggest that the adsorption capacity of biochar is lower than previously reported. The primary reason for this was found to be inconsistency in the methodology for quantification of sorption capacity. After doing a standardized remodeling of published batch sorption studies, Weldon et al. (2022) reported a maximum sorption capacity of 22.8 mg NH_4^+-N g^{-1} for unmodified biochar. To take a conservative approach, the adsorptive capacity for biochar chosen for this study is decided to not exceed the reported maximum capacity of 22.8 mg NH_4^+-N per g biochar.

Published studies on NH_4^+ adsorption of biochar are laboratory experiments looking isolated on adsorption of NH_4^+, often from a chemical solution. In real-life, adsorption from pig manure storage will mean that other nutrients and gases are present, which most likely will lower the NH_4^+ adsorption capacity. In studies that have compared NH_4^+ adsorption of wood biochar from both chemical solution and piggery anaerobic digestate slurry, the adsorption from piggery slurry was 15-23% lower compared to the chemical solution, depending on N concentration (Kizito et al., 2015). It would be appropriate to scale down the adsorption capacity with 20%, but as two of the five chosen studies from the literature review are doing adsorption experiments on piggery slurry, it is assumed that it is sufficient to further reduce the adsorption capacity with 10% to obtain a more realistic adsorption capacity, from 22.8 to 20.5 mg NH_4^+-N per gram biochar.

2.6.2 Scenario 3a: Biochar Addition to Manure - Emission changes

For this study, an application rate of 4.56 kg biochar per m^2 pig manure storage was chosen, based on a study investigating biochar adsorption (Maurer et al., 2017). It is further assumed that biochar application to pig manure over 30 days leads to a 17% reduction in NH_3 emissions, a 32% increase in CH_4 emissions, and that CO_2 and N_2O emissions are not affected. The values have been estimated based on a literature review for studies including wood biochar addition to pig manure storage (Table 2), where the weighted average of two studies was used. Explanation of the main steps is presented in the following paragraphs.

A literature review was conducted to estimate the adsorption capacity of liquid NH_4^+ to biochar from pig manure. Since Maurer et al. (2017) used softwood biochar pyrolyzed at 500°C and they only found a significant reduction in NH_3 emissions for an application rate of 4.56 kg biochar per m^2, this is the chosen application rate of biochar to manure for this study (app_{manure}). The same cut-off is used for emission changes as for N adsorption, which is soft- and hardwood biochar pyrolyzed at 400-600°C, resulting in two studies. It is not known which temperature Meiirkhanuly et al. (2020) used, and their application rate of biochar is lower than 4.56 kg m^{-2}, so the study of Maurer et al. (2017) is more relevant for the current study in terms of biochar feedstock and pyrolysis temperature. The approach used to evaluate changes in NH_3 and GHGs was to give the two studies different weighting based on their relevance. The reported values from Maurer et al. (2017) are weighted 70% and values from Meiirkhanuly et al. (2020) are weighted 30%. This results in a reduction of NH_3 emissions of 17% and an increase in CH_4 emissions of 32% from manure storage after biochar application. Since the two studies did not agree on a significant effect on CO_2 and N_2O emissions, it is assumed for this study that these emissions are not affected by biochar addition.

Table 2: Literature review of how biochar application affects ammonia and GHG emissions from pig manure.

Biochar feedstock	**Pyrolysis temperature**	**Application rate**	**Emission changes 30 days**[*]	**Source**
Pinewood	500°	4.56 kg/m^2	NH_3: **- 13%** CH_4: **+ 22%** CO_2: No sign. effect N_2O: No sign. effect	Maurer et al. (2017)
Pinewood	500°	2.28 kg/m^2	NH_3: No sign. effect CH_4: No sign. effect CO_2: No sign. effect N_2O: No sign. effect	Maurer et al. (2017)
Red oak wood	Not informed	1.65 kg/m^2	NH_3: **- 25%** CH_4: **+ 54%** CO_2: **- 13.5%** N_2O: No sign. effect	Meiirkhanuly et al. (2020)
Wood shavings	900°	50 g/kg	NH_3: **- 20%** CH_4: No sign. effect CO_2: No sign. effect N_2O: - 12%	Pereira et al. (2022)

[*] Negative numbers indicate a reduction in emissions, and positive numbers indicate an emission increase. Bold numbers indicate a significant effect across all trials.

2.7 Scenario 3b: Nitrogen-enriched Biochar by Using Air Adsorption

2.7.1 Scenario 3b: Biochar Adsorption of Gaseous Ammonia

An adsorption capacity of 25.2 mg NH_3-N per gram of biochar was chosen for this study. This value is based on a literature review, where a value of 31.5 mg NH_3-N per gram of biochar was used for acid-activated biochar, and then reduced by 20% to obtain a more realistic estimation. A detailed explanation of the approach is given in the following paragraphs.

To estimate the adsorptive capacity of NH_3 of spruce wood biochar, a literature review was conducted. Few studies in the current literature have investigated biochar adsorption of gaseous NH_3, and out of them, several have investigated NH_3 adsorption by keeping biochar and NH_3 gas in a sealed jar over a period of time. These are not comparable to this study, where biochar is imagined attached over a source of NH_3 emissions (Figure A1). One study was found in the literature review, where they passed NH_3 gas through a column with biochar and measured how much NH_3 the biochar adsorbed (Ro et al., 2015). In their study, they used NH_3 standard gas with a concentration of 103 ppm, and as NH_3 gas from pig manure storage is reported to have a significantly higher concentration of NH_3 Meiirkhanuly et al. (2020), it is considered comparable to this study.

Non-activated biochar is found to have a limited capacity of adsorbing NH_3 (Rasse et al., 2022). Ro et al. (2015) investigated both activated and non-activated biochar, where they found an adsorption capacity of 0.84 mg NH_3-N per gram of non-activated biochar, and 31.5 mg NH_3-N per gram of activated biochar. Acid-activated biochar was therefore chosen for this study and the adsorption capacity of 31.5 mg NH_3-N per gram of biochar was used from literature (Ro et al., 2015).

The adsorption study of Ro et al. (2015) was conducted as a small-scale laboratory study where they investigated the adsorption of NH_3 isolated. In real life, NH_3 adsorption to biochar from pig manure storage is assumed to be different, as there are not only NH_3 emissions from pig manure storage but also GHGs, pollutants like H_2S and SO_2, volatile organic compounds (VOCs) Gwenzi et al. (2021), and moisture. In a realistic context, biochar will not only adsorb NH_3, but also other gases and compounds, so it is plausible that the adsorption capacity of NH_3 will be lower compared to the study of Ro et al. (2015). As mentioned earlier, for liquid adsorption the reported values were 15-23% lower for adsorption of piggery slurry compared to adsorption from a chemical N solution. Therefore, the adsorption capacity is reduced by 20%, from 31.5 to 25.2 mg NH_3-N g^{-1} biochar.

For the acid activation, Ro et al. (2015) used phosphoric acid (H_3PO_4), which is commonly used as an activating agent (Zhao et al., 2017), and is proven to be efficient for increasing the adsorptive capacity of biochar (Takaya et al., 2019). The mechanism behind this is that phosphoric acid activation enhances the surface area of the biochar and results in a high abundance of porous structure (Chu et al., 2018). Acid-activation with phosphoric acid is therefore also chosen for this study.

In the study of Ro et al. (2015), they soaked the biochar in 30% (*w/w*) phosphoric acid (H_3PO_4) overnight with a ratio of 1:1 (m/v) biochar and acid, and then activated the biochar under breathing air at 450°C for 60 minutes. The same method was used for this study, and the information above was used to calculate the amount of phosphoric acid and water needed to activate 1 kg of biochar (Table A2 in the appendix). Potential fugitive emissions from handling phosphoric acid are neglected.

It is assumed that the acid-soaked biochar is dried and activated in a heater run by wood pellets. Estimation of energy requirements for the drying process was done by using an equation for calculating heat transfer (Equation 4), similarly applied by de Souza et al. (2023). Q is the energy need in MJ, M_{mix} is the material that is going to be heated in kg, C_p is the heat capacity of the material in MJ kg^{-1} K^{-1}, and T_r and T_0 are the goal temperature and room temperature (25°C) given in kelvin, respectively.

$$Q = M_{mix} \cdot C_p \cdot (T_r - T_0) \tag{4}$$

Air adsorption of NH_3 is imagined to be conducted by attaching biochar in the air above pig manure, using it as an adsorbent when the NH_3 gas passes by (Figure A1). Attachment of the biochar above the pig manure storage can be done in different ways, for example by placing biochar on top of a net or making biochar into a filter and having several layers. To be comparable to Scenario 3a, it is imagined that the biochar is attached above the manure in amounts of 4.56 kg m^{-2}, and kept there for 30 days at a time. Material and energy inputs connected to attaching the biochar above the pig manure are neglected, as it is assumed that the related emissions will be minimal in the system. The LCA inventory for pig manure emissions for Scenario 3a can be found in the appendix (Table A5).

2.7.2 Scenario 3b: Biochar Air Adsorption of Ammonia - Emission changes

For this study, it is assumed that acid-activated biochar can reduce NH_3 emissions by 50%, CO_2 by 10%, while CH_4 and N_2O emissions are not affected. The estimation of emission changes of CO_2, CH_4 and N_2O was based on a literature review, while the estimation of changes in NH_3 emissions is based on calculations for NH_3 removal efficiency across a 30 day time period. This is further explained in the following paragraphs.

Air filtering of NH_3 by using biochar is a novel topic, so the literature review did not result in any measurements for how much NH_3 emissions from pig manure storage are affected by using wood biochar as a filter. Calculations based on NH_3 emissions from manure storage and biochar adsorption capacity have therefore been used to estimate the emission changes. Equation 5 was used to calculate the total amount of NH_3 biochar can adsorb in kg across one year ($ads_{NH3-tot}$), where ads_{aab} is the adsorption rate of NH_3 by biochar in mg g^{-1} $year^{-1}$, and app_{field} is the amount of biochar applied per hectare per year in kg (see Section 2.4.2). Equation 6 was used to find the percentage reduction in NH_3 emissions per year (red_{NH3}), where E_{NH3} is the total emissions of NH_3 from manure storage in kg per year (see Section 2.3.1).

$$ads_{NH3-tot} = \frac{ads_{aab} \cdot app_{field}}{10^3} \tag{5}$$

$$red_{NH3} = \frac{ads_{NH3-tot}}{E_{NH3}} \cdot 100\% \tag{6}$$

To be able to compare the two N-enriched biochar methods (Scenario 3a and 3b), the same application rate of 4.56 kg biochar per m^2 of manure storage is used, and the biochar is used as an air filter above the manure for 30 days. Mark that since the treatment period is 30 days, it is possible to use a large area of manure storage and enrich all biochar across 30 days, or one can divide it and have several treatment periods across one year with a smaller area of manure storage. The NH_3 emissions from pig manure storage across a 30-day period are 96.77 kg (E_{NH3}), while the NH_3 adsorption capacity across the same time period is 64.31 kg ($ads_{NH3-tot}$). This results in a removal capacity of 66.5% (red_{NH3}). To take a conservative approach, the emission reduction capacity of biochar is estimated to be 50% of NH_3 emissions from pig manure storage. This approach has been presented to biochar experts at the research institute NIBIO who are working on the topic of N-enriched biochar (AgriCascade project). In discussion with them, it was concluded that this approach is sufficient for this study until more research and experiments are done on this topic.

Findings from literature show that CH_4 is not well adsorbed by biochar, where explanations could be that the pore size of biochar is not small enough to separate CH_4 from CO_2, or because of the competition between H_2S and CO_2, as both gases have smaller molecular sizes than CH_4 (Sethupathi et al., 2017). Therefore, it is assumed for this study that biochar does not adsorb, and subsequentially affects CH_4 emissions from pig manure storage when using air adsorption. Biochar has the ability to adsorb CO_2 (Francis et al., 2023), but with a low capacity when other gases are present (Sethupathi et al., 2017). Based on these findings, the adsorption capacity of CO_2 is assumed to be 10% for air adsorption onto biochar.

Literature is clear that biochar has the ability to reduce N_2O emissions from soil, but few studies have proved that biochar can directly adsorb N_2O. However, some studies have suggested that biochar can adsorb N_2O approximately as strongly as CO_2, but the adsorption effect of N_2O was not investigated with other gases present (Cornelissen et al., 2013). Therefore, to take a conservative approach, it is assumed for this study that biochar does not adsorb N_2O. Compared to liquid adsorption of biochar, air adsorption is assumed to not increase NH_3 and GHG emissions from pig manure storage, as it filters the gases after emitted from manure storage and does not interact with the manure directly. The LCA inventory for manure emissions for is in the appendix (Table A7).

2.8 Scenarios 3a and 3b: N-enriched Biochar Application

Investigating the nitrogen dynamics in biochar involves more than just looking at biochar adsorption capacity. It is important to also consider how nitrogen is retained in the biochar and subsequently released for plant uptake in agricultural fields. For this study, a N desorption rate from biochar was chosen to be 70% of adsorbed NH_4^+ for non-activated biochar, and 80% of adsorbed NH_3 for acid-activated biochar. These estimations are based on a literature review and conversations with biochar experts. A detailed explanation is given in the following sections.

Plants can take up nitrogen in the form of NH_4^+ or NO_3^- (Zayed et al., 2023), but NH_3 adsorbed by biochar is also found to be available for plant uptake (Taghizadeh-Toosi et al., 2012). Nitrogen adsorbed by biochar is found to be stored in a stable form (Taghizadeh-Toosi et al., 2012), but the nitrogen release capacity from biochars is dependent on different factors, both biochar properties and soil characteristics (Jellali et al., 2022).

A literature review was conducted to estimate the N desorption capacity of biochars pyrolyzed at 350-600°C (Table 3). Biochar does not only adsorb N physically but also chemically by reacting with the biochar through co-occurring mechanisms (Jellali et al., 2022). This indicates that the adsorbed N is more difficult to release. The reported values on biochar N desorption from the literature are spread, from 3.9% to 62%. To prevent the results from being skewed by outliers, the median values from the studies were used. The median value for NH_4^+ desorption is 33%, while for NH_3 the median desorption rate is 48.5%. By looking at the desorption studies, it seems like in general, increased N concentration in the source that biochar is adsorbing from is correlated with an increased desorption rate. Therefore, it is plausible that for this system where N is adsorbed from pig manure with high amounts of N, the desorption rate can be assumed even higher than the reported values in the literature.

In addition to biochar properties, the desorption of N from biochar can be significantly influenced by soil conditions, including pH, contact time, and the presence of ions (Jellali et al., 2022). Under alkaline conditions, the N desorption process from biochar is enhanced (Chintala et al., 2013; Nguyen et al., 2019), however, it is important to note that very high pH values may promote the conversion of NH_4^+ into NH_3 gas and its evaporation (Rozïc et al., 2000; Saleh et al., 2012). In the context of barley fields in Norway, the soil pH is recommended to be within the range of 6.0-6.3 (NIBIO, 2020) and liming

Table 3: Literature review of nitrogen desorption from biochar

Biochar feedstock	**Pyrolysis temperature**	**N concentration**	**Desorption rate**	**Source**
		NH_4^+		
Pinewood	550°C	40 mg NH_4^+-N L^{-1}	33%	Hina et al. (2015)
Pinewood	350-600°C	100 mg NH_4^+ L^{-1}	53%	Aghoghovwia et al. (2022)
Oak wood Oak wood	450°C 450°C	43 mg NH_4^+ L^{-1} 450 mg NH_4^+ L^{-1}	12% 40%	Takaya et al. (2019)
Oak wood	400°C	1 000 mg NH_4^+-N L^{-1}	3.9%	Takaya et al. (2016)
		NH_3		
Oak wood Oak wood	450°C 450°C	43 mg NH_3 L^{-1} 450 mg NH_3 L^{-1}	35% 62%	Takaya et al. (2019)

is used to obtain this value, so a significant NH_3 volatilization is not expected.

Previous research indicates that extraction with potassium chloride (KCL) effectively removes nearly all adsorbed NH_4^+ from biochar (B. Wang et al., 2015). This is relevant for biochar application in agricultural soils, since biochar will typically be added combined with mineral fertilizers, and KCL represents the primary form of K fertilizers (Zörb et al., 2014).

In addition to pH, desorption of NH_4^+ is found to increase as time increases (Yin et al., 2020). The studies from the literature review are of laboratory scale, with desorption experiments usually lasting less than 24 hours. When biochar is applied to agricultural soil it is kept there permanently, which is plausible to have a positive impact on N desorption.

By talking to experts in the field, they state that generally speaking, the N that is adsorbed by biochar is also released. Therefore, a desorption rate of 70% is assumed appropriate for non-activated biochar. Acid-activated biochars are reported to have significantly higher desorption rates compared to non-activated biochars (Chintala et al., 2013), so for the acid-activated biochars a desorption rate of 80% is chosen.

The total amount of desorbed NH_4^+-N from biochar to soil per year in kg was calculated by using Equation 7 and Equation 8 for non-activated and activated biochar, respectively. Ads$_b$ is adsorbed NH_4^+-N in mg per g per year by non-activated biochar, and des$_b$ is the desorption rate of NH_4^+-N of non-activated biochar in percentage. Ads$_{aab}$ is adsorbed NH_3-N in mg per kg year by acid-activated biochar, and des$_{aab}$ is the desorption rate of NH_4^+-N of acid-activated biochar in percentage. App$_{field}$ is as explained before, the amount of biochar applied yearly per hectare of barley field.

$$des_b tot = \frac{ads_b \cdot app_{field} \cdot des_b}{10^3} \tag{7}$$

$$des_{aab} tot = \frac{ads_{aab} \cdot app_{field} \cdot des_{aab}}{10^3} \tag{8}$$

In the LCA inventory (Table A4), N fertilizer is represented by ammonium nitrate ($NH_4^+NO_3^-$), as the inventory for biochar application to soil is largely based on Tisserant et al. (2022), and they used ammonium nitrate as N fertilizer input in their inventory. According to the ecoinvent v3.9 database, 1 kg of $NH_4^+NO_3^-$ is equivalent to 0.35 kg of N. The amount of adsorbed and desorbed nitrogen from biochar is estimated based on the pure N content (NH_4^+-N, NH_3-N), similar to most of the studies in the literature. Therefore, when calculating the amount of N provided for plant uptake from biochar, the N is divided by 0.35 to obtain the amount in $NH_4^+NO_3^-$. The input of packaging for fertilizer is adjusted based on the total amount of NPK mineral fertilizer in kg needed for each scenario. This was done by calculating the percentage reduction in mineral fertilizer input compared to Scenario 1 and reducing the input of fertilizer packaging by the same percentage.

2.9 Sensitivity analysis

As the literature reviews in this study show a large variability for both biochar N adsorption, N desorption, and changes in manure emissions after adding biochar, a sensitivity analysis was conducted to assess how this variability might impact the performance of the N-enriched biochar. The analysis is based on a scenario approach, where a "low-performance" and a "high-performance" scenario were added alongside the "default" scenario (values from Section 2.6-2.8). The high-performance scenario includes high emission mitigation from manure emissions, and high biochar N adsorption and desorption, while the low-performance scenario represents low emission mitigation from manure storage and low N adsorption and desorption from biochar. The values for the sensitivity analysis were obtained from the literature review performed for this study, representing the ranges in the literature, alongside with estimations (Table A8 in the Appendix). As plants have a maximum uptake capacity of N, a restriction is chosen for the N desorption from biochar in kg (des$_b$tot and des$_{aab}$tot) for the high-performance scenario. It is set to not exceed the N fertilizer requirement per hectare of barley field (127.5 kg N).

3 Results and Discussion

As the N-enriched biochar system is the focus of this study, the results and discussion is mainly focused on how Scenarios 3a and 3b are performing compared to Scenarios 1 and 2. As explained in Section 2.3.1, the impact from pig manure emissions will be displayed as the difference from baseline emissions (Scenario 1). The LCA results with total impact from manure emissions for the impact categories that are affected by emissions from pig manure can be found in the appendix (Figure A2). The full LCA results for the default scenarios are also in the appendix (Table A9 and A10). The graphs below show the impact of the default scenarios, and high-performance (high) and low-performance (low) sensitivity scenarios. The sensitivity scenarios are blurred to increase the visibility of the default scenarios.

The main takeaways from the LCA results are that both the N-enriched biochar scenarios (3a and 3b) provide significant benefits for terrestrial acidification, with 120% and 170% lower impact for Scenario 3a and 3b, respectively, compared to Scenario 1. Scenario 3a also provides benefits for particulate matter formation and marine eutrophication, with 53% and 8% lower impact, respectively, compared to Scenario 1. The main reason for these benefits is due to the mitigation of NH_3 emissions from manure storage and lower impact from fertilizer production due to reduced demand for mineral fertilizer. Scenario 3a provides trade-offs for climate change due to increased methane emissions from manure storage, with an 18% higher impact compared to Scenario 1. Scenario 3b provides trade-offs for freshwater eutrophication, particulate matter formation, and human toxicity with 109%, 87%, and 132% higher impact, respectively, compared to not using biochar (Scenario 1). The main reason for these increased impacts is due to the acid-activation process in Scenario 3b.

3.1 LCA results

3.1.1 Climate change

All the biochar scenarios perform better than Scenario 1 for climate change (Figure 2) due to negative emissions from carbon sequestering of the biochar. However, the N-enriched biochar system performs badly for climate change compared to Scenario 2. Scenarios 3a and 3b have an impact of 1 862 and -1 925 kg CO_2-eq ha^{-1} year^{-1}, respectively, while Scenario 2 is performing best across the four scenarios with an impact of -2 437 kg CO_2-eq ha^{-1} year^{-1}. Scenario 3b performs second best for climate change and is reducing climate change due to carbon sequestration. Even though Scenario 3b is reducing CO_2 emissions from manure storage by 10%, this does not show an impact on climate change as the CO_2 emissions from manure storage are biogenic, meaning that the CO_2 have been taken up from the atmosphere and is emitted back. Therefore, CO_2 emissions from manure storage have no impact on climate change for this system. This is the same for biochar production, where the CO_2 that is emitted from the wood has already been sequestered by the trees from the atmosphere, so it does not impact climate change when emitted back. Scenario 3b has increased emissions from biochar production compared to Scenario 2 and 3a due to the biochar activation process, where the impacts come primarily from extra energy use, and phosphate rock, sulfur, and quicklime use related to phosphoric acid production.

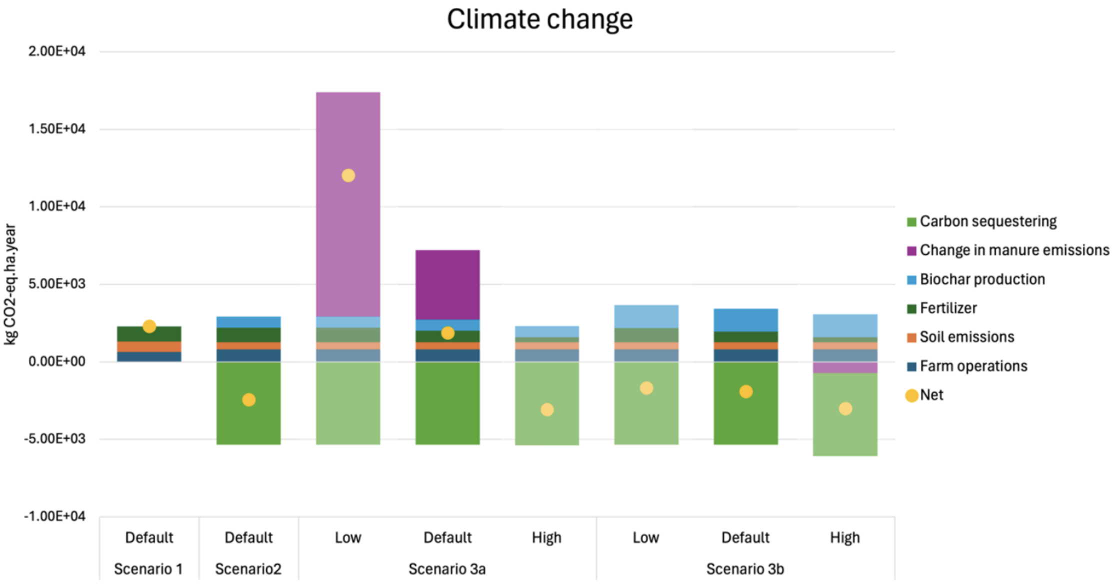


Figure 2: Impact on climate change (GWP100)

Reduced synthetic N fertilizer need results in 23% and 30% lower impact on climate change for fertilizer production for Scenario 3a and 3b, respectively. Scenario 3a has a high impact from change in manure emissions, as the increased CH_4 emissions from biochar addition have a high impact on climate change, and NH_3 emissions do not impact climate change directly. The impact of the increased manure emissions almost evens out the benefit from carbon sequestering, resulting in that Scenario 3a has an almost as high impact as Scenario 1 (1 862 vs. 2 278 kg CO_2-eq ha^{-1} year^{-1}). CH_4 is the second most anthropogenic GHG after CO_2, and it constitutes approximately 16% of global GHG emissions and is 28 times more potent compared to CO_2. It is therefore a large trade-off that Scenario 3a contributes to additional CH_4 emissions.

For the sensitivity analysis for climate change, it is clear that the large range of findings in the literature is reflected in the impact of climate change. For Scenario 3a, the high-performance view results in the highest mitigation effect (-3 076 kg CO_2-eq ha^{-1} year^{-1}) across all scenarios, while the low-performance perspective results in the highest impact (12 040 kg CO_2-eq ha^{-1} year^{-1}) across all scenarios. This is due to the variability in how much CH_4 emissions can increase from adding biochar to the manure. For Scenario 3b, the range is lower, from -1 682 kg CO_2-eq ha^{-1} year^{-1} for low-performance perspective, to -3 015 kg CO_2-eq ha^{-1} year^{-1} for the high-performance perspective. This scenario performs well as the biochar is assumed to release the optimal N needed per hectare so that no mineral N fertilizer is required. In addition, Scenario 3b high-performance is assumed to reduce N_2O emissions from manure storage, leading to a negative impact of 12.3 kg CO_2-eq ha^{-1} year^{-1}.

It is crucial to identify why CH_4 emissions from manure increase after biochar addition, and how these increased emissions can be avoided. Meiirkhanuly et al. (2020) found that freshly added biochar can reduce CH_4 emissions, while after three weeks when the biochar had sunk into the manure, CH_4 emissions were generated instead of being mitigated. Their hypothesis was that biochar adsorbs CH_4, and releases it again once it is incorporated into the manure. Maurer et al. (2017) argues that the reason for increased CH_4 emissions following biochar application is most likely due to the addition of nutrients and labile carbon into the manure, which stimulates CH_4-producing microbes. Scenario 3b deals with this problem, where the biochar is attached above the manure, hindering the biochar to incorporate into the manure. One potential strategy to deal with the increased CH_4 emissions after biochar addition to manure is to change the duration for which biochar is added to the manure. Experiments conducted by Meiirkhanuly et al. (2020) indicate that during the first week after biochar application, CH_4 emissions were mitigated by 54% compared to not using biochar. After two weeks, a reduction of 33% was observed. By reducing the treatment time to 1-2 weeks, one can potentially avoid or even mitigate CH_4 emissions from manure storage compared to standard manure storage. It is necessary to investigate further the mechanisms that happen when biochar is added to manure, and how the increased CH_4 emissions can be avoided.

When including NTCFs (Figure 3), Scenario 2 is still performing best with a net impact of -2 710 kg CO_2-eq ha^{-1} year^{-1}, which is a larger negative impact compared to only including long-lived GHGs. This is due to the cooling effect from NO_x, SO_x, organic carbon, and NH_3 emissions. The negative impact of NO_x comes primarily from soil emissions, and the negative impact of NH_3 is also mainly coming from soil emissions, as well as from fertilizer production. Scenario 1 performs slightly better compared to only including GHGs, due to the cooling effect of NO_x, SO_x, organic carbon, and NH_3 emissions from fertilizer production and soil management, but it is still performing worst out of the four scenarios.

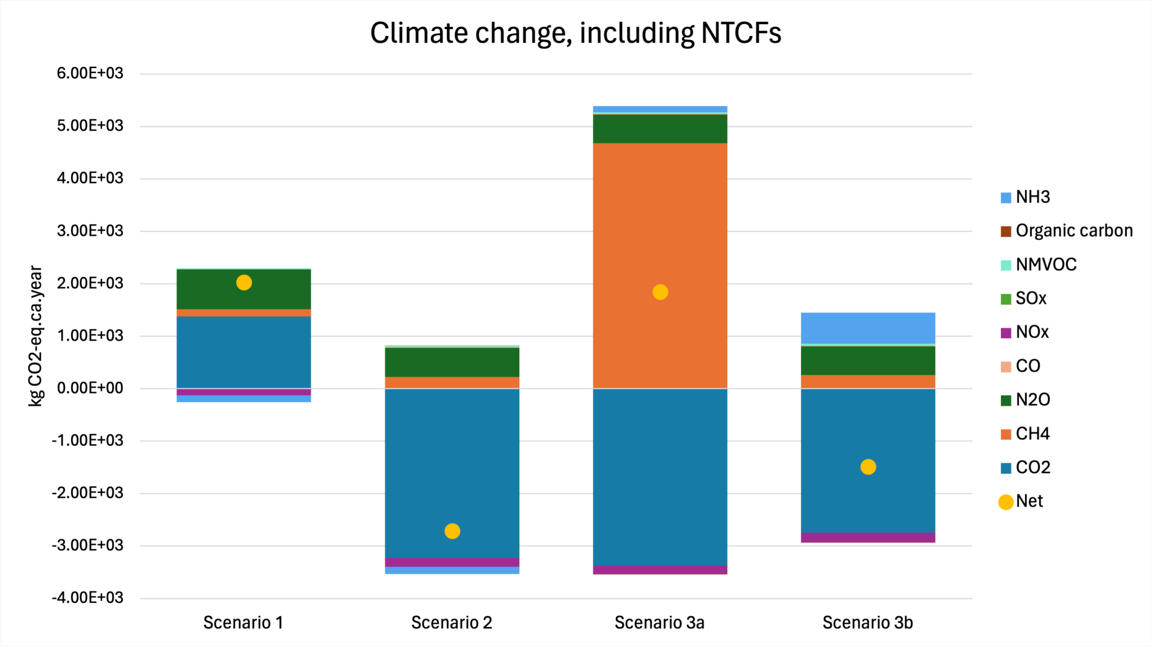


Figure 3: Climate change effects including with GHGs and NTCFs

Scenario 3a has a net positive impact of 1 847 kg CO_2-eq ha^{-1} year^{-1}, which is a slightly lower impact compared to only including GHGs (1 862 kg CO_2-eq ha^{-1} year^{-1}). This is due to a significant cooling effect from N_2O from both soil emissions, biochar production, and fertilizer production and a cooling effect from organic carbon emissions from biochar production. CH_4 contributes to a strong warming effect for Scenario 3a, mainly due to the increased CH_4 emissions from pig manure storage. Interestingly, the reduced NH_3 emissions from pig manure storage are contributing to a warming effect on climate, as NH_3 in general has a cooling effect since it is a precursor to aerosol formation (Szopa et al., 2021). As N-enriched biochar reduces the need for mineral N fertilizer, the reduction of ammonia emissions in fertilizer production due to reduced demand also contributes to an additional warming effect.

Scenario 3b performs second best out of the four scenarios, with a net negative impact of -1 483 kg CO_2-eq ha^{-1} year^{-1}. When including the effect of NTCFs, Scenario 3b is performing worse compared to only GHGs (-1 925 kg CO_2-eq ha^{-1} year^{-1}), by 23%. The main reason for this is that the scenario contributes to a large reduction (50%) in NH_3 emissions from pig manure storage, so the mitigation of NH_3 is resulting in a significant warming effect on climate. Scenario 3b is also contributing to a larger reduction in mineral N fertilizer use since the N-enriched biochar for Scenario 3b has the highest provision of N for plants through biochar. The reduced mineral N fertilizer leads to reduced NH_3 emissions from N fertilizer production, which result in an even higher warming effect.

After including the impact of NTCFs, nothing changes for the ranking of the scenarios in terms of impacts on climate change compared to only GHGs. What is interesting is that the reduction in NH_3 emissions from manure storage and the reduced need for mineral N fertilizer, which were hypothesized as the main benefits of the N-enriched biochar system, are both contributing to global warming short-term due to reduced aerosols. However, these effects are smaller than expected and does not make a big difference on how the scenarios perform for climate change.

3.1.2 Terrestrial acidification

The N-enriched biochar system (Scenario 3a and 3b) performs well when it comes to terrestrial acidification (Figure 4). Scenario 3b performs best out of the four scenarios, with a net impact of -16.9 kg SO_2-eq. ha^{-1} $year^{-1}$ due to a large reduction in impact due to NH_3 emission mitigation from pig manure storage. Even though Scenario 3b has the lowest impact is the impact from biochar production drastically increased compared to Scenario 2 and 3a because of the acid-activation process. The impact from the biochar production for activated biochar is 16 times higher compared to non-activated biochar, due to the production process of phosphoric acid, more specifically the use of sulfuric acid and waste treatment of gypsum. Scenario 2 is performing worst, with an impact of 28.8 kg SO_2-eq. ha^{-1} $year^{-1}$.

Scenario 3a performs second best with a net impact of -4.85 kg SO_2-eq. ha^{-1} $year^{-1}$, also due to reduced impact from NH_3 emissions from manure storage. The net impacts on terrestrial acidification for Scenario 3a and 3b are 1.2 and 1.7 times lower compared to Scenario 1 (24.7 kg SO_2-eq. ha^{-1} $year^{-1}$). If only looking at the impact on terrestrial acidification from manure emissions, Scenario 3a and 3b contribute to a reduction in the impact of 32.9 and 95.1 kg SO_2-eq. ha^{-1} $year^{-1}$, respectively, which represents a 17% and 50% reduction in the total impact from manure emissions.

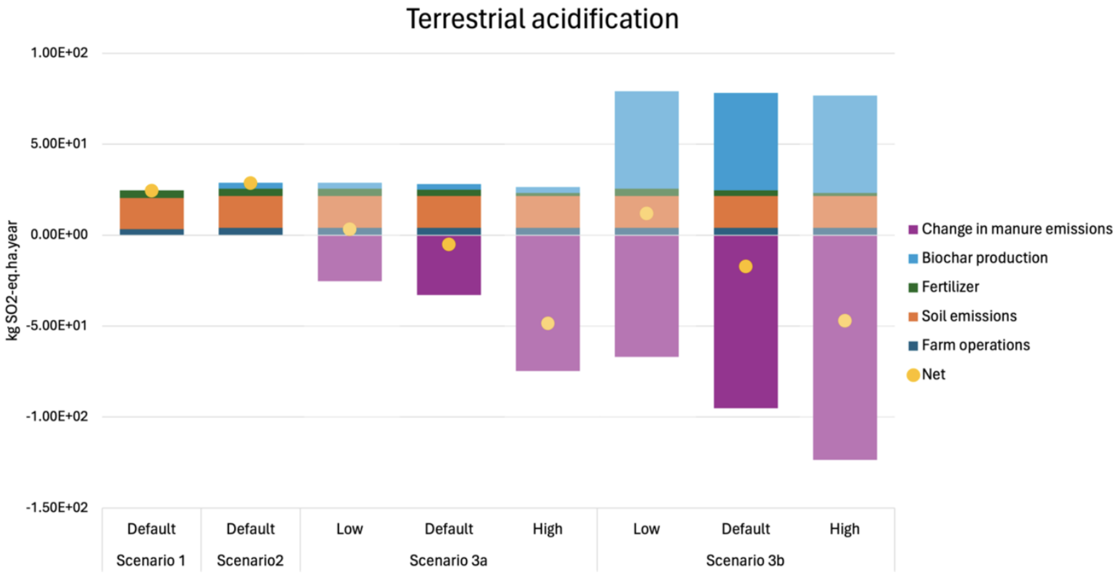


Figure 4: Impact on terrestrial acidification.

When including the low-performance and high-performance scenarios for the N-enriched biochar system, we can see that Scenario 3a goes from having a net negative impact for the default scenario to having a net positive impact of 3.49 kg SO_2-eq. ha^{-1} year^{-1} for the low-performance scenario. However, the low-performance scenario still performs better than Scenario 1 and 2. The high-performance view for Scenario 3a performs well with a net negative impact of 48.2 kg SO_2-eq. ha^{-1} year^{-1}, which is the best-performing scenario across all scenarios. For Scenario 3b, the low-performance view has a net positive impact of 1.21 kg SO_2-eq. ha^{-1} year^{-1}, while the high-performance view performs the second best with a high net negative impact of 46.8 kg SO_2-eq. ha^{-1} year^{-1}. As we can see, the ranges are large between the high-performance and low-performance perspectives for Scenario 3a and 3b, but even for the low-performance perspective the N-enriched biochar system performs better than using normal biochar (Scenario 2) or not using biochar at all (Scenario 1). Therefore, we can conclude that the N-enriched biochar system provides a benefit for terrestrial acidification even when considering low-performance values, and that liquid adsorption (Scenario 3a) has the potential to obtain the largest benefit.

Out of the compounds that lead to terrestrial acidification, NH_3 emissions have the largest effect, with 1.96 kg SO_2-eq per kg NH_3. At the EU-28 level, animal manure management contributes to 65% of anthropogenic NH_3 emissions in Europe (Hou et al., 2017), so using biochar to reduce these emissions has the potential to significantly reduce the total anthropogenic NH_3 emissions, and subsequentially reduce terrestrial acidification.

3.1.3 Particulate matter formation

For particulate matter formation (Figure 5), Scenario 3a is performing best across the four scenarios, with an impact of 2.43 kg PM2.5-eq ha^{-1} $year^{-1}$. This is due to the reduction of NH_3 emissions from manure storage after biochar addition. The reduction in manure emissions has a negative impact of -4.03 kg PM2.5-eq ha^{-1} $year^{-1}$, which represents a 17% reduction of total manure emissions. Even though Scenario 3b has higher emission mitigation of NH_3 compared to Scenario 3a, the impact on particulate matter formation is high, due to the production of phosphoric acid, where sulfuric acid, sulfur, and waste treatment of gypsum are the main contributors. The impact of biochar production on particulate matter formation is over 12.5 times higher for acid-activated biochar (Scenario 3b) compared to normal biochar production (Scenario 2 and 3a). Scenario 3b has a total impact of 9.64 kg PM2.5-eq ha^{-1} $year^{-1}$, which is 87% higher compared to Scenario 1 (5.14 kg PM2.5-eq ha^{-1} $year^{-1}$). Although Scenario 3b contributes to a large reduction in NH_3 emissions from manure storage, the use of phosphoric acid significantly reduces the potential Scenario 3b could have had on terrestrial acidification, making it the worst-performing scenario for particulate matter formation.

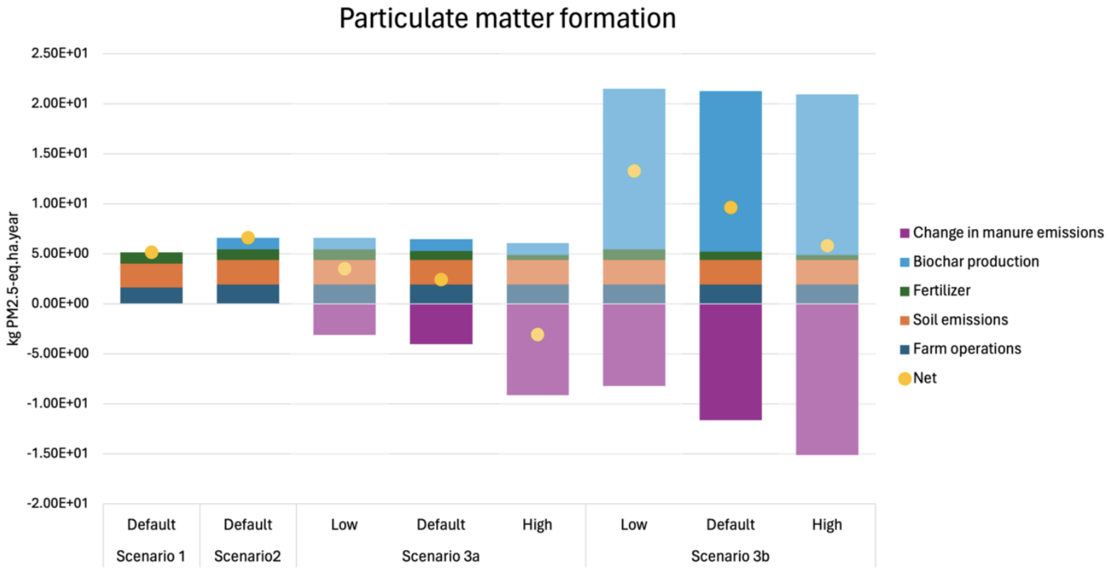


Figure 5: Impact on particulate matter formation.

When including the sensitivities for particulate matter formation, the high-performance view for Scenario 3a is performing best across all scenarios with a net negative impact of -3.07 kg PM2.5-eq ha^{-1} year^{-1}, compared to the default view for Scenario 3a that has a positive impact. This is due to a higher capacity for biochar to mitigate NH_3 emissions from pig manure storage, and results in 1.6 times lower impact compared to Scenario 1. The low-performance view for Scenario 3a has an impact of 3.52 kg PM2.5-eq ha^{-1} year^{-1}, which is still lower than Scenario 1 and 2. For Scenario 3b, the high-performance view has an impact of 5.83 kg PM2.5-eq ha^{-1} year^{-1}, which is slightly better than Scenario 2 (6.62 kg PM2.5-eq ha^{-1} year^{-1}), but still higher impacts than Scenario 1. The low-performance view for Scenario 3b has a high impact of 13.3 kg PM2.5-eq ha^{-1} year^{-1}, which is 1.6 times higher than Scenario 1. This means that only the high-performance view of Scenario 3b can compete with using normal biochar (Scenario 2).

Particulate matter poses a large risk to human health. World Health Organization (WHO) stated that in 2019, 99% of the global population was living in places with high levels of air pollution (World Health Organization (WHO), 2022), and in the same year it was estimated that 4.2 million people died prematurely due to fine particulate matter (World Health Organization (WHO), 2022). Scenario 3a has the potential to significantly reduce the impact on particulate matter formation through its mitigation of NH_3 emissions from manure storage.

3.1.4 Marine eutrophication

For marine eutrophication (Figure 6, all the biochar scenarios (Scenarios 2, 3a, and 3b) perform better than Scenario 1, mainly due to reduced soil emissions when applying biochar to agricultural soil. N-enriched biochar is not assumed to have an additional benefit compared to non-enriched biochar when it comes to soil emissions. Scenario 3a and 3b have a lower impact from N fertilizer production emissions, as the N-enriched biochar reduces the need for mineral N fertilizer per hectare. Scenario 3a and 3b have 23% and 30% lower impact from fertilizer production and use compared to Scenario 1, respectively. Scenario 3a performs best with an impact of 2.28 kg N-eq ha^{-1} year^{-1}, just slightly better than Scenario 2 that has an impact of 2.30 kg N-eq ha^{-1} year^{-1}. Scenario 3b performs worse than Scenario 2, with an impact of 2.32 kg N-eq ha^{-1} year^{-1} due to upstream activities related to phosphoric acid production.

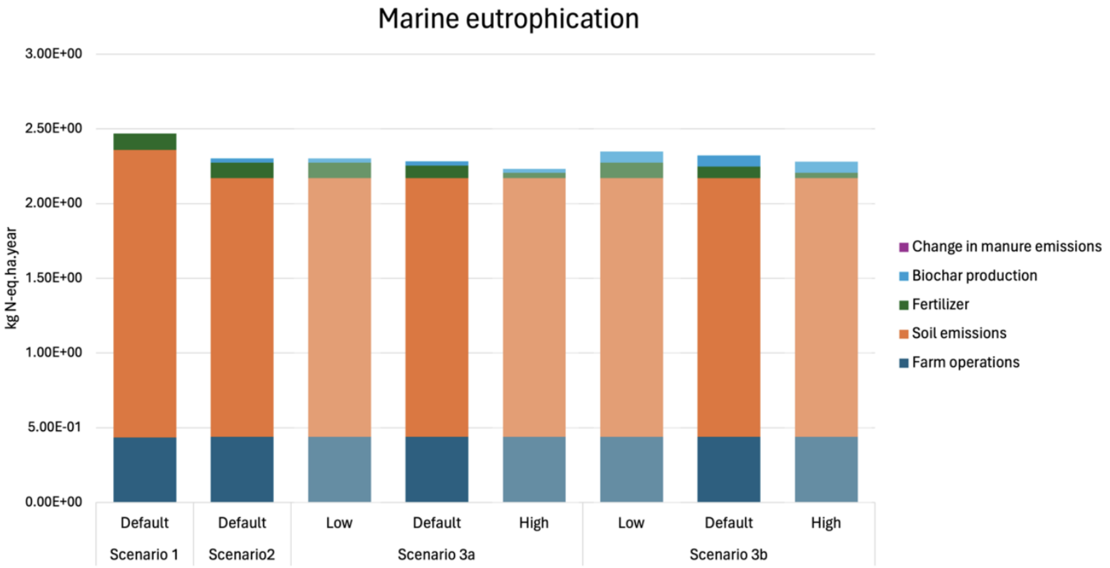


Figure 6: Impact on marine eutrophication.

When including sensitivities for marine eutrophication, the relative change is not very visible in the graph (6), as only the impact related to fertilizer production is affected. However, the high-performance perspective for both Scenario 3a and 3b results in that all the needed N requirements per hectare can be provided by biochar desorption. This leads to a 68% reduction in the impact from fertilizer production on marine eutrophication for both scenarios. The low-performance view of Scenario 3a has an impact equal to Scenario 2, of 2.30 kg N-eq ha^{-1} year^{-1}, while the low-performance view for Scenario 3b is performing worse than Scenario 2, with an impact of 2.35 kg N-eq ha^{-1} year^{-1}. The high-performance perspective for Scenario 3a is performing best across all scenarios for marine eutrophication, with an impact of 2.23 kg N-eq ha^{-1} year^{-1}, which is a 10% reduction in impact compared to Scenario 1. For Scenario 3b, the impact from the high-performance perspective is 2.28 kg N-eq ha^{-1} year^{-1}, which is equal to the impact from default for Scenario 3a. All in all, Scenario 3a is performing best for marine eutrophication, as it has the lowest impact, and even with a low-performance view, Scenario 3a is still performing better than Scenario 1 and 2. Scenario 3b is only performing better than Scenario 1 and 2 when the optimistic view is applied.

As the N-enriched biochar system primarily contributes to the recycling of N, one could think that the reduction of the impact on marine eutrophication would be higher than a maximum potential of 10% reduction in impact. The substances that impact marine eutrophication are ammonium, nitrate, and nitrogen leaching into water or soil, and not ammonia emissions. However, adding biochar to liquid manure can have the potential to reduce the NH_4^+ content of the manure, and sequentially reduce the amount that will go into the environment when the manure is further used. This is not modeled in the system, but it is plausible that if it was, one could see an even higher reduction in marine eutrophication. If biochar addition actually leads to a reduction in the NH_4^+ content of the manure, or if it just adsorbs what would be emitted from the manure anyway, and which benefits or trade-offs this can have for the environment when the manure is used is something that should be further investigated.

Marine pollution has in the last 50 years abruptly accelerated as an environmental concern (Horta et al., 2021). Marine eutrophication is related to the impairment of seawater quality which significantly affects marine ecosystem services, with nutrient runoff from human activities being the primary factor (Horta et al., 2021). Using biochar to recycle nitrogen instead of letting it be emitted into waterbodies can contribute to reducing this issue.

3.1.5 Freshwater eutrophication

For freshwater eutrophication (Figure 7), all the biochar scenarios have a higher impact compared to Scenario 1, since both biochar production and application of biochar to field contributes to freshwater eutrophication. Out of the biochar scenarios, Scenario 3a is performing best, with an impact of 0.392 kg P-eq. ha^{-1} $year^{-1}$, due to less impact of mining operations related to fertilizer production. Scenario 3b has an even lower impact from fertilizer production but is still performing worst out of the four scenarios, with an impact on freshwater eutrophication of 0.602 kg P-eq. ha^{-1} $year^{-1}$, 109% higher compared to Scenario 1. This is due to the acid-activation process, more specifically extraction of phosphate rock and related mining impacts.

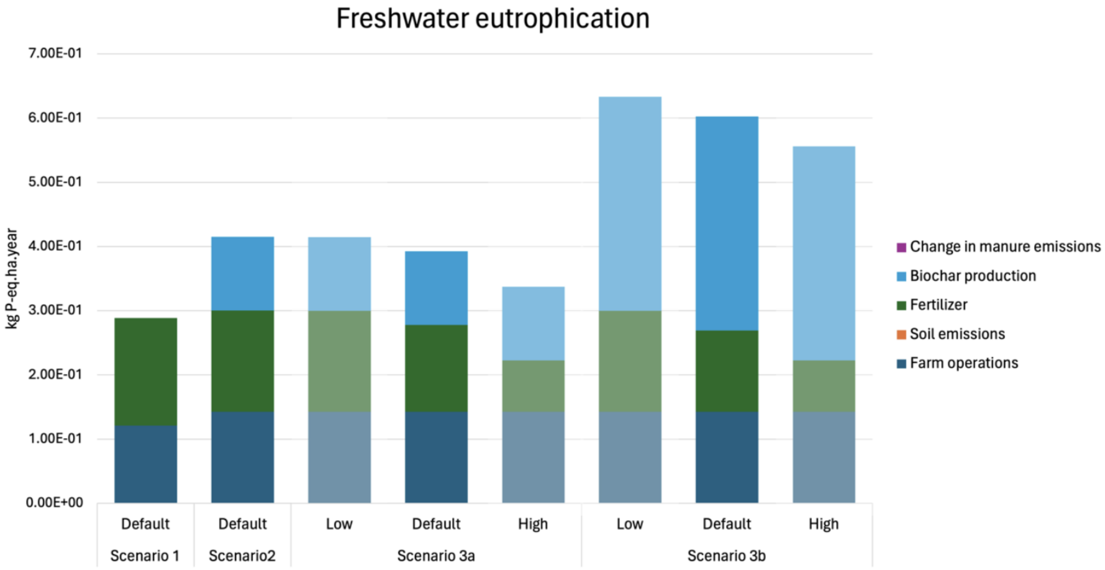


Figure 7: Impact on marine eutrophication.

When including the sensitivities, Scenario 3a is still performing better than Scenario 2. Even with a low-performance perspective, Scenario 3a is performing better than Scenario 2 (0.414 vs. 0.415 kg P-eq. ha^{-1} year^{-1} for Scenario 3a and Scenario 2, respectively). The high-performance perspective for Scenario 3a has an impact of 0.337 kg P-eq. ha^{-1} year^{-1} due to low impact related to N fertilizer production. For Scenario 3b, the low-performance view is performing the worst across all scenarios with an impact of 0.633 kg P-eq. ha^{-1} year^{-1}, due to high impacts from fertilizer production since almost no N is adsorbed and released for plant uptake by biochar. The high-performance view of Scenario 3b is performing well for fertilizer impact, but due to the high impact from the acid-activated biochar production, it still performs worse than the low-performance 3a, and Scenario 1 and 2.

Freshwater eutrophication has become an increasing global problem since the 1960s, and the number of eutrophic lakes increased from 41 to 61% between the late 1970s and late 1990s. In 2012, 63% of the world's inland waterbodies were eutrophic (Y. Zhang et al., 2021). The fact that the N-enriched scenarios, especially Scenario 3b, have such high impacts on freshwater eutrophication is

3.1.6 Human toxicity

For human toxicity (Figure 8), Scenario 1 performs best with an impact of 0.00159 CTU ha^{-1} year^{-1}. This is because both biochar production and application have a significant impact on human toxicity, due to direct emissions of heavy metals to air and soil. Out of the three biochar scenarios, Scenario 3a is performing second best with an impact of 0.00318 CTU ha^{-1} year^{-1}, since the reduced need for synthetic fertilizer reduces the impact on human toxicity from fertilizer production. Scenario 3b is performing worst, with an impact of 0.00390 CTU ha^{-1} year^{-1}, which is 1.5 times higher impact compared to Scenario 1. The reason why Scenario 3b performs badly for human toxicity is similar to freshwater eutrophication, due to the production process of the phosphoric acid, such as mining activities, use of zinc monosulfate, and waste treatment of gypsum.

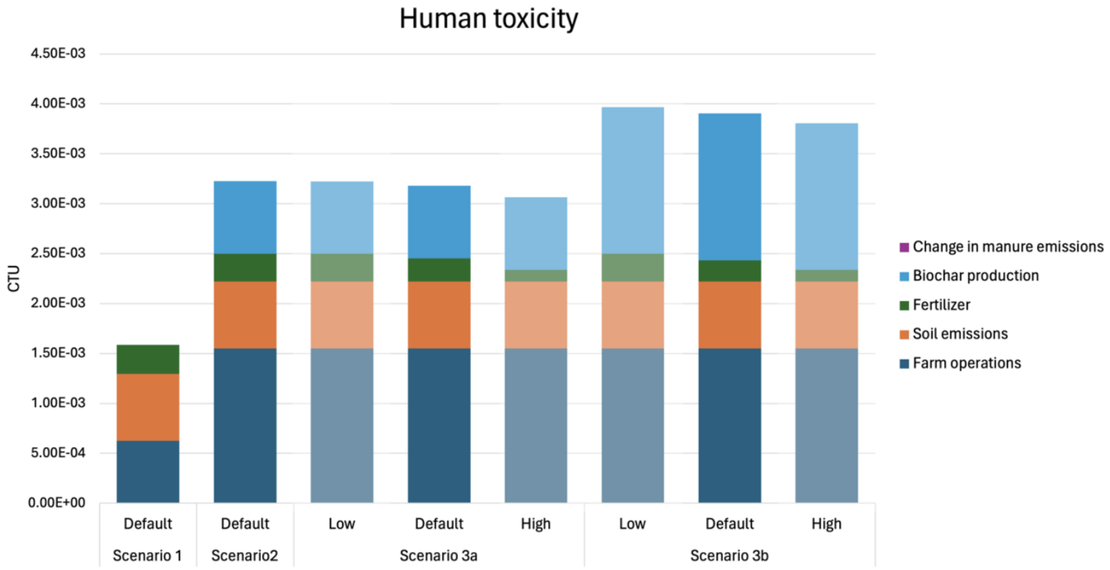


Figure 8: Impact on human toxicity.

When including sensitivities for human toxicity, the ranking of the impact of the four scenarios remains the same as for the default scenarios. Scenario 1 is still performing best, while Scenario 3a is performing second best, even for the low-performance perspective. This is because it is only the impact on fertilizer production and use that changes between the low and high-performance scenarios for Scenario 3a and 3b, and the changes are not large enough to change the ranking of performance. For Scenario 3a, the high-performing perspective has an impact of 0.00306 CTU ha^{-1} year^{-1}, while the low-performance perspective has an impact of 0.00322 CTU ha^{-1} year^{-1}, slightly better than Scenario 2 (0.00323 CTU ha^{-1} year^{-1}). For Scenario 3b, even the high-performance perspective is still performing worse than the low-performance perspective for Scenario 3a, with an impact of 0.00381 CTU ha^{-1} year^{-1}. The low-performance view for Scenario 3b is performing worst across all scenarios with an impact of 0.00397 CTU ha^{-1} year^{-1}, which is a 1.5 higher impact compared to Scenario 1.

The use of phosphoric acid for biochar activation leads to significant impacts on all the chosen impact categories. A strategy to deal with this issue is to use a different activation method. Jellali et al. (2022) lists several methods to activate biochar to enhance the recovery of NH_4^+ and NO_3 from aqueous solutions. Other acids that could be used for activation are hydrogen peroxide (H_2O_2 or sulfuric acid (H_2SO_4). Besides acids, chemical modification of biochar can also be done by using alkaline solutions or salts (Jellali et al., 2022). Physical modification can also be used to enhance N recovery, such as grinding, activation with CO_2, or activation with steam (Jellali et al., 2022). These methods are mostly tested for liquid adsorption of N from aqueous solutions, so further research has to be conducted to see if these methods are also applicable for air filtering of NH_3, and to identify which method has the lowest environmental impact while still maintaining the enhanced adsorption effect.

Comparing the environmental impact of the production of 1 kg of phosphoric acid with 1 kg of sulfuric acid from ecoinvent v3.9, showed that sulfuric acid has a 65-89% lower impact on all six impact categories chosen for this study, with the strongest reduced impact on human toxicity (89%). This indicates that sulfuric acid can be a better choice for acid activation compared to phosphoric acid. However, this presumes that the same amount of acid is needed to activate the biochar and that the sulfuric acid is as efficient to enhance the adsorptive NH_3 capacity of biochar. More research has to be done to identify an acid-activation method that leads to increased adsorption capacity of NH_3, which has low environmental trade-offs.

3.2 Identification of benefits and trade-offs based on the LCA results

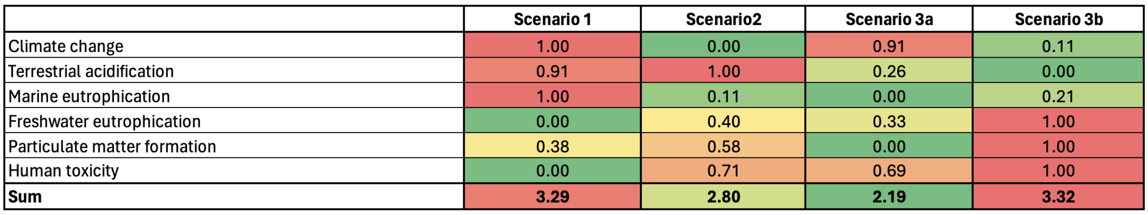
After analyzing the results from the LCA, the first thing to notice is that there is not one single scenario that performs better across all impact categories. Looking at which scenario performs better per impact category (Table 4), Scenario 1 performs best for freshwater eutrophication, Scenario 2 performs best for human toxicity and climate change, Scenario 3a performs best for particulate matter formation and marine eutrophication, while Scenario 3b performs best for terrestrial acidification. Therefore, the choice of scenario has to be based on which environmental concern is in focus. If the main aim of introducing the system is to reduce terrestrial acidification, Scenario 3b should be chosen, while if climate change is the main concern, Scenario 2 should be chosen.

Table 4: Ranking of the scenarios based on which performs better for each impact category.

	Scenario 1	**Scenario 2**	**Scenario 3a**	**Scenario 3b**
Climate change		x		
Terrestrial acidification				x
Marine eutrophication			x	
Freshwater eutrophication	x			
Particulate matter formation			x	
Human toxicity	x			

An attempt has been done to try to identify the scenario that performs better across all six impact categories by allocating a score to each scenario per impact category (Table 9). This was done by normalizing each impact category from 0 to 1 by using Equation 9, where 0 represents the lowest impact (best), and 1 represents the highest impact (worst). Per scenario, the minimum total score is 0, while the maximum total score is 6. By using this method, one is able to express the range in the performance between the different scenarios. The scores are displayed in a heat map to increase the understanding of the scores. This method is highly qualitative, assuming that the impact categories are equally important.

$$x' = \frac{x - min(x)}{max(x) - min(x)} \tag{9}$$



	Scenario 1	**Scenario2**	**Scenario 3a**	**Scenario 3b**
Climate change	1.00	0.00	0.91	0.11
Terrestrial acidification	0.91	1.00	0.26	0.00
Marine eutrophication	1.00	0.11	0.00	0.21
Freshwater eutrophication	0.00	0.40	0.33	1.00
Particulate matter formation	0.38	0.58	0.00	1.00
Human toxicity	0.00	0.71	0.69	1.00
Sum	**3.29**	**2.80**	**2.19**	**3.32**

Figure 9: Performance scores based on normalization and aggregation of the LCA results. Red = high impact, green = low impact.

Looking at the scores, Scenario 3b performs worst in total, with a score of 3.32, while Scenario 3a performs best with a score of 2.19. Although, as previously mentioned this is a qualitative approach, some thoughts can be made up from this ranking. The scope of this thesis was to analyze the environmental performance of the "N-enriched biochar system", where two different approaches of N-enrichment were investigated (Scenario 3a and 3b). After analyzing the environmental impact on the six impact categories, we can see that in total, the N-enriched system is performing both best (Scenario 3a) and worst (Scenario 3b), given equal weighting. This indicates that it is not possible to give the conclusion that "N-enriched biochar" is the better choice, because the two methods differ so significantly in terms of environmental impacts and benefits. Another reflection regarding these scores is that even though Scenario 3b has the highest score, it performs just slightly worse than Scenario 1 (3.32 vs 3.29). This can indicate that overall, by implementing the improvements discussed in this thesis for Scenario 3b (change of biochar activation method), it is plausible that this scenario can perform significantly better compared to not using biochar (Scenario 1).

The results from this study have identified some key benefits and trade-offs for the N-enriched biochar system. The main benefit from the system is the mitigation of NH_3 emissions from pig manure storage, leading to a net negative impact on terrestrial acidification for both Scenario 3a and 3b, and a reduced impact on particulate matter formation for Scenario 3a, compared to Scenario 1 and 2. In addition, another benefit is the reduction of mineral N fertilizer needed per hectare, due to that the N-enriched biochar provides N for plant uptake. The main trade-offs are the increased CH_4 emissions from pig manure storage from adding biochar to the manure (Scenario 3a), which increases the impact on climate change to the extent that it almost evens out the benefit of the carbon sequestration. The other trade-off is the large impact of the acid production used for the activation of the biochar for Scenario 3b. This leads to especially increased impacts on terrestrial acidification, particulate matter formation, freshwater eutrophication, and human toxicity. Although it is a commonly used method for biochar activation, the results of this study indicate that phosphoric acid is not suitable for large-scale use due to large environmental trade-offs.

Even though the N-enriched biochar system provides significant trade-offs in addition to benefits, these findings can be used to further improve the system. The increase in CH_4 emissions from pig manure storage for Scenario 3a is a large problem for climate change, and knowledge from current literature can be used to deal with this problem. In addition, the results indicate that on a large scale, phosphoric acid is not a good choice of activation method for biochar, due to the large environmental impacts it poses. Other activation methods with lower impacts on the environment can be identified, which can result in improved performance of Scenario 3b. This is an early-stage development of a N-enriched biochar system, and it seems realistic that improvements can be made to significantly reduce the trade-offs identified in this study.

Including the sensitivities, the high-performance and low-performance scenarios for Scenario 3a and 3b shows that the performance of the system can vary greatly, depending on the values chosen from the literature. For some of the impact categories, Scenario 3a and 3b have large ranges in performance. For climate change, the impact of Scenario 3a ranges from -3 076 to 12 040 kg CO_2-eq ha^{-1} $year^{-1}$ for the high-performance and low-performance scenario, respectively. For Scenario 3b, the impact on terrestrial acidification ranges from -46.8 to 12.1 kg SO_2-eq. ha^{-1} $year^{-1}$. Given these large uncertainties in impact, it makes it difficult to identify which scenario is performing better. More research is needed to become more certain about the performance of the N-enriched biochar system.

3.3 Potential additional benefits from N-enriched biochar

3.3.1 Adsorption of other plant nutrients

The scope of this study was to look at how much nitrogen biochar can adsorb and provide for plant uptake, but it is also realistic that biochar can adsorb and provide other nutrients for plant growth. Wood biochar has been proven to have the ability to adsorb both phosphate (P) (Dugdug et al., 2018; Takaya et al., 2016) and potassium (K) (Rens et al., 2018). For instance, biochar made from pine waste biochar pyrolyzed at 600°C has shown an adsorption capacity of 20 mg $PO_4{}^{3-}$ (Vijayaraghavan & Balasubramanian, 2021). It is therefore plausible that biochar can adsorb these nutrients in addition to N from pig manure, as pig manure contains both P and K (Kizito et al., 2015; Nunes et al., 2023).

Mineral P fertilizer is related to several environmental concerns, as it is dependent on phosphate rock mining, and these concerns include among other things land use, waste generation from mining activities, and the slow natural fixation of phosphate (Daramola & Hatzell, 2023). In 2019, 227 million tons of phosphate rock was mined globally (U.S. Geological Survey, 2022), so by recycling P from pig manure, one can reduce the demand for P from phosphate rock. Traditional production of mineral K fertilizer is usually done by using potassium chloride (KCl), and the KCl production is related to high energy use and complicated operations (Ji et al., 2022). Recycling P and K from pig manure through biochar adsorption can further reduce the environmental impacts of mineral fertilizer production. However, research is needed to identify how wood biochar with different pyrolysis settings adsorbs P and K from pig manure, and if these forms of P and K are available for plant uptake.

3.3.2 Tailoring of biochar

In literature, there is an agreement that the adsorption capacity of biochar varies with biochar properties such as feedstock, pyrolysis conditions, and pre- or post-treatment of the biochar (Jellali et al., 2022). By doing further research and tests, it is realistic that one can design a biochar that performs optimally for N adsorption and desorption for the given system. For the system in this study, with an application rate of 2 552 kg biochar per hectare, the optimal biochar will be one that can provide 50 mg N per gram biochar, as this fulfills the N requirement per hectare so that no input of mineral N fertilizer is needed. The knowledge on several of these mechanisms is already provided in the literature, especially for liquid adsorption of NH_4^+ from aqueous solution (Jellali et al., 2022). Findings such as increased pyrolysis temperature are negatively correlated with NH_4^+ adsorption (Jellali et al., 2022), that increased residence time can significantly increase the NH_4^+ adsorption capacities (Xue et al., 2019), and that out of pyrolysis temperatures in the range between 200-700°C, 450-500°C shows the highest desorption capacity (Cai et al., 2016; Singh et al., 2020) can be used to develop the most efficient biochar for the system through testing. The same knowledge basis does not exist for air adsorption of NH_3 onto wood biochar, so more research is needed in this field.

3.4 Potential additional trade-offs from N-enriched biochar

3.4.1 Excess provision of nitrogen to field

Findings in the literature about N adsorption onto biochar vary significantly, and some studies have found high adsorption capacities of above 100 mg NH_4^+-N g^{-1} biochar (Kizito et al., 2016; Takaya et al., 2016). As mentioned before, a desorption rate of 50 mg NH_4^+-N per g biochar results in meeting the N requirements per hectare (127.5 kg) for this system. This means that if the biochar releases more than 50 mg NH_4^+-N per g biochar, there is a risk that the plants are not able to take up all the N provided, and excess N will be released into the environment, and lead to water eutrophication. Therefore, it is crucial to do experiments on the chosen biochar to gain more certainty on how much N the biochar is adsorbing and releasing to the field. This can be challenging, as both biochar and manure are biomasses, where variations in composition and properties must be expected.

4 Limitations and Suggestions for Future Research

The main limitation of this study is the data availability for biochar adsorption of nitrogen from pig manure, especially for the air adsorption of NH_3. Very few studies have been conducted on air adsorption of NH_3, and they are all on the experimental level, where the measurements are done with pure NH_3 gas. In this study, it is assumed that gaseous emissions from pig manure are filtered by biochar. As mentioned previously, emissions from pig manure will consist of a mix of different gases, odors, moisture etc., so it is plausible that the adsorption mechanism of NH_3 will be different when NH_3 is not adsorbed alone. In addition, no studies investigating the emission changes from biochar air filtering were found in the literature review, so a very simplified approach for estimating emission changes was used in this study. This is a significant limitation, as the approach used can be far from the reality. Further research is needed to estimate the adsorptive capacity of softwood biochar for gaseous NH_3 emissions from pig manure storage and measurements of which effect it has on manure emissions.

The literature is more extensive on liquid adsorption of NH_4^+ onto biochar, but the findings vary significantly in terms of deciding the adsorptive capacity of biochar. As there were not enough studies done on softwood biochar, other types of wood biochar were also included to estimate the adsorption capacity for this study. This is a limitation, as biochar from different feedstocks varies in their adsorptive capacity, and the results can differ. The methodology in the studies chosen from the literature review can vary significantly, both on properties of the biochar used, and the experimental designs of the adsorption tests. This is an aspect that were neglected in this study, which is a limitation. Research is needed to provide more knowledge on the adsorptive capacity of NH_4^+ from biochars with standardized methods. In addition, measurements should be done for the specific context in the biochar is going to be used, to be able to assess the performance of the system, and the environmental impacts related to the system more precisely.

Another limitation of this study is the estimation of the desorption capacity of NH_4^+ to agricultural soil. Several studies have looked into the desorption capacity of NH_4^+ from biochar, but these studies are on the experimental scale, where chemical solutions have been used to extract the adsorbed N from the biochar, over a short time of period. No studies found in the literature review were looking at desorption of N from biochar in an agricultural context, where water, soil, and other nutrients are present, which can impact the desorption mechanism. Experiments of biochar's desorption capacity of N in agricultural soils are needed. In addition, it was assumed for this study that the adsorbed NH_3 was released as NH_4^+ for plant uptake. There is no certainty in current literature about this mechanism, so more research is needed to gain knowledge on the adsorption and desorption mechanism of NH_3 and biochar.

For the acid-activation process of biochar (Scenario 3b), the biochar was soaked in phosphoric acid, and dried in an oven for 450°C for 60 minutes. The estimation of energy use related to the drying process is a limitation of this study. The approach that was used was to look at the specific heat capacity, in other words, how much energy is needed to heat biochar to 450°C, and after choosing an energy source for the heating. This is not an accurate method for energy estimation, and it is plausible that by using this approach, the energy need is underestimated. More realistic measurements should be done to better assess the energy use of the drying process.

5 Conclusion

In this study, biochar was used as an adsorbent of nitrogen from pig manure, and provider of the adsorbed nitrogen to agricultural soils. Two methods for nitrogen enrichment were investigated, adsorption of liquid ammonium (NH_4^+) where the biochar is added to the manure, and adsorption of gaseous ammonia (NH_3) where biochar is attached above the manure. A life cycle assessment (LCA) was conducted to quantify the environmental impacts of applying nitrogen-enriched (N-enriched) biochar to barley fields, comparing it to using non-enriched biochar, and not using biochar in barley cultivation.

The main benefits of the nitrogen-enriched biochar system are reduced terrestrial acidification, and depending on the enrichment method, reduced particulate matter formation, and marine eutrophication. N-enriched biochar using liquid adsorption and air adsorption have 120% and 170% lower impact on terrestrial acidification, respectively, compared to not using biochar. N-enriched biochar by liquid adsorption has a 53% lower impact on particulate matter formation compared to barley cultivation without biochar. The main trade-offs of N-enriched biochar are increased freshwater eutrophication and human toxicity, and depending on the enrichment method, climate change and particulate matter formation. For human toxicity, N-enriched biochar with liquid adsorption and air adsorption have 100% and 146% higher impact, respectively, compared to not using biochar. For climate change, biochar using liquid adsorption of N has an 18% higher impact compared to not using biochar in barley fields. No scenario is performing better across all chosen impact categories, so the choice has to be made based on which environmental concern is in focus.

Some hotspots were identified in this analysis. Nitrogen-enrichment from pig manure can significantly reduce ammonia emissions from pig manure storage. On the other hand, biochar addition to manure can significantly increase the impact on climate change through increased methane emissions from manure storage, and almost even out the benefit of carbon sequestration. Another hotspot is that the acid-activation of biochar, used to increase the adsorptive capacity of nitrogen, has significant environmental impacts. An additional potential benefit that were not analyzed in this study is that biochar can in addition to nitrogen, also adsorb other macronutrients (phosphorus and potassium), reducing the need for mineral fertilizer even more. A potential trade-off for the system is that the N-enriched biochar can have the ability to adsorb and provide more nitrogen than the plants are able to take up, leading to nutrient runoff and water eutrophication.

The results of this study shows that there are both benefits and trade-offs related to enriching biochar with nitrogen from pig manure storage and applying it to agricultural fields. Several improvement points have been identified in this study, that can be used to obtain better environmental performance for the N-enriched biochar system. Nitrogen enrichment of biochar from livestock manure is a novel topic, and the literature is scarce in this field, especially for the adsorption of gaseous NH_3. This study contributes to showing the large variability in the current literature on biochar adsorption and desorption of nitrogen, and the lack of large-scale studies, so more research is needed to increase the knowledge of this topic.

6 References

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A Appendix

Table A1: Characterization factors for WMGHGs and SLCFs

Compound	**GWP100**	**Source**
Carbon dioxide, fossil	1	IPCC 2021
Carbon dioxide, non-fossil	0	IPCC 2021
Methane, fossil	29.8	IPCC 2021
Methane, non-fossil	27	IPCC 2021
Dinitrogen oxide	273	IPCC 2021
Carbon monoxide	2.1	Levasseur et al. (2016)
Nitrogen oxides	-10.7	Levasseur et al. (2016)
Sulfur oxides	-38	Levasseur et al. (2016)
Non methane volatile organic compounds	5.5	Levasseur et al. (2016)
Black carbon	846	Levasseur et al. (2016)
Organic carbon	-43	Levasseur et al. (2016)
Ammonia	-15*	Aamaas et al. (2016)

*The characterization factor "NH3, Europe, Summer" from their study is chosen, since the geographical area for this study is Norway, and that the N-enriched biochar is meant to be applied when the crops are cultivated, which is between spring and autumn.

Table A2: Calculation phosphoric acid-solution for acid-activation of biochar

Acid-solution in volume	**Acid-solution in weight**
1 000 ml	1.139 kg
203 ml	0.342 kg
797 ml	0.797 kg

Acid-activation of biochar was done by soaking biochar in 30% (*w/w*) phosphoric acid overnight. The biochar was added to the acid solution in a ratio of 1:1 (*m:v*), meaning that per kg of biochar, 1 000 ml of acid solution is needed. As 30% of the solution is acid and the rest is water (in weight), and taking into consideration that phosphoric acid has a density of 1.685 g/ml and water has a density of 1 g/ml, 1.139 kg of acid-solution is needed per kg biochar, where 0.342 kg is phosphoric acid and 0.797 kg is water. This equals 203 ml of phosphoric acid and 797 ml of water per kg biochar. These values were found by using the Goal Seek function in Excel. From ecoinvent, "phosphoric acid, fertiliser grade" was used, as it is assumed that purified, industrial grade phosphoric acid is necessary.

Table A3: Parameters for quantification of the system

Parameter	**Value**	**Unit**	**Description**
app_{field}	2 552	kg	Application rate of biochar per ha field
app_{manure}	4.56	kg m^{-2}	Application rate of biochar to manure
e_{NH3}	240	mg $\text{m}^{-2}\text{h}^{-1}$	Ammonia emissions from pig manure storage
e_{CH4}	1 270	mg $\text{m}^{-2}\text{h}^{-1}$	Methane emissions from pig manure storage
e_{CO2}	8 000	mg $\text{m}^{-2}\text{h}^{-1}$	Carbon dioxide emissions from pig manure storage
e_{N2O}	2.00	mg $\text{m}^{-2}\text{h}^{-1}$	Ammonia emissions from pig manure storage
t	720	h	Treatment period for biochar, 30 days
F_{K}	75.9	kg	Fertilizer K need per ha
b_{K}	0.0051	factor	Amount of K_2O in softwood biochar
M_{mix}	2 552	kg	Application rate of biochar per ha field
C_{p}	0.001	MJ kg^{-1} K^{-1}	Heat capacity wood charcoal, from The Engineering ToolBox (2003)
T_{r}	723.15	K	Biochar activation temperature (450°C)
T_{0}	298.15	K	Room temperature (25°C)
ads_{b}	20.5	mg g^{-1}	Adsorption capacity of NH_4^+-N by biochar
ads_{aab}	25.2	mg g^{-1}	Adsorption capacity of NH_3-N by acid-activated biochar
des_{b}	70	factor	Desorption rate of NH_4^+-N by biochar
des_{aab}	80	factor	Desorption rate of NH_4^+-N by acid-activated biochar

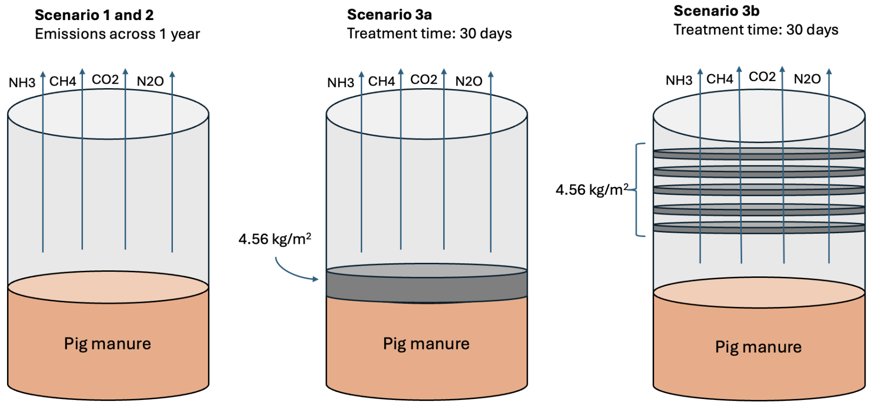

Figure A1: Envisioning of how biochar adsorption from pig manure can be applied.

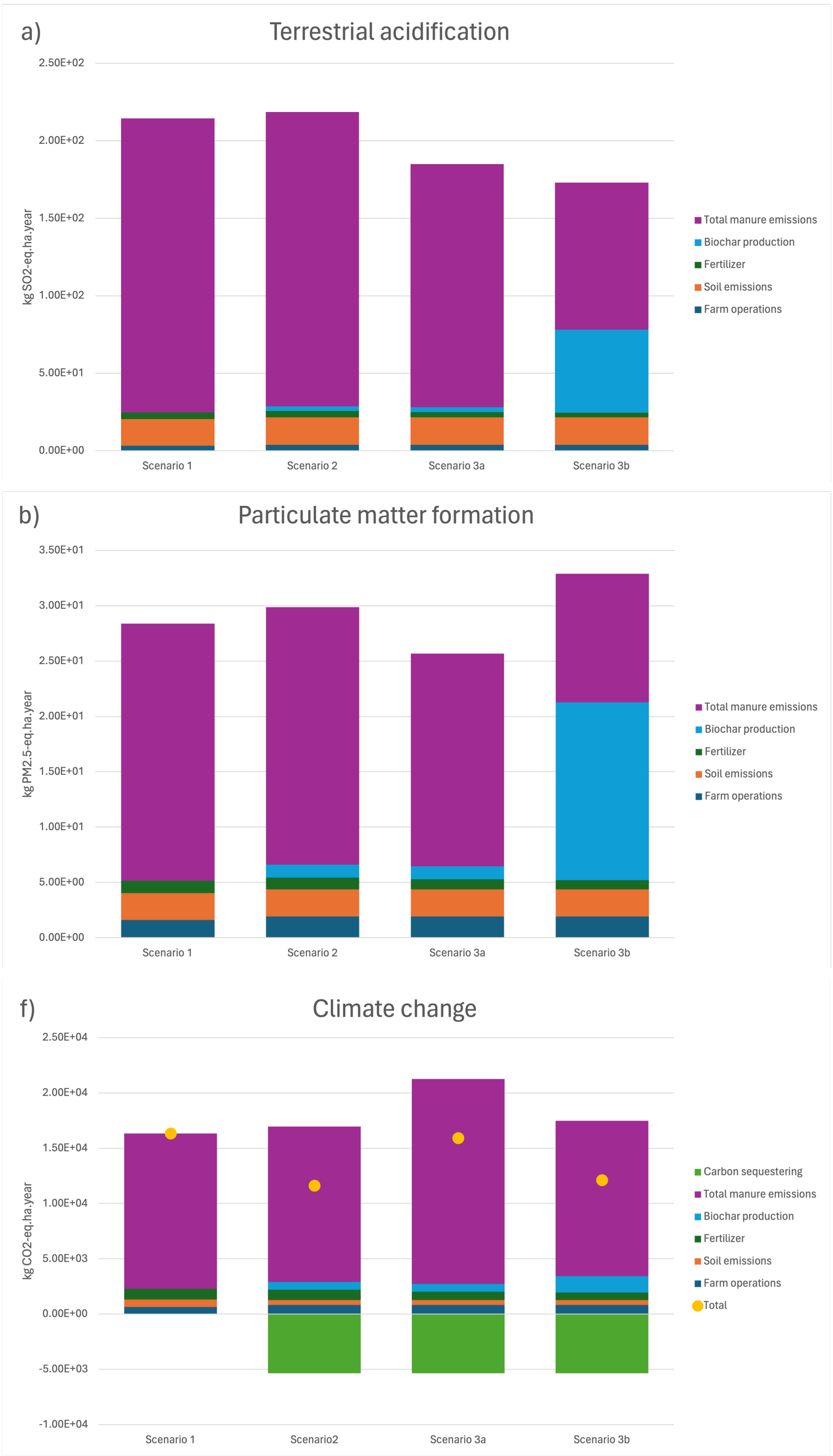

Figure A2: Environmental impacts of the biochar and N-enriched biochar scenarios against the reference system with total emissions from manure storage.

Table A4: Life cycle inventory: Barley cultivation and biochar application

	Scenario 1	**Scenario 2**	**Scenario 3a**	**Scenario 3b**	**Unit**	**Source**
Inputs from techno-sphere						
Tillage, ploughing	1	1	1	1	ha^{-1} yr^{-1}	Tisserant et al. (2022)
Sowing	1	1	1	1	ha^{-1} yr^{-1}	Tisserant et al. (2022)
Tillage, harrowing, by rotary harrow	2	2	2	2	ha^{-1} yr^{-1}	Tisserant et al. (2022)
Fertilizing by broad-caster	1	1	1	1	ha^{-1} yr^{-1}	Tisserant et al. (2022)
Application of plant protection product, by field sprayer	2.333	2.333	2.333	2.333	ha^{-1} yr^{-1}	Tisserant et al. (2022)
Combine harvesting	1	1	1	1	ha^{-1} yr^{-1}	Tisserant et al. (2022)
Rolling	1	1	1	1	ha^{-1} yr^{-1}	Tisserant et al. (2022)
Liming	447	302	302	302	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Cyclic N-compound	0.075	0.075	0.075	0.075	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Pesticide, unspecified	0.01	0.01	0.01	0.01	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Triazine-compound, unspecified	0.011	0.011	0.011	0.011	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Pyrethroid-compound	0.017	0.017	0.017	0.017	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Benzoic-compound	0.088	0.088	0.088	0.088	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)

Organo-phosphorus-compound, unspecified	0.98	0.98	0.98	0.98	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Packaging, for pesticides	1.818	1.818	1.818	1.818	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Ammonium nitrate	364.3	364.3	364.3	364.3	kg ha^{-1} yr^{-1}	Section 2.8
Ammonium nitrate			-104.6	-147.0	kg ha^{-1} yr^{-1}	Section 2.8
Phosphate, as P_2O_5	39.5	39.5	39.5	39.5	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Potassium chloride, as K_2O	75.9	62.9	62.9	62.9	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Packaging for fertilizer	426.3	414.7	321.8	284.1	kg ha^{-1} yr^{-1}	Tisserant et al. (2022), Table 2.8
Biochar application		2 552	2 552	2 552	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Biochar		2 552			kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
N-enriched biochar (liquid ads.)			2 552		kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
N-enriched biochar (air ads.)				2 552	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Barley seed, for sowing	160	160	160	160	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Transport, tractor and trailer, agricultural	30	30	30	30	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Manure storage	560	560	560	560	m^2 yr^{-1}	Section 2.3.1

Emissions to air						
Ammonia (NH_3)	7.75	8.13	8.13	8.13	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Nitrous oxide (N_2O)	2.43	1.65	1.65	1.65	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Nitrogen oxides (NO_x)	5.10	4.59	4.59	4.59	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Emissions to soil						
Fludioxonil	0.01	0.01	0.01	0.01	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Tribenuron-methyl	0.011	0.011	0.011	0.011	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Alpha-cypermethrin	0.017	0.017	0.017	0.017	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Trifloxystrobin	0.088	0.088	0.088	0.088	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Prothioconazol	0.075	0.075	0.075	0.075	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Ethephon	0.05	0.05	0.05	0.05	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Glyphosate	0.93	0.93	0.93	0.93	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Cadmium	4.83E-03	4.83E-03	4.83E-03	4.83E-03	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Chromium	3.79E-03	3.79E-03	3.79E-03	3.79E-03	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Lead	9.85E-04	9.85E-04	9.85E-04	9.85E-04	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Nickel	4.14E-03	4.14E-03	4.14E-03	4.14E-03	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)

Emissions to water						
Cadmium, ion, groundwater	4.39E-05	4.39E-05	4.39E-05	4.39E-05	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Cadmium, ion, river	2.81E-05	2.81E-05	2.81E-05	2.81E-05	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Chromium, ion, groundwater	1.86E-02	1.86E-02	1.86E-02	1.86E-02	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Chromium, ion, river	2.81E-03	2.81E-03	2.81E-03	2.81E-03	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Copper, ion, groundwater	2.60E-03	2.60E-03	2.60E-03	2.60E-03	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Copper, ion, river	1.93E-03	1.93E-03	1.93E-03	1.93E-03	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Lead, groundwater	1.65E-04	1.65E-04	1.65E-04	1.65E-04	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Lead, river	3.82E-05	3.82E-05	3.82E-05	3.82E-05	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Nickel, ion, river	1.61E-03	1.61E-03	1.61E-03	1.61E-03	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Nitrate	28.7	25.8	25.8	25.8	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Zinc, ion, groundwater	2.27E-03	2.27E-03	2.27E-03	2.27E-03	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)
Zinc, ion, river	1.13E-02	1.13E-02	1.13E-02	1.13E-02	kg ha^{-1} yr^{-1}	Tisserant et al. (2022)

Table A5: Life cycle inventory: Biochar production

	Non-activated biochar	**Activated biochar**	**Unit**	**Source**
Outputs to technosphere				
Biochar	1	1	kg	
Inputs from technosphere				
Wood chips, wet, measured as dry mass	3.571	3.571	kg	Tisserant et al. (2022)
Synthetic gas factory	1.23E-09	1.23E-09	unit	Tisserant et al. (2022)
Electricity	0.2818	0.2818	kWh	Tisserant et al. (2022)
Heat, central or small scale, wood pellet, at furnace 9kW		0.425	MJ	Section 2.7.1, Equation 4
Phosphoric acid, fertiliser grade, without water, in 70% solution state		0.3417	kg	Table A2
Water		0.7972	kg	Table A2
Transport, freight, lorry >32 metric ton, EURO6	1.1129	1.1129	tkm	Tisserant et al. (2022)
Emissions to air				
Carbon monoxide, non-fossil	3.57E-06	3.57E-06	kg	Tisserant et al. (2022)
Carbon dioxide, non-fossil	3.45E+00	3.45E+00	kg	Tisserant et al. (2022)
Hydrochloric acid	1.13E-04	1.13E-04	kg	Tisserant et al. (2022)
Water	2.94E-03	2.94E-03	m^3	Tisserant et al. (2022)
Nitrogen oxides	5.59E-04	5.59E-04	kg	Tisserant et al. (2022)
Dinitrogen monoxide	4.21E-08	4.21E-08	kg	Tisserant et al. (2022)
Sulfur dioxide	4.87E-04	4.87E-04	kg	Tisserant et al. (2022)
Chlorine	3.53E-11	3.53E-11	kg	Tisserant et al. (2022)
Hydrogen	2.21E-07	2.21E-07	kg	Tisserant et al. (2022)
PAH, polycyclic aromatic hydrocarbons	3.98E-08	3.98E-08	kg	Tisserant et al. (2022)

Arsenic	3.50E-06	3.50E-06	kg	Tisserant et al. (2022)
Cadmium	5.40E-07	5.40E-07	kg	Tisserant et al. (2022)
Chromium	9.60E-06	9.60E-06	kg	Tisserant et al. (2022)
Copper	2.00E-06	20E-06	kg	Tisserant et al. (2022)
Lead	1.10E-06	1.10E-06	kg	Tisserant et al. (2022)
Mercury	1.40E-07	1.40E-07	kg	Tisserant et al. (2022)
Molybdenum	5.70E-07	5.70E-07	kg	Tisserant et al. (2022)
Nickel	1.80E-06	1.80E-06	kg	Tisserant et al. (2022)
Tin	3.00E-08	3.00E-08	kg	Tisserant et al. (2022)
NMVOC, non-methane volatile organic compounds	3.40E-04	3.40E-04	kg	Tisserant et al. (2022)
Particulates > 2.5 μm, and < 10 μm	1.79E-03	1.79E-03	kg	Tisserant et al. (2022)
VOC, volatile organic compounds	5.6E-05	5.6E-05	kg	Tisserant et al. (2022)

Table A6: Life cycle inventory: Biochar application

	Non-activated biochar	**Activated biochar**	**Unit**	**Source**
Outputs to technosphere				
Biochar application (spreading and harrowing)	1	1	kg	
Inputs from technosphere				
Fertilizing, by broadcaster	3.90E-04	3.90E-04	ha	Tisserant et al. (2022)
Tillage, harrowing, by rotary harrow	3.90E-04	3.90E-04	ha	Tisserant et al. (2022)
Transport, freight, lorry >32 metric ton, EURO6	2.90E-01	2.90E-01	tkm	Tisserant et al. (2022)
Emissions to soil				
Cadmium	1.62E-05	1.62E-05	kg	Tisserant et al. (2022)
Lead	7.02E-05	7.02E-05	kg	Tisserant et al. (2022)
Zinc	4.32E-04	4.32E-04	kg	Tisserant et al. (2022)

Table A7: Life cycle inventory: Manure storage

	Scenario 1 and 2	**Scenario 3a**	**Scenario 3b**	**Unit**
Outputs to technosphere				
Manure storage	1	1	1	m^2
Inputs from technosphere				
Emissions to air				
Ammonia	1.73E-01	1.43E-01	8.64E-01	kg yr^{-1}
Methane	9.14E-01	1.21E+00	9.14E-01	kg yr^{-1}
Dinitrogen monoxide	1.44E-03	1.44E-03	1.44E-03	kg yr^{-1}
Carbon dioxide	5.76E+00	5.76E+00	5.76E+00	kg yr^{-1}

Table A8: Parameters Sensitivity analysis

Parameter	**Default**	**Pessimistic**	**Optimistic**	**Unit**	**Reference**
Emission changes from manure storage - Scenario 3a					
NH_3	-17	-13	-39	%	Default: Section 2.6.2, pessimistic: Maurer et al. (2017), optimistic: Meiirkhanuly et al. (2020)
CH_4	32	104	0	%	Default: Section 2.6.2, pessimistic: Meiirkhanuly et al. (2020), optimistic: Maurer et al. (2017)
N_2O	0	10	-12	%	Default: Section 2.6.2, pessimistic: +10%, optimistic: Pereira et al. (2022)
CO_2	0	10	-25	%	Default: Section 2.6.2, pessimistic: +10%, optimistic: Meiirkhanuly et al. (2020)
Emission changes from manure storage - Scenario 3b					
NH_3	-50	-35	-65	%	Default: Section 2.7.2, pessimistic: +30%, optimistic: -30%
CH_4	0	10	-10	%	Default: Section 2.7.2, pessimistic: +10%, optimistic: -10%
N_2O	0	10	-10	%	Default: Section 2.7.2, pessimistic: +10%, optimistic: -10%
CO_2	0	10	-10	%	Default: Section 2.7.2, pessimistic: +10%, optimistic: -10%

N adsorption of biochar					
Non-activated biochar	20.5	2.5	57.1	mg NH_4^+-N g^{-1}	Default: Section 2.6.1, pessimistic: Weldon et al. (2022), optimistic: Takaya et al. (2016)*
Activated biochar	25.2	3.1	50.0	mg NH_3-N g^{-1}	Default: Section 2.7.1, pessimistic and optimistic same difference from Default as for non-activated biochar*
N desorption from biochar					
Non-activated biochar	70	52.5	87.5	%	Default: Section 2.8, pessimistic: -25%, optimistic: +25%
Activated biochar	80	60	100	%	Default: Section 2.8, pessimistic: -25%, optimistic: +25%

Table A9: LCA results

Climate change	**Scenario 1** **Default**	**Scenario 2** **Default**	**Low**	**Scenario 3a** **Default**	**High**	**Low**	**Scenario 3b** **Default**	**High**
Farm operations	6.42E+02	8.14E+02	8.14E+02	8.14E+02	8.14E+02	8.14E+02	8.14E+02	8.14E+02
Soil emissions	6.63E+02	4.50E+02	4.50E+02	4.50E+02	4.50E+02	4.50E+02	4.50E+02	4.50E+02
Fertilizer	9.73E+02	9.29E+02	9.26E+02	7.53E+02	3.17E+02	9.25E+02	6.82E+02	3.17E+02
Biochar production	0.00E+00	7.18E+02	7.18E+02	7.18E+02	7.18E+02	1.48E+03	1.48E+03	1.48E+03
Manure emissions	0.00E+00	0.00E+00	1.45E+04	4.48E+03	-2.60E+01	0.00E+00	0.00E+00	-7.25E+02
Carbon sequestering	0.00E+00	-5.35E+03	-5.35E+03	-5.35E+03	-5.35E+03	-5.35E+03	-5.35E+03	-5.35E+03
Total	**2.278E+03**	**-2.437E+03**	**1.204E+04**	**1.862E+03**	**-3.076E+03**	**-1.682E+03**	**-1.925E+03**	**-3.015E+03**
Terrestrial acidification	**Scenario 1** **Default**	**Scenario 2** **Default**	**Low**	**Scenario 3a** **Default**	**High**	**Low**	**Scenario 3b** **Default**	**High**
Farm operations	3.36E+00	3.91E+00	3.91E+00	3.91E+00	3.91E+00	3.91E+00	3.91E+00	3.91E+00
Soil emissions	1.70E+01	1.76E+01	1.76E+01	1.76E+01	1.76E+01	1.76E+01	1.76E+01	1.76E+01
Fertilizer	4.28E+00	4.10E+00	4.09E+00	3.43E+00	1.75E+00	4.09E+00	3.15E+00	1.75E+00
Biochar production	0.00E+00	3.15E+00	3.15E+00	3.15E+00	3.15E+00	5.35E+01	5.35E+01	5.35E+01
Manure emissions	0.00E+00	0.00E+00	-2.52E+01	-3.29E+01	-7.46E+01	-6.70E+01	-9.51E+01	-1.23E+02
Total	**2.47E+01**	**2.88E+01**	**3.49E+00**	**-4.85E+00**	**-4.82E+01**	**1.21E+01**	**-1.69E+01**	**-4.68E+01**
Marine eutrophication	**Scenario 1** **Default**	**Scenario 2** **Default**	**Low**	**Scenario 3a** **Default**	**High**	**Low**	**Scenario 3b** **Default**	**High**
Farm operations	4.34E-01	4.39E-01	4.39E-01	4.39E-01	4.39E-01	4.39E-01	4.39E-01	4.39E-01
Soil emissions	1.93E+00	1.73E+00	1.73E+00	1.73E+00	1.73E+00	1.73E+00	1.73E+00	1.73E+00
Fertilizer	1.10E-01	1.05E-01	1.04E-01	8.47E-02	3.51E-02	1.04E-01	7.66E-02	3.51E-02
Biochar production	0.00E+00	2.78E-02	2.78E-02	2.78E-02	2.78E-02	7.46E-02	7.46E-02	7.46E-02
Manure emissions	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Total	**2.47E+00**	**2.30E+00**	**2.30E+00**	**2.28E+00**	**2.23E+00**	**2.35E+00**	**2.32E+00**	**2.28E+00**

Freshwater eutrophication	Scenario 1	Scenario 2		Scenario 3a			Scenario 3b	
	Default	Default	Low	Default	High	Low	Default	High
Farm operations	1.21E-01	1.42E-01	1.42E-01	1.42E-01	1.42E-01	1.42E-01	1.42E-01	1.42E-01
Soil emissions	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Fertilizer	1.67E-01	1.58E-01	1.58E-01	1.36E-01	8.01E-02	1.57E-01	1.27E-01	8.01E-02
Biochar production	0.00E+00	1.14E-01	1.14E-01	1.14E-01	1.14E-01	3.33E-01	3.33E-01	3.33E-01
Manure emissions	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Total	**2.88E-01**	**4.15E-01**	**4.14E-01**	**3.92E-01**	**3.37E-01**	**6.33E-01**	**6.02E-01**	**5.56E-01**
Particulate matter formation	**Scenario 1**	**Scenario 2**		**Scenario 3a**			**Scenario 3b**	
	Default	**Default**	**Low**	**Default**	**High**	**Low**	**Default**	**High**
Farm operations	1.62E+00	1.92E+00	1.92E+00	1.92E+00	1.92E+00	1.92E+00	1.92E+00	1.92E+00
Soil emissions	2.42E+00	2.46E+00	2.46E+00	2.46E+00	2.46E+00	2.46E+00	2.46E+00	2.46E+00
Fertilizer	1.11E+00	1.06E+00	1.05E+00	8.97E-01	5.04E-01	1.05E+00	8.33E-01	5.04E-01
Biochar production	0.00E+00	1.18E+00	1.18E+00	1.18E+00	1.18E+00	1.61E+01	1.61E+01	1.61E+01
Manure emissions	0.00E+00	0.00E+00	-3.09E+00	-4.03E+00	-9.14E+00	-8.20E+00	-1.16E+01	-1.51E+01
Total	**5.14E+00**	**6.62E+00**	**3.52E+00**	**2.43E+00**	**-3.07E+00**	**1.33E+01**	**9.64E+00**	**5.83E+00**
	Scenario 1	**Scenario 2**		**Scenario 3a**			**Scenario 3b**	
Human toxicity	**Default**	**Default**	**Low**	**Default**	**High**	**Low**	**Default**	**High**
Farm operations	6.24E-04	1.55E-03	1.55E-03	1.55E-03	1.55E-03	1.55E-03	1.55E-03	1.55E-03
Soil emissions	6.71E-04	6.71E-04	6.71E-04	6.71E-04	6.71E-04	6.71E-04	6.71E-04	6.71E-04
Fertilizer	2.93E-04	2.77E-04	2.77E-04	2.31E-04	1.16E-04	2.76E-04	2.12E-04	1.16E-04
Biochar production	0.00E+00	7.27E-04	7.27E-04	7.27E-04	7.27E-04	1.47E-03	1.47E-03	1.47E-03
Manure emissions	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Total	**1.59E-03**	**3.23E-03**	**3.22E-03**	**3.18E-03**	**3.06E-03**	**3.97E-03**	**3.90E-03**	**3.81E-03**

Manure emissions are change in impact from manure emissions compared to Scenario 1.

Table A10: LCA results on climate change with NTCFs

Climate change with NTCFs	**Scenario 1**	**Scenario 2**	**Scenario 3a**	**Scenario 3b**
CO2	1.37E+03	-3.23E+03	-3.37E+03	-2.75E+03
CH4	1.38E+02	2.21E+02	4.68E+03	2.61E+02
N2O	7.63E+02	5.58E+02	5.44E+02	5.43E+02
CO	9.77E+00	1.77E+01	1.75E+01	1.94E+01
NOx	-1.26E+02	-1.66E+02	-1.63E+02	-1.80E+02
SOx	-7.79E-02	-1.17E-01	-1.09E-01	-1.34E-01
NMVOC	1.29E+01	2.91E+01	2.81E+01	3.26E+01
Organic carbon	-8.15E-02	-2.65E+00	-2.64E+00	-2.69E+00
NH3	-1.35E+02	-1.40E+02	1.15E+02	5.91E+02
Total	**2.035E+03**	**-2.710E+03**	**1.847E+03**	**-1.483E+03**

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