KNOWLEDGE SYNTHESIS ON BIOCHAR IN DANISH AGRICULTURE

LARS ELSGAARD (ED.), ANDERS PETER S. ADAMSEN, HENRIK B. MØLLER, ANNE WINDING, UFFE JØRGENSEN, ESBEN Ø. MORTENSEN, EMMANUEL ARTHUR, DIEGO ABALOS, MATHIAS N. ANDERSEN, HENRIK THERS, PETER SØRENSEN, ADDISU ANTENEH DILNESSA & KATARINA ELOFSSON.

DCA REPORT NO. 208 · SEPTEMBER 2022 · ADVISORY REPORT

AARHUS UNIVERSITY

DCA - DANISH CENTRE FOR FOOD AND AGRICULTURE

Knowledge synthesis on biochar in Danish agriculture

- Biochar production, use and effect in soil agroecosystems (part 1)

- Economic assessment of biochar production and use (part 2)

Advisory report from DCA – Danish Centre for Food and Agriculture

AUTHORS:

Lars Elsgaard[1] (ed.), Anders Peter S. Adamsen[2], Henrik B. Møller[2], Anne Winding[3], Uffe Jørgensen[1], Esben Ø. Mortensen[1], Emmanuel Arthur[1], Diego Abalos[1], Mathias N. Andersen[1], Henrik Thers[1], Peter Sørensen[1], Addisu Anteneh Dilnessa[3], Katarina Elofsson[3]

1) Department of Agroecology, Aarhus University

2) Department of Biological and Chemical Engineering, Aarhus University

3) Department of Environmental Science, Aarhus University

Data sheet

Title:	Knowledge synthesis on biochar in Danish agriculture - Biochar production, use and effect in soil agroecosystems (part 1) and Economic assessment of biochar production and use (part 2)
Series and number:	DCA report No. 208
Report type:	Advisory report
Year of issue:	September 2022, 1st PDF edition, 1st printing
Authors, part 1:	Senior researchers Anders Peter Adamsen and Henrik B. Møller, Dept. of Biological and Chemical Engineering, AU, professor Anne Winding, Dept. of Environmental Science, AU, professor Uffe Jørgensen, PhD student Esben Ø. Mortensen, tenure track researchers Emmanuel Arthur and Diego Abalos, professor Mathias N. Andersen, postdoc Henrik Thers, senior researcher Peter Sørensen and associate professor Lars Elsgaard, Dept. of Agroecology, AU. Author contribution is specified in each chapter
Authors, part 2:	Professor Katarina Elofsson and PhD student Addisu Anteneh Dilnessa, Dept. of Environmental Science, AU.
Review, part 1:	Professors Jørgen E. Olsen and Mathias N. Andersen, associate professor Lars Elsgaard, tenure track researcher Emmanuel Arthur, Dept. of Agroecology, AU, professor Anne Winding, Dept. of Environmental Science, AU and associate professor Tobias Pape Thomsen, Dept. of People and Technology, Roskilde University (RUC). Reviewer contribution is specified in each chapter
Review, part 2:	Professor Berit Hasler from Dept. of Environmental Science, AU
Quality assurance, DCA:	Chief consultant Lene Hegelund, DCA Centre Unit, AU
Commissioned by:	Danish Agricultural Agency
Date for request/submission:	15.11.2021 / 10.06.2022 (hearing) / 01.09.2022 (part 1) 15.11.2021 / 06.07.2022 (hearing) / 19.09.2022 (part 2) /21.02.2024 (present revision of the knowledge synthesis (part 1+2)
File no.:	2021-0220775
Funding, part 1:	This report has been prepared as part of the "Framework Agreement on the Provision of research-based Policy Support" between the Danish Ministry of Food, Agriculture and Fisheries (FVM) and Aarhus University (AU) according to ID no. 2.28 in "Performance Agreement Plant production 2021-2024"
Funding, part 2:	This report has been prepared as part of the "Framework Agreement on the Provision of research-based Policy Support between the Danish Ministry of Food, Agriculture and Fisheries (FVM) and Copenhagen University (KU) and Aarhus University (AU) according to ID no. 3.7 in "Performance Agreement Resource- and Social economy (Ressource- og Samfundsøkonomi) 2022-2025"

External comments,	
part 1 and 2:	The report has been in public hearing as described in the Preface. The comment sheet, incl. the response from AU can be found here (part 1) and here (part 2)
External contributions,	
part1:	Review of Chapter 1 and 2 was performed by associate professor Tobias Pape Thomsen, RUC. For Chapter 2 information and data on practical processes was collected from Stiesdal SkyClean A/S and AquaGreen ApS. For Chapter 8 contributions (text boxes) are included from professor Søren Krogh Jensen, Dept. of Animal Science, AU (Box 8.1) and Associate professor Dorette S. Müller-Stöver, postdoc Esben W. Bruun and Associate professor Carsten T. Petersen, Dept. of Plant and Environmental Sciences, KU (Box 8.2)
Comments to the report,	
part 1 and 2:	The report presents results, which – at the time of the publication of this present report – have not been peer reviewed by external parties or published elsewhere. In case of subsequent publishing in journals with external peer review, changes may appear. The combined knowledge synthesis (part 1+2) was initially delivered 19.09.2022. In the present report an error in the calculated nutrient value in Tabel 1.1, in Part 2 of the initial report is corrected. Consequently, the resulting net cost after consideration of revenues for nutrients and energy side streams (section 1.2.5, Part 2) has also been adjusted. The changes have not affected the conclusions or any other elements of the knowledge synthesis.
To be cited as:	Elsgaard L., Adamsen S. A. P., Henrik B. Møller B. H., Winding A., Jørgensen U., Mortensen Ø. E., Arthur E., Abalos D., Andersen N. M., Thers H., Sørensen P., Dilnessa A. A. & Elofsson K. Knowledge synthesis on biochar in Danish agriculture. 166 pg. – DCA advisory report No. 208
Layout:	Jette Ilkjær, DCA – Danish Centre for Food and Agriculture, Aarhus University
Photo front page:	Henning Carlo Thomsen, Aarhus University
Print:	Digisource.dk
ISBN:	Printed version: 978-87-93998-93-3. Electronic version: 978-87-93998-94-0
ISSN:	2245-1684
Pages:	166 (Part 1: 144 pages. Part 2: 22 pages)
Internet version:	https://dcapub.au.dk/djfpublikation/djfpdf/DCArapport208.pdf
Policy support from DCA:	Read more here https://dca.au.dk/raadgivning/

Preface

The knowledge synthesis on biochar in Danish agriculture biochar has been prepared by researchers at Aarhus University at the request of the Danish Agricultural Agency (LBST) under the Performance Agreement on Plant Production between DCA and LBST (part 1) and under the Performance Agreement on Resource and Social Economics between FVM and the University of Copenhagen and Aarhus University (part 2).

As described in the request, LBST wants to outline the conditions necessary for production and use of biochar in an agricultural context in order to ensure that it is a cost-effective climate and environmental tool. Thus, the purpose of this knowledge synthesis is to collect current knowledge about, e.g., yield, environmental and climate effects of biochar, as well as economic aspects of its use on agricultural land, and also to inform on potential barriers for such use of biochar in a Danish context. The knowledge synthesis also lists what is currently considered as knowledge gaps.

The knowledge synthesis is structured according to the request by LBST, and is focused on biochar produced from three main streams of biomass (straw, biogas digestate and sewage sludge) as well as general use in agricultural soil. Before finalizing the request LBST organized a start-up seminar for stakeholders (October 4th, 2021) in order to obtain input for the request. The work on the knowledge synthesis follows a timeline agreed between LBST and DCA, which included a mid-term seminar with status orientation for ministries and agencies (April 5th, 2022). Part 1 of the knowledge synthesis was sent for external hearing by LBST before the final report was delivered. The economic analysis (part 2) was prepared during the hearing phase of Part 1, and afterwards part 2 was also sent for external hearing. See the data sheet for a link to the documents with the comments and AUs considering and handling.

Both parts of the knowledge synthesis are based on a literature study of relevant scientific publications primarily from Denmark, but also including other countries. Results from unpublished research projects and experiments, as well as personal communications, are also included when these were found to be relevant.

Abbreviations

AD	Anaerobic digestion
AM	Arbuscular mycorrhiza
AW	Plant available water
CEC	Cation exchange capacity
CO_2e	CO_2 equivalents, CO_2eq
DDSS	Dewatered digested sewage sludge
DOC	Dissolved organic carbon
EBC	European Biochar Certificate
EC	Electrical conductivity
FC	Field capacity
F_{perm}	Fraction of biochar C that remains in soil after 100 years
GHG	Greenhouse gas
H/C_{org}	Molar ratio of hydrogen to organic C
HHV	Higher heating value
HTT	Highest treatment temperature
K_s	Saturated hydraulic conductivity
Mg	Mega gram, equal to 1000 kg or 1 tonne (used interchangeably)
MJ	Mega Joule
PAH	Polycyclic aromatic hydrocarbons
PFAS	Per- and polyfluoroalkyl substances
PFOA	Perfluorooctanoic acid
PFOS	Perfluorooctanesulfonic acid
PTE	Potentially toxic elements
SOC	Soil organic carbon
SOM	Soil organic matter
TS	Total solids
VOC	Volatile organic compounds
WHC	Water holding capacity
WP	Wilting point
Wt	Weight

Contents

1 Selection of biochar feedstock scenarios and estimation of carbon sequestration and emissions

Anders Peter S. Adamsen and Henrik Bjarne Møller (reviewer, Tobias Pape Thomsen, RUC).

1.1 Introduction

This chapter will briefly define and describe biochar and provide an introduction to the recommendations from the European Biochar Certificate (EBC, 2022). Relevant feedstocks for a Danish production of biochar are highlighted and the chosen feedstocks for this synthesis report are defined and characterized. Finally, the emissions and carbon sequestration for the feedstock and the uses in the reference situation are defined.

1.2 Definition and description

Thermochemical conversion of biomass is traditionally referring to processes in which the biomass is decomposed at high temperatures. Among the thermochemical conversion processes, the most widely used are combustion, torrefaction, pyrolysis, and gasification. A general prerequisite for thermochemical conversion processes (except for hydrothermal liquefaction and hydrothermal carbonization) is an input stream with a dry matter content of at least 85 percent (Meers et al. 2020). Conversion of feedstock with a lower dry matter content is possible, but this will increase the retention time in the reactor and decrease reactor capacity. In this report, the focus will be on the pyrolysis process. The word “pyrolysis” is used for the conversion of a substance upon exposure to high temperatures and without or with little initial presence of oxygen. Fast and slow pyrolysis of biomass are thermochemical processes that produce pyrolytic bio-oils and gases as well as solids (char). Fast pyrolysis is generally employed to maximize the liquid bio-oil product yield, the benefit being that the bio-oil has a higher calorific value and the liquid may be handled with greater ease than conventional biomass (Prins and Dahmen 2015).

The bio-oil may be combusted directly or upgraded for use as a transportation fuel. Fast pyrolysis often results in biochar where the feedstock is only partly degraded (Bruun et al. 2011). Pyrolysis with retention times in the reactor of a few minutes and up to hours is considered as slow pyrolysis and can produce uniform biochar. In the present scenario analyses, focus will be on slow pyrolysis that can be used for production of uniform biochar, pyrolysis oil and combustible gas. Other thermochemical conversion processes are microwave-assisted pyrolysis where the biomass is heated by microwaves by adding energy to the chemical bonds, in particular water, and thus heat up the biomass from the inside. Hydrothermal carbonization and hydrothermal liquefaction are processes where wet biomass is converted into biochar, bio-oil and gases at high temperature and pressure.

1.3 The European Biochar Certificate

Biochar is not a single well defined material, but rather can have highly different properties, which makes it difficult to set up simple regulations for its use in agricultural soils including, e.g., the carbon sequestration potential. Therefore, the Ithaca Institute has proposed the European Biochar Certificate (EBC) which is a guideline for secure control and assessment for production and analysis of biochar. According to EBC (2022) biochar is defined as:

> *"Biochar is a porous, carbonaceous material that is produced by pyrolysis of biomass and is applied in such a way that the contained carbon remains stored as a long-term C sink or replaces fossil carbon in industrial manufacturing. It is not made to be burnt for energy generation."*

There is a growing number of biochar uses and the EBC has consequently introduced a number of certification classes. According to the requirements and safety regulations of the different applications, different parameters are controlled, and different limit values apply. The definition of a certification class (e.g., EBC-AgroOrganic or EBC-Agro) is a statement of admissibility of biochar for a given purpose regarding applicable laws, regulations, and relevant industry standards. Each application and thus certification class has its specific requirements. Every biochar and biochar-based product must be labelled according to the EBC certification class under which it is traded (EBC, 2022).

In the present analyses we have focused on the agricultural use of biochar and the requirements are shown in Table 1.1. EBC (2022) includes a positive list of permissible biomasses for the production of biochar. This positive list includes straw and digestate, but the proportion of animal source materials for the biogas plant must be less than 40 percent which is often not the case for Danish biogas plants. Furthermore sewage sludge is at the moment not included in the positive list. However, in a recent online version of the positive list by EBC it is stated that work is in progress in relation to positive lists for further biomasses, including sewage sludge: "*The pyrolysis of non-plant biomasses such as sewage sludge, livestock manure, manure containing biogas digestates or bones and slaughterhouse wastes may also produce valuable raw materials that could be used in the interests of the bioeconomy and climate protection. It is planned to include these raw materials mid 2022 in the EBC feedstock list following a key review publication about the product safety and conditions of use*." (EBC, 2022).

The most important criterion in EBC is that the ratio of hydrogen to organic carbon (H/C_{org}) for biochar should be lower than 0.7 on a molar basis. In comparison, the H/C_{org} ratio for biomass like straw and wood of are in the range of 1.4 – 1.5 (Brown, 2003), while sewage sludge and food waste have an even higher H/C_{org} ratio (see also Chapter 6 for details on H/C_{org} ratios). The EBC also set up a number of additional requirements like declaration of elemental and physical parameters, nutrients, and limit values for heavy metals and organic contaminants like polycyclic aromatic hydrocarbons (PAH). It should be noted that the EBC hitherto is a voluntary industry standard in Europe.

Table 1.1. Overview of the most important analytical parameters for EBC biochar used in organic farming (EBC-AgroOrganic) and conventional farming (EBC-Agro). Modified from EBC (2022).

Properties	EBC Certification Class		
	EBC-AgroOrganic		EBC-Agro
Elemental analysis	Declaration of C_{tot}, C_{org}, H, N, O, S, ash		
H/C_{org}	<0.7		
Physical parameters	Water content, dry matter (<3 mm particle size), bulk density (TS), WHC[2], pH, salt content, electrical conductivity of the solid biochar		
TGA[1]	Needs to be presented for the first production batch of a pyrolysis unit		
Nutrients	Declaration of N, P, K, Mg, Ca, Fe		
Heavy metals (g/Mg dry matter) (limit values)	Pb	45	120
	Cd	0.7	1.5
	Cu	70	100
	Ni	25	50
	Hg	0.4	1
	Zn	200	400
	Cr	70	90
	As	13	13
Organic contaminants (g/Mg dry matter (limit values)	16 EPA[3] PAH[4]	4 ± 2	6.0 ± 2.2
	8 EFSA[5] PAH	<1	
	benzo[e]pyrene benzo-[j]fluoran-	<1	

[1]TGA, thermal gravimetric analysis; [2]WHC, water holding capacity; [3]EPA, US Environmental Protection Agency; [4]PAH, Polycyclic Aromatic Hydrocarbons; [5]EFSA, European Food Safety Authority

1.4 Feedstock for pyrolytic biochar production

In principle all types of biomasses can be used as feedstock for pyrolysis, but not all pyrolysis processes are suitable for all types of biomass. In general, the dry matter content should be high to facilitate a fast process on an industrial scale. However other factors like low wear and tear (corrosion, fouling etc.), simple exhaust gas cleaning (low levels of Hg, Cd, S, Cl, etc.), suitable particle size distribution, high density and heating value etc. are important.

The feedstock should be available locally and to an affordable cost. By-products from the pyrolysis plant should be easy to use and capitalize. In Table 1.2, the most obvious feedstock for a Danish production of biochar are listed. A common factor is that all of them can be considered as by-products.

Table 1.2. Biomass origins and types

Origin	**Types**
Agricultural residues	Livestock manure Straw Hulls, brans etc. from processing of cereals
Energy and non-food crops	Digestate from biogas plants Fibre fraction from bio-refining of grass
Forest residues	Willow (as energy crop) Wood
Industrial and municipal waste	Woody fraction from garden and park waste Residues from feed and food production
Wastewater treatment plants	Sewage sludge

In the present analysis, straw from cereals, fibres from separation of digestate and digested and dewatered sewage sludge (DDSS) will be used for assessment of energy- and mass balances by pyrolysis and compared with reference situations. In Table 1.3, the reference uses of biomass for the three scenarios are listed. In Table 1.4, the assumptions for the biomass used are listed. The reasons for the feedstock selection are that straw is produced in large quantities with a well-known logistic, while fibres from digested manure and digested and dewatered sewage sludge are surplus materials with low or negative price.

Table 1.3. References in pyrolysis scenarios

Biomass	Reference situation
Straw from cereals	Straw is left after harvest and incorporated into agricultural soil.
Separated fibre fraction from digestate from biogas production	The separated fibre fraction from digestate is stored, applied and incorporated into agricultural soil
Digested and Dewatered sewage sludge (DDSS)	The anaerobic digested and dewatered sewage sludge will be stored, applied and incorporated into agricultural soil

Table 1.4. Assumptions on dry matter, ash and nutrient content for feedstock in the three scenarios

Biomass scenario	Dry matter	Ash	Org C	Total N	NH4-N	P	K	Energy HHV*	Literature source
	(%)	(%)	(kg/Mg DM)					(GJ/Mg)	
Straw from cereals	91	5	420	4.2	0	0.72	13.6	16.4	1, 2, 7
Fibre fraction from digestate	30	20	360	12	3.5	14	3.2	16.4	3, 4, 5
DDSS†	25	42	290	44	4.7	32	1.4	13.2	6

*HHV, Higher Heating Value. †DDSS, digested and dewatered sewage sludge.
Literature sources: (1) Ea Energianalyse (2020), (2) Møller et al. (2021), (3) Møller et al. (2022), (4) Cathcart et al. (2021), (5) Poulsen et al. (2019), (6) Thomsen (2018), (7) Brown (2003).

1.5 Input and output of energy

There are two energy outputs from the pyrolysis process, namely the pyrolysis oil and non-condensable gas. The energy outputs leave the reactor as a mixture of condensable and non-condensable gasses, and after cooling the condensable part of the gas can be taken out as pyrolysis oil leaving the non-condensable gas. The raw gas from the pyrolysis process, containing the oil components, can also be used directly as a hot fuel for the pyrolysis process and/or other processes requiring high temperature heat. In our calculations we assume that the energy output will displace natural gas or heating. It is assumed that the methane substituted is of fossil origin with emissions of 68 g CO2e/MJ from the BioGrace II database for an EU mix of natural gas (www.biograce.net). Electricity is also needed for operation of the plant, e.g., for conveyers, pumps etc. Heat is required for the pyrolysis process; this heat is normally generated by combustion of a part of the combustible gasses generated in the main pyrolysis process, meaning that less energy will be available for substitution of natural gas or heating. The plant electricity demand is assumed to be covered by a mix of the Danish electricity production, which is predicted to be 0.048 kg CO_2/kWh for 2022 (ENS, 2022).

1.6 Carbon sequestration in soil

The main interest for biochar in relation to Danish agriculture is caused by the fact that biochar due to a high degree of carbonization will make the biomass more recalcitrant towards biological degradation thus preserving the carbon in the soil and avoiding re-emission of CO_2 back into the atmosphere where it contributes to climate change.

1.6.1 Carbon sequestration based on hydrogen and organic carbon ratio

IPCC has described a preliminary method for estimating the sequestering of carbon in soils after 100 years (IPCC, 2019). This methodology defines a factor F_{perm} denoting the estimated carbon sequestering after 100 years, which depends on the feedstock type and pyrolysis temperature. This IPCC methodology, and variations thereof, is described in more detail in Chapter 6.

An indirect method to estimate F_{perm} is based on H/C_{org} molar ratio and yearly soil temperature in the depth of 10 cm as:

$$F_{perm} = c_{hc} - m_{hc} \times H/C_{org} \qquad \text{(Eq. 1.1)}$$

where chc and mhc are regression coefficients, intercept and slope, respectively, as defined in the publication by Woolf et al. (2021). The chc is 1.13 and 1.10 and mhc is 0.46 and 0.59 for mean annual soil temperatures of 5 are 10°C, respectively (Woolf et al. 2021).

Based on a 5-year average, the yearly soil temperature in Denmark in the depth of 10 cm is 9.8°C (see Chapter 6). For the cut-off value for the H/C_{org} ratio in the EBC certificate of 0.7, the resulting F_{perm} values are 0.81 and 0.63 at soil temperatures of 5 and 10°C, respectively. Lower H/C_{org} ratios can be found in the literature, e.g., Laghari et al. (2021) who found H/C_{org} ratios less than 0.4 for sewage sludge, biogas fibre, cattle manure, poultry manure, woodchips and wheat straw pyrolysed in lab scale at 600°C and retention time of 60 minutes. However, full scale production is a trade-off between reactor capacity, dry matter and particle size of feedstock, resulting in ratios lower than lab-scale results. In the estimates in the present analyses, a H/C_{org} ratio of 0.7 will be used, which will result in a conservative estimate of the stability of biochar in soil, meaning that high-quality biochars produced with lower H/C_{org} ratio expectedly will have larger C sequestration potential (see Chapter 6 for further details on C sequestration from biochar).

1.6.2 Carbon sequestration of straw, separated fibre fraction and dewatered sewage sludge

The potential carbon storage from mulched straw is based on an exponentially decreasing projection of test results for soil carbon content after straw mulching up to 18 years from Christensen and Schjønning (1987). Based on these data, it is estimated with a simple single-pool exponential decay function that after 2 years about 26 percent of the added carbon remains, after 20 years about 7.5 percent remains, while after 100 years only 3 percent of the carbon remains from the straw added to soil in year 0. Thus, in our calculation we assume that 3 percent of soil carbon is left after 100 years for straw. This estimate is similar to the output from the C-TOOL model, based on straw decomposition in a Danish soil type with 12.5% clay, a C/N ratio of 10, and an annual air temperature of 8°C (see Jensen et al. 2022, where also C retention at 0-100 year time scales is shown).

For the potential carbon storage of separated fibres from biogas, data are generally missing from the literature on the storage of soil carbon. One of the few useful sources, Thomsen et al. (2013), estimated that 42 percent of the carbon in bio-gasified grass silage decomposes over 1-2 years, whereby 58 percent of the carbon is retained in the soil after 2 years. This corresponds to just over twice the retention of the carbon compared to the soil-incorporated straw described above (with 26 percent retained after 2 years). By comparison with 30% C remaining after 20 years for straw (see previous paragraph), we therefore assume that 15 percent of the carbon produced in the bio-gasified straw is left behind after 20 years. This corresponds very well to the 13 percent retention that Thomsen et al. (2013) estimated for 'long-term' carbon storage, without, however, indicating which time horizon long-term covers. In our calculation, we assume that 10 percent of the carbon amended to soil with the digestate fibres still persists in the soil after 100 years. This estimate aligns with a model run in C-TOOL, which indicated retention of ca. 8 percent C in digestate fibres on a 100 year time scale under selected Danish soil conditions (see Jensen et al. 2022, where also C retention at 0-100 years is shown). For sewage sludge, a value of 12.5 percent for carbon sequestration based on simulation in the Daisy-model is used (Larsen et al. 2018 and supplementary information therein).

1.7 Emission of methane and nitrous oxide from use of feedstock in the reference situations

The biomass from the three scenarios will in the reference situation be applied on agricultural fields (Table 1.3). Both fibre fraction from digestate as well as dewatered, separated sewage sludge will be stored before application the fields. During storage, methane and nitrous oxide (N_2O) will be produced and emitted. The IPCC emission factors will be used to assess these emissions (IPCC, 2006). When applied on the fields, the nitrogen will generate nitrous oxide, which will be assessed according to chapter 11 in the IPCC guidelines from 2006.

1.8 Emission during production of biochar

It is assumed that production of biochar in Denmark fulfils the current regulations with respect to emission of gas, wastewater and solids, and these emissions are very dependent on the degree of gas cleaning. For gas emissions, the Legal order for waste combustion plants (Miljøstyrelsen 2017) can be used as a guideline.

1.9 References

Brown RC 2003. Biorenewable Resources. Engineering New Products from Agriculture (1st edition). Blackwell Publishing.

Bruun EW, Hauggaard-Nielsen H, Ibrahim N, Egsgaard H, Ambus P, Jensen PA, Dam-Johansen K, 2011. Influence of fast pyrolysis temperature on biochar labile fraction and short-term carbon loss in a loamy soil. Biomass Bioenergy 35, 1182–1189.

Cathcart A, Smyth BM, Lyons G, Murray ST, Rooney D, Johnston CR, 2021. An economic analysis of anaerobic digestate fuel pellet production: can digestate fuel pellets add value to existing operations? Cleaner Engineering and Technology 3, 100098.

Christensen BT, Schjønning P, 1987. Nedmuldning af halm. Tidsskrift for Planteavls Specialserie, S1911.

Ea Energianalyse, 2020. SkyClean pyrolyse af halm og nedmuldning af biokul som klimavirkemiddel, 34 pp.

EBC (2022) European Biochar Certificate - Guidelines for a Sustainable Production of Biochar. European Biochar Foundation (EBC), Arbaz, Switzerland. (http://european-biochar.org). Version 10.1 from 10th Jan 2022.

IPCC (2006). 2006 IPCC Guidelines for National Greenhouse Gas Inventories Volume 4 Agriculture, Forestry and Other Land Use. Chapter 11: N_2O Emissions from Managed Soils, and CO_2 Emissions from Lime and Urea Application. https://www.ipcc-nggip.iges.or.jp/public/2006gl/vol4.html

IPCC, 2019. 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Calvo Buendia E., Tanabe K, Kranjc A, Baasansuren J, Fukuda M, Ngarize S, Osako A, Pyrozhenko Y, Shermanau P, Federici S (Eds). IPCC, Switzerland. https://www.ipccnggip.iges.or.jp/public/2019rf/index.html

Jensen JL, Thers H, Elsgaard L, 2022. Afklaring om videns- og ressourcebehov ved at integrere biokul i C-TOOL modellen til brug for emissionsopgørelser, Nr. 2022-0341679, 10 s., maj 17, 2022

Larsen, J. D., ten Hoeve, M., Nielsen, S., & Scheutz, C. (2018). Life cycle assessment comparing the treatment of surplus activated sludge in a sludge treatment reed bed system with mechanical treatment on centrifuge. Journal of Cleaner Production 185, 148-156.

Laghari M, Muller-Stover DS, Puig-Arnavat M, Thomsen TP, Henriksen UB, 2021. Evaluation of biochar post-process treatments to produce soil enhancers and phosphorus fertilizers at a single plant. Waste and Biomass Valorization 12, 5517-5532.

Meers E, Velthof G, Michels E, Rietra R, 2020. Biorefinery of Inorganics: Recovering Mineral Nutrients from Biomass and Organic Waste, (1st Edition). John Wiley & Sons Ltd.

Miljøstyrelsen 2017. Bekendtgørelse om anlæg, der forbrænder affald. BEK nr 1271 af 21/11/2017 https://www.retsinformation.dk/eli/lta/2017/1271

Møller HB, Sørensen P, Olesen JE, Petersen SO, Nyord T, Sommer SG, 2022. Agricultural biogas production - climate and environmental impacts. Sustainability 14, 1849.

Møller HB, Romio C, Rasmussen MD, Jensen LS, Schwartzkopff M, Holm-Nielsen JB, Agdal EBL, Larsen SU, Hinge J, 2021. Halm til biogas – Erfaringer og klimaaspekter. Teknologisk Institut. Rapport udarbejdet for INBIOM – Innovationsnetværket for Bioressourcer.

Prins W, Dahmen N, 2015 (Eds.). Processes for thermochemical conversion of biomass. 10th European Conference on Industrial Furnaces and Boilers. Porto Gaia, Portugal.

Poulsen HD, Møller HB, Klinglmair M, Thomsen M, 2019. Husdyrs fosforudnyttelse og fosfors værdikæde fra husdyrgødning, bioaffald og spildevand: Faglig baggrundsrapport for fosforvidensyntese. Aarhus Universitet, DCE – Nationalt Center for Miljø og Energi, 82 s. - Videnskabelig rapport nr. 325.

Thomsen IK, Olesen JE, Møller HB, Sørensen P, Christensen BT, 2013. Carbon dynamics and retention in soil after anaerobic digestion of dairy cattle feed and faeces. Soil Biology and Biochemistry 58, 82–87.

Thomsen TP, 2018. Drying and pyrolysis of sludge – in a climate perspective. AquaGreen PCP phase 2, WP 5: Carbon Footprint Analysis of AquaGreen system. DTU Chemical Engineering, Risø, Denmark, 59 pp.

Woolf D, Lehmann J, Ogle S, Kishimoto-Mo AW, McConkey B, Baldock J, 2021. Greenhouse gas inventory model for biochar additions to soil. Environmental Science & Technology 55, 14795-14805.

2 Production of biochar based on straw, digestate fibers and sewage sludge

Anders Peter S. Adamsen and Henrik Bjarne Møller (reviewer, Tobias Pape Thomsen, RUC).

2.1 Introduction

This chapter will define and describe the different process types and the consequences for the properties of the biochar and give examples of mass balances for the three selected biomasses used in the scenarios. Economic and environmental impact besides emission of climate gases will not be covered. Trends and variation in published data on biochar are shown based on database values, and three examples of scenarios are presented based on the selected feedstocks and reference uses of the feedstocks.

2.2 Process types

The thermal decomposition process of pyrolysis using lignocellulosic biomass takes place in the absence of oxygen under an inert atmosphere. As an inert atmosphere, argon or nitrogen gas flow is usually needed. The fundamental chemical reaction is very complex and consists of several steps. The end products of biomass pyrolysis consist of biochar, bio-oil and gases. The pyrolysis gases are mainly methane (CH_4), hydrogen (H_2), carbon monoxide (CO) and carbon dioxide (CO_2). The organic materials present in the biomass substrate starts to decompose at around 350 – 550°C and it can proceed until 700 – 800°C without the presence of air/oxygen. The proportion of each end product depends on the temperature, retention time, heating rate, pressure, use of catalysts, char beds, partial oxidation, and reactor design and configuration.

For biochar production, slow pyrolysis is often chosen, whereas fast or flash pyrolysis is applied when the focus in on optimization of oil or gas yields. The hot biochar will leave the reactor at high temperature and needs to be cooled by using, e.g., a water mantled cooler, generating warm water or, in some cases, directly cooled by adding water directly on the hot char to minimize the risk of self-ignition. Alternately, the cooling can be done by use of inert gases. To optimize the biochar quality and reduce contamination with toxic polycyclic aromatic hydrocarbons (PAH), it is crucial to flush the pyrolytic gases away from the biochar before cooling (Madej et al. 2016). Cooling the biochar in the proximity of the pyrolysis gases may lead to the re-condensation of PAH and other tar species onto the biochar.

2.3 Properties of produced biochar

The properties of the biochar depend on the feedstock and process conditions. In general, higher pyrolysis temperatures tend to produce biochar with higher pH-value, organic carbon and ash contents. On the other hand, higher pyrolysis temperature reduces the yield of biochar. Figures 2.1 – 2.9 are compiled based on data extracted from the UC Davis Biochar database (http://biochar.ucdavis.edu/) for dry grass (straw etc.), manure and sludge. This database included in March 2022 data from 1177 experiments. Filtering the data to include "manure", "sludge" and "grass" gave 406 experiments, which are used for the figures. Manure encompasses different livestock manure and grass includes straw from cereals. The responsibility for adding data to the UC Davis Biochar database is on the contributors. As the database is based on inputs from many sources with various feedstocks, process conditions and analyses of biochar, but the figures can be used to illustrate general trends and the variations encountered for various biochars.

2.3.1 Total ash and volatile matter contents

Figure 2.1 shows the total ash content as a function of pyrolysis temperature. Most notable is the high ash content of biochar from sewage sludge in the majority of cases, which can be attributed to the feedstock and use of chemicals for dewatering. Manure-based biochar also has high ash contents, whereas dry grasses (incl. straw) show moderate ash contents. In general, the ash contents increase with higher pyrolysis temperatures, which degasses volatiles with oxygen and hydrogen, and thus increase the carbon and ash contents in the resulting biochar.

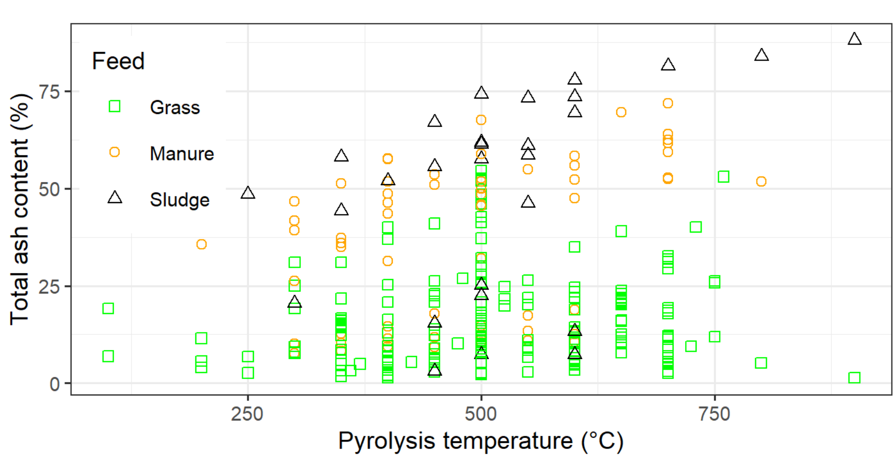

Figure 2.1. Total ash content in biochar versus pyrolysis temperature. Data from UC Davis Biochar database.

The volatile matter in biochar, as reported in the UC Davis Biochar database, is generally measured after combustion in an oven at 900°C for 7 minutes in air. The volatile matter is small in sewage sludge and high in manure at pyrolysis temperatures about 500°C (Figure 2.2).

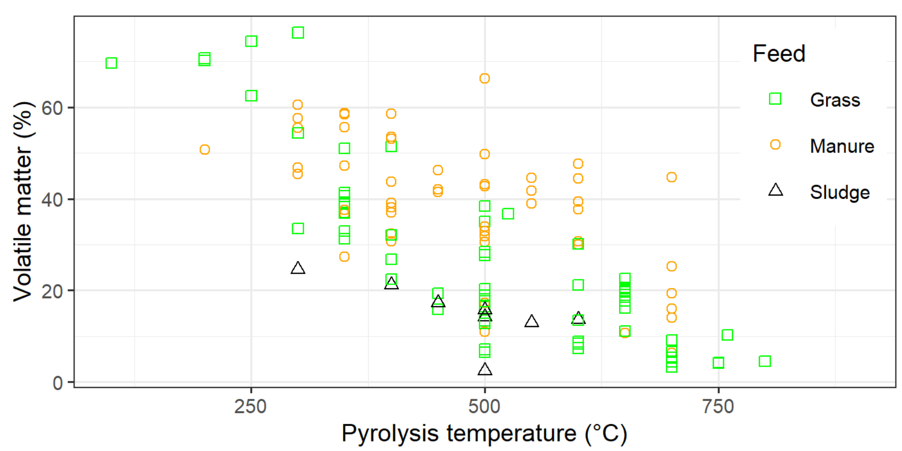

Figure 2.2. Content of volatile matter in biochar versus pyrolysis temperature. Data from UC Davis Biochar database.

2.3.2 pH of produced biochar

The pH of biochar (measured in an aqueous suspension) increases with increasing pyrolysis temperatures as shown in Figure 2.3. This is attributed formation of carbonates and deprotonated carboxyl and alcohol groups during the pyrolysis (see Chapter 4, section 4.2.1. The data also highlight that most produced biochars are alkaline, although this is not always the case.

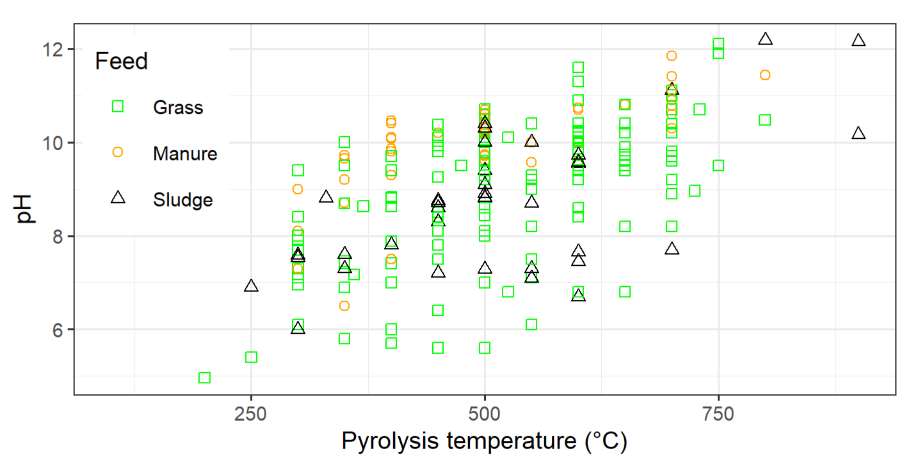

Figure 2.3. The pH of biochar versus pyrolysis temperature. Data from UC Davis Biochar database.

2.3.3 Contents of organic carbon, nitrogen and phosphorus

The content of carbon in biochar vary with pyrolysis temperature and feedstock (Figure 2.4). In general the content is high for grasses, and low for sludge due to the high ash content (Figure 2.1). Manure biochar has a carbon content that lies between the other two types, although a few manure biochars samples have lower carbon content than found in sludge biochar.

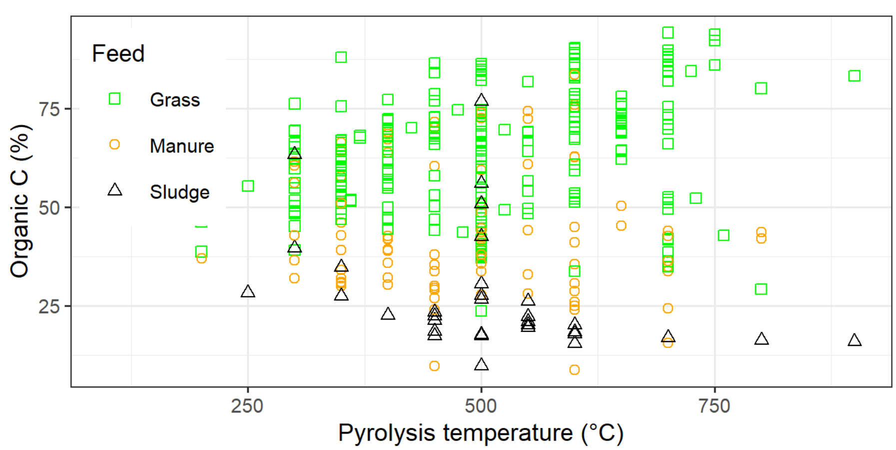

Figure 2.4. Organic carbon (C) content versus pyrolysis temperature. Data from UC Davis Biochar database.

The total nitrogen content in biochar reflects the nitrogen in the feedstock with low content (0 – 2 weight percent) in biochar from grasses and higher contents in sewage sludge and manure (Figure 2.5). However, it should also be noted that a large proportion of the feedstock N is lost during the pyrolysis process as described in Chapter 7.

For total phosphorus in biochar the distribution is more pronounced than for nitrogen with high contents in manure-based biochar, less in sewage sludge and in general low contents of phosphorus in biochar from grasses (Figure 2.6).

It is of importance that nitrogen and phosphorus can be used as nutrients for plants when applied on agricultural fields. Therefore, the bio-availability of these nutrients in biochar is also of importance, which is discussed in Chapter 7. Furthermore, it must be considered that there are losses (emissions) of nitrogen during the processing of feedstock, where ammonium-nitrogen will follow the water fraction during separation (dewatering), and gaseous ammonia (NH_3) can be emitted during drying.

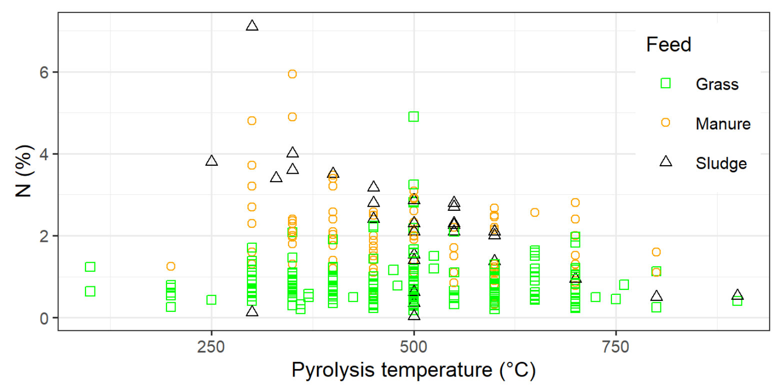

Figure 2.5. Total nitrogen (N) content in biochar versus pyrolysis temperature. Data from UC Davis Biochar database.

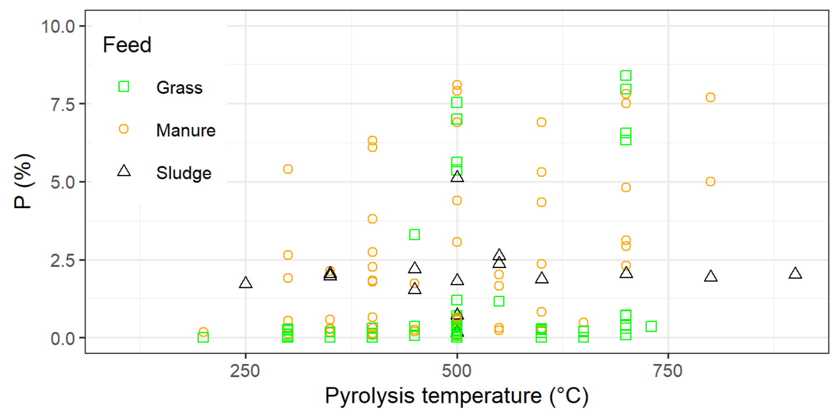

Figure 2.6. Phosphorus (P) content in biochar versus pyrolysis temperature. Data from UC Davis Biochar database.

2.4 Molar ratios of carbon, hydrogen and oxygen

Molar ratios of carbon and hydrogen or oxygen give information about the severity or effectiveness of the pyrolysis process (EBC, 2022). In fact, the hydrogen-carbon ratio (H/C ratio) has been suggested as the primary parameter to characterize biochar and its related carbon sequestration potential in soils. This ratio can also be calculated based on only the organic C (C_{org}) in biochar (thus excluding carbonate contents), in which case it is denoted the H/C_{org} ratio (see Chapter 6). There is an inverse proportionality between the hydrogen-carbon ratios in biochar as a function of pyrolysis temperature (Figure 2.7). According to the EBC, the molar hydrogen-carbon ratio of biochar should be less than 0.7. Many biochars produced at temperatures below 500°C in the UC Davis Biochar database cannot fulfil this criterion as shown in Figure 2.7, while almost all biochars produced above temperature of 500°C have a molar H/C ratio that is below the threshold of 0.7. The molar ratio of oxygen and carbon is shown in Figure 2.8, where the oxygen content decrease with increasing pyrolysis temperature as for the hydrogen-carbon ratios, giving a linear relationship between the oxygen-carbon and the hydrogen-carbon ratios as shown in Figure 2.9.

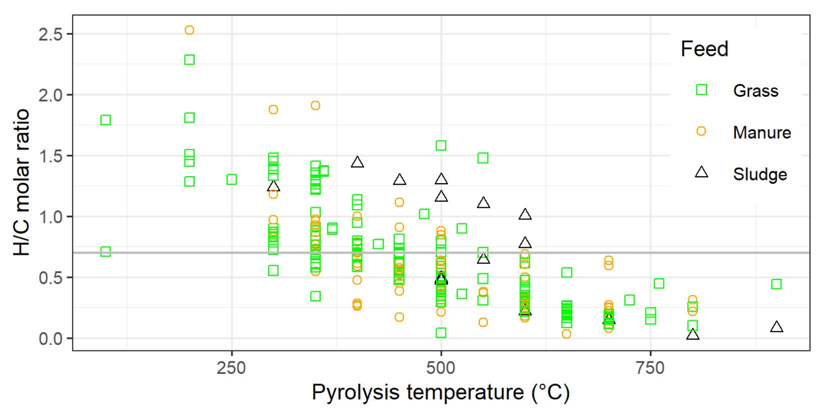

Figure 2.7. Hydrogen/carbon (H/C) molar ratio versus pyrolysis temperature. The horizontal line shows the cut-off value of 0.7 from the European Biochar Certificate (EBC, 2022). Data from UC Davis Biochar database.

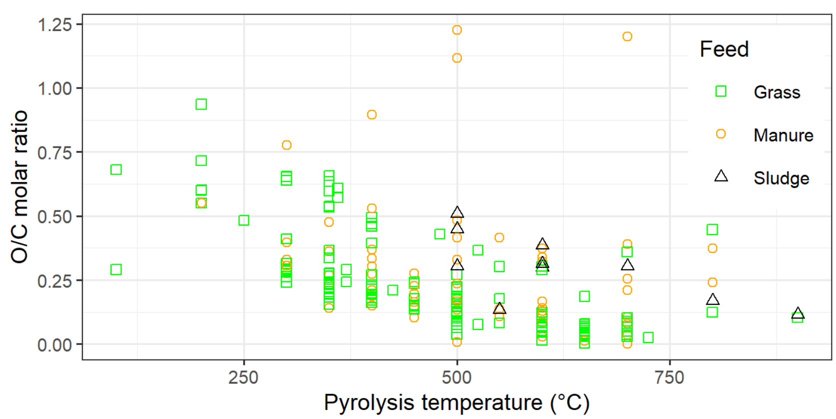

Figure 2.8. Oxygen/carbon (O/C) molar ratio versus pyrolysis temperature. Data from UC Davis Biochar database.

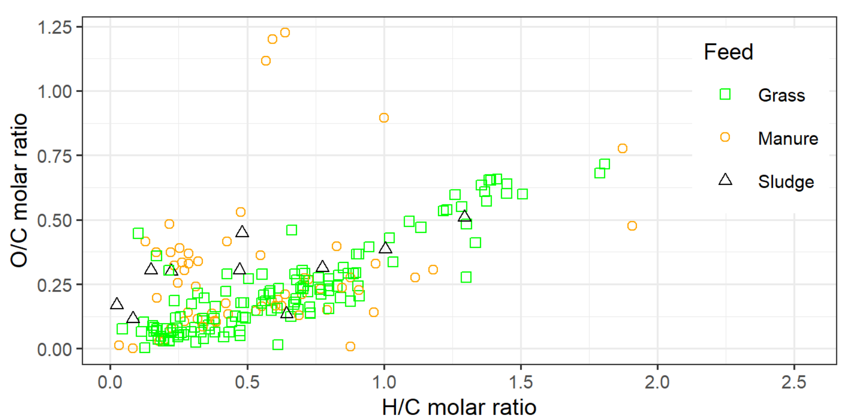

Figure 2.9. Oxygen/carbon (O/C) molar ratio versus hydrogen/carbon (H/C) molar ratio. Data from UC Davis Biochar database.

2.4.1 Impact of pyrolysis process retention times

In a study of the effect of temperature and retention time on carbon yields by Ronsse et al. (2013), pyrolysis temperatures varied between 300 and 750°C with pyrolysis process retention times of 10 and 60 min for wood, straw, green waste and dry algae. For straw, there was a clear difference with the applied retention times at the low pyrolysis temperatures of 300°C. However at 450, 600 and 750°C the hydrogen-carbon ratio did not differ a lot between 10 and 60 min (Figure 2.10). Another interesting outcome of their study was that the fixed carbon yields (the carbon yield based on the carbon content in the feedstock) on dry and ash-free basis were between 22 and 25% for the various process conditions for straw even though the biochar mass yields differed between 24 and 95% on a dry and ash-free basis (Figure 2.11). The high yields obtained at pyrolysis temperatures at 300°C were offset by a corresponding reduction in carbon contents in the biochar, thus resulting in a similar fixed carbon yields at all temperatures (Figure 2.12). It should be noted, that the carbon yields from this study is somewhat lower than other studies, where, e.g., 40% was found for straw (Laghari et al. 2021). The hydrogen-carbon ratios decreased with increasing temperatures, which will affect the carbon sequestration potentials of the biochar.

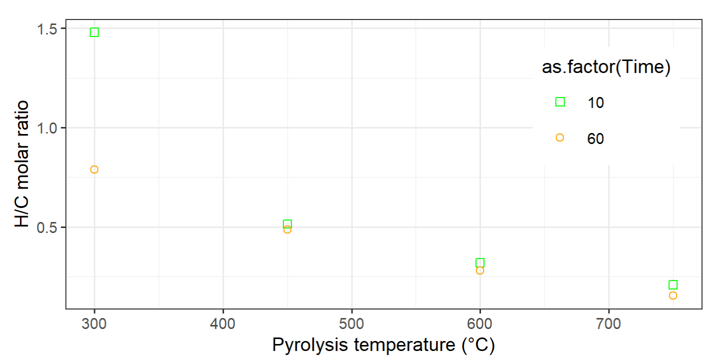

Figure 2.10. Hydrogen carbon (H/C) molar ratio versus pyrolysis temperature for one batch of straw. Only pyrolysis temperature and time were varied. Green square: 10 min, orange circle: 60 min. Data from Ronsse et al. (2013).

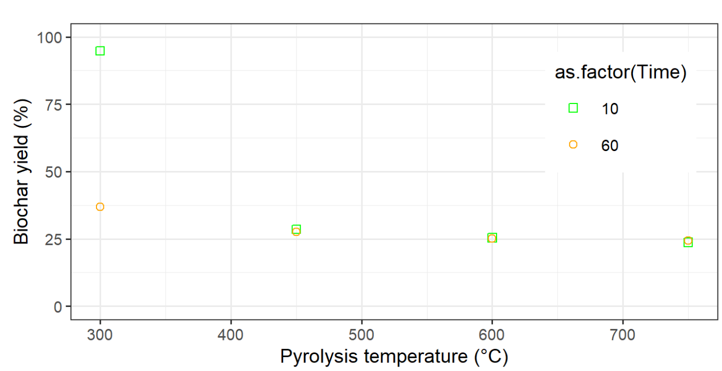

Figure 2.11. Biochar yield versus pyrolysis temperature for one batch of straw. Only pyrolysis temperature and time were varied. Data from Ronsse et al. (2013).

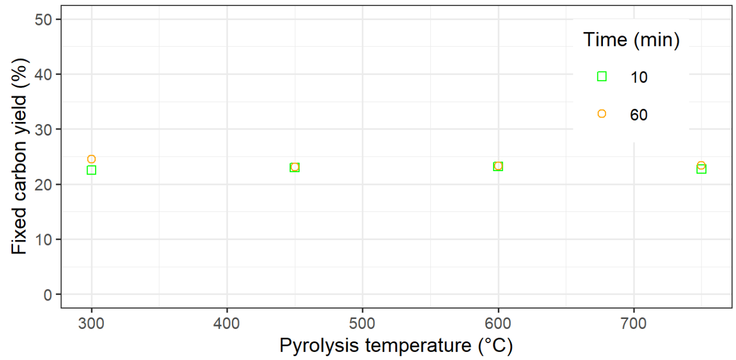

Figure 2.12. Fixed carbon yield in straw biochar (on a dry and ash-free basis) versus pyrolysis temperature. Only pyrolysis temperature and time were varied. Data from Ronsse et al. (2013).

2.4.2 Self-ignition

Stored biochar like other organic materials can continue to decompose with generation of heat, which under some conditions can result in self-ignition (Phounglamcheik et al. 2022). This can be circumvented by adding water to the stored biochar or storing it in an inert atmosphere

2.4.3 Application of biochar and dust emission

Production and spreading of biochar as a powder can generate huge emission of dust (e.g., Gelardi et al. 2019). Therefore methods to spread and incorporate or inject the biochar into the soil should be tested and documented (such as pelletizing). This is relevant in relation to wide-scale application of biochar to the plough layer of agricultural soils, but also in relation to potential alternative applications of biochar, e.g., for improvement of the physical structure in subsoils (see Chapter 8, Box 8.2). Addition of water and incorporation into the soil is a possible solution for dust problems (Gelardi et al. 2019). The biochar could also be mixed with manure slurry and applied with slurry tankers with trailing hoses, trailing shoes, or injection. However, these technologies should be tested under Danish conditions, also in relation to the amount of biochar that could be added in this way without damaging the equipment. Besides the dust emission during spreading of biochar, dust can also be a problem during handling and storage of biochar.

2.5 Biochar from digested and dewatered sewage sludge

Dewatered sewage sludge is a troublesome and costly biomass to handle. Currently, the majority is stored for up to one year and subsequently spread on agricultural fields. Another fraction is mineralized in reed beds, and the final fraction is combusted. Using dewatered sewage sludge as a feedstock for biochar production is thus an interesting alternative to the present handling.

In Denmark, the company AquaGreen markets a combined drying and pyrolysis plant to treat dewatered wastewater (sewage) sludge. All the pyrolysis gas is used for superheated steam, which dry the dewatered sludge. The rest of the energy in the steam after drying is used for heating (e.g., district heating) after being condensed in a heat exchanger. Figure 2.13 shows a diagram of an AquaGreen plant for production of biochar based on dewatered sewage sludge.

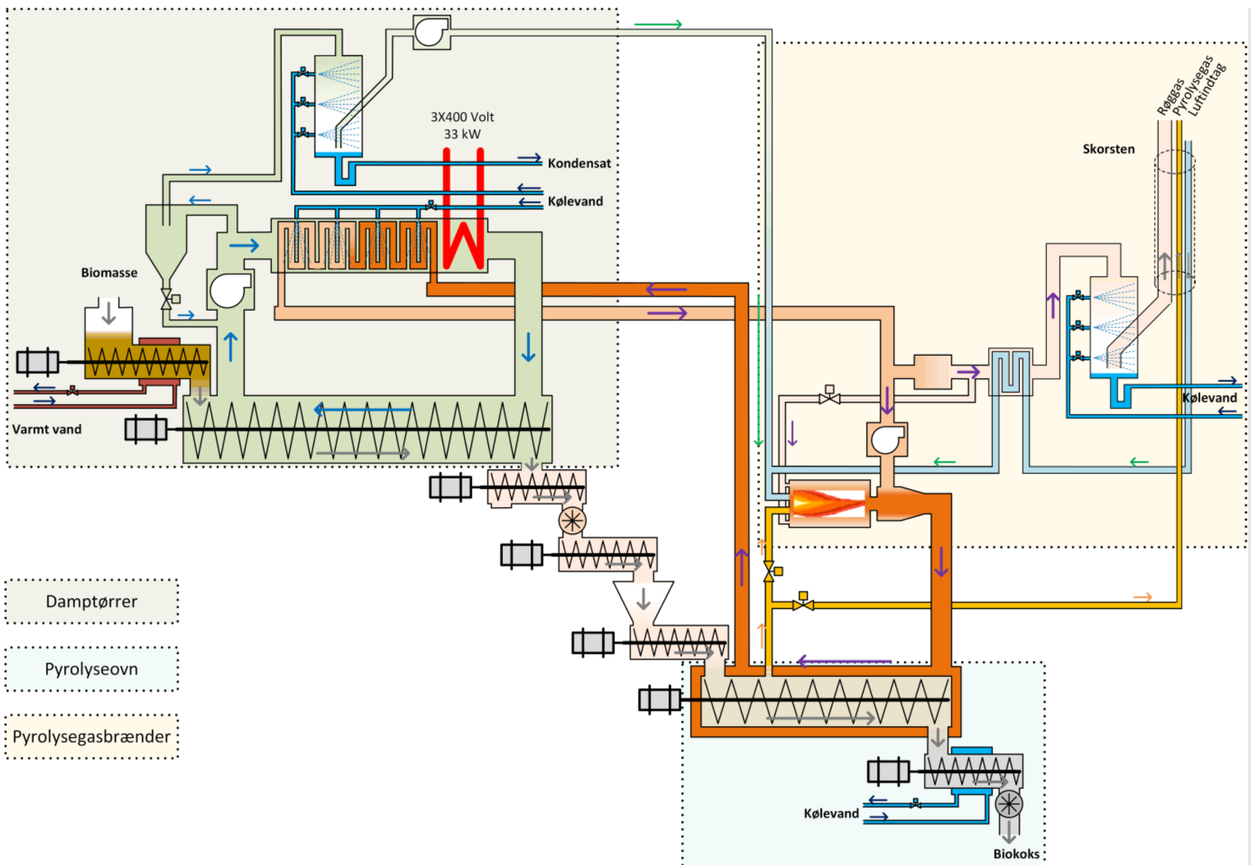

Figure 2.13. Diagram of a combined sludge and pyrolysis plant from AquaGreen. From Odsherred Kommune (2021). Damptører = steam dryer; Pyrolyseovn = pyrolysis reactor; Pyrolysegasbrænder = pyrolytic gas burner: Kølevand = cooling water; Biokoks = biochar.

The contents of dry matter and organic matter in sewage sludge vary significantly and depend on wastewater treatment method, dewatering method and whether flocculants has been used. In a dewatered sludge analysis used for calculation in Thomsen (2018) the dry matter was 24.8 percent and the organic matter was 57.6 percent of the dry matter. Similarly, results from another Danish sewage sludge using a decanting centrifuge for dewatering were found to 23.5 percent dry matter and 62.1 percent organic matter of the dry matter (Larsen et al. 2018).

AquaGreen expects 500 kg biochar to be produced from 1000 kg 100-percent dry sewage sludge and that the surplus heat is used for heating where it is expected to substitute natural gas (AquaGreen 2022, personal communication). In Figure 2.14, an example of an energy and mass balance is shown. We emphasize that the data can vary considerably and has to be evaluated in each case.

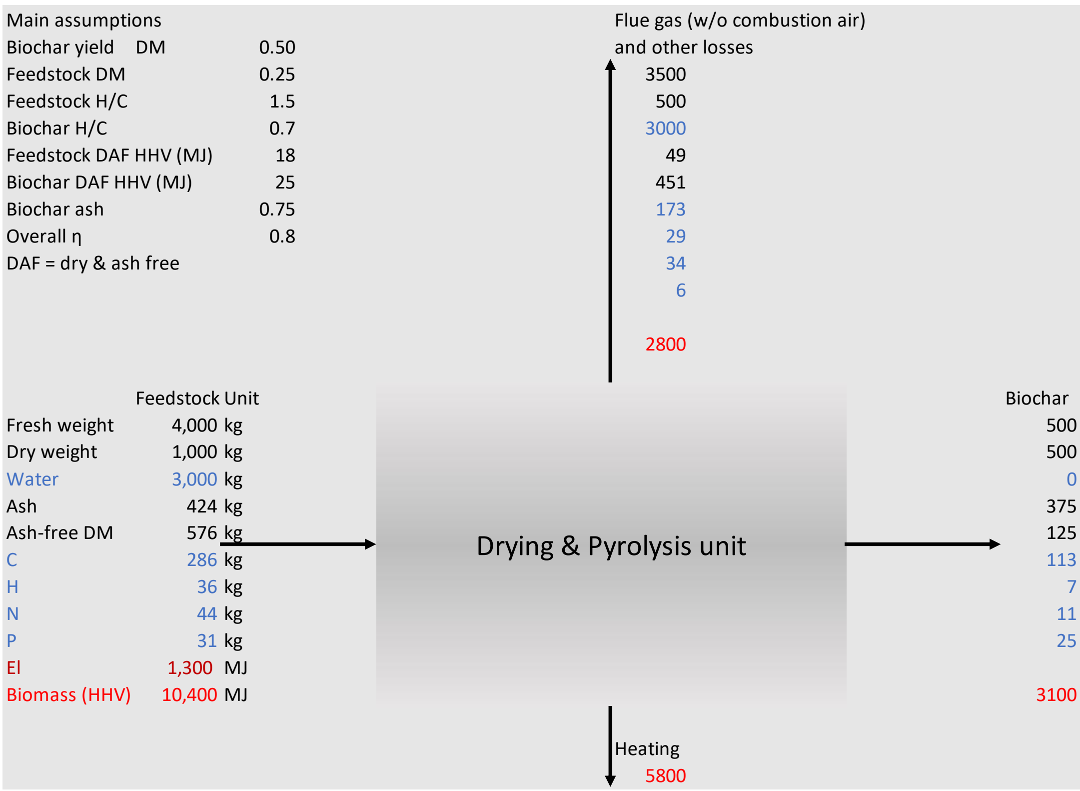

Figure 2.14. Tentative mass and energy balances for biochar production based on digested and dewatered sewage sludge (DDSS) with an ash content of 42.4 percent. The hydrogen-carbon rations are assumed as well as the biochar yield on dry matter basis. The colours are to ease the reading. Blue colour shows water and elements, and red colour shows energy. Data for feedstock and biochar from Thomsen (2018).

2.5.1 Sequestration of sewage sludge-based biochar in soil

With respect to calculation of the sequestration of biochar carbon into the soil, we have used a model described in Woolf et al. (2021) using a H/C_{org} ratio of 0.7 and a yearly soil temperature of 10°C in the depth of 10 cm (see Chapter 6 for details). The model predicts that 63 percent of the organic carbon is sequestered after 100 years. With 113 kg C per tonne (Mg) dry matter feedstock as shown in Figure 2.14, this results in 71 kg sequestered carbon, which corresponds to 260 kg CO_2-eq (Table 2.1). We emphasize that there is considerable uncertainty on this estimate as it is based on severe extrapolation. Furthermore, it can be considered that the estimated long-term decomposition of biochar based on an H/C_{org} ratio of 0.7 represents a conservative estimate, since biochars with a lower H/C_{org} are common and will have higher stability and contribute more to carbon sequestration when added to soil.

The reference situation is also influenced by severe uncertainty. The long-time sequestering of dewatered sewage sludge applied on the fields has been estimated by the Daisy model to 12.5 percent of carbon after 100 year (Larsen et al. 2018, supplementary information). The uncertainty of estimating the carbon sequestration potential will be further discussed in Chapter 6

Table 2.1. Estimated long-time (100 years) sequestering of sewage sludge-based biochar in soil assuming a hydrogen to carbon ratio (H/Corg) of 0.7. The F_{perm} value for sludge is from Larsen et al. (2018). The mass of carbon from one tonne (Mg) feedstock dry matter is calculated as 286 kg per tonne minus 10 percent for mineralization of carbon in during storage (i.e., 257 kg C). The regression coefficients, c_{hc} and m_{hc} are the intercept and slope, respectively, in the model described in section 1.6.

Biomass	Soil temp (°C)	c_{hc}	m_{hc}	H/C_{org} ratio	F_{perm}	C (kg)	$C_{100\ yr}$ (kg)	CO_2-eq (kg)	Source
Biochar	10	1.040	0.590	0.7	0.627	113	71	260	Woolf et al. (2021)
Sludge	-	-	-	-	0.125	257	32	118	Larsen et al. (2018)

2.5.2 Emission of climate gasses

2.5.2.1 Production of biochar

During production of biochar, energy is used for transportation of feedstock and biochar as well as running the combined drying and pyrolysis plant. The produced pyrolysis gas is used for drying the dewatered sewage sludge and subsequent heat exchanged with water that can be used for other purposes, e.g. district heating where we assume it substitutes natural gas. The emission from the use of electricity and production of surplus heat is shown in Table 2.2.

Table 2.2. Emission from production of biochar from digested and dewatered sewage sludge. EF, emission factor; GWP, global warming potential.

	Values	Unit	EF	Unit	GWP	kg CO_2-eq Mg^{-1} biochar	Comments
Electricity	360	kWh	0.070	kg CO_2-eq kWh^{-1}	1	25	Consumption
Surplus heat	5800	MJ	0.068	kg MJ^{-1}	1	-390	Substitutes natural gas
Net emission						-365	

2.5.2.2 Use of feedstock in the reference situation

Dewatered sewage sludge is typically stored for up to one year in stockpiles (Larsen et al. 2018). It is not mandatory to cover the stockpile in contrast to storage of deep litter or fibre fraction from separated livestock manure. During the storage time, the stored sludge will emit methane, carbon dioxide and nitrous oxide. Carbon dioxide is not included in the emission inventory as the carbon in biomass has been fixed from the atmosphere recently.

Methane is produced during storage of dewatered sewage sludge. In a study under Danish conditions, the average methane emission during one year in a 1.5 meter high heap was measured and indicated that an emission factor of 3 percent of the stored carbon can be estimated assuming an average storage time of 6 months (Larsen et al. 2018 and supplementary information therein).

Nitrous oxide is produced in the interphases of aerobic and anaerobic conditions, typically by denitrification of nitrate to gaseous N_2O, which is an intermediate in the full conversion to N_2 (dinitrogen). The production and the ratio between nitrous oxide and dinitrogen depends on various factors including the density of the biomass (Bernal et al. 2017). Due to the lack of better data, the IPCC values has been used for the estimates in Table 2.3 (IPCC, 2006; Chapter 11 therein).

Table 2.3. Emission from digested and dewatered sewage sludge in the reference situation (storage for up to one year and field application). DM, dry matter.

Location and climate gas	Length (mo.)	Initial C or N (kg/Mg DM)	EF	Unit	Emission (kg/Mg DM)	GWP AR6†	CO_2-eq (kg)‡	Comments
Storage								
CH_4	6	286	0.030	kg CH_4 kg C^{-1}	8.6	27	232	Larsen et al. (2018)
N_2O	6	44	0.005	kg N_2O-N kg N^{-1}	0.22	273	94	IPCC (2006) Table 10.21
Field								
CH_4	-		0	kg CH_4 kg C^{-1}	0			Willen et al. (2016)
N_2O*	-	42	0.010	kg N_2O-N kg N^{-1}	0.42	273	180	IPCC (2006) Table 11.1
Net emission							506	

**Calculated from the remaining nitrogen after storage assuming 9 N_2 molecules are emitted per N_2O molecule.*
†Values for global warming potential over 100 years from IPCC Assessment Report 6, Table 7.15 (IPCC, 2021).
‡For conversion of kg N_2O-N to kg N_2O, the values are multiplied with 44/28.

In a Swedish LCA analysis on digested and dewatered sewage sludge the energy to transport feedstock from the wastewater treatment plant to the storage facilities (at a distance set to 100 km) and subsequent spreading of the sludge on the fields using a tractor, was offset by the energy needed to produce nitrogen and phosphorus fertilizer otherwise used (Willen et al. 2017). This is in contrast to the finding of Thomsen

(2018), who found that emission from transport of the digested and dewatered sewage sludge was much less than found in the Swedish study.

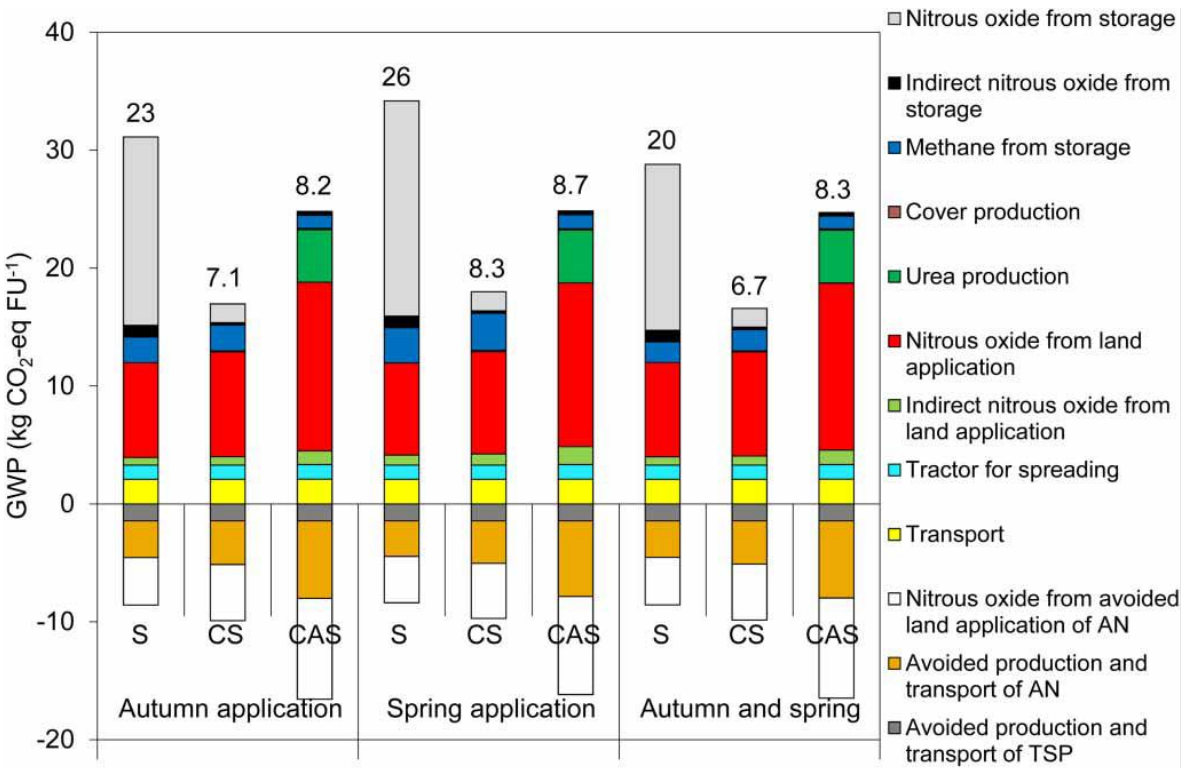

Figure 2.15. Greenhouse gas emission from storage of digested and dewatered sewage sludge (S), sewage sludge covered during storage (CS), and sewage sludge treated with urea to reduce nitrous oxide and methane emission (CAS). One functional unit (FU) is the phosphorus (1.67 kg) corresponding to 1 kg chemical phosphorus fertilizer. The S has a dry matter content of 29 percent, 100 kg C per tonne (Mg) wet weight, 12 kg nitrogen per tonne wet weight, and 9.3 kg phosphorus per tonne wet weight. The autumn application was after 1 – 9 months of storage, the spring application within 1 -12 months of storage, and the autumn-spring application within 1 – 6.5 and 5 – 6.5 months. The negative numbers are emission avoided due to saved fertilizers. From Willen et al. (2017).

The major contributors to greenhouse gas emission in the study by Willen et al (2017) were nitrous oxide from storage and land application followed by methane emission from storage (Figure 2.15). Covering of the storage reduces the production of nitrous oxide around 10 times, whereas the methane production is unaltered or slightly higher (Figure 2.15). The importance of covering with respect to reducing nitrous oxide emission is known from manure heaps, where covering the heap with an air-tight membrane reduces the emission by 99 percent (Hansen et al. 2006).

2.6 Fibre fraction from separated digestate from biogas plant

In Denmark, more than 25 percent (weight basis) of animal manure is today anaerobically co-digested on centralized biogas plants with organic wastes from the food industry, slaughterhouses, dairies, and the fish industry with the aim to produce methane for bioenergy. The residues from anaerobic digestion (AD) are recycled and today the AD plants use almost all industrial residues available in Denmark. Furthermore, increasing amounts of straw, grass, deep litter, etc. are used in the co-digestion of animal manure. The emissions of methane from storage and application of wet, biologically active biomass can be mitigated by production of biogas from manure, and the coupling of the AD and a pyrolysis plant using the fibre fraction can further reduce the emission of methane. In a future biogas scenario, pyrolysis can be integrated in several ways, and one of the solutions proposed by company Stiesdal SkyClean A/S include digestate separation, drying and pelletizing. Instead of delivering the degassed slurry directly to the field, the slurry is first separated mechanically in a fibre and a liquid fraction. The solid fraction is then dried, pelletized and used in a pyrolysis plant, where part of the dry matter is converted into pyrolysis gas and the rest comes out as biochar. This solution can help solve a phosphorus surplus problem at the biogas plants, where separation can produce a liquid fraction that does not present challenges with the phosphorous application limits of the customers of the degassed slurry (see also Chapter 7). In terms of weight, the biochar is a small proportion in relation to the digestate and can therefore be easily transported further away to areas without phosphorus surplus. If the solid fraction of the digestate fibres is not used for pyrolysis it needs to be stored and organic matter will be degraded in the manure heap as a result of increasing temperatures, and emissions of ammonia, nitrous oxide and methane will take place.

The separation of dry matter in the digestate is depending on the technology and the composition of the digestate. In Denmark, screw presses and decanting centrifuges are the most common technologies. The decanting centrifuge is more effective at partitioning both total solids (TS) and total phosphorous (TP) compared to the screw press. Moller et al. (2000) explained that increased N separation efficiency was due to the centrifuge's ability to partition fine solids in the solid phase. During separation by screw press, particles smaller than the screen size pass through to the liquid fraction. Screw press screen size varies (from 0.5 to 3 mm) depending on the particular separator, while a decanting centrifuge has been shown to partition particles as small as 0.02 mm into the solid fraction (Moller et al. 2000). These small particles contain organic nitrogen compounds, while the majority of the nitrogen in the liquid fraction is inorganic, dissolved ammonia nitrogen (NH_3–N) (Cathcart et al. 2021).

In our calculations it is assumed that a decanting centrifuge is used and the separation has an efficiency of 60 percent of the dry matter i.e., 60 percent of the dry matter ends up in the fibre fraction and 40 percent in the fluid fraction. The dry-matter concentration of the fibre fraction varies, but is often around 30%.

It is assumed that the pyrolysis of the fibre fraction is carried out with an installation such as that commercially available from Stiesdal SkyClean A/S, after drying and pelletizing (Figure 2.16).

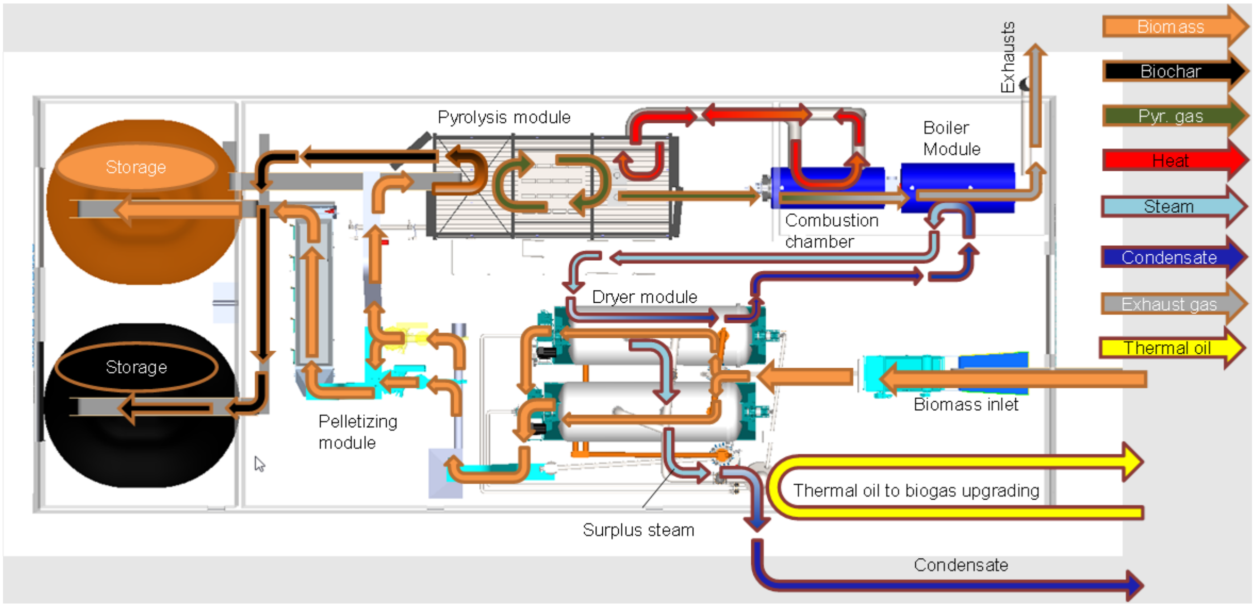

Figure 2.16. Biochar production from fibre fraction from digestate. Figure from Stiesdal SkyClean A/S (2022).

Table 2.4. Mass, energy and carbon balance by conversion of 1 tonne (Mg) of dried digestate fibres (90 percent dry matter, DM) in a SkyClean pyrolysis plant (data from Stiesdal SkyClean A/S, 2022). Energy as Higher Heating Value (HHV). HHV for biochar, 28.26 MJ/kg; HHV for oil, 32.8 MJ/kg.

		Mass balance			Energy balance		Carbon balance	
		kg	(AR basis)	(DM basis)	GJ		Kg C	
Input	Digestate fibre AR[1]	1000	100%		15.9	100%	392	100%
	Digestate fibre DM	900	90%	100%	15.9	100%	392	100%
	Moisture	100	10%	11%	0.0	0%	0	0%
Output	Biochar	390	39%	43%	8.1	51%	200	51%
	Oil	70	7%	8%	2.2	14%	49	13%
	Gas	540	54%	60%	5.6	35%	143	36%

[1]AR = as received

2.6.1 Energy and climate gas balances for digestate fibres

It is assumed that the digestate will be separated in both the reference and pyrolysis scenarios. For drying a belt dryer system was chosen as it is continuous and separated solids can be fed directly, reducing the requirement for the wet solids storage needed in a batch drying system. Belt drying requires approximately 1 MWh energy per tonne of water removed (Cathcart et al. 2021; Bolzonella et al. 2018). This consumption is higher than the theoretical energy need, which is 0.7 MWh per tonne of water removed (own calculations). In our calculations the theoretical energy consumption for evaporating water has been used. From this it can be calculated that around 4.6 GJ is needed to produce 1 tonne dried fibres with 90 percent DM. However in modern, pressurized heat pump based steam drying systems, the energy can be much lower since a large share of the energy can be recovered. By using more efficient drying systems the energy input can be reduced and the energy balance improved. For the pelletizing process it is assumed that 0.8 GJ of energy is used to produce 1 tonne of pellets.

The digestate will be separated in both the reference and pyrolysis scenario. In the pyrolysis process, no emission of climate gases is expected since the process is completely closed. However, during storage of the solid fraction of digestate, organic matter will be degraded in the manure heap and as a result of increasing temperatures, emissions of ammonia, nitrous oxide and methane will take place. Air exchange through the heap is essential for the extent and composition of gaseous emissions, where high air exchange rates contribute to intense biological degradation and resulting heat production. In the warm core, the oxygen has often been depleted, and the transformation of organic matter under anaerobic conditions may result in the production of methane, which is transported to the surface and surroundings (Møller et al. 2022). The knowledge about quantitative emissions from separated digestate is poor, but there are some results for solid pig manure (Hansen et al. 2006). Because of the lack of data on solid digestate, we use the same emissions as used by Olesen et al. (2021 with an estimated loss at 0.0075 kg CH_4 kg^{-1} VS (volatile solids) and 0.05 kg of N_2O kg^{-1} N. However these values are very uncertain and there is a need for validation of the emission factors for digestate fibres. Currently Aarhus University is partner in an ongoing research project (STABIL) investigating greenhouse gas emissions from separated and non-separated pig manure and biogas digestate samples, which is expected to create more solid data within the next couple of years.

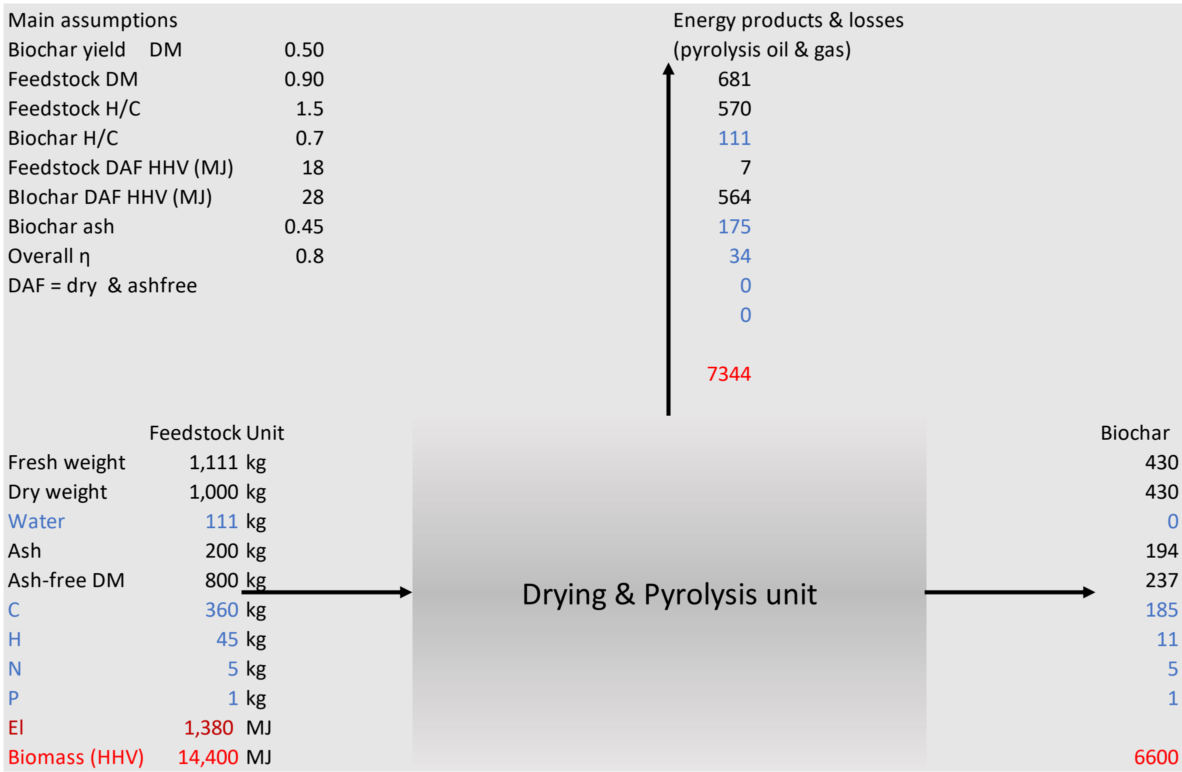

Figure 2.17. Tentative mass and energy balance for biochar production based on fibre fraction from digestate. The hydrogen-carbon rations are assumed. Data from Stiesdal SkyClean A/S (2022). The colours are to ease the reading. Blue colour shows water and elements, and red colour shows energy.

The following tables (Table 2.5-2.7) show the estimated sequestration of carbon (Table 2.5), the emission from the production of biochar (Table 2.6) and the climate gas emissions from the reference scenario of the fibre fraction (Table 2.7).

Table 2.5. Estimation of long-time (100 years) carbon sequestration from fibre fraction and digestate-based biochar based on one tonne feedstock dry matter. It is assumed that 10 percent of the carbon in the feedstock has been mineralized during storage. The regression coefficients, c_{hc} and m_{hc} are the intercept and slope, respectively, in the model described in section 1.6.

Biomass	Soil temp (°C)	c_{hc}	m_{hc}	H/C	F_{perm}	C (kg)	$C_{100\ yr}$ (kg)	CO_2-eq (kg)	Source
Biochar	10	1.040	0.590	0.7	0.627	185	116	425	Woolf et al. (2021)
Fibre fraction	10	-	-	-	0.10	324	36	132	See text

Table 2.6. Emission from production of biochar from fibre fraction from digestate. EF, emission factor; GWP, global warming potential

	Values	Unit	EF	Unit	GWP	kg CO_2-eq Mg^{-1} biochar	Comments
Electricity	383	kWh	0.070	kg CO_2-eq · kWh^{-1}	1	27	Consumption
Surplus energy	5878	MJ	0.068	kg · MJ^{-1}	1	-400	Substitutes natural gas
Net emission						-373	

Table 2.7. Emission from fibre fraction from digestate (storage up to one year and land application). EF, emission factor; GWP, global warming potential

Location	Length (mo.)	Initial C or N (kg/Mg DM)	EF	Unit	Emission (kg/Mg DM)	GWP AR6†	CO_2-eq (kg)‡	Comments
Storage								
CH_4	6	360	0.017	kg CH_4 kg C^{-1}	6.1	27	165	
N_2O	6	5.0	0.005	kg N_2O-N kg N^{-1}	0.025	273	11	IPCC (2006) Table 10.21
Field								
CH_4	-		0	kg CH_4 kg C^{-1}	0			Willen et al. (2016)
N_2O*	-	4.8	0.010	kg N_2O-N kg N^{-1}	0.048	273	21	IPCC (2006) Table 11.1
Net emission							197	

*Calculated from the remaining nitrogen after storage assuming 9 N_2 molecules are emitted per N_2O molecule.
†Values for global warming potential over 100 years from IPCC Assessment Report 6, Table 7.15 (IPCC 2021).
‡For conversion of kg N_2O-N to kg N_2O, the values are multiplied with 44/28.

2.7 Straw

In Denmark the company Stiesdal SkyClean A/S is the main actor on the market for straw pyrolysis and has recently commissioned a new installation at Greenlab Skive. The technology (SkyClean) converts straw into biochar, gas and oil. Biomass in the form of straw pellets is heated in a reactor to 500 – 600°C, thereby converting it into biochar, gas and oil. The technology used is illustrated in Figure 2.16.

2.7.1 Production of biochar from straw

In Table 2.8 the mass, energy and carbon balances for 1 tonne of straw that is converted by the SkyClean plant is shown.

Table 2.8. Mass, energy and carbon balance by conversion of 1 tonne of straw (90 percent DM) by SkyClean pyrolysis plant (Stiesdal SkyClean A/S, 2022). Energy as Higher Heating Value (HHV). HHV for biochar, 28.26 MJ/kg; HHV for oil, 32.8 MJ/kg

		Mass balance			Energy balance		Carbon balance	
		kg	(AR basis)	(DM basis)	GJ		Kg C	
Input	Straw AR[1]	1000	100%		16.5	100%	426	100%
	Straw DM	900	90%	100%	16.5	100%	426	100%
	Moisture	100	10%	11%	0.0	0%	0	0%
Output	Biochar	263	26%	29%	7.4	45%	189	44%
	Oil	107	11%	12%	3.5	21%	76	18%
	Gas	630	63%	70%	5.5	34%	161	38%

[1]AR = as received

A tentative example of a mass and energy balance is shown in figure 2.18. By-products as pyrolysis oil and gas as well as energy loss are shown in one stream as the ratio can vary due to process conditions. It is estimated that the biochar yield is 29 percent of dry matter, and the carbon yield is 44 percent.

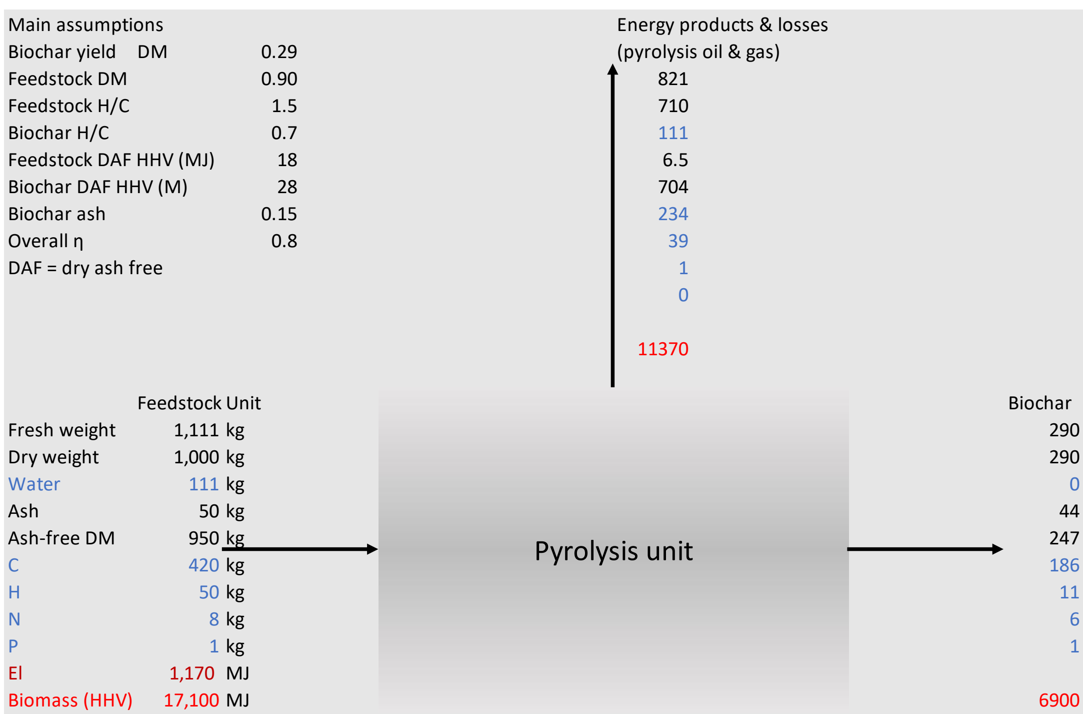

Figure 2.18. Tentative mass and energy balance for biochar production based on straw. The hydrogen-carbon ratios are assumed. Data from Stiesdal SkyClean A/S (2022). The colours are to ease the reading. Blue colour shows water and elements, and red colour shows energy.

2.7.2 Sequestering of straw-based biochar in soil

The biochar yield is relatively high for straw, and the amount sequestered after 100 years is around 63 percent assuming a yearly soil temperature of 10°C in the depth of 10 cm and a hydrogen-carbon ratio of 0.7 using the model by Woolf et al. (2021), see Table 2.9.

For the straw used as feedstock for biochar, only 3 percent of the carbon is assumed to be sequestered after 100 years (see Chapter 1).

Table 2.9. Estimation of long-term (100 years) sequestering of straw-based biochar in soil. The regression coefficients, C_{hr} and m_{hc} are the intercept and slope, respectively, in the model described in section 1.6.

Biomass	Soil temp (°C)	c_{hc}	m_{hc}	H/C	F_{perm}	C (kg)	$C_{100\ yr}$ (kg)	CO_2-eq (kg)	Source
Biochar	10	1.040	0.590	0.7	0.627	186	116	427	Woolf et al. (2021)
Straw	10	-	-	-	0.030	420	13	46	See section 1.7.3

2.7.3 Emission of climate gases

The emission from the use of electricity and surplus energy that is assumed used for district heating is shown in table 2.10. We assume that application of straw directly on the fields does not give additional emission of methane or nitrous oxide.

Table 2.10. Emission from production of biochar from straw. EF, emission factor; GWP, global warming potential

	Values	Unit	EF	Unit	GWP	kg CO_2-eq Mg^{-1} biochar	Comments
Electricity	325	kWh	0.070	kg CO_2-eq · kWh^{-1}	1	23	Consumption
Surplus energy	9100	MJ	0.068	kg · MJ^{-1}	1	-619	Substitutes natural gas
Net emission						-596	

2.8 Uncertainties

During this work, major uncertainties have been identified, which will be described briefly in the next sections (knowledge gaps will also be specifically addressed in Chapter 8). A general finding is that much of the data in the literature, and those used for our estimation, are uncertain. Many data comes from lab scale experiments and are based on feedstock with small particle sizes for practical reasons. In the future, biochar from full-scale production will be optimized economically with the use of by-products, feedstock etc. Several Danish companies are in the phase of producing biochar on demonstration or full scale, which in the next years will provide better data for evaluation of the process and products.

2.8.1 Carbon sequestering

Carbon storage of biochar is one of the most important parameters for the carbon dioxide reduction by using the pyrolysis technology. The impact on climate gas emission reduction depends both on the biochar and what is happening in terms of carbon sequestration if the biomass is used on farmland without pyrolysis. Although only few studies are available, we assume a slower carbon release from manure fibres and sewages sludge compared to the straw based on the assumption that manure fibres and sewage sludge already have been degraded in previous processes and thus consist of more recalcitrant materials.

For the biochar, several factors affect the fraction of carbon remaining (un-mineralized) after 100 years. According to IPCC this is very dependent on the temperature used in the process but the residence time during pyrolysis time will also be an important factor. This is, however, not always well defined and this information will usually not follow the biochar product. If the product has an EBC certificate there will be valuable information in this regard that follows the biochar product. However, if more precise information on carbon storage over long time should be calculated more information about process conditions are important, as discussed further in Chapter 6.

The estimated carbon sequestration values in this chapter is based on a hydrogen-carbon ratio of 0.7 which is the threshold limit set by the EBC certificate. Lower values of the biochar H/C_{org} ratio will result in higher long-term carbon sequestration.

2.8.2 Emissions from stored digestate fibres and sewage sludge

Avoided emissions of methane and nitrous oxides are very important for the climate gas balance for digestate fibres and sewage sludge. There are still considerable uncertainties and need for knowledge concerning the estimation of methane and nitrous oxide emissions from digestate fibres. Almost all current data is based on uncovered heaps, but in Denmark storage should take place in heaps covered with a tight material. Further studies on such systems are clearly needed.

2.8.3 Separation of digestate

In the biogas industry, decanting centrifuges and screw presses are the most common separation technologies used. In the present chapter, a decanting centrifuge has been considered for separation of digestate. This technology is much more efficient in separation of dry matter, phosphorous and organic nitrogen compared to a screw press. This means that the choice of separator is important for the amount of dry matter that will be available for pyrolysis. Furthermore, the avoided emissions of especially nitrous oxide will be much higher by using a decanting centrifuge since the nitrogen fraction in the fibre will be higher. Therefore, the choice of separator and efficiency is very important. Further studies of the impact of choice of separation on climate gas emissions in relation to pyrolysis are needed.

2.8.4 Emission from combustion of pyrolysis gas and oil

Combustion of pyrolysis gas and oil can produce nitrogen oxides (NO_x) and polycyclic aromatic hydrocarbons and we expect that this should be controlled by regulations. However, limiting these emissions to lower levels than the permit could have an overall positive effect on the environmental impact of the technology.

2.8.5 Pyrolysis parameter

Usually very few process parameters for biochar production are described in the literature. Temperature are (almost) always mentioned, but temperature cannot stand alone. A better parameter for describing process conditions would be a "severity factor" as used for describing pre-treatment conditions for lignocellulosic biomass. Guizani et al. (2019) has suggested a Heat Treatment Severity Index (HTSI) as a new metric describing the properties of pyrolytic biochar. The HTSI consists of two terms, the temperature in Kelvin and the reactor retention time. The HTSI showed promising results for the test samples in that paper (Guizani et al. 2019), but remains to be tested on other biomass sources and with other pyrolysis systems. The hydrogen-organic carbon molar ratio can be more challenging to establish, but is independent of the biochar production parameters, and is so far a better way to characterize biochar with regard to carbon persistence and C-sink potential (Chapter 6).

2.9 References

Bernal, M. P., Sommer, S. G., Chadwick, D., Qing, C., Li, G. X., & Michel, F. C. (2017). Current Approaches and Future Trends in Compost Quality Criteria for Agronomic, Environmental, and Human Health Benefits. Advances in Agronomy, Vol 144, 144, 143-233. https://doi.org/10.1016/bs.agron.2017.03.002

Bolzonella, D., Fatone, F., Gottardo, M., & Frison, N. (2018). Nutrients recovery from anaerobic digestate of agro-waste: Techno-economic assessment of full scale applications. Journal of Environmental Management, 216, 111-119. https://doi.org/10.1016/j.jenvman.2017.08.026

Brown, R. C. (2003). Biorenewable Resources. Engineering New Products from Agriculture (1st edition). Blackwell Publishing.

Cathcart, A., Smyth, B. M., Lyons, G., Murray, S. T., Rooney, D., & Johnston, C. R. (2021). An economic analysis of anaerobic digestate fuel pellet production: can digestate fuel pellets add value to existing operations? Cleaner Engineering and Technology, 3, 100098. https://doi.org/https://doi.org/10.1016/j.clet.2021.100098

Christensen, B. T., & Schjønning, P. (1987). Nedmuldning af halm. Tidsskrift for Planteavls Specialserie, S1911.

Ea Energianalyse. (2020) SkyClean pyrolyse af halm og nedmuldning af biokul som klimavirkemiddel.

EBC (2022) 'European Biochar Certificate - Guidelines for a Sustainable Production of Biochar.' European Biochar Foundation (EBC), Arbaz, Switzerland. (http://european-biochar.org). Version 10.1 from 10th Jan 2022

Gelardi, D. L., Li, C. Y., & Parikh, S. J. (2019). An emerging environmental concern: Biochar-induced dust emissions and their potentially toxic properties. Science of the Total Environment, 678, 813-820. https://doi.org/10.1016/j.scitotenv.2019.05.007

Guizani, C., Jeguirim, M., Valin, S., Peyrot, M., & Salvador, S. (2019). The Heat Treatment Severity Index: A new metric correlated to the properties of biochars obtained from entrained flow pyrolysis of biomass. Fuel, 244, 61-68. https://doi.org/10.1016/j.fuel.2019.01.170

Hansen, M. N., Henriksen, K., & Sommer, S. G. (2006). Observations of production and emission of greenhouse gases and ammonia during storage of solids separated from pig slurry: Effects of covering. Atmospheric Environment, 40(22), 4172-4181. https://doi.org/10.1016/j.atmosenv.2006.02.013

Hansen, M. N., Sommer, S. G., Hutchings, N., & Sørensen, P. (2008). Emissionsfaktorer til beregning af ammoniakfordampning ved lagring og udbringning af husdyrgødning. Aarhus Universitet, Det Jordbrugsvidenskabelige Fakultet. DJF Husdyrbrug No. 84

IPCC (2006). 2006 IPCC Guidelines for National Greenhouse Gas Inventories Volume 4 Agriculture, Forestry and Other Land Use. Chapter 11: N_2O Emissions from Managed Soils, and CO_2 Emissions from Lime and Urea Application. https://www.ipcc-nggip.iges.or.jp/public/2006gl/vol4.html

IPCC, 2019: 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Calvo Buendia, E., Tanabe, K., Kranjc, A., Baasansuren, J., Fukuda, M., Ngarize, S., Osako, A., Pyrozhenko, Y., Shermanau, P. and Federici, S. (eds). Published: IPCC, Switzerland. https://www.ipccnggip.iges.or.jp/public/2019rf/index.html (26-01-22)

IPCC (2021). Forster, P., T. Storelvmo, K. Armour, W. Collins, J. L. Dufresne, D. Frame, D. J. Lunt, T. Mauritsen, M. D. 38 Palmer, M. Watanabe, M. Wild, H. Zhang, 2021, The Earth's Energy Budget, Climate Feedbacks, and Climate Sensitivity. In: Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Masson-Delmotte, V., P. Zhai, A. Pirani, S. L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M. I. Gomis, M. Huang, K.Leitzell, E. Lonnoy, J.B.R. Matthews, T. K. Maycock, T. Waterfield, O. Yelekçi, R. Yu and B. Zhou (eds.)]. Cambridge University Press. In Press.

Jensen, J. L., Thers, H., & Elsgaard, L., (2022). Afklaring om videns- og ressourcebehov ved at integrere biokul i C-TOOL modellen til brug for emissionsopgørelser, Nr. 2022-0341679, 10 s., maj 17, 2022

Laghari, M., Muller-Stover, D. S., Puig-Arnavat, M., Thomsen, T. P., & Henriksen, U. B. (2021). Evaluation of Biochar Post-Process Treatments to Produce Soil Enhancers and Phosphorus Fertilizers at a Single Plant. Waste and Biomass Valorization, 12(10), 5517-5532. https://doi.org/10.1007/s12649-021-01358-5

Larsen, J. D., ten Hoeve, M., Nielsen, S., & Scheutz, C. (2018). Life cycle assessment comparing the treatment of surplus activated sludge in a sludge treatment reed bed system with mechanical treatment on centrifuge. Journal of Cleaner Production, 185, 148-156. https://doi.org/10.1016/j.jclepro.2018.02.193

Madej, J., Hilber, I., Bucheli, T. D., & Oleszczuk, P. (2016). Biochars with low polycyclic aromatic hydrocarbon concentrations achievable by pyrolysis under high carrier gas flows irrespective of oxygen content or feedstock. Journal of Analytical and Applied Pyrolysis, 122, 365-369. https://doi.org/10.1016/j.jaap.2016.09.005

Meers E., Velthof G., Michels E. & Rietra R. (2020). Biorefinery of Inorganics: Recovering Mineral Nutrients from Biomass and Organic Waste, First Edition. Edited by. © 2020 John Wiley & Sons Ltd. Published 2020 by John Wiley & Sons Ltd.

Moller, H.B., Lund, I., Sommer, S.B. (2000). Solid-liquid separation of livestock slurry: efficiency and cost. Bioresour. Technol. 74, 223–229. https://doi.org/10.1007/ s00595-015-1112-8.

Moller, H. B., Sorensen, P., Olesen, J. E., Petersen, S. O., Nyord, T., & Sommer, S. G. (2022). Agricultural Biogas Production-Climate and Environmental Impacts. Sustainability, 14(3). https://doi.org/ARTN 184910.3390/su14031849

Møller, H. B., Nielsen, A., Murto, M. & Christensson, K. (2008). Manure and energy crops for biogas production. Nordic Council of Ministers TemaNord, ISSN 0908-6692; 2008:544.

Møller, H. B., Romio, C., Rasmussen, M. D., Jensen, L. S., Schwartzkopff, M., Holm-Nielsen, J. B., Agdal, E. B. L., Larsen, S. U. & Hinge, J. (2021). HALM TIL BIOGAS – Erfaringer og klimaaspekter. Teknologisk Institut. Rapport udarbejdet for INBIOM – Innovationsnetværket for Bioressourcer.

Nielsen, O-K., (2022). Indregning af Carbon Capture and Storage, Carbon Capture and Use og biokul i de nationale emissionsopgørelser, 23 s., Notat fra DCE - Nationalt Center for Miljø og Energi Bind 2022 Nr. 15 https://dce.au.dk/fileadmin/dce.au.dk/Udgivelser/Notater_2022/N2022_15.pdf

Odsherred Kommune. (2021). Environmental approval of pyrolysis plant. (in Danish). Ændring af anlæg til pyrolyseanlæg. https://odsherred.dk/sites/default/files/PDF/miljoegodkendelse.pdf

Olesen, J. E., Møller, H. B., Petersen, S. O., Sørensen, P., Nyord, T., & Sommer, S. G. (2021). Sustainable biogas - climate and environmental effects of biogas production. DCA - Nationalt Center for Fødevarer og Jordbrug. DCA rapport Nr. 181 https://dcapub.au.dk/djfpublikation/index.asp?action=show&id=1463

Ok, Y.S., Ucimija, S.M, Chang, S.X., Bolan, N. (Eds). (2020). Biochar - Production, Characterization and Application. CRC Press.

Prins W, Dahmen N, eds. Processes for thermochemical conversion of biomass. 10th European Conference on Industrial Furnaces and Boilers; 2015; Porto Gaia, Portugal

Ronsse, F., van Hecke, S., Dickinson, D., & Prins, W. (2013). Production and characterization of slow pyrolysis biochar: influence of feedstock type and pyrolysis conditions. Global Change Biology Bioenergy, 5(2), 104-115. https://doi.org/10.1111/gcbb.12018

Phounglamcheik, A., Johnson, N., Kienzl, N., Strasser, C., & Umeki, K. (2022). Self-Heating of Biochar during Postproduction Storage by O-2 Chemisorption at Low Temperatures. Energies, 15(1). https://doi.org/ARTN 38010.3390/en15010380

Poulsen, H. D., Møller, H. B., Klinglmair, M., & Thomsen, M. (2019). Husdyrs fosforudnyttelse og fosfors værdikæde fra husdyrgødning, bioaffald og spildevand: Faglig baggrundsrapport for fosforvidensyntese. Aarhus Universitet, DCE – Nationalt Center for Miljø og Energi, 82 s. - Videnskabelig rapport nr. 325. Aarhus University, DCE - Danish Centre for Environment and Energy. Videnskabelig rapport fra DCE - Nationalt Center for Miljø og Energi Nr. 325

Stiesdal SkyClean A/S (2022). Personal communication with development engineer Giacomo Butera.

Thomsen, I. K., Olesen, J. E., Møller, H. B., Sørensen, P., & Christensen, B. T. (2013). Carbon dynamics and retention in soil after anaerobic digestion of dairy cattle feed and faeces. Soil Biology and Biochemistry, 58, 82–87. https://doi.org/10.1016/j.soilbio.2012.11.006

Thomsen, T. P. (2018). Drying and pyrolysis of sludge – in a climate perspective. AquaGreen PCP phase 2, WP 5: Carbon Footprint Analysis of AquaGreen system

Willen, A., Jonsson, H., Pell, M., & Rodhe, L. (2016). Emissions of Nitrous Oxide, Methane and Ammonia After Field Application of Digested and Dewatered Sewage Sludge With or Without Addition of Urea. Waste and Biomass Valorization, 7(2), 281-292. https://doi.org/10.1007/s12649-015-9456-2

Willen, A., Junestedt, C., Rodhe, L., Pell, M., & Jonsson, H. (2017). Sewage sludge as fertilizer - environmental assessment of storage and land application options. Water Science and Technology, 75(5), 1034-1050. https://doi.org/10.2166/wst.2016.584

Woolf, D., Lehmann, J., Ogle, S., Kishimoto-Mo, A. W., McConkey, B., & Baldock, J. (2021). Greenhouse Gas Inventory Model for Biochar Additions to Soil. Environmental Science & Technology, 55(21), 14795-14805. https://doi.org/10.1021/acs.est.1c02425

3 Biomass potentials

Uffe Jørgensen and Esben Øster Mortensen (reviewer, Mathias N. Andersen)

This chapter gives an overview of the currently available biomass resource from agriculture that may be used for pyrolysis, and the potential increase of this resource in 2030 based on different land-use scenarios. The focus of this analysis is on three types of feedstock for biochar production: straw, biogas fibre, and sewage sludge. These biomasses are already to some extent treated industrially for direct combustion, biogas production and as fertilizer to agricultural fields. This means that there are experiences on logistics, and that the centralized location (especially for biogas fibre and sewage sludge) may reduce further transport costs if a pyrolysis plant is co-located at the biogas or sewage water treatment plant. However, this also means that there are competing technology interests in using these types of biomass as a stable and abundant feedstock. One example is Vordingborg Biofuel that proposes to use 300.000 tonne (Mg) of straw for methanol production annually by 2025 (www.vordingborgbiofuel.dk). Straw may also be used for biogas production, after which the remaining fibre can be separated and used for biochar. In order to reduce risks of double-counting, we have kept the three resources separate in the estimation of resources now and in the future.

3.1 Current biomass resources

3.1.1 Straw

Straw is a large resource from arable farming, as the main part of the agricultural landscape is used for grain crops at present. A significant fraction of the straw is already used for energy production (30% for cereal straw, 15% for rape seed straw, Table 3.1), which has historically been for farm and local district heating purposes, but later also for centralized combined heat and power, and now with initial uses for boosting biogas production.

Annual variation in the total production of straw can be large; in the very dry year 2018, the estimated total production was only 4.00 Mt (million tonne) for cereal straw and 0.44 Mt for rapeseed straw. However, in the more productive season 2017, the total production was estimated at 5.72 and 0.67 Mt, respectively, for cereal and rapeseed straw. The drought in 2018 caused a lack of straw resources, not only in Denmark, and thus the “not harvested” fraction was reduced to 0.60 for cereal straw and to 0.22 Mt for rapeseed straw. It may therefore be estimated, that with current practices, where farmers have different priorities, approx. 0.82 Mt of straw (cereal and rape seed) will always be left on the field. An estimation of the current additional straw availability for industrial applications (energy, biochar or material) may be based on the minimum use for other purposes in 2018 (not harvested), see Table 3.1. For cereal straw the estimated mean availability is then 2.20-0.60 = 1.60 Mt, and for rape seed straw it is 0.49 – 0.22 = 0.27 Mt. Apart from these amounts, the currently used resource for energy purposes (i.e., 1.56 Mt cereal straw and 0.06 Mt rape seed straw) may be diverted into pyrolysis if the environmental, climatic, agronomic and economic conditions favour such a shift. By adding the not harvested fraction and the energy use, the total resource available for pyrolysis is 3.16 Mt of cereal straw and 0.33 Mt of straw from rapeseed. These numbers are with 15% water content and converts to 2.69 Mt of dry matter cereal straw and 0.28 Mt of dry matter rapeseed straw. Grass seed straw is another abundant resource, which is however not reported by Statistics Denmark. Mortensen & Jørgensen (2022a) estimate a resource of 0.40 Mt of dry matter seed straw, which together with grain and rapeseed straw gives a total potential of 2.69 + 0.28 + 0.40 = 3.37 Mt of dry matter straw for industrial conversion.

Table 3.1. Total amount of straw and its uses as a mean of 2016-20 (www.statistikbanken.dk, visited 15/3 2022). Mt, million tonne (1000 Mg).

Straw type	Category	Mean 2016, 2017, 2019 and 2020 Mt straw (at 15% water content)	2018 Mt straw (at 15% water content)
Cereal straw	Total	5.46	4.00
	Energy use	1.56	1.43
	Feed use	0.83	0.86
	Animal bedding	0.86	1.11
	Not harvested	2.20	0.60
Rape seed straw	Total	0.57	0.44
	Energy use	0.06	0.17
	Feed use	0.01	0.01
	Animal bedding	0.02	0.04
	Not harvested	0.49	0.22

The above amounts are calculated from general statistics, and with current harvest techniques. Studies, where all non-grain material is harvested (also spikes, awns, smaller leaves), have shown that this can provide as much or even more biomass than in the grain (Andersen et al. 1992; Jørgensen et al. 2007). This means that almost twice the amount of the total shown in Table 3.1 may potentially be available for harvest. Practical trials with adapted or new harvest equipment have shown that between 12-30% increases in straw collection may be obtained (Kristensen, 2012). However, this has so far not been of economic interest.

Removal of straw from the field reduces the content of soil organic carbon (Olesen et al. 2018). At some soil types, especially where the clay content is high relative to the carbon content, this reduction is detrimental to soil quality as soil physical parameters will become less favourable for creating a proper soil structure that supports plant growth, root development, and oxygen and water transport in the soil matrix (Jensen et al. 2019). On top of this, especially organic farmers are unwilling to export the nutrient content of the straw from the farm, and farmers in favour of Conservation Agriculture are using straw to cover the soil surface and increase soil microbial biomass (Munkholm et al. 2020). It may be that returning biochar can provide similar soil quality support (see Chapter 4), which may release a higher share of straw for pyrolysis if farmers believe in the benefits of biochar.

3.1.2 Biogas fibre

The potential amount of biogas fibre is difficult to assess; it depends on the current level of animal manure used for biogas, which is increasing rapidly, and on the fraction of the digestate that is separated into a liquid and a fibre fraction. Aarhus University data extracted from farmer fertilizer accounts for 2018 indicated that 14.2% of the manure utilized on fields was biogas digestate. According to the Energy Statistics (Energistyrelsen, 2021) total biogas production increased from 13.3 PJ (Peta Joule = 10^{15} Joule) in 2018 to 21.4 PJ in 2020, which suggests that 22.8% of animal manure was used for biogas in 2020, if a linear relation between energy production and manure use is assumed. We therefore anticipate that by 2022 roughly 25% of animal manure is used for biogas.

Based on farmers fertilizer accounts from 2016-2018 and typical relations between manure N-content and dry matter contents (Mortensen & Jørgensen 2022b), we estimated a total manure dry matter mass of 4.044 Mt, which is quite stable over short time. We then anticipate that 25% (1,011 Mt) is used for biogas for 2022, and that manure is added with approx. 25% other feedstock, which gives in total 1.264 Mt biogas mix. Anticipating 20% ash content in the biogas mix, and that 40% of organic matter is converted into biogas means that 0.607 Mt organic matter is left. Fibre fractionation with a decanter centrifuge is anticipated to extract 60% (0.364 Mt) of organic matter in a fibre fraction with 20% ash content (Henrik B. Møller, personal communication). Including the ash content, this means that in total 0.455 Mt dry matter can be available for pyrolysis, if all biogas digestate (as of 2022) will be separated.

3.1.3 Sewage sludge

The amount of sewage sludge produced in Denmark is relatively stable, varying between 107,000 and 121,000 tonne of dry matter between 2015 and 2019 (Table 3.2). The data on the final use of sewage sludge have been collected from different sources by the Danish Environmental Protection Agency (Miljøstyrelsen, 2020) and include substantial uncertainty, e.g., related to variations in dry matter contents of the materials. A more in-depth analysis of the sewage sludge resource in 2018, using data directly from the sewage plants, indicated that the amount was 140,000 tonne of dry matter as compared to the 107,000 tonne in Table 3.2 (Miljøstyrelsen, 2020). In this 2018 analysis, 91,000 tonne were applied to agricultural land, 15,000 tonne were composted, and 34,000 tonne of dry matter sludge were combusted or disposed of. This indicates that the total amount of sewage sludge is >30% higher than indicated from available statistics with especially the agricultural land application and combustion/landfill categories being too low.

Table 3.2. Total production of sewage sludge in kilotonne (kt, 1000 Mg) of dry matter in Denmark from 2015-2019 and its distribution for different uses (Miljøstyrelsen, 2020). Yearly data also shown in percent (%).

Category	2015		2016		2017		2018		2019	
	kt	(%)	kt	(%)	kt	(%)	kt	(%)	kt	(%)
Utilization at agricultural land	86	72	85	70	77	65	71	67	88	78
Composting and other reutilization	10	8	12	10	22	19	22	21	9	8
Combustion	22	19	23	19	18	16	13	12	16	14
Depositing	1	1	1	1	1	0	1	0	0	0
Total	119	100	121	100	113	100	107	100	113	100

The variation in the amount used for application to agricultural land (65-78%) is judged to be an artefact caused by the uncertainty of the statistics (Miljøstyrelsen, 2020). In general, the level is lower than it was in the 1990s, where approx. 80% of sludge was used as fertilizer for agricultural land. However, from 1997-2000, the regulation on contents of organic contaminants such as PAHs, LAS, NPEs and DEHP became stricter and caused an increase in the amount of sludge directed for combustion (Miljøstyrelsen, 2020). Lately, focus has increased on the group of PFAS-compounds and limits for their contents in sewage sludge were circulated to Danish municipalities in 2021 (Miljøstyrelsen, 2021). It is possible, that these different organic substances can be decomposed by pyrolysis, although documentation for specific pyrolysis conditions are at an early stage. This makes the resource of 13,000 to 23,000 tonne (or possibly 30% higher) of dry matter, that is combusted today, even more interesting for pyrolysis, if it will be possible to compete economically with combustion. It will be important, though, to avoid that the concentration of heavy metals will increase to levels above legal limits after pyrolysis, as the metals are not decomposed by pyrolysis. However, some heavy metals may evaporate or follow the oil phase, which is a topic that calls for further investigation and documentation.

3.2 Future biomass resources

To our knowledge, large changes in future sewage sludge total amounts are not expected. However, there may well be changes in which uses are preferred or allowed by legislation, taking into account, e.g., new problematic compounds detected in the sludge such as PFAS.

Land use in agriculture may change significantly over time due to, e.g., productivity improvements, market conditions, and environmental and climate legislation. The same can happen with livestock production for similar reasons, but in addition also for health and sanitary reasons, exemplified by the recent removal of the mink sector. Aarhus University and University of Copenhagen are about to finalize a project on scenarios for future biomass resources in Denmark. Three main scenarios – Business As Usual (BAU), Biomass, and Extensification – are defined (Mortensen & Jørgensen, 2022a,b). The two latter ones are combined with ±20% increase/reduction in animal production in Denmark creating in total seven scenarios (Table 3.3). In both the Biomass and Extensification scenarios, significant changes in land use are implemented in order to contribute to fulfilling environmental and climate and biodiversity demands for the agricultural sector, which are otherwise difficult to reach. The major land use changes included are rewetting of organic soils, and a substantially increased grassland area with perennial grass-clover production for biorefining. In the extensification scenario significant land is taken out of production for establishing nature areas. In the stables, improved handling of manure is implemented (Adamsen et al. 2021) with an expected 7.5% increase in manure resources due to lower gaseous and energy losses before storage. The new grassland areas are expected to deliver feedstock for green biorefineries that can also utilize sugar beet leaves (Jørgensen et al. 2021). The product output in focus is a protein concentrate that can substitute imported soy products, while the main output by volume from green biorefineries is a fibre fraction that can substitute other roughage crops for ruminants or it can be used for biogas, packaging, textiles, etc. It may, however, also be used for biochar production as investigated in the ongoing "Grass Biochar" GUDP project.

Table 3.3. Future agricultural biomass resources (Mt DM, million tonne of dry matter) in seven scenarios for 2030 compared with their mean use for bioenergy in 2015-2019 (based on Mortensen & Jørgensen, 2022b). BAU, Business As Usual; Ext, Extensification. The scenarios Biomass and Ext are also shown for the assumption of 20% increase/reduction in animal production in Denmark (i.e., -20% and +20%).

Biomass type	Bioenergy use 2015-2019	BAU	Biomass	Ext	Biomass -20%	Ext -20%	Biomass +20%	Ext +20%
Straw (cereals, rape, grass seed)	1.49	3.55	3.40	3.54	3.71	3.85	3.09	3.23
Green biomass fibres	0.00	0.00	5.79	3.50	8.32	3.50	4.70	2.41
Woody biomass on farmland	0.09	0.09	0.09	0.09	0.09	0.09	0.09	0.09
Animal manure	0.47	2.95	3.17	3.17	2.54	2.54	3.81	3.81
Mt of dry matter in total	2.04	6.58	12.45	10.30	14.66	9.97	11.68	9.53

The available straw resources by 2030 in the scenarios – taking into account other uses for animal feed and bedding etc., as well as a 13% "unusable" fraction – varies between 3.1 and 3.9 Mt of dry matter. This should be seen as maximal achievable amounts that can only be achieved if new adaptations (e.g., use of cereal varieties with 15% higher straw production and 15% higher straw pick-up by altered harvesting technique) to support the production is applied (Mortensen & Jørgensen, 2022b). Comparing the scenario results with the currently available resource of 3.37 Mt of dry matter straw today (see section 3.1.1) shows that it may be possible to sustain, or slightly increase, the available straw resource over time even with significant changes in the composition of the agricultural land use. It is likely, however, to require significant adaptation in variety selection, harvest technology etc. to achieve this.

For animal manure in the scenarios for 2030, we have applied the same conversion and separation factors as for the current situation in section 3.1.2 (Table 3.4). We anticipate that also in the future, manure is added with additional 25% other feedstock; this could be straw in which case the total potential resource for pyrolysis is mixed up with the direct straw resource available for pyrolysis. However, it may also be grass fibre that supplements the manure resource, of which there is plenty to fulfil the 25% addition in the scenarios (see Table 3.3). The future estimates in Table 3.4, which anticipates maximal technical biogas use of manure, are approximately three times higher than the current potential estimated in section 3.1.2. However, the potential is slightly less if animal production is reduced by 20%, and slightly increased if animal production is increased.

Table 3.4. Estimated future resources of biogas fibre for 2030-scenarios (Mt DM, million tonne dry matter) in case of technical maximal use of manure for biogas (90% use for slurry types and 50% use for deep litter, Mortensen & Jørgensen, 2022b) and similar biogas conversion and fibre separation efficiency as today. BAU, Business As Usual; Ext, Extensification. The scenarios Biomass and Ext are also shown for the assumption of 20% increase/reduction in animal production in Denmark (i.e., -20% and +20%).

	BAU	Biomass	Ext	Biomass -20%	Ext -20%	Biomass +20%	Ext +20%
Biogas fibre available in 2030 (Mt DM)	1.33	1.43	1.43	1.14	1.14	1.71	1.71

3.3 References

Adamsen APS, Hansen MJ, Møller HB, 2021. Effekt af hyppig udslusning af gylle på metanproduktion, Policy Note Aarhus University, DCA No. 2020-0166155, 9 pp.

Andersen MN, Jensen CR, Lösch R, 1992. The interaction effects of potassium and drought in field-grown barley. I. Yield, water-use efficiency and growth. Acta Agriculturae Scandinavica, Sect B 42, 34-44.

Energistyrelsen, 2021. Energistatistik 2020. ISSN 0906-4699. https://ens.dk/sites/ens.dk/files/Statistik/energistatistik2020.pdf

Jensen JL, Schjønning P, Watts CW, Christensen BT, Peltre C, Munkholm LJ, 2019. Relating soil C and organic matter fractions to soil structural stability. Geoderma 337, 834-843.

Jørgensen JR, Deleuran LC, Wollenweber B, 2007. Prospects of whole grain crops of wheat, rye and triticale under different fertilizer regimes for energy production. Biomass & Bioenergy 31, 308-317.

Jørgensen U, Kristensen T, Jørgensen JR, Kongsted AG, De Notaris C, Nielsen C, Mortensen EØ, Ambye-Jensen M, Jensen SK, Stødkilde-Jørgensen L, Dalsgaard TK, Møller AH, Grøn Sørensen,C, Asp T, Lehmann Olsen F, Gylling M, 2021. Green biorefining of grassland biomass. DCA - Nationalt Center for Fødevarer og Jordbrug. DCA rapport Nr. 193. https://dcapub.au.dk/djfpublikation/index.asp?action=show&id=1480

Kristensen EF, 2012. Tekniske muligheder for at bjerge en større del af den producerede halmmængde. Baggrundsnotat til +10 mio. tons planen. https://ifro.ku.dk/publikationer/saerlige_ifro-udgivelser/10miotons/

Miljøstyrelsen, 2020. Affaldsstatistik 2019. Miljøprojekt nr. 2152, Miljøstyrelsen, Miljø- og Fødevareministeriet, 80 pp.

Miljøstyrelsen, 2021. PFAS i spildevandsslam – en foreløbig vejledende grænseværdi. Letter for all Danish Municipalities, Miljøstyrelsen. https://mst.dk/media/228003/bilag-1-vejledende-graensevaerdi-for-pfas-i-spildevandsslam.pdf.

Mortensen EØ, Jørgensen U, 2022a. Danish agricultural biomass production and utilization in 2030, Advisory memorandum from DCA, 24 s.

Mortensen EØ, Jørgensen U, 2022b. Forudsætninger for og beregninger af 2030 scenarier for arealanvendelse og biomasseproduktionen i landbruget. Rådgivningsrapport fra DCA - Nationalt Center for Fødevarer og Jordbrug, 38 s.

Munkholm LJ, Hansen EM, Melander B, Kudsk P, Jørgensen LN, Heckrath GJ, Ravnskov S, Axelsen JA, 2020. Vidensyntese om Conservation Agriculture. DCA - Nationalt Center for Fødevarer og Jordbrug. DCA rapport Nr. 177. https://dcapub.au.dk/djfpublikation/index.asp?action=show&id=1455

Olesen JE, Petersen SO, Lund P, Jørgensen U, Kristensen T, Elsgaard L, Sørensen P, Lassen J. 2018. Virkemidler til reduktion af klimagasser i landbruget. DCA - Nationalt Center for Fødevarer og Jordbrug, DCA rapport Nr. 130 http://web.agrsci.dk/djfpublikation/index.asp?action=show&id=1273

4 Effect of biochar on soil physical and chemical properties

Emmanuel Arthur and Mathias N. Andersen (reviewer, Anne Winding)

Biochar represents a stable carbon form that can be added to agricultural soils and contribute to carbon sequestration as described in the preceding chapters and in Chapter 6. However, in addition to carbon sequestration, biochar amendment has been reported to contribute to changes, and often improvements, in soil physical and chemical properties. However, these changes depend on the biochar feedstock source and temperature used in pyrolysis. Likewise, the resulting changes may be dependent on the interaction between soil type and biochar properties. In the following sections an introduction is given to the importance of soil physical and chemical properties and how biochar may interact with these properties.

4.1 Soil physical properties

4.1.1 Introduction and methodological approach

Soil physical properties reflect how strong or flexible the soil components (solid, liquid and gas) relate to each other. Soil bulk density describes how much of the solid component is packed within a given soil volume and is directly related to the soil structure. High bulk density is linked to a strong, compact, or dense structure, and low bulk density is related to a weak or loose structure. A good soil structure that is moderately compact is crucial for soil functions, such as support for plant growth and storage and distribution of air and water. Conversely, a compact soil with a very high density is inimical to crop growth due to germination inhibition, restricted root growth, and waterlogging risks that cause anaerobic conditions. The retention of water in soils under a given set of environmental conditions reflects how big or small the soil pores are, and how they are distributed in the soil matrix. An important aspect of soil water retention (or water holding capacity) is the concept of plant-available water, defined as the amount of water that is available for use by crops. An increase in soil water retention does not necessarily mean higher plant available water (AW) content. This is because the available water content is the difference between the water content after drainage (i.e., at field capacity, FC) and at the permanent wilting point (WP). Therefore, when an amendment or management practice increases water retention at both FC and WP proportionally, AW will be unaffected. On the other hand, any intervention that improves soil structural pores, and consequently increases water content at FC, but has no effect on WP, causes an increase in AW.

The effect of biochar on soil physical properties was in the present chapter assessed by a meta-analysis of 31 relevant papers (listed under Table 4.1). The papers report on 67 paired comparisons that were included in the meta-analysis. Most experiments were conducted in the laboratory or greenhouse (84%), while only 16% of them represented field experiments. The analyses also considered only top soils or the ploughed layer. The soils included all soil types: coarse-textured (sands and loamy sands), medium-textured (sandy loam, loam, silty loam, sandy clay loam), and fine-textured (clay loam and clay) soils. The three categories represented, respectively, 34, 55, and 10% of the considered studies. The organic carbon content of the soils ranged from 0.08% to 10.8% (average of 1.90%). The biochar feedstocks included (i) straw derived from maize, wheat, miscanthus, switchgrass and rice, (ii) animal manure, and (iii) wastewater sewage sludge. The proportion of each biochar type in the studies is shown in Table 4.1.

Biochar total carbon content and "added biochar carbon (BCad)" are of relevance to improvements in soil physical properties. The concept BCad was introduced due to the differences in the carbon content of the biochars applied and is estimated as suggested by Razzaghi et al. (2020):

$$BC_{ad}\ (\%) = C_b \times A_{rate} \qquad \text{(Eq. 4.1)}$$

where C_b is the biochar carbon (kg C kg^{-1} dry weight biochar) and A_{rate} is the biochar application rate (weight% on soil dry weight basis). The averages of these biochar properties within the three categories (straw, manure, and sludge) are provided in Table 4.1. The particle sizes of the applied biochars ranged from 0.045 to 2.0 mm, with an average of 1.35 mm, and the application rates ranged from 0.25 to 10% (wt/wt), with an average of 2.5%. The pyrolysis temperature for all biochars, regardless of feedstock, ranged from 300 to 750°C.

Published studies that included other biochar feedstocks (e.g., wood) or soil types that are mineralogically different from Danish soils (e.g., well-weathered tropical soils, volcanic soils or heavy clays such as Vertisols) were not included in the meta-analyses.

Table 4.1 Median values of the pyrolysis temperature and carbon contents of the applied biochar[a]. Numbers in square brackets represent the minimum and maximum values of the variable.

Biochar type	Pyrolysis Temperature (°C)	Total carbon (%)	Biochar added carbon (%)
Straw (86.4%)[b]	512 [300 – 750]	63.3 [20.5 – 85.8]	1.16 [0.09 – 7.17]
Manure (9.1%)[b]	450 [300 – 750]	47.2 [19.0 – 74.9]	1.12 [0.37 – 1.97]
Sludge (4.5%)[b]	600 [550 – 650]	26.2 [22.3 – 47.7]	0.95 [0.07 – 1.00]

[a] Included references are: Abel et al. (2013); Herath et al. (2013); Lei & Zhang (2013); Alburquerque et al. (2014); Hansen et al. (2015); Mollinedo et al. (2015); Ojeda et al. (2015); Burrell et al. (2016); Esmaeelnejad et al. (2016); Gamage et al. (2016); Glab et al. (2016); Hansen et al. (2016a,b); Jin et al. (2016); Liu et al. (2016); Ma et al. (2016); Petersen et al. (2016); Aller et al. (2017); Arthur & Ahmed (2017); Hansen et al. (2017); Kelly et al. (2017); Moragues-Saitua et al. (2017); Bornø et al. (2018a,b); Gunal et al. (2018); Mohan et al. (2018); Zong et al. (2018); Fu et al. (2019); Thers et al. (2020); Fu et al. (2021); Wang et al. (2021); Kassaye et al. (2022).
[b] Fraction of studies that included the feedstock.

4.1.2 Bulk density

Disregarding the soil or biochar type, soil bulk density decreases by 9.7% on average after applying biochar (Figure 4.1). This is similar to an earlier global review by Razzaghi et al. (2020) that also reported an average decrease of 9% in bulk density after biochar addition. This decrease in bulk density after biochar addition was highest for medium-textured soils (–13.5%), compared to fine-textured clays (–5.3%) or coarse sands (–6.7%).

Added biochar carbon at rates of up to 2% caused a similar decrease (5-10%) in bulk density, whereas higher rates of ca. 2.5% caused a decrease of 20-30% (Fig 4.2a). High soil organic carbon content affects how biochar affected bulk density. The addition of miscanthus biochar (BC_{ad} = 0.81%) to sandy loam-textured soil with 10.8% organic carbon content led to a 7% increase in bulk density (Figure 4.2a).

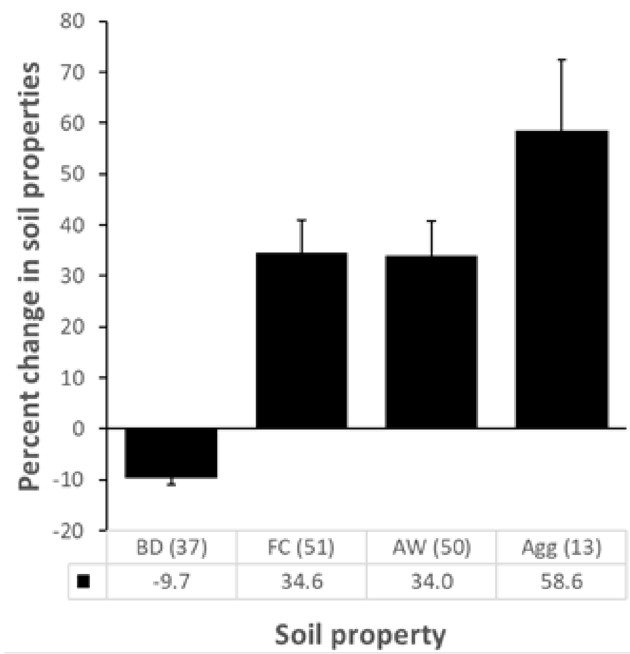

Figure 4.1. Average percent changes in soil physical properties after addition of biochar. BD: bulk density, FC: soil water content at field capacity or water holding capacity, AW: plant available water content, Agg: soil aggregation indicator. The number of studies for each soil property is indicated in parentheses. Error bars represent standard errors.

The reduction in bulk density following biochar application can be attributed to the (i) lower density (<0.6 g cm^{-3}) and higher porosity of biochar compared to soil mineral particles, and (ii) promotion of biological activity and aggregation and increased macroporosity (Blanco-Canqui 2021).

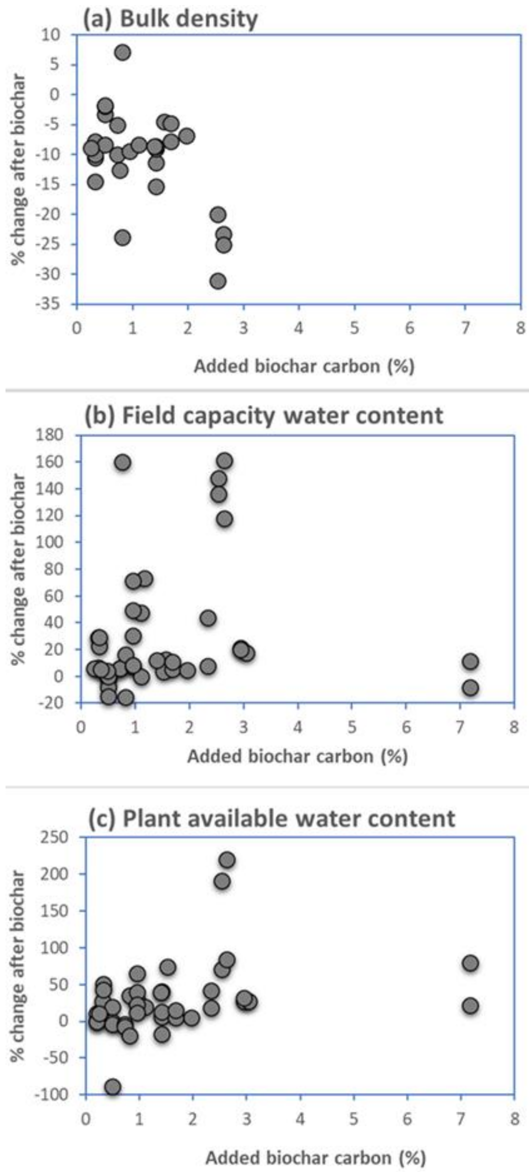

Figure 4.2. Percent changes in (a) bulk density, (b) soil water content at field capacity and (c) plant available water content with added biochar carbon.

4.1.3 Soil water retention

The retention and availability of water in soils under specific environmental conditions reflect the size of the soil pores, and how they are distributed in the soil matrix. Two concepts of water retention, as defined in section 4.1.1, are (i) field capacity (FC), which defines the water that remains in the soil after excess water has drained off by gravity, and (ii) plant available water (AW), which is the fraction of water plants can access at a given point. The fraction of field capacity that is inaccessible to plants is often termed the water content at wilting point and is the fraction of water that is tightly held by the soil matrix. Therefore, an amendment that improves soil water availability to plants is the one that increases the FC and AW content at the same time. Adding biochar to soil leads to an approximately 35% increase in both the water held at FC and the water available to plants. This effect of biochar on soil water retention and availability is dependent on soil type and the BC_{ad}. In general, there was a weak positive correlation between BC_{ad} and the changes in FC (Figure 4.2b). The relationship was stronger for BC_{ad} and AW (Figure 4.2c). The relationship between BC_{ad} and water retention is indirect (Sun et al. 2013). Increased soil organic C from biochar addition leads to the creation of new aggregates and the increased stability of existing aggregates. These two processes potentially increase water retention.

Table 4.2. Effect of biochar on water contents (FC, field capacity; AW, plant available water) in different soil types

Soil type[a]	BC_{ad} (%)[b]	ΔFC (%)[c]	ΔAW (%)[d]
Sandy	1.3	69.7	56.2
Loamy	1.2	17.0	19.1
Clayey	2.8	2.9	23.3

[a] Sandy = sand + loamy sand; Loamy = loam, sandy loam, sandy clay loam, and silt loam; Clayey = clay + clay loam
[b] BC_{ad} = added biochar carbon
[c] ΔFC = change in water content at field capacity
[d] ΔAW = change in plant available water content.

According to Table 4.2, the soil type strongly influenced the biochar effect on the water retention variables. For the studies included in this synthesis, biochar-induced increases in FC ranged from 2.9% in fine-textured soils to 69.7% in coarse-textured soils. Similarly, global reviews (Edeh et al. 2020; Razzaghi et al. 2020) have reported that biochar increases FC by 20.4% and AW by 28.5% on average for all soil types together. Furthermore, AW in the present analysis increased by 19.1-56.2% across the soil types with highest increase for the coarse-textured soils, which agree with previous reports (Razzaghi et al. 2020). Thus, for soil water retention, coarse-textured soils benefit more from biochar addition compared to fine-textured soils.

Other biochar properties that are relevant to soil water retention are biochar surface area, surface chemistry and hydrophobicity. Pyrolysis conditions that produce biochar with high surface area, acidic and oxygenated functional groups and hydrophilic biochar are optimal for soil water retention improvements (Edeh et al. 2020).

4.1.4 Soil aggregation

Soil aggregation describes how the solid components (clay, silt, sand, and organic matter particles) are combined and interact with each other. Soil aggregation is crucial for soil functions such as stabilization and decomposition of organic matter, water and airflow and storage – which are all necessary for plant growth. Poorly aggregated soils easily break down when exposed to water and have poor water holding capacity, whereas well-aggregated soils are resistant to breakdown, and can combat climate change by long-term

C sequestration (Zhao et al. 2018). Evaluating the stability of soil aggregates is often done by immersing the aggregates in water and quantifying the remaining aggregates after the test. The effect of biochar on soil aggregation was quantified by changes in both the fraction of aggregates and the mean weight diameter after wet sieving. Biochar addition led to an average increase of 58.6% in soil aggregate stability. However, large variation in this average outcome is obvious in the standard error of the biochar effect (Figure 4.1) and may be attributed partly to the small number of studies. Increases in aggregation arising from biochar application can be attributed to enhanced soil organic C and microbial activities, both of which are necessary for binding soil aggregates.

For the same biochar it is also possible that effects on aggregate stability are controlled by the recipient soil type. For example, Moragues-Saitua et al. (2017) found that miscanthus biochar addition reduced aggregate stability by 7% in loam soil, while it increased aggregation by 19% in sandy loam soil.

A recent meta-analysis (Ul Islam et al. 2021) reported that biochar application significantly improved soil aggregation by 16.4 %, regardless of biochar/experimental/soil conditions. The study also found that biochar strengthened soil aggregation in the loam-textured soils (19.9%) relative to sandy soils (13.4%). The limited data available for this synthesis precluded the comparison of the biochar effect on aggregation for different soil textures. Aside from the soil type, other soil properties such as the initial organic C contents, the ratio of clay to organic C, and other biochar properties such as electrical conductivity determine how biochar may affect aggregation (Khademalrasoul et al. 2014; Burrell et al. 2016).

4.1.5 Saturated hydraulic conductivity and soil wettability

The saturated hydraulic conductivity (K_s) indicates how a soil conducts water under saturation conditions. This is essential for soil functions such as infiltration, groundwater recharge and nutrient transport. The addition of biochar to soil can potentially increase or decrease K_s, depending on the biochar properties or initial K_s of the soil. Herath et al. (2013) found that K_s increased by 40 and 140% when 1% rice straw biochar pyrolysed at 350 and 550°C, respectively, were added to silt loam. Conversely, for sandy loam soil, Esmaeelnejad et al. (2016) reported approximately 14-17% reduction in K_s at 180 days after applying 2% of rice straw biochar, probably due to clogging of soil pores by biochar. In other instances, adding wheat straw biochar to loamy sand or manure biochar to a loam soil had no significant effect on K_s, regardless of pyrolysis temperature or application rates (Lei & Zhang 2013; Glab et al. 2016).

The wettability of soils is crucial, particularly, during the dry seasons where hydrophobicity can impede water adsorption and reduce infiltration. Several studies including sludge and straw biochar show that there is no effect of biochar on the wettability of soils (Abel et al. 2013, Herath et al. 2013, Glab et al. 2016, Petersen et al. 2016). This is probably because the biochar used in the mentioned studies were pyrolysed at temperatures between 300 and 750°C, in a process that resulted in hydrophilic biochar.

4.2 Soil chemical properties

The present section covers the interaction between biochar and soil chemical properties related to pH, cation exchange capacity (CEC) and electrical conductivity (EC) as a measure of soil salinity. Interactions between biochar and the major plant nutrients nitrogen, phosphorous and potassium is described in Chapter 7.

4.2.1 Soil pH and cation exchange capacity

Soil pH and CEC represent two crucial soil properties that influence both nutrient and water availability, and thus the crop uptake of these. Low soil pH (less than 6) can severely restrict the uptake of some nutrients such as phosphorus. Both the composition of the feedstock used for biochar production as well as pyrolysis conditions, especially temperature, influence the alkalinity of the biochar and thus its effect on soil pH (see, e.g., Chapter 2, Figure 2.3). When plant material is heated to 400-800°C, carbonates are formed in the presence of alkali metal ions (K, Na, Ca, Mg) - a process that may be further catalysed in the presence of silicon (Watanabe et al. 2014). In addition, negatively charged oxygen-containing functional aromatic and aliphatic organic groups such as de-protonated carboxyl groups and alcohols are formed (Haider et al. 2022). Pyrolysis temperature in general increases the formation of carbonates (Yuan et al. 2011, El-Naggar et al. 2019) and thus the biochar's ability to increase soil pH. It has been suggested that especially potassium oxide and hydroxide is formed during pyrolysis and react with CO_2 to form carbonate. This would indicate that biochar based on crop residues in general have higher carbonate content and alkalinity than wood-based biochars as the K content in crop residues is often higher. Hansen et al. (2016b) compared straw and wood based biochars and found 3.5 times higher K concentration in the straw based biochar. The pH was 11.6 and 11.1, respectively, and although pH of soil amended with the two biochars was initially equal, the pH of the wood biochar amended soil decreased faster upon incubation. A potential contributing factor to lower alkalinity in wood biochar is the higher lignin content that seems to decrease the amounts of carbonates formed during pyrolysis (Huang et al. 2018). In line with the above reported effects of pyrolysis temperature, Yang et al. (2020) found that the pH of a sandy loam from Foulum was increased more when two biochars produced at a high temperature (700°C) were incorporated in the soil than when biochars from the same feedstocks of either wheat or miscanthus straw but pyrolysed at 550°C were incorporated. Thers et al. (2020) reported no significant difference (<1%) in soil pH after applying 15 Mg/ha of wheat straw and manure fiber biochars to sandy loam. Conversely, Hansen et al. (2015) showed for sandy loam that adding 5% of wheat straw biochar increased soil pH from 7.5 to 8.6 (15.1% increase). While this may seem contradictory, it must be considered that 5% biochar is a much higher amount (around 200 Mg/ha) than 15 Mg/ha. In general changes in soil pH would be predictable based on soil properties and the alkalinity of the biochar. In another study, Nissen et al. (2021) applied high biochar concentrations, far beyond realistic field application rates, to study potential biochar effects on soil chemical and microbial properties. These studies substantiated that biochar based on straw may in general have higher pH than wood-based biochars, but also that the pH in soil after biochar amendment depends on individual biochars and, for some biochars, large application rates are needed to substantially affect soil pH (Figure 4.3).

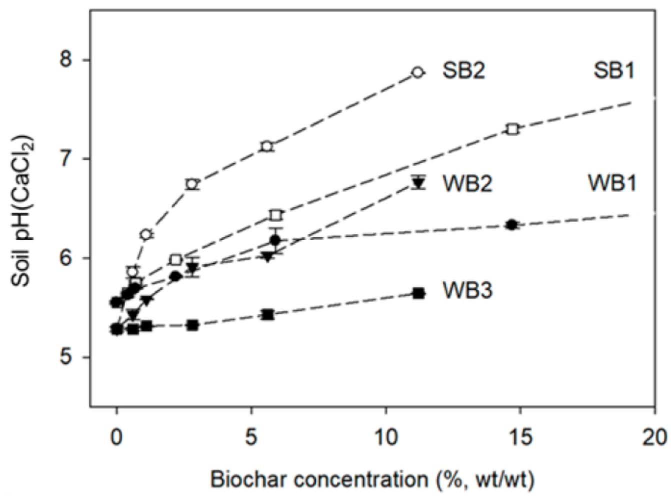

Figure 4.3. Effect of biochar concentration on soil pH after 1 week of soil/biochar interaction. SB1, SB2 - straw biochars; WB1, WB2 - pine wood biochars; WB3 - oak wood biochar. Adapted from Nissen et al. (2021)

Under Danish conditions, surplus precipitation and the supply of fertilizers and manure to fields leads to a continuous decrease in soil pH, which is usually corrected by the application of lime. The extent to which biochar addition can replace this lime input represents an economic value and as well a replacement of a fossil-derived CO_2 emission. However, further studies are needed to document the longevity of the liming effect of biochar, i.e., how many years the mitigation of soil acidification will persist.

The functional groups that are created during pyrolysis are to some extent amphoteric, meaning that depending on the pH of the soil, the surface of the biochar particles may carry a negative or positive electrical charge. This in turn means that biochar contributes to the CEC of the soil. The CEC of soil is a crucial property, which influences the soil's ability to retain positively charged nutrients and other cations. Therefore, the CEC of soils is an indicator of soil fertility because it determines the potential of soil to supply soil nutrients, particularly calcium, magnesium, and potassium. Soil CEC influences nutrient availability, soil pH and soil reaction to added fertilizers. Soil constituents that contribute substantially to CEC are clay content, clay type, and the content of organic matter. CEC is measured in $cmol_c$/kg indicating how many centimol (negative charge) per kg the soil contains. Pure clay may have a CEC of up to 250 $cmol_c\ kg^{-1}$, but the typical value for the commonly found clay minerals in Danish soils is in the range of 10-40 $cmol_c\ kg^{-1}$. Humified organic matter has a very high CEC that may reach 400 $cmol_c\ kg^{-1}$. Danish coarse sandy soils with low clay content have low CEC in the range of 2-8 $cmol_c\ kg^{-1}$ while the clay containing sandy loams have larger CEC in the range of 10-20 $cmol_c\ kg^{-1}$ when both topsoil and subsoil are considered (Nielsen & Møberg 1985). Biochar has been reported to have CEC values in the same range as clay, e.g., 14-17 $cmol_c\ kg^{-1}$ (Kharel et al. 2019), i.e., much lower than that of organic matter. Other studies, however, have found higher values in the range of 100-170 $cmol_c\ kg^{-1}$ (Sarfaraz et al. 2020). The common range for CEC of biochar is from 11 to 120 $cmol_c\ kg^{-1}$ depending on the feedstock, pyrolysis temperatures or other conditions (Munera-Echeverri et al. 2018). For a Danish sandy loam, adding 5% wheat straw biochar (BC_{ad} = 2.42%) resulted in a 30% increase in CEC (Hansen et al. 2016b). Similarly, 1% rice husk biochar added to sand and sandy loam soil increased CEC by 21 and 14%, respectively (Gamage et al. 2016). Confirming the impact of the feedstock, Mohan et al. (2018) reported that for soil with exceedingly low CEC of 4.2 $cmol_c\ kg^{-1}$ (similar to Danish coarse sands), adding 1.5% of corn stover biochar increased the CEC to 5.3 $cmol_c\ kg^{-1}$ (26% increase), but adding a similar rate of rice husk biochar caused a drastic increase in CEC to 23.2 $cmol_c\ kg^{-1}$ (452% increase). Thus, the impact of biochar on soil CEC may be strongly dependent on feedstock as well as pyrolysis temperature. Some of the variations between studies may be due to differences in the pH value at which CEC was measured. Preferably this should be at soil pH rather than biochar pH. Nevertheless, it appears that biochar would not be able to change soil CEC to any large extent, perhaps apart from at high application rates and to very sandy soils. There is, however, ongoing research on how to increase the CEC of biochar by oxidation or other post-production treatments that increase the content of negative surface charge. This would open avenues for producing designer biochars with tailored properties optimized for specific applications (Chacon et al. 2020). Kharel et al. (2019) found that it might be relatively straightforward to increase CEC of biochar by a factor of 10 by simple oxidation protocols. This step may be needed if biochar should contribute essentially to increasing CEC in Danish sandy soils, which would be desirable and add value to biochar application to agricultural soils.

4.2.2 Soil salinity (EC)

High electrical conductivity of the soil water solution, i.e., the salt level, increases the osmotic potential of the soil water and reduces plant water uptake and may induce salt stress This, in turn, limits the productivity of crops when crop-depending thresholds are exceeded. Salt concentrations in soil water higher than 5-10 dS/m corresponding roughly to K concentrations of 50-100 mM will reduce the yield of most crops (Maas & Hoffman 1977; Razzaghi et al. 2015). From the survey on plant nutrient content in biochar (Table 7.1) it appears that K is often the ion present in the largest amounts in biochars based on straw, manure fibre or sludge. Salinity is thus a factor to consider if extreme amounts of biochar, e.g., 100 t/ha are added to soils. This would correspond to adding between 2-10 Mg K/ha and would create a concentration of 200 to 500 mM of K in the soil water solution of the topsoil if applied to a typical Danish coarse sandy soil with limited CEC, exceeding the threshold values for salinity inhibition of crop growth with a factor of 2-10. However, when a phosphorous ceiling of 30 kg P/ha is respected in relation of permissible amounts of biochar added to agricultural soils (Chapter 7), it would generally prevent salinity effects, although for straw-based biochars it could still result in up to 875 K/ha (mean value, 350 kg K/ha) as deduced from the data compiled in Table 7.1 (Chapter 7). These levels exceed the crop requirement of typically 50-100 kg K/ha/year.

4.3 Conclusions

The effect of straw, manure or sludge biochar on soil physical properties strongly depends on the type of soil and the added biochar carbon. Coarse-textured soils are improved the most in terms of their hydraulic properties (water retention and availability) and compactness (bulk density). There is evidence to suggest that soil aggregation is also improved after the addition of biochar. Changes to soil chemical properties are more predictable based on the chemical properties of biochar and soil. Concerning the pyrolysis conditions, high temperature increases the pH and alkalinity of the biochar, but usually decreases the plant availability of the P content (as discussed further in Chapter 7). The K content is mostly plant-available and could potentially create salinity problems if high amounts of straw-based biochars are added to soils.

4.4 References

Abel, S., Peters, A., Trinks, S., Schonsky, H., Facklam, M. & Wessolek, G. 2013. Impact of biochar and hydrochar addition on water retention and water repellency of sandy soil. Geoderma 202: 183-191.

Alburquerque, J.A., Calero, J.M., Barron, V., Torrent, J., del Campillo, M.C., Gallardo, A. & Villar, R. 2014. Effects of biochars produced from different feedstocks on soil properties and sunflower growth. Journal of Plant Nutrition and Soil Science 177: 16-25.

Aller, D., Rathke, S., Laird, D., Cruse, R. & Hatfield, J. 2017. Impacts of fresh and aged biochars on plant available water and water use efficiency. Geoderma 307: 114-121.

Andersen, M.N., Jensen, C.R. & Lösch, R. 1992a. The interaction effects of potassium and drought in field-grown barley. I. yield, water-use efficiency and growth. Acta Agriculturae Scandinavica, Section B - Soil & Plant Science 42: 34-44.

Andersen, M.N., Jensen, C.R. & Lösch, R. 1992b. The interaction effects of potassium and drought in field-grown barley. II. nutrient relations, tissue water content and morphological development. Acta Agriculturae Scandinavica, Section B — Soil & Plant Science 42: 45-56.

Arthur, E. & Ahmed, F. 2017. Rice straw biochar affects water retention and air movement in a sand-textured tropical soil. Archives of Agronomy and Soil Science 63: 2035-2047.

Blanco-Canqui, H. 2021. Does biochar application alleviate soil compaction? Review and data synthesis. Geoderma 404., 115317

Bornø, M.L., Eduah, J.O., Muller-Stover, D.S. & Liu, F. 2018a. Effect of different biochars on phosphorus (P) dynamics in the rhizosphere of *Zea mays* L. (maize). Plant and Soil 431: 257-272.

Bornø, M.L., Muller-Stover, D.S. & Liu, F.L. 2018b. Contrasting effects of biochar on phosphorus dynamics and bioavailability in different soil types. Science of the Total Environment 627: 963-974.

Burrell, L.D., Zehetner, F., Rampazzo, N., Wimmer, B. & Soja, G. 2016. Long-term effects of biochar on soil physical properties. Geoderma 282: 96-102.

Chacon, F.J., Sanchez-Monedero, M.A., Lezama, L. & Cayuela, M.L. 2020. Enhancing biochar redox properties through feedstock selection, metal preloading and post-pyrolysis treatments. Chemical Engineering Journal 395, 125100

Clough, T.J., Condron, L.M., Kammann, C. & Müller, C. 2013. A review of biochar and soil nitrogen dynamics. Agronomy 3: 275-293.

Edeh, I.G., Masek, O. & Buss, W. 2020. A meta-analysis on biochar's effects on soil water properties - New insights and future research challenges. Science of the Total Environment 714, 136857.

El-Naggar, A., Lee, S.S., Rinklebe, J., Farooq, M., Song, H., Sarmah, A.K., Zimmerman, A.R., Ahmad, M., Shaheen, S.M. & Ok, Y.S. 2019. Biochar application to low fertility soils: A review of current status, and future prospects. Geoderma 337: 536-554.

Esmaeelnejad, L., Shorafa, M., Gorji, M. & Hosseini, S.M. 2016. Enhancement of physical and hydrological properties of a sandy loam soil via application of different biochar particle sizes during incubation period. Spanish Journal of Agricultural Research 14.

Fu, G.Q., Qiu, X.N., Xu, X.Y., Zhang, W., Zang, F. & Zhao, C.Y. 2021. The role of biochar particle size and application rate in promoting the hydraulic and physical properties of sandy desert soil. Catena 207.

Fu, Q., Zhao, H., Li, T., Hou, R., Liu, D., Ji, Y., Zhou, Z. & Yang, L. 2019. Effects of biochar addition on soil hydraulic properties before and after freezing-thawing. Catena 176: 112-124.

Gamage, D.N.V., Mapa, R.B., Dharmakeerthi, R.S. & Biswas, A. 2016. Effect of rice-husk biochar on selected soil properties in tropical Alfisols. Soil Research 54: 302-310.

Glab, T., Palmowska, J., Zaleski, T. & Gondek, K. 2016. Effect of biochar application on soil hydrological properties and physical quality of sandy soil. Geoderma 281: 11-20.

Gunal, E., Erdem, H. & Celik, I. 2018. Effects of three different biochars amendment on water retention of silty loam and loamy soils. Agricultural Water Management 208: 232-244.

Haider, F.U., Coulter, J.A., Cai, L.Q., Hussain, S., Cheema, S.A., Wu, J. & Zhang, R.Z. 2022. An overview on biochar production, its implications, and mechanisms of biochar-induced amelioration of soil and plant characteristics. Pedosphere 32: 107-130.

Hansen, V., Hauggaard-Nielsen, H., Petersen, C.T., Mikkelsen, T.N. & Muller-Stover, D. 2016a. Effects of gasification biochar on plant-available water capacity and plant growth in two contrasting soil types. Soil & Tillage Research 161: 1-9.

Hansen, V., Muller-Stover, D., Ahrenfeldt, J., Holm, J.K., Henriksen, U.B. & Hauggaard-Nielsen, H. 2015. Gasification biochar as a valuable by-product for carbon sequestration and soil amendment. Biomass & Bioenergy 72: 300-308.

Hansen, V., Müller-Stöver, D., Imparato, V., Krogh, P.H., Jensen, L.S., Dolmer, A. & Hauggaard-Nielsen, H. 2017. The effects of straw or straw-derived gasification biochar applications on soil quality and crop productivity: A farm case study. Journal of Environmental Management 186: 88-95.

Hansen, V., Müller-Stöver, D., Munkholm, L.J., Peltre, C., Hauggaard-Nielsen, H. & Jensen, L.S. 2016b. The effect of straw and wood gasification biochar on carbon sequestration, selected soil fertility indicators and functional groups in soil: An incubation study. Geoderma 269: 99-107.

Herath, H.M.S.K., Camps-Arbestain, M. & Hedley, M. 2013. Effect of biochar on soil physical properties in two contrasting soils: An Alfisol and an Andisol. Geoderma 209: 188-197.

Huang, Y.W., Li, F.C., Meng, J. & Chen, W.F. 2018. Lignin Content of Agro-forestry Biomass Negatively Affects the Resultant Biochar pH. Bioresources 13: 5153-5163.

Jin, Y., Liang, X.Q., He, M.M., Liu, Y., Tian, G.M. & Shi, J.Y. 2016. Manure biochar influence upon soil properties, phosphorus distribution and phosphatase activities: A microcosm incubation study. Chemosphere 142: 128-135.

Kassaye, K.T., Boulange, J., Kurebito, S., Tokunari, T., Saito, H. & Watanabe, H. 2022. The role of biochar in improving soil properties, water retention and potato yield in a Fluvisol under temperate monsoon climate. Soil Use and Management 38: 1069-1083.

Kelly, C.N., Benjamin, J., Calderon, F.C., Mikha, M.M., Rutherford, D.W. & Rostad, C.E. 2017. Incorporation of Biochar Carbon into Stable Soil Aggregates: The Role of Clay Mineralogy and Other Soil Characteristics. Pedosphere 27: 694-704.

Khademalrasoul, A., Naveed, M., Heckrath, G., Kumari, K.G.I.D., de Jonge, L.W., Elsgaard, L., Vogel, H.J. & Iversen, B.V. 2014. Biochar Effects on Soil Aggregate Properties Under No-Till Maize. Soil Science 179: 273 283.

Kharel, G., Sacko, O., Feng, X., Morris, J.R., Phillips, C.L., Trippe, K., Kumar, S. & Lee, J.W. 2019. Biochar Surface Oxygenation by Ozonization for Super High Cation Exchange Capacity. Acs Sustainable Chemistry & Engineering 7: 16410-16418.

Lei, O. & Zhang, R.D. 2013. Effects of biochars derived from different feedstocks and pyrolysis temperatures on soil physical and hydraulic properties. Journal of Soils and Sediments 13: 1561-1572.

Li XX, Rubaek GH, Muller-Stove, DS, Thomsen TP, Ahrenfeldt J, Sorensen P, 2017. Plant availability of phosphorus in five gasification biochars. Front Sustain Food Syst 1, 2.

Liu, C., Wang, H.L., Tang, X.Y., Guan, Z., Reid, B.J., Rajapaksha, A.U., Ok, Y.S. & Sun, H. 2016. Biochar increased water holding capacity but accelerated organic carbon leaching from a sloping farmland soil in China. Environmental Science and Pollution Research 23: 995-1006.

Ma, N.N., Zhang, L.L., Zhang, Y.L., Yang, L.J., Yu, C.X., Yin, G.H., Doane, T.A., Wu, Z.J., Zhu, P. & Ma, X.Z. 2016. Biochar improves soil aggregate stability and water availability in a mollisol after three years of field application. PLoS ONE 11, e0154091.

Maas, E.V. & Hoffman, G.J. 1977. Crop salt tolerance - current assessment. Journal of the Irrigation and Drainage Division 103: 115-134.

Mohan, D., Abhishek, K., Sarswat, A., Patel, M., Singh, P. & Pittman, C.U. 2018. Biochar production and applications in soil fertility and carbon sequestration - a sustainable solution to crop-residue burning in India. Rsc Advances 8: 508-520.

Mollinedo, J., Schumacher, T.E. & Chintala, R. 2015. Influence of feedstocks and pyrolysis on biochar's capacity to modify soil water retention characteristics. Journal of Analytical and Applied Pyrolysis.

Moragues-Saitua, L., Arias-González, A. & Gartzia-Bengoetxea, N. 2017. Effects of biochar and wood ash on soil hydraulic properties: A field experiment involving contrasting temperate soils. Geoderma 305: 144-152.

Munera-Echeverri, J.L., Martinsen, V., Strand, L.T., Zivanovic, V., Cornelissen, G. & Mulder, J. 2018. Cation exchange capacity of biochar: An urgent method modification. Science of the Total Environment 642: 190-197.

Nielsen, J.D. & Møberg, J.P. 1985. Klassificering af jordprofiler fra forsøgsstationer i Danmark. Tidsskr. Planteavl 89: 157-167.

Nissen R, Khanal G, Elsgaard L, 2021. Microbial ecotoxicity of biochars in agricultural soil and interactions with linear alkylbenzene sulfonates. Agronomy 11, 828.

Ojeda, G., Mattana, S., Avila, A., Alcaniz, J.M., Volkmann, M. & Bachmann, J. 2015. Are soil-water functions affected by biochar application? Geoderma 249: 1-11.

Petersen, C.T., Hansen, E., Larsen, H.H., Hansen, L.V., Ahrenfeldt, J. & Hauggaard-Nielsen, H. 2016. Pore-size distribution and compressibility of coarse sandy subsoil with added biochar. European Journal of Soil Science 67: 726-736.

Razzaghi, F., Jacobsen, S.E., Jensen, C.R. & Andersen, M.N. 2015. Ionic and photosynthetic homeostasis in quinoa challenged by salinity and drought - mechanisms of tolerance. Functional Plant Biology 42: 136-148.

Razzaghi, F., Obour, P.B. & Arthur, E. 2020. Does biochar improve soil water retention? A systematic review and meta-analysis. Geoderma 361, 114055.

Sarfaraz, Q., Silva, L., Drescher, G., Zafar, M., Severo, F., Kokkonen, A., Molin, G., Shafi, M., Shafique, Q. & Solaiman, Z. 2020. Characterization and carbon mineralization of biochars produced from different animal manures and plant residues. Scientific Reports 10, 955.

Sun, Z.C., Moldrup, P., Elsgaard, L., Arthur, E., Bruun, E.W., Hauggaard-Nielsen, H. & de Jonge, L.W. 2013. Direct and Indirect Short-term Effects of Biochar on Physical Characteristics of an Arable Sandy Loam. Soil Science 178: 465-473.

Thers, H., Abalos, D., Dorsch, P. & Elsgaard, L. 2020. Nitrous oxide emissions from oilseed rape cultivation were unaffected by flash pyrolysis biochar of different type, rate and field ageing. Science of the Total Environment 724, 138140.

Ul Islam, M., Jiang, F.H., Guo, Z.C. & Peng, X.H. 2021. Does biochar application improve soil aggregation? A meta-analysis. Soil & Tillage Research 209, 104926.

Wang, T., Arbestain, M.C., Hedley, M. & Bishop, P. 2012. Chemical and bioassay characterisation of nitrogen availability in biochar produced from dairy manure and biosolids. Organic Geochemistry 51: 45-54.

Wang, Z.P., Shen, R., Ji, S.B., Xie, L.K. & Zhang, H.B. 2021. Effects of biochar derived from sewage sludge and sewage sludge/cotton stalks on the immobilization and phytoavailability of Pb, Cu, and Zn in sandy loam soil. Journal of Hazardous Materials 419, 126468.

Watanabe, H., Shimomura, K. & Okazaki, K. 2014. Experimental investigation of carbonate formation characteristics during coal and biomass pyrolysis under CO_2. Energy & Fuels 28: 4795-4800.

Xu, G., Zhang, Y., Shao, H. & Sun, J. 2016. Pyrolysis temperature affects phosphorus transformation in biochar: Chemical fractionation and 31P NMR analysis. Science of the Total Environment 569-570: 65-72.

Yang, Q., Ravnskov, S. & Andersen, M.N. 2020. Nutrient uptake and growth of potato: Arbuscular mycorrhiza symbiosis interacts with quality and quantity of amended biochars. Journal of Plant Nutrition and Soil Science 183: 220-232.

Yang, Q., Ravnskov, S., Pullens, J.W.M. & Andersen, M.N. 2022. Interactions between biochar, arbuscular mycorrhizal fungi and photosynthetic processes in potato (*Solanum tuberosum* L.). Science of the Total Environment 816: 151649.

Yuan, J.H., Xu, R.K. & Zhang, H. 2011. The forms of alkalis in the biochar produced from crop residues at different temperatures. Bioresource Technology 102: 3488-3497.

Zhao, H., Shar, A.G., Li, S., Chen, Y., Shi, J., Zhang, X. & Tian, X. 2018. Effect of straw return mode on soil aggregation and aggregate carbon content in an annual maize-wheat double cropping system. Soil and Tillage Research 175: 178-186.

Zong, Y.T., Wang, Y.F., Sheng, Y., Wu, C.F. & Lu, S.G. 2018. Ameliorating soil acidity and physical properties of two contrasting texture Ultisols with wastewater sludge biochar. Environmental Science and Pollution Research 25: 25726-25733.

5 Effects of biochar on soil biology

Anne Winding and Lars Elsgaard (reviewer, Emmanuel Arthur)

The living organisms in soil are responsible for the mineralization of organic matter and recycling of nutrients such as N, P and K for plant growth. The soil food web consists of several trophic layers and sizes of organisms, where the bacteria and fungi generally mineralize organic matter and are grazed by protists and micro-invertebrates. Mycorrhizal fungi specifically form symbiotic associations with plant roots and directly supply nutrients for plant growth. Also, larger invertebrates, such as earthworms, feed on microorganisms, micro-invertebrates as well as soil organic matter. The earthworms feeding and burrowing activities are important for the incorporation of organic matter into soil and for soil aeration and water infiltration capacity. A healthy soil supporting good plant growth has high biodiversity and relatively high organic carbon content, e.g., 1-2% on a soil dry weight basis.

Biochar added to soil as a soil improver, fertilizer or for C-sequestration will affect the soils physical and chemical properties (Chapter 4) and the soil biology in many different ways. Biochar will affect the water holding capacity, pH, salinity, soil erosion, surface area, CEC and nutrient availability (Chapter 4 and 7) among others. Furthermore, the biochar might contain and release problematic organic compounds and heavy metals as well as adsorb these from the soil matrix. The latter has led to the use of biochar for soil remediation, and in agricultural soil the absorption processes may also include adsorption of organic agrochemicals, such as pesticides. The soil biology will be affected by the biochar-induced changes to the physical-chemical parameters as well as directly by the biochar constituents. Overall, the effects of biochar on soil biology are strongly dependent on the biochar feedstock and pyrolysis conditions.

Effects of biochar on soil biology are inevitably difficult to separate from effects on soil chemical and physical parameters, and often the effects have to be considered in a holistic way. The alternative is to test the effects of biochar on soil biology without the presence of soil (e.g., Godlewska et al. 2021). For example, some microbial ecotoxicity tests have been performed with isolated bacteria exposed to water extracts of biochars, but the interpretation of such data is challenging in an ecological context, since microbial responses, e.g., in agricultural ecosystems, depend on native microbiomes and realistic routes of biochar exposure (Godlewska et al. 2021). Therefore, tests with biochar-amended soils are necessary to assess the biological and environmental impacts (Nissen et al. 2021)

Here, we aim to report on the effects of biochar on soil biology, but we continuously have to bear in mind that the effects reported might be due to proximal effects on soil physico-chemical parameters with effects on biology. Biochar can have positive effects on the soil biology, due to macro- and micronutrients, available labile carbon fractions, liming effect, water holding capacity and habitable pore spaces within the biochar particles (Siedt et al. 2021) while negative effects are also reported (Godlewska et al. 2021, Brtnicky et al.

2021). The increase in pH, as mediated by alkaline biochars, will result in changes in the microbial communities (Xiang et al. 2021), since pH is a master variable affecting the majority of biogeochemical processes in the soil ecosystem (Abalos et al. 2020). These changes may to some extent resemble the effects of liming, which is a common practice in Danish agricultural soils to mitigate soil acidity and increase crop yields.

The resulting impact after adding biochar to soil depends on the biochar properties as affected by feedstock and pyrolysis conditions, but also on the amount of biochar applied. This means that if certain positive effects on, e.g., soil aggregates (see Figure 4.1) or soil pH (see Figure 4.3) is the aim of biochar amendment, then a sufficiently high biochar rate should be administered to the soil to obtain the desired effect. However, the protection and maintenance of soil biology is of high relevance for maintaining a healthy soil with high agricultural value and hence application rates and effects of biochar should not compromise and deteriorate the soil biology.

5.1 Soil fauna

The effect of biochar on soil fauna can be due to, e.g., direct physical interaction or indirect effects on feeding. Moreover, the fauna can affect the biochar spreading and transportation in the soil. Earthworms are among the most significant modulators of the soil environment through their burrowing activities during which they also consume soil with biochar resulting in vertical mixing of biochar (Lehmann et al. 2011). Generally, the effects of biochar on soil fauna are less studied than the effects on microorganisms and plants. Detrimental effects on soil fauna, such as earthworms and micro-invertebrates, are of high importance as these organisms are essential for many ecosystem services (e.g., water infiltration, soil aeration, comminution and degradation of litter and soil organic matter). Furthermore, any effects on the soil fauna will inevitably lead to changes in the soil microbiome diversity and activity. No matter whether the changes increase or decrease soil microbiome activities, they will affect the soil microbiome recycling of organic matter.

5.1.1 Earthworms

Most of the studies on soil invertebrates have focused on earthworms. Biochar may affect earthworms directly or indirectly through changes in soil physico-chemical properties, as soil moisture content and pH, which can affect different stages of the earthworm life, such as survival, growth, and reproduction (Domene 2016, Sanchez-Hernandez et al. 2019). Further, there may be synergistic effects between biochar and earthworms, which can lead to improved soil structure, microbial abundance and activity (Zhang et al. 2021). For example, in a six month mesocosm experiment with biochar amended soil, earthworms were found to have a higher effect on clover growth than biochar, while synergistic effects of combined biochar and earthworms were seen on the abundance of spring tails (collembolans) and fungal biomass (Garbuz et al. 2020). In a review, Brtnicky et al. (2021) concluded that the effects of biochar on earthworms are contradictory. The adverse effects included lower reproduction and activity, genotoxicity, slower growth and decreased weight. However, also increased or unaffected reproduction has been reported, as well as increase in digestive enzyme activities. Some of the adverse effects have been attributed to the physical effect of biochar particles sticking to the earthworm surfaces and earthworms avoiding feeding on biochar. Notably, the review by Brtnicky et al. (2021) does not distinguish between lab and field studies, and concentrations of biochar tend to vary in these two types of experimental setups, i.e., often with higher experimental biochar rates applied under lab conditions. In contrast to this, a Danish three-year field experiment with straw-based biochar applied at a total rate of either 3 Mg/ha or 16 Mg/ha reported that earthworm abundance was unaffected compared both to straw-amended soil and control treatment (Hansen et al. 2017).

5.1.2 Micro-arthropods

ISO standard ecotoxicity tests of biochars from four different feedstocks on the springtail *Folsomia candida* in an artificial standard soil revealed negative effects at high biochar concentrations and differences in toxicity among biochars (Conti et al. 2018). In line with this, Gruss et al. (2019b) found short-term toxic effects on *F. candida* in studies of biochar-amended soil using ISO tests; this was attributed to the pH effect while biochar additions at realistic field conditions did not affect the springtails.

Llovet et al. (2021) reported harmful effects on nematode and micro-arthropod communities after adding 50 Mg/ha of biochar to agricultural soil but not after adding 12 Mg/ha. In contrast, Gruss et al. (2019b) found an increase in the abundance of mesofauna when applying 50 Mg/ha of biochar. In a field experiment, biochar increased the abundance of collembolans, however, the choice of crop species (maize vs. oil seed rape) affected collembolans even more (Gruss et al. 2019a).

Brtnicky et al. (2021) reported negative effects of biochar on micro-arthropods and linked the effect to biochar concentration, while citing Bielská et al. (2018) for showing positive effects of biochar at 1-5% concentration while negative effects were seen at 10% concentration. This was explained by the lower biochar dose adsorbing toxic compounds, while at higher biochar concentrations, this effect was overrun by the toxic effects of biochar itself. In this respect, it should be noted that 10% biochar in soil is an unrealistically high rate under normal field conditions.

5.2 Soil microorganisms

Studies on the effects of biochar on soil microorganisms are abundant and Lehmann et al. (2011) concluded in a review that generally the effects of biochar on soil organisms were positive with higher microbial biomass, microbial diversity and activity. Since then, several publications have reached similar conclusions as reviewed in Palansooriya et al. (2019), but there are, however, also disturbing results of negative effects on arbuscular mycorrhiza (AM) and contradictory effects on microbial community diversity and functions (reviewed in Godlewska et al. 2021 and Brtnicky et al. 2021). Siedt et al. (2021) concluded that our understanding of the effects of biochar on soil microbial communities is still limited, despite many studies of biochar effects on soil microbial communities, nutrient cycling and soil properties. This can partly be due to the close interaction between nutrient availability and microorganisms and the fact that biochar affects microorganisms directly and indirectly at the same time.

Whereas biochar amendment repeatedly has been reported to change microbial communities, it is often more uncertain (i) whether this has positive or negative impact on soil quality, (ii) whether the changes are larger or comparable to normal agricultural practices, and (iii) how long-lasting the changes are.

In a Danish three-year field experiment with straw-based biochar applied at total rates of either 3 or 16 Mg/ha, the abundance of cultivable bacteria and protists (only for the 16 Mg/ha amendment) increased compared to the control with no biochar amendment, but was unchanged compared to straw amended soil (Hansen et al. 2017). Detailed studies of the effect on the microbial communities by analysis of ATP, ten enzymatic activities, substrate-induced respiration with seven carbon substrates, and bacterial community diversity by 16S rDNA amplicon sequencing, revealed increased phenol oxidase activity and decreased cellulose activity in the 16 Mg/ha treatment when analysed three months after the last amendment (Imparato et al. 2016). Effects on the bacterial diversity were limited to changes in relative abundance of some bacteria without a clear ecological significance. Also, Llovet et al. (2021) reported negligible effects on soil microorganisms. However, Han et al. (2017) found that 21 of 45 bacterial genera were affected by biochar, e.g., with decreasing abundance of the genus *Nitrospira*. The nitrite-oxidizing *Nitrospira* plays an important role in the soil N-cycle, where they contribute to nitrification. Likewise, N-fixing bacteria can be affected by biochar (Orr et al. 2016) and Liu et al. (2019) reported that addition of biochar could increase the abundance and alter the community structure of diazotrophs, which may benefit N-fixation in agricultural soil. Biochar has also been found to affect the microbial rhizosphere communities and in this way the nutrient cycling and crop health (Siedt et al. 2021). This is definitely an area that needs more research to fully understand the dynamics of microorganisms, nutrients and biochar in the rhizosphere and the effects on crop production.

The effects of biochar on the soil microbiome should be separated into short-term and long-term effects as the short-term effects are often more pronounced than the long-term effects. Stimulatory short-term effects may be due to factors such as (i) a pool of easily degradable organic matter released from the biochar right after introduction into soil, (ii) amelioration of acidic soil pH or (iii) improved conditions for microbial activity in the soil due to better aeration. Further, stimulatory effects may occur on longer term, if the soil microorganisms colonize the biochar surfaces and establish an active community there. However, the importance of biochar as niche for active microorganism needs to be better understood. Negative short-term effects may be due to potentially toxic elements (PTEs) released from biochar, such as polycyclic aromatic hydrocarbons (PAHs), heavy metals or volatile organic compounds (VOCs), which are not persistent in the soil, though. Also longer-term negative effects could be caused by PAHs or heavy metals, which are known potential contaminants in biochars (e.g., Nissen et al. 2021), where heavy metals results from the feedstock whereas PAHs originate from re-condensation of pyrolysis liquids and gases. Brtnicky et al. (2021) listed a range of different effects of biochar on soil microorganisms; effects that are counted as positive, negative or indifferent. In contrast, Palansooriya et al. (2019) listed a range of effects of biochar on microbial communities that are all considered positive. This includes increased microbial biomass and activity, increased ratio of Gram-positive/Gram-negative bacteria (G+/G- ratio), increased ratio of AM/saprotrophic fungi, and increased dehydrogenase and alkaline phosphatase activity. Generally, the microbiome in different soils may respond differently to biochar amendments and different microbial groups have a different response to biochar and the changes induced by biochar in the soil physical and chemical environment (Chapter 4).

Bruun and El-Zehery (2012) found biochar addition to soil to decrease the soil organic matter (SOM) degradation in lab experiments testing separately fresh SOM and 40 year old SOM, where the SOM was ^{14}C labelled at the time of the experiment and 40 years ago. Bruun et al. (2014) found that mineralization of biochar (measured as CO_2 production) was partly mediated by microorganisms and partly by chemical processes during 40 days of lab incubation. Hence, they concluded that microorganisms take part in the biochar degradation, but also chemical processes are important. Certainly, biochar interacts with indigenous soil organic matter, and the net effects in terms of increases or decreases in soil organic matter degradation seem to depend on the biochar feedstock and pyrolysis conditions in combination with the soil organic matter properties (Palansooriya et al. 2019). This is currently studied in Danish >1 year lab studies of CO_2 release from Danish agricultural soils with different biochars, and include studies of changes in microbial communities.

5.2.1 Microbial enzymatic activities

Biochar amendment generally increases the sorption capacity in soil (Chapter 4), which could further influence the microbial ecosystem, for example, by stabilizing soil-enzyme interactions. The latter mechanisms are known for clay-enzyme and humus-enzyme interactions (Burns 1982), but an analogous role for biochar needs to be further documented. Stabilization of extracellular enzymes could increase (accumulate) the pool of catalytic sites in the soil, although such interactions could also reduce the specific activity of individual enzymes depending on the orientation of the active sites in the biochar-enzyme complexes. Thus, biochar may interact with soil enzymes by adsorption processes, but biochar may also induce the microbial production of certain extracellular enzymes and increase their activity (Lehmann et al. 2011). Specifically, enzymatic activities such as phenol oxidase related to the degradation of aromatic compounds as found in PAHs and highly lignified organic matter have been found to increase (e.g., Imparato et al. 2016). However, the changes are often found to be related to the feedstock and pyrolysis conditions.

Wood-based biochar in Danish field soils increased potential ammonia oxidation compared to no biochar amendment, but without relation to the amount of biochar added while soil pH was a driving parameter. The arylsulfatase activities in the same soils were not affected by biochar (Sun et al. 2014).

Llovet et al. (2021) found arylsulfatase and phosphodiesterase to respond the most to biochar addition (12 and 50 Mg/ha) six years after the amendment. The effects were dependent on the fertilization regime during the growing season with negative effects in spring followed by positive effects later in the growing season. Tea bag decomposition studies showed no effect of biochar on green tea mineralization and a slight positive effect on rooibos tea mineralization (Llovet et al. 2021). In these studies, the tea bags are used as mesh bags for in situ studies of degradation of relatively labile organic carbon material (green tea) and relatively recalcitrant organic carbon material (rooibos tea), thus indicating if the microorganisms adapted to biochar in soil may have an increased enzymatic capacity to decompose these carbon fractions. The overall effects of biochar addition on soil quality, using a range of indicators, indicated carbon sequestration to increase and soil food web functioning to decrease with increasing biochar rate (Llovet et al. 2021). Over a three-year field study, Brtnicky et al. (2019) found biochar amendment increased microbial biomass and ammonia-oxidizing bacteria, especially in co-amendment with manure, while dehydrogenase activities decreased when biochar was applied without manure. The latter observation was explained by enzyme adsorption to biochar.

5.2.2 Biochar effect on specific soil microorganisms

Microorganisms such as N-fixing rhizobia, biological control agents and other beneficial microorganisms are added to soil as plant growth promoting agents, and biochar has been suggested as a beneficial carrier of these inoculants increasing and extending the survival and successful inoculation by providing a protective and habitable pore space. In the pores of biochar, microorganisms are suggested to be protected against predation and have easier access to micronutrients in an aerated environment (Lehmann et al. 2011). However, such specific use may require designed and optimized biochars, and indeed the role of biochar as habitable space for active microorganisms needs to be better understood.

Biochar has been reported to reduce pathogen attacks which can be due to biochar-mediated actions such as change in competition for nutrients and space competition, biochar sorption of inhibiting compounds and enhanced plant disease resistance due to improved plant health (Lehmann et al. 2011). These are interesting perspectives that need further study. However, the well-established beneficial symbiosis between AM and plant roots might be affected by biochar. Negative effects of biochar on this relationship have been reported in several cases (Lehmann et al. 2011, Liu et al. 2017, 2018a,b, Yang et al. 2022). In contrast, a meta-analysis found no significant change in mycorrhizal colonization in biochar amended soil (Biederman and Harpole, 2013), and other studies have even shown that biochar promotes root colonization by mycorrhizal fungi (Siedt et al. 2021). Various results are published, and generally, it appears that the biochar properties and effects on pH and nutrient availability in the soil are crucial for the resulting effects of biochar on AM fungi. Hence, as of today, no firm conclusions on the effect of biochar on AM can be reached.

Biochar application and the increase in soil pH have been reported to promote hydrolysis of N-acyl-homoserine lactone (AHL), a signaling compound used by Gram-negative bacteria for cell communication (Masiello et al. 2013). Further, biochar produced from maize was reported to bind AHL to an extent that increased with higher pyrolysis temperature (Sheng et al. 2022). This indicates that biochar might affect the soil microorganisms by altering the cell-to-cell communication.

5.3 Problematic compounds

During pyrolysis, the increased temperatures will degrade many of the problematic compounds in the feedstock such as microplastics, organic agrochemicals, and pharmaceuticals, including antibiotics. However, other organic compounds such as PAHs, dioxins, PFOA and PFOS, and VOCs can be produced depending on the pyrolysis conditions (Brtnicky et al. 2021, Godlewska et al. 2021, Xiang et al. 2021). VOCs such as acetone, benzene, organic acids, methanol and phenol are produced during lower pyrolysis temperatures and will bind to biochar (Godlewska et al. 2021). Godlewska et al. (2021) also reviewed the pyrolysis conditions and the resulting concentrations and bioavailability of PAHs and a range of heavy metals. This indicated how strong many of the compounds are adsorbed to biochar. PAHs might be produced and condensed on the biochar during production, but this depends on the feedstock, pyrolysis conditions and the cooling process, which should be optimized to avoid PAHs formation (Brtnicky et al. 2021). Han et al. (2022) reviewed formation of PAH during pyrolysis and reported 2-3 ringed PAHs to be formed at temperatures lower than 500°C and PAHs with 5 or 6 rings to be formed at temperatures higher than 500°C and through recombination of reactive free radicals. The concentration of PAH produced seems to be increasing with higher temperatures, but this is not clearly evident as contrasting findings are also reported. Also, fast pyrolysis and gasification biochar exhibited higher total PAHs concentration than slow pyrolysis biochars, while evaporation at high temperatures results in much of the PAHs to end up in the gas phase and not settle in the biochar (Han et al. 2022). This is emphasizing the importance of the pyrolysis reactor conditions and control thereof.

Biochar has been found to be able to increase PAH content of soil, which, however, might be decreased with time due to leaching or microbial degradation (Brtnicky et al. 2021). But biochar can also function as an absorbent of contaminating organic compounds in soils, including PAHs (from other sources than biochar) and organic agrochemicals, and biochar has in this way been used for remediation of polluted soils. The adsorption capacity of biochar is typically higher than for the un-pyrolysed organic feedstock compounds. The remediation potential of biochar is not particularly relevant for Danish agricultural soils. However, the relative high adsorption capacity of biochar towards agrochemicals (e.g., pesticides, herbicides and nitrification inhibitors) might reduce their efficacy in agriculture and contribute to adverse effects of biochars (Fuertes-Mendizábal et al. 2019; Brtnicky et al. 2021).

In the case of interaction between biochar and nitrification inhibitors (which are increasingly used to mitigate N_2O emissions from agricultural soils) Fuertes-Mendizábal et al. (2019) showed that the effect of the common nitrification inhibitor DMPP was significantly reduced when it was combined with the application of biochar, probably due to the adsorption of DMPP to biochar surfaces. No studies on such interactions are published for Danish soil conditions and Fuertes-Mendizábal et al. (2019) also concluded that further experiments are needed to understand the basis and temporal longevity of the adsorption mechanism, which could reduce the efficiency of N_2O mitigation by nitrification inhibitors.

PAHs should be avoided in agricultural soil ecosystems and in Denmark cut-off values are implemented for soil amendments such as sewage sludge and ashes. This will also be the case for the marketing of CE-labelled biochars (European Commission, 2021). Thus, although studies with PAH contaminated biochar does not always indicate PAHs in biochar as a likely source of short-term ecotoxicity to soil microorganisms (Nissen et al. 2021), there will be a requirement for producers of biochar to comply with PAH cut-off values that will protect soil biological processes for the marketing of CE-labelled biochars. The Danish national legislation does not today require CE-labelling of marketed biochars.

Heavy metals in the biochar feedstock will generally be concentrated in the biochar produced and is especially an issue when using sewage sludge and household waste as feedstock. Increasing pyrolysis temperature increases the heavy metal concentration due to the degradation of organic matter. However, it is reported that the heavy metal bioavailability decreases with increasing pyrolysis temperature (Xiang et al. 2021). Additionally, certain heavy metals such as Hg and Cd will enter a volatile form and provided optimal pyrolysis conditions (including cooling), the heavy metal can be captured and deposited away from the biochar. Despite this, heavy metals are repeatedly reported in biochar (Godlewska et al. 2021), and the feedstock should be carefully selected to minimize heavy metal content.

At all circumstances, the heavy metal content in the biochar should be monitored prior to incorporation in soil. As described in Chapter 1, voluntary standards, such as the European Biochar Certificate (EBC 2022) are issued in this respect with focus on suggested cut-off values for problematic compounds in biochar for use on agricultural fields. As discussed in Chapter 1, the positive list for EBC biochar includes straw and digestate, but the proportion of animal source materials for the biogas plant must currently be less than 40 percent (see also Chapter 1, section 1.3). Biochar produced from pure sources of, e.g., pig manure digestate may contain relatively large amounts of Cu and Zn, and such biochars were found to increase heavy metals in an agricultural soil by 55% for Cu and 70% for Zn, when applied at a rate of 15 Mg biochar per ha (Ayaz et al. 2021).

Dioxins might be produced during pyrolysis at temperatures of 200-900ºC and also by volatilization followed by re-condensation (Han et al. 2022). Dioxin formation decreases with increasing pyrolysis temperature and increases with increasing chlorine concentration as in household waste. However, Hale et al. (2012) found dioxin levels in 14 biochars produced from plant materials at 200-900ºC to be in the range of 0.008-1.2 pg/g which is below the Swedish guideline values.

Per- and polyfluoroalkyl substances (PFAS) as perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid (PFOS) have been discovered in biochar produced from sewage sludge in the concentrations of ca. 16 ng/g and unchanged during pyrolysis of 300-700ºC (Kim et al. 2015) while they were not discovered in biochar produced from plant materials. In a review by Buss (2021), it was stated that pyrolysis at 500°C removed >91% of PFOA and PFOS along with many other organic contaminants, like antimicrobials, dioxins, and PAH from sewage sludge (Table 1 in Buss 2021). Silvani et al. (2019) reported designed biochar to reduce leaching of PFAS and heavy metals from soil depending on total organic carbon content, and Askeland et al. (2020) suggested biochar to be used as sorbent of PFOS, PFOA and congener in contaminated soils. They showed that soil type and texture affected the sorption capacity. These results indicate many unresolved questions regarding fate of organic contaminants, depending on feedstock, pyrolysis conditions (especially temperature) and soil type.

Aging of biochar after amendment to soil by physical, chemical and biological interactions has the potential to change the properties of biochar in soil. Biochar may break up and shrink in size, exposing new surface areas. Change in cation exchange capacity over time and chemical oxidation may change the chemical structure and surface properties generating oxygen-containing functional groups. Adsorption of PAHs and other organic contaminants might weaken over time, causing release of the contaminants. Microorganisms may use biochar as a substrate, form biofilm on the surface with bacteria and fungal hyphae penetrating the biochar pores, and through this process excrete enzymes changing and degrading the biochar (Xiang et al. 2021, and references therein).

PAHs and other problematic organic compounds introduced with biochar may directly harm the soil biology by ecotoxicological effects, but on the other hand, microorganisms able to degrade the organic compounds might increase their activity. Also, biochar might adsorb nutrients and, in this way, affect microorganisms (Xiang et al. 2021) by restricting nutrient availability. Likewise, cell-to-cell communication between microorganisms can also be hampered by biochar adsorbing the signalling molecules.

5.4 Effects on aquatic organisms and leaching

Due to the theoretical risk of leaching of biochar to aquatic environments (e.g., via drains and surface water), ecotoxicological tests performed using aquatic organisms according to the OECD and ISO guidelines are also reported in the literature in relation to the effects on aquatic organisms. Several harmful effects are reported on fish (*Oreochromis niloticus,* tilapia), crustaceans (*Daphnia magna*), phytoplankton, plants (*Vallisneria spiralis*, eel grass) and chlorophyll-a activity (Xiang et al. 2021 and references therein). However, the effect of biochar after interaction with soil particles prior to leaching has not been thoroughly studied. The leaching of heavy metals and organic contaminants is difficult to generalize as it depends on the biochar feedstocks, pyrolysis conditions, biochar particle size, biochar application rates as well as the recipient soil texture and organic matter content. However, while leaching of contaminants cannot be excluded, biochar has been reported to adsorb both heavy metals and organic contaminants after application to soil and is being used to remediate contaminated soils (Zhao et al. 2019). The risk of leaching under Danish agricultural conditions are not well studied.

5.5 Concluding remarks

Biochar can affect soil biology very differently and in both negative and positive ways. The chemical and physical properties of the biochar are of high importance and hence the feedstock and the pyrolysis conditions including temperature, residence time during pyrolysis, and cooling processes, are crucial for the resulting effects of the produced biochar, e.g., also in relation to concentrations of contaminants such as PAHs and heavy metals, which affect soil biology. Also, the amount and particle size of biochar added, the soil texture, structure and organic matter and nutrient content are important factors to consider.

In summary, the literature reporting effects of biochar on soil biology repeatedly presents the same conclusion: the effects on soil biology are dependent on the feedstock, pyrolysis temperature and cooling conditions. Also, the biochar concentration of easily degradable organic carbon compounds (see Chapter 6), the pore and aggregate structure, and the adsorption capacity of nutrients, minerals and organic contaminants rely on the feedstock and production conditions. The feedstock may vary from sewage sludge and household waste with potential higher content of xenobiotic compounds to different plant materials, including straw. However, even for the same feedstock type, different batches of, e.g., sewage sludge may result in biochars with different properties and also biochar from different plant materials may have different resulting effects on soil biology (Brtnicky et al. 2021; Lehmann et al. 2011).

Furthermore, the effects of biochar on soil biology are strongly dependent on the amount of biochar added and the soil quality including physical and chemical parameters. The evident significance of feedstock and production conditions on environmental effects makes it very difficult to reach a general conclusion from the large number of primary publications on the effects of biochar on soil biology. It also highlights the need for standardized characterization of biochar production conditions or at least chemical and physical characterization and minimal standards for the produced biochar prior to soil amendment (EBC, 2022).

As stated by Nissen et al. (2021), biochar currently attracts attention as a negative carbon emission technology and therefore, large-scale application of biochar to agricultural land could become a significant parameter in meeting the Danish 2030 goals on climate change mitigation (see Chapter 8). This will require a firm knowledge base on safe application in agricultural soils. So far, biochar quality criteria, such as those suggested by the European Biochar Certificate (2020) mainly concerns physical and chemical properties of biochar. It is therefore timely to strengthen the research emphasis on soil biological tests with biochar-amended soils to better understand both short-term and long-term effects in the agricultural soil ecosystem. Some of these knowledge gaps are further addressed in Chapter 8.

5.6 References

Abalos D, Liang Z, Dörsch P, Elsgaard L (2020). Trade-offs in greenhouse gas emissions across a liming-induced gradient of soil pH: Role of microbial structure and functioning. Soil Biol Biochem 150, 108006.

Askeland M, Clarke BO, Cheema SA, Mendez A, Gasco G, Paz-Ferreiro J, Biochar sorption of PFOS, PFOA, PFHxS and PFHxA in two soils with contrasting texture, 2020. Chemosphere, 249, 126072, https://doi.org/10.1016/j.chemosphere.2020.126072.

Ayaz M, Stulpinaite U, Feiziene D, Tilvikiene V, Akthar K, Baltrenaite-Gediene E, Striugas N, Rehmani U, et al. (2021). Pig manure digestate-derived biochar for soil management and crop cultivation in heavy metals contaminated soil. Soil Use Manag 38, 1307-1321.

Biederman LA, Harpole WS, 2013. Biochar and its effects on plant productivity and nutrient cycling: a meta-analysis. GCB Bioenergy 5, 202-214.

Bielská L, Škulcová L, Neuwirthová N, Cornelissen G, Hale SE, 2018. Sorption, bioavailability and ecotoxic effects of hydrophobic organic compounds in biochar amended soils. Sci Total Environ 624, pp. 78-86.

Brtnicky M, Datta R, Holatko J, Bielska L, Gusiatin ZM, Kucerik J, Hammerschmiedt T, Danish S, Radziemska M, Mravcova L, Fahad S, Kintl A, Sudoma M, Ahmed N, Pecina V, 2021. A critical review of the possible adverse effects of biochar in the soil environment. Sci Tot Environ 20, 796, 148756.

Brtnicky M, Dokulilova T, Holatko J, Pecina V, Kintl A, Latal O, Vyhnanek T, Prichystalova J, Datta R, 2019. Long-term effects of biochar-based organic amendments on soil microbial parameters. Agronomy 9, 747.

Bruun S, Clauson-Kaas S, Bobuľská L, Thomsen IK, 2014. Carbon dioxide emissions from biochar in soil: role of clay, microorganisms and carbonates. Eur J Soil Sci 65, 52–59.

Bruun S, EL-Zehery T, 2012. Biochar effect on the mineralization of soil organic matter. Pesquisa Agropecuária Brasileira 47, 665–671.

Burns R, 1982. Enzyme activity in soil: location and a possible role in microbial ecology. Soil Biol Biochem 14, 423–427.

Buss W, 2021. Pyrolysis Solves the Issue of Organic Contaminants in Sewage Sludge while Retaining Carbon—Making the Case for Sewage Sludge Treatment via Pyrolysis. ACS Sus. Chem Eng. 9: 10048-10053. doi: 10.1021/acssuschemeng.1c03651

Conti FD, Visioli G, Malcevschi A, Menta C, 2018. Safety assessment of gasification biochars using *Folsomia candida* (Collembola) ecotoxicological bioassays. Environmental Science and Pollution Research 25, 6668–6679.

Domene X, 2016. A critical analysis of meso- and macrofauna effects following biochar supplementation. In: Biochar Application: Essential Soil Microbial Ecology (Ralebitso-Senior TK, Orr CH, Eds). Elsevier, Amsterdam, pp. 268–292.

EBC (2022) European Biochar Certificate - Guidelines for a Sustainable Production of Biochar. European Biochar Foundation (EBC), Arbaz, Switzerland. (http://european-biochar.org). Version 10.1 from 10th Jan 2022.

European Commission, 2021. Annexes to the Commission Delegated Regulation Amending Annexes II, III and IV to Regulation (EU) 2019/1009 of the European Parliament and of the Council for the Purpose of Adding Pyrolysis or Gasification Materials as a Component Material Category in EU Fertilising Products, European Commission, Ref. Ares(2021)44211—04/01/2021. https://eur-lex.europa.eu/legalcontent/EN/TXT/?uri=PI_COM:Ares(2021)44211

Fuertes-Mendizábal T et al. 2019. Biochar reduces the efficiency of nitrification inhibitor 3,4-dimethylpyrazole phosphate (DMPP) mitigating N_2O emissions. Sci. Rep 9, 2346.

Garbuz S, Camps-Arbestain M, Mackay A, DeVantier B, Minor M, 2020. The interactions between biochar and earthworms, and their influence on soil properties and clover growth: a 6-month mesocosm experiment. Appl. Soil Ecol 147, 103402.

Godlewska P, Ok YS, Oleszczuk P, 2021. The dark side of black gold: Ecotoxicological aspects of biochar and biochar-amended soils. J. Hazard Mater 403, 123833.

Gruss I, Twardowski JP, Latawiec A, Kròlczyk J, Medynska-Juraszek A, 2019a. The effect of biochar used as soil amendment on morphological diversity of Collembola. Sustainability 11, 5126.

Gruss I, Twardowski JP, Latawiec A, Medyńska-Juraszek A, Królczyk J. 2019b. Risk assessment of low-temperature biochar used as soil amendment on soil mesofauna. Environ Sci Pollut Res Int 26, 18230-18239.

Hale, S.E., Lehmann, J., Rutherford, D., Zimmerman, A.R., Bachmann, R.T., Shitumbanuma, V., O'Toole, A., Sundqvist, K.L., Arp, H.P.H., & Cornelissen, G. 2012. Quantifying the total and bioavailable polycyclic aromatic hydrocarbons and dioxins in biochars. Environmental Science and Technology 46: 2830–2838.

Han H, Buss W, Zheng Y, Song P, Rafiq MK, Liu P, Mašek O, Li X. 2022. Contaminants in biochar and suggested mitigation measures – a review, Chemical Engineering Journal 429, 132287.

Han G, Lan J, Chen Q et al. 2017. Response of soil microbial community to application of biochar in cotton soils with different continuous cropping years. Sci Rep 7, 10184.

Hansen V, Müller-Stöver D, Imparato V, Krogh PH, Jensen LS, Dolmer A, Hauggaard-Nielsen H, 2017. The effects of straw or straw-derived gasification biochar applications on soil quality and crop productivity: A farm case study. J Environ Manage 186, 88-95.

Imparato V, Hansen V, Santos S, Nielsen TK, Giagnoni L, Hauggaard-Nielsen H, Johansen A, Renella G, Winding A, 2016. Gasification biochar has limited effects on functional and structural diversity of soil microbial communities in a temperate agroecosystem. Soil Biol Biochem 99, 128-136.

Kim JH, Ok YS, Choi G-H, Park B-J, 2015. Residual perfluorochemicals in the biochar from sewage sludge. Chemosphere 134, 435-437,

Lehmann J, Rillig MC, Thies J, Masiello CA, Hockaday WC, Crowley D, 2011. Biochar effects on soil biota – A review. Soil Biol Biochem 43, 1812-1836

Liu C, Liu F, Ravnskov S, Rubæk G, Sun Z, Andersen MN, 2017. Impact of wood biochar and its interactions with mycorrhizal fungi, phosphorus fertilization and irrigation strategies on potato growth. J Agron Crop Sci 203, 131-145.

Liu C, Ravnskov S, Liu F, Rubæk GH, Andersen MN, 2018a. Arbuscular mycorrhizal fungi alleviate abiotic stresses in potato plants caused by low phosphorus and deficit irrigation/partial root-zone drying. Sci Tot Environ 156, 46-58.

Liu L, Li JW, Yue FX, Yan XW, Wang FY, Bloszies S, Wang YF, 2018b. Effects of arbuscular mycorrhizal inoculation and biochar amendment on maize growth, cadmium uptake and soil cadmium speciation in Cd-contaminated soil. Chemosphere 194, 495-503.

Liu X, Liu C, Gao W, Xue C, Guo Z, Jiang L, et al. 2019. Impact of biochar amendment on the abundance and structure of diazotrophic community in an alkaline soil. Science of the Total Environment 688, 944–951

Llovet A, Mattana S, Chin-Pampillo J, Gascó G, Sánchez S, Mondini C, Briones MJI, Márquez L, Alcañiz JM, Ribas A, Domene X, 2021. Long-term effects of gasification biochar application on soil functions in a Mediterranean agroecosystem: Higher addition rates sequester more carbon but pose a risk to soil faunal communities. Sci Tot Environ 801, 149580.

Masiello CA, Chen Y, Gao X, Liu S, Cheng HY, Bennett MR, Rudgers JA, Wagner DS, Zygourakis K, Silberg JJ, 2013. Biochar and microbial signaling: production conditions determine effects on microbial communication. Environ Sci Technol 47, 11496−11503.

Nissen R, Khanal G, Elsgaard L, 2021. Microbial ecotoxicity of biochars in agricultural soil and interactions with linear alkylbenzene sulfonates. Agronomy 11, 828.

Orr CH, Komang Ralebitso-Senior T, Prior S, 2016. Chapter 8 - Microbial Ecology of the Rhizosphere and Its Response to Biochar Augmentation. In: Biochar Application, (Eds: Komang Ralebitso-Senior T, Orr CH), pp 199-220.

Palansooriya KN, Ok YS, Awad YM, Lee SS, Sung JK, Koutsospyros A, Moon DH, 2019. Impacts of biochar application on upland agriculture: A review. J Environ Manage 234, 52-64.

Sanchez-Hernandez JC, Rios JM, Maximiliano Attademo A, Malcevschi A, Andrade Cares X, 2019. Assessing biochar impact on earthworms: Implications for soil quality promotion. J Hazard Mater 366: 582–591.

Sheng H, Yin Y, Xiang L, Wang Z, Harindintwali JD, Cheng J, Ge J, Zhang L, Jiang X, Yu X, Wang F, 2022. Sorption of N-acyl homoserine lactones on maize straw derived biochars: Characterization, kinetics and isotherm analysis. Chemosphere 299, 134446.

Siedt M, Schäffer A, Smith KEC, Nabel M, Roß-Nickoll M, van Dongen JT, 2021. Comparing straw, compost, and biochar regarding their suitability as agricultural soil amendments to affect soil structure, nutrient leaching, microbial communities, and the fate of pesticides. Sci Tot Environ 751, 141607.

Silvani L, Cornelissen G, Smebye AB, Zhang Y, Okkenhaug G, Zimmerman AR, Thune G, Sævarsson H, Hale SE. 2019. Can biochar and designer biochar be used to remediate per- and polyfluorinated alkyl substances (PFAS) and lead and antimony contaminated soils? Science of The Total Environment, 694, 133693, https://doi.org/10.1016/j.scitotenv.2019.133693.

Sun Z, Bruun EW, Arthur E, de Jonge LW, Moldrup P, Haugaard-Nielsen H, Elsgaard L, 2014. Effect of biochar on aerobic processes, enzyme activity, and crop yields in two sandy loam soils. Biol Fertil Soils 50, 1087-1097.

Xiang L, Liu S, Ye S, Yang H, Song B, Qin F, Shen M, Tan C, Zeng G, Tan X, 2021. Potential hazards of biochar: The negative environmental impacts of biochar applications. J. Hazard Mater 420, 126611.

Yang Q, Ravnskov S, Pullens JWM, Andersen MN, 2022. Interactions between biochar, arbuscular mycorrhizal fungi and photosynthetic processes in potato (*Solanum tuberosum* L.). Sci Tot Environ 816, 151649.

Zhang Q, Li S, Saleem M, Ali MY, Xiang J, 2021. Biochar and earthworms synergistically improve soil structure, microbial abundance, activities and pyraclostrobin degradation. Appl Soil Ecol 168, 104154.

Zhao L, Nan H, Kan Y, Xu X, Qiu H, Cao X, 2019. Infiltration behavior of heavy metals in runoff through soil amended with biochar as bulking agent, Environmental Pollution, 254, Part B, 113114, https://doi.org/10.1016/j.envpol.2019.113114.

6 Effects of biochar on the soil greenhouse gas emission balance

Diego Abalos and Henrik Thers (reviewer, Lars Elsgaard)

6.1 Carbon storage

Biochar has a C sequestration potential when applied to soil (Greenberg et al. 2019; Joseph et al. 2021; Chagas et al. 2022), which is due to the stabilization of biochar C in the pyrolysis process. This results in slower degradation of biochar C as compared to the untreated initial biomass (Bruun et al. 2012; Hansen et al. 2015). Studies have shown that this is the case, even though a part of the initial biomass C is lost in the pyrolysis process (Thers et al. 2019; Thomsen 2021). In an assessment of optimal conditions for the performance of greenhouse gas removal methods, Asibor et al. (2021) found biochar to be suitable for removing C from the atmosphere in temperate zones, although based on an assumption of a mean residence time (MRT) of 1000-4000 years for the biochar. However, despite the biophysical potential for soil C sequestration, the feasibility of biochar as a global tool is believed to be limited due to limitations on pyrolysis facilities and biomass (Asibor et al. 2021). In addition, attention should be payed to whether the biomass collected for pyrolysis purposes already are providing ecosystem services, which could differ between regions (Gelardi & Parikh 2021).

The recalcitrant nature of biochar is believed to result mainly from intrinsic chemical properties in terms of aromatic structures as a consequence of carbonization in the heating process (Joseph et al. 2021). However, organo-mineral complexes are also important for biochar-C stability in soil (Fang et al. 2014).

6.1.1 Biochar stability and degradation

Multiple factors are believed to influence the degradation rate and profile of biochar (e.g., the proportion of the biochar C that is emitted as CO_2 shortly after field application). These include both abiotic and biotic processes, which are affected by for instance soil type and soil environmental conditions e.g., clay content, soil moisture and temperature, physical effects on biochar of for instance the management (e.g., soil tillage) and biochar characteristics, which depend on the feedstock biomass and pyrolysis conditions (Bruun et al. 2011; Bruun et al. 2012; Bruun et al. 2014; Fang et al. 2014; Wang et al. 2016; Ippolito et al. 2020; Wang et al. 2022).

The variety of conditions affecting biochar stability/degradation results in an extreme range in reported degradation rates, which are main controllers of the biochar C sequestration potential. Most research on biochar degradation has been conducted as laboratory studies, which has the drawback that it does not include all the above factors, like for instance physical impact and freeze-thaw cycles (Wang et al. 2022).

Higher pyrolysis process temperature is generally correlated to increased biochar-C stability (Fang et al. 2014; Joseph et al. 2021), and there are indications pointing at a lower C stability for manure based biochars as compared to, e.g., herbaceous based biochars (Joseph et al. 2021).

As a simplification, it is generally conceived that biochar (before soil amendment) consists of two pools – a minor labile fraction and a larger recalcitrant fraction (Fang et al. 2014; IPCC 2019; Leng et al. 2019; Wang et al. 2022). Bruun et al. (2011) found that the proportion of biochar C emitted within 115 days increased from 3 to 12% in biochar treated by 575 to 475°C, respectively, in a fast pyrolysis process. This result supports the relationship between increasing pyrolysis temperature and decreasing labile fraction. A global meta-analysis reported average pool sizes to be 3% for the labile fraction and 97% for the recalcitrant fraction (Wang et al. 2016). This means, e.g., that 1000 Kg of dry mass biochar with a C content of 40% would contain 12 kg labile C and 388 kg stabile C, equal to 0.044 and 1.423 Mg CO_2eq, respectively.

It is often suggested that the degradation profile of biochar C can be expressed by an exponentially decreasing function. Therefore, the remaining fraction of the biochar C in the soil after a certain period of time of can be described by equation 6.1 for a one-pool model of biochar (e.g., Bai et al. 2014):

$$C_{rem}(t) = C_0 \times e^{-kt} \qquad \text{(Eq. 6.1)}$$

where $C_{rem}(t)$ is the remaining biochar C in the soil after *t* years since application, C_0 is the initial amount of applied biochar C, and k is the degradation rate constant. For a two-pool biochar, the equation becomes:

$$C_{rem}(t) = C_1 \times e^{-k1\,t} + C_2 \times e^{-k2\,t} \qquad \text{(Eq. 6.2)}$$

where $C_{rem}(t)$ is the remaining biochar C in the soil after *t* years since application, C_1 is the initial amount of the labile fraction of the applied biochar C, k_1 is the degradation rate constant for the labile fraction, C_2 is the initial amount of the recalcitrant fraction of the applied biochar C, and k_2 is the degradation rate constant for the recalcitrant fraction.

In the scientific terminology, different terms are used to express degradation times, such as mean residence time (MRT), half-life and the degradation rate constant k (from above equations 6.1 and 6.2). These are mathematically related by the following formulas:

$$\text{MRT} = 1/k \qquad \text{(Eq. 6.3)}$$

$$\text{Half-life} = \ln(2)/k \qquad \text{(Eq. 6.4)}$$

which in combination means that the half-life is approximately 70% of MRT.

6.1.2 Degradation rates

In a long-term laboratory experiment on biochar degradation (running for 8.5 years), It was found that the MRT was 402 years for biochar made from perennial ryegrass (*Lolium perenne*) (Kuzyakov et al. 2014). Considering that the results were obtained under laboratory conditions, where decomposition is faster than under field conditions, it was estimated that the MRT of the biochar in field soils of temperate climates would be about 4000 years, although this extrapolation is reported as speculative and needs further confirmation (Kuzyakov et al. 2014). In comparison, a meta-analysis found average MRT's to be 108 days and 556 years for the labile and recalcitrant fractions of biochar, respectively, based on 24 incubation and field studies employing stable isotopic techniques (^{13}C natural abundance or ^{13}C labeling) or ^{14}C labeling (Wang et al. 2016), although this was not related to specific feedstock and pyrolysis conditions. In order to connect biochar degradation rates to specific biochar properties, a large number of studies have examined the relationship between biochar stability and biochar molar H/C and O/C ratios.

The relationship between biochar degradation rates and biochar properties in terms of O/C ratio, H/C ratio and ash content as well as the highest temperature treatment (HTT) in the pyrolysis process were examined in a study by Bai et al. (2014). Here, the O/C molar ratio was found to have the strongest correlation with the degradation rate, although it was suggested that the H/C molar ratio could be a better indicator of the stability of biochars that were subjected to long-time aging in soil. Spokas (2010) also pointed at the O/C ratio as a valuable predictor of biochar stability and listed that biochar with an O/C molar ratio of less than 0.2 was typically the most stable, with an estimated half-life of more than 1000 years; biochar with an O/C ratio of 0.2–0.6 had intermediate half-lives (100–1000 years); and finally, biochar with an O/C ratio greater than 0.6 had a half-life in the order of over 100 years. Another study found a correlation ($r = 0.73$) between O/C ratio and an accelerated ageing method, namely H_2O_2 treatment, which was used as a proxy for the oxidative biochar degradation in soil (Cross & Sohi 2013). A drawback for the O/C ratio could be that the O content is normally not measured but estimated by difference, which may lead to an overestimation of the O content (Wang et al. 2022). Joseph et al. (2021) pointed at the H/C ratio as a simple and reliable parameter for characterizing biochar persistence.

As an alternative to H/C and O/C, the molar ratios of H, O and the C in the organic proportion of biochar (i.e. H/C_{org} and O/C_{org}) was applied, which excludes the inorganic biochar C, such as carbonates (Fidel et al. 2017). Lehmann et al. (2021) examined relationships between reported values on biochar stability and pyrolysis temperature and the biochar properties of molar H/C_{org} and O/C_{org} ratios, and found that the H/C_{org} ratio could explain the highest proportion of the variance ($r^2 = 0.33$) in the dataset. Based on those results, they developed the following linear formula:

$$BC_{100} = -63.3H/C_{org} + 104.6 \qquad \text{(Eq. 6.5)}$$

where BC_{100} is the percentage (%) of initial biochar C remaining in the soil 100 years after biochar application. Budai et al. (2013) reported a similar relationship (r^2 = 0.50):

$$BC_{100} = -74.3H/C_{org} + 110.2 \qquad \text{(Eq. 6.6)}$$

From this relationship, Budai et al. (2013) derived conservative estimates of biochar stability after 100 years: 70% for H/C_{org} ratios of 0.4 or lower, and 50% for ratios within the range of 0.4 to 0.7. A large meta-analysis on biochar properties reported that average C, H and O contents are dependent on pyrolysis type, feedstock source and pyrolysis temperature (Ippolito et al. 2020). From the reported C, H and O contents obtained in that study, molar H/C and O/C ratios were calculated by the authors of this chapter and presented in Table 6.1. From this it can be seen that biochar from crop wastes (i.e., straw residues) generally have lower H/C ratios as compared to biochars from manures/biosolids, and thus also must be expected to have a slower degradation and thus a higher stability. Data on biochar based on sewage sludge was not available in the analysis. According to the International Biochar Initiative (IBI, 2022) and the European Biochar Certificate (EBC, 2022), biochars need to have H/C_{org} ratios below or equal to 0.7 to be considered biochar. Although the listed values are H/C ratios and not H/C_{org} ratios, the reported values underpin that focus should be on individual batches of biochar to secure that they comply with biochar definitions, which is not automatically the case for pyrolysis outputs.

.

Table 6.1. Values of H/C and O/C ratios recalculated from Ippolito et al. (2020)

Category	Molar H/C ratio	Molar O/C ratio
Pyrolysis type		
Fast	0.66	0.24
Slow	0.66	0.23
Feedstock source		
Wood based	0.57	0.19
Crop wastes	0.64	0.22
Other grasses	0.96	0.25
Manures/biosolids	0.78	0.30
Pyrolysis Temp (°C)		
< 300	1.24	0.42
300–399	0.97	0.31
400–499	0.72	0.22
500–599	0.54	0.17
600–699	0.42	0.14
700–799	0.33	0.15
> 800	0.28	0.12

The IPPC has proposed a preliminary framework to estimate biochar stability in relation to pyrolysis temperature on a 100-years scale (Figure 6.1). According to this, the fraction of biochar remaining in soil after 100 years (F_{perm}; equivalent to BC_{100}) for the pyrolysis temperature groups of 350-450, 450-600, and >600 ºC, can be tentatively estimated to 65, 80, and 89%, respectively (IPCC 2019). A function for calculating biochar stability was also supplied in the figure shown by IPCC (Figure 6.1, inserted below):

$$F_{perm} = 0.000173(T) - 0.346 \qquad \text{(Eq. 6.7)}$$

where F_{perm} is the fraction of biochar C remaining in the soil after 100 years and T is the pyrolysis temperature (°C). However, we find that this equation is erroneous and should not be used (there is an error the linear regression coefficients). Yet, this error is not propagated in the cited F_{perm} values by IPCC.

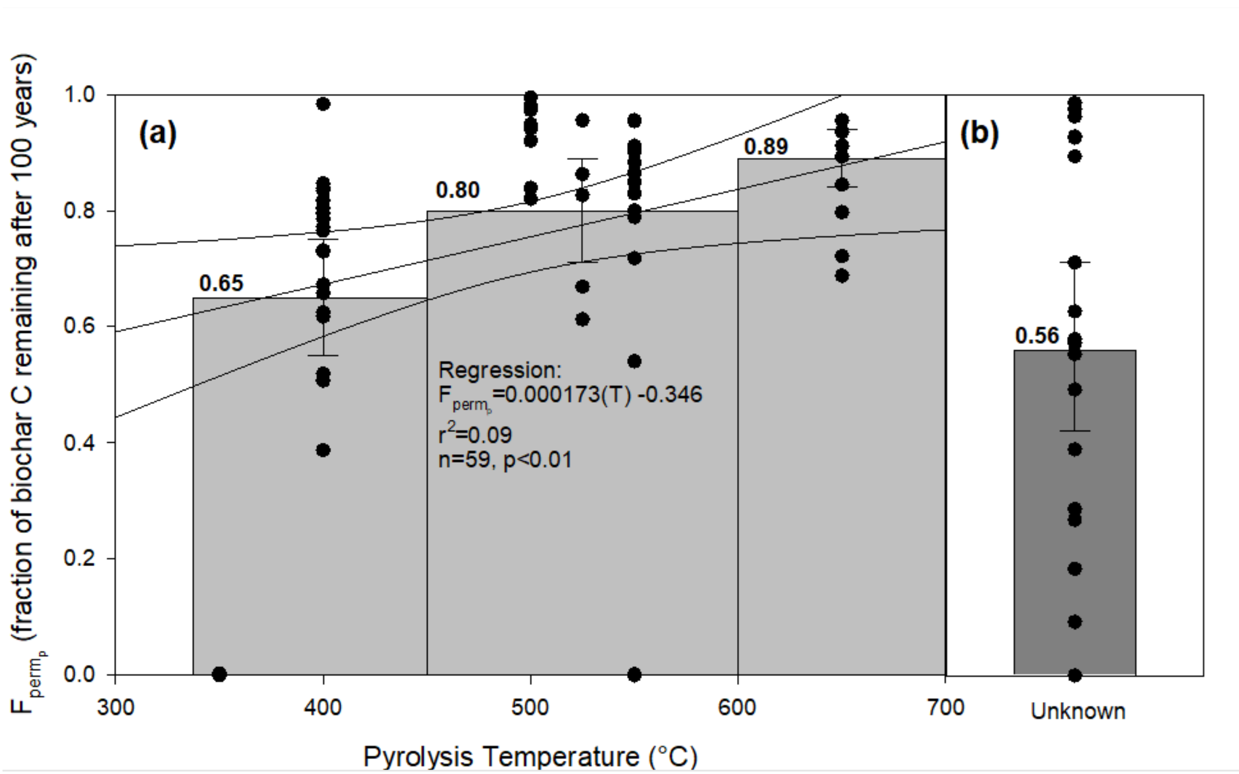

Figure 6.1. (a) Relationship between pyrolysis temperature (ºC) and biochar stability. F_{perm} is the fraction of biochar remaining in the soil after 100 years. Pyrolysis temperature is divided into three temperature groups, namely 350-450, 450-600 and >600 ºC. (b) Biochars produced at unknown temperatures for instance in a wildfire event. Figure from IPCC (2019).

The IPCC approach represents a very simple way to estimate C sequestration from biochar, since pyrolysis temperature is often known and does not require laboratory analysis. In addition, the effect of feedstock type is ignored and so is the retention time at the respective pyrolysis temperature. The values are only valid for biochar application on mineral soils in croplands and grasslands. Woolf et al. (2021) pointed out that although the IPCC approach was presented as an annex to the 2019 refinement report on IPCC GHG Guidelines, it is still not a part of good practice for national inventories due to lack of sufficient support in the scientific literature. In addition, Woolf et al. (2021) extended the dataset used in the IPCC (2019) report and added information on the H/C_{org} ratio and provided two equations for calculating biochar stability after 100 years, calibrated to soil temperatures of 14.9 ºC (representing the world average). One equation was based on the H/C_{org} ratio (approximately similar to equation 6.5 retrieved from Lehmann et al. 2021):

$$F_{perm} = -0.635H/C_{org} + 1.04 \qquad \text{(Eq. 6.8)}$$

where F_{perm} is the remaining fraction of biochar C in soil after 100 years. The other equation was based on pyrolysis temperature:

$$F_{perm} = 0.000856(T) + 0.28 \qquad \text{(Eq. 6.9)}$$

where F_{perm} is the remaining fraction of biochar C in soil after 100 years and T is pyrolysis temperature (°C). Woolf et al. (2021) recommended to include data on H/C_{org} if possible and thus to prioritize equation 6.8 over equation 6.9.

Furthermore, Woolf et al. (2021) calculated biochar stability at five soil temperatures (i.e. 5, 10, 15, 20, and 25 ºC) for different feedstock and time perspectives (100, 500, and 1000 years), revealing that soil temperature has a strong influence on biochar stability. For instance, in a 100-year time perspective, soil temperature of 15 ºC results in biochar stability of 6-9 percentage points less than soil temperature of 10 ºC across the low, medium and high pyrolysis-temperature classes. A tentatively derived Danish average soil temperature (10 cm depth under grass) of 9.8 ºC (range: 9.3 – 10.2 ºC) was calculated as a simple five year (2017 - 2021) average based on seven climate stations. Although this should not be taken as a scientifically derived value, it does indicate that the 10 ºC class fits better Danish conditions that the 5 or 15 ºC classes, which may be kept in mind when deriving biochar stability values under Danish conditions. Despite the various suggestions, it has been stated that no method can yet support accurate estimation of biochar stability over time (Leng et al. 2019). The data at hand so far relates largely to research on biochar that is conducted internationally. Further studies are needed under Danish pedoclimatic conditions and with the most common feedstock sources available in Denmark to have a better understanding of biochars stability under Danish agroecological conditions. Such research is now ongoing, e.g., in the SkyClean Scale-up project funded by EUDP (2022-2025).

6.1.3 Biochar and native soil organic matter (SOM)

It is believed that sequential additions of biochar C will continue to build C stocks in soil, whereas un-pyrolysed organic matter (e.g., plant residues and manure) will be rapidly mineralized and will increase C stocks only until an equilibrium is reached, in which decomposition equals the input rate (Joseph et al. 2021). Nevertheless, the potential effect of biochar application on native soil organic matter (i.e., priming effect) has been examined in several studies. Priming deals with the phenomena of biochar application altering the degradation rate of the existing SOM pool. It is named positive priming when biochar increases degradation rates of the existing SOM pool, and negative priming when biochar decreases the degradation rate of the existing SOM pool.

Bruun and EL-Zehery (2012) examined potential priming effects of biochar on existing SOM that was isotope-labeled 40 years ago, and found no significant positive or negative priming effects in a lab incubation experiment lasting 451 days. Wang et al. (2016) reported an average negative priming of 3.8%, based on a meta-analysis. In addition, various examples of both positive and negative priming have been reported in the literature (Fang et al. 2014; Gelardi & Parikh 2021; Joseph et al. 2021), and especially a potential positive priming on SOM on sandy soils, which are very common in Denmark, has been suggested (Wang et al. 2016). More knowledge is needed in order to be able to include the biochar priming effects in C sequestration calculations and modelling.

6.1.4 Co-application of biochar and organic material

A practical suggestion for biochar application has been co-amendment with liquid manure, which could also deal with dust problems connected to application at field scale. However, the application rate that can be applied must be assumed to be lowered as compared to application of pure biochar. Some studies found interactions from co-application of biochar with easily degradable organic compounds (glucose) pointing at an increased biochar decomposition/mineralization when co-applied (Hamer et al. 2004; Kuzyakov et al. 2009). However, there is no clear evidence of whether these interactions are also valid for co-application with manure, and whether a potentially faster decomposition of biochar would be long- or short-term. More knowledge is needed to clarify whether co-application of biochar and organic fertilizers/amendments affect the degradation rate for either of the co-applied products.

6.1.5 Cleaching from biochar in soil

Some of the biochar C can be leached after field application. Bruun et al. (2012b) compared biochar C leaching from biochar applied to repacked sandy soil columns. The three biochars were produced from wheat straw (*Triticum aestivum* L.), two resulting from fast pyrolysis and one from slow pyrolysis. The study found high mobility of C compounds from fast pyrolysis biochar and none from slow pyrolysis biochar. The finding was explained by the content of dissolved (or fine particulate <0.20 μm) C from biochars made from fast pyrolysis. The C leaching from the fast pyrolysis biochars was found to increase proportionally with the content of extractable C (dissolved organic carbon; DOC) and carbohydrates. In comparison, the slow pyrolysis biochar contained limited amounts of DOC and no carbohydrates, which was suggested as the reason for no C leaching in this case. Thus, it appears that to avoid biochar C leaching, the pyrolysis conditions should aim for biochar output with likewise limited or no content of DOC and carbohydrates.

6.1.6 Degradation of biomass feedstock

In order to account for the 'lost' C sequestration when biomass feedstock is removed from the field (here for pyrolysis purposes) instead of being left in or incorporated into the field, information on the degradation rates for parental feedstock is needed. According to the Danish model C tool (Taghizadeh-Toosi et al. 2014), the non-degraded proportion of straw C after 20 and 100 years are in the range of 11-17% and 2.5-4%, respectively, across a span of clay contents covering most Danish soils (JB 1-8), and with simplified assumptions on future air temperatures (kept constant representing historical temperature at Foulum; Denmark) and soil C/N ratio set to 10 (Jensen 2022). The corresponding values for the non-degraded proportion of manure C are equal or higher than for straw. However, the parameter settings for manure in C-TOOL does assumingly need calibration in order to apply to biogas-digested manure (Hansen et al. 2020). The potential "lost" C sequestration from not incorporating this feedstock into the soil needs to be accounted for when estimating the biochar C sequestration potential, however, this requires a life cycle assessment approach which is beyond the scope for this synthesis.

6.1.7 Approaches for including biochar C sequestration in inventories and LCA

When including biochar C sequestration in life cycle assessments (LCA), most often a specific time perspective is included since it is too difficult to account for the running biochar C stock in soil (IPCC 2019). Time spans of 20 and 100 years are the normal choices although values for biochar stability across 30, 500 and 1000 years are sometimes also reported. In most cases, the reported values are based on the actual remaining biochar fraction in the soil. However, the C sequestration can also be reported as the avoided atmospheric CO_2 load, which includes the temporal distribution of CO_2 emissions from the degradation of organic material as well as the Bern Cycle, i.e., the decay pattern of CO_2 in the atmosphere (Petersen et al. 2013). Thers et al. (2019) applied and explained the method in relation to biochar C sequestration and found the remaining proportion of C in soil after 100 years for two generic biochars to be 57 and 79%, whereas the avoided atmospheric CO_2 load was reported as 72 and 87%, thus being 15 and 8 percentage point higher that concluded from the C sequestration as such. This suggests that the actual climate effect from the C sequestration is higher as compared to the proportion of C remaining in the soil after, e.g., 100 years. However, this methodology is not part of the current IPCC approach and care should be taken when the method is applied.

In relation to the concept of using biochar C stability after 100 years, Woolf et al. (2021) states that: *"If biochar were to form a substantial component of mitigation efforts over the coming century, then future inventory systems in the 22nd century and onward would need to recognize the ongoing emissions from biochar decay as a net CO_2 source."* This reveals some still unresolved challenges in relation to accounting for biochar C sequestration, which require further detailed studies in LCA perspectives.

The Danish Centre for Environment and Energy, Aarhus University, (DCE) has previously described potentials and limitations in the IPCC methodology for inclusion of biochar C sequestration in the Danish national inventory (Nielsen 2022). As discussed in section 6.1.2, IPCC presents a Tier 1 approach in the IPCC (2019) report, and furthermore indicate a range of parameters that should be considered if a higher Tier approach should be applied. The presented Tier 1 approach is simplistic and DCE states that the scientific robustness behind the standard values are too vague to be used in the Danish national inventory and thus a more comprehensive development work is needed.

6.2 Nitrous oxide emissions

6.2.1 Mechanisms of biochar mitigation of N_2O emissions

Biochar has been found to mitigate soil N_2O emissions in lab and field experiments globally (Cayuela et al. 2015; He et al. 2017; Borchard et al. 2019). Multiple mechanisms have been suggested to explain how biochar reduces N_2O emissions, though no single mode of action seems to apply (Cayuela et al. 2014). Several meta-analyses have synthesized the response of soil N_2O emissions to biochar application. The effects of biochar in N_2O mitigation were reported to be on average 28% for laboratory studies and 54% for field studies (Cayuela et al. 2015), whereas He et al. (2017) reported average N_2O reductions of 33% for fertilized soils due to biochar application. Borchard et al. (2019) concluded an overall 38% reduction on N_2O, but indicating that the effect after more than one year from field application was uncertain and maybe negligible, i.e., indicating that the effect of biochar on mitigation of N_2O emissions was not persistent. Verhoeven et al. (2017) reported an average mitigation of 12.4% from field studies only and reported that the significance of the mitigation was dependent on the method applied in the data analysis.

The hypotheses invoked to explain the N_2O mitigation effect of biochar comprise increased soil pH (liming effect) (Lehmann 2007; Horák et al. 2021), reduction of inorganic soil N (Clough et al. 2013), increased cation exchange capacity (CEC) (Liang et al. 2006), enhanced soil aeration (Case et al. 2012), addition of labile C to soil (Cayuela et al. 2014), alteration of microbial activities (Lehmann et al. 2011), release of toxic/inhibitory compounds (Wang et al. 2013), biochar acting as an electron shuttle (Cayuela et al. 2013) and adsorption of N_2O (Cornelissen et al. 2013; He et al. 2019).

The liming effect is due to the predominant alkaline nature of biochar, although acidic biochar types also occur (see, e.g., Chapter 2, Figure 2.3), especially depending on the feedstock and pyrolysis temperature (Yuan et al. 2011), where increasing temperature generally entails increasing pH (Ippolito et al. 2020). The soil pH has been shown to affect the nitrification and denitrification processes and the proportional N_2O output, since increased pH led to decreases in both $N_2O/(NO_2^- + NO_3^-)$ and $N_2O/(N_2 + N_2O)$ ratios (Simek & Cooper 2002; Mørkved et al. 2007). The logic outcome of this is that if biochar should mitigate N_2O due to the liming hypothesis, the combined effect of biochar pH and application rate needs to be large enough to increase soil pH, which may not always be the case (Thers et al. 2020).

Reduction of inorganic soil N concentration when biochar is used as a soil amendment is caused either by biochar induced microbial immobilization or by adsorption (Clough et al. 2013). Adsorption of inorganic soil N (i.e., NH_4^+ and NO_3^-) could be due to chemical adsorption or physical entrapment. The chemical adsorption of the positive charged NH_4^+ ions are closely related to the hypothesis of increased CEC following biochar amendment (Clough et al. 2013), whereas the sorption of NO_3^- has been found to be caused by ionic binding with basic functional groups (Kameyama et al. 2012; Clough et al. 2013). Both NH_4^+ and NO_3^- might be physically entrapped into the biochar pore structure (Clough et al. 2013; Haider et al. 2017). The corresponding decrease of inorganic soil N caused by the above mechanisms presumably lead to N_2O mitigation since inorganic N is the main substrate for N_2O-producing microorganisms (Butterbach-Bahl et al. 2013).

Enhanced soil aeration from biochar field application is believed to be due to the highly porous structure of biochar, which decreases soil bulk density and thus increases soil water holding capacity, although the biochar application rates apparently need to be substantial (Rogovska et al. 2011; Case et al. 2012; Clough et al. 2013). Fine grained-biochar application to coarse sandy soils has been reported to strongly increase the available water capacity, which is suggested to be mainly caused by fine biochar particles forming interstitial soil pores, which converts drainable pores into moisture-retaining pores (Bruun et al. 2021). The possible impact of biochar on soil aeration could affect N_2O emissions, since both denitrification and N_2O production from nitrification are affected by soil oxygen and moisture conditions (Granli & Bøckman 1994).

Biochar soil amendment entails addition of a labile C pool, although minor as compared to the larger recalcitrant biochar C pool (Wang et al. 2016). In addition, biochar might affect the mineralization of existing SOC (as treated in the above section on priming). In cases where denitrification is C limited, the additional available C will increase denitrification and possibly N_2O emissions, although the extra C source could also increase the final process of N_2O reduction to N_2 and thereby mitigate N_2O formation (Granli & Bøckman 1994; Morley & Baggs 2010). The effect of biochar on the soil labile C pool is believed to be rather short-lived, such as weeks or possibly a few years (Zimmerman 2010; Bruun, Müller-Stöver, et al. 2011; Singh et al. 2012) and is suggested to be correlated to the biochar volatile matter content (Ameloot et al. 2013).

Polycyclic aromatic hydrocarbons (PAH) has been suggested to play an important role in N_2O mitigation in soils, since they might inhibit nitrifier and denitrifier activities (Wang et al. 2013). A more recent study could not confirm these findings, and suggested instead that other structural and compositional properties were causing the biochar mitigating effect on soil N_2O emissions (Alburquerque et al. 2015).

A conceptual distinction between manure fiber-based and straw and wood-based (lignocellulosic) biochar has been suggested, since the properties differs systematically, for example regarding nutrient contents, such as N and P (Novak & Johnson 2019; Ippolito et al. 2020). This indicates that such biochars interact differently with soils in relation to the above mentioned hypotheses for N_2O mitigation mechanisms, and the lignocellulosic biochar should generally be better suited for N_2O mitigation (Mandal et al. 2016).

Racek et al. (2020) reported C contents in biochar produced from sewage sludge from a wide range of pyrolysis conditions to be 10-40%, indicating that using sewage sludge as feedstock results in relatively lower C contents. Molar H/C ratios from the same study were reported to be in the range 0.03 – 2.35, revealing a huge span and stressing the need to have concrete biochar batches analysed before the effects of those biochars for field application can be assessed, including the potential effect on N_2O emissions.

Despite the general finding of N_2O mitigation following biochar application, some studies reported no effect (Wang et al. 2015) or increased N_2O emissions due to biochar application (Saarnio et al. 2013). Borchard et al. (2019) summarized that biochars produced by slow pyrolysis, with high degree of carbonization (low H/C_{org} molar ratio), high pH, and high surface area, are most effective in suppressing N_2O emissions and in addition, a correlation between dose and mitigation effect is believed to be evident. The importance of pH, surface area and H/C ratio was confirmed by a Norwegian lab study testing biochar produced from four temperatures and tested on two soil types (Weldon et al. 2019).

6.2.2 Field experiments

Bamminger et al. (2018) applied 30 Mg ha^{-1} miscanthus (*Miscanthus × giganteus*) biochar in a German field study at both ambient temperatures and temperatures elevated by 2.5°C. Across the two-year study period, no significant N_2O mitigating effect of biochar was concluded. When focusing on shorter periods within the full study period, both increased and decreased effects from biochar on N_2O emissions were reported. A Slovak field trail reported significant reduction by applying 20 Mg ha^{-1} biochar and 80 kg N (applied as calcium-ammonium nitrate), which could be explained by a soil pH increase of 0.9 units due to biochar application, from approximately pH 5 to 6 (Horák et al. 2017). A Swiss field trial compared liming to biochar application and found a 52% N_2O mitigation from the biochar treatment as compared to the control and no mitigation from the limed treatment as compared to the control, although showing great variation, indicating that the liming theory did not apply to the mitigation potential found in that study (Hüppi et al. 2015). A Swiss lysimeter study found a 15% N_2O mitigation from applying 20 Mg ha^{-1} biochar to silty and sandy loam soils (Hüppi et al. 2016). A thorough Danish field study including 65 measuring campaigns across 402 days found no mitigating effect of 1.5 and 15 Mg ha^{-1} biochar from wheat straw and manure fibers when this biochar was applied immediately before and approximately two years before the study year (Thers et al. 2020). However, the study was only conducted on one location in one growing season (winter oilseed rape). A German field trial applying 10 Mg ha^{-1} biochar reported a significant reduction of N_2O at an over-sufficient fertilizer N level of 195 kg N ha^{-1}, but not at N levels of 75 and 150 kg N ha^{-1} (Sun et al. 2017). In conclusion, there is a further need for comprehensive studies that clarifies the conditions under which biochar mitigates N_2O emissions at field scale, in order to improve our understanding of the mechanisms behind such reductions, and how they can be optimized under Danish conditions.

6.2.3 Co-application with manure

Combining biochar application with application of organic fertilizers could potentially mitigate the N_2O emissions from fertilizer application. A Danish incubation study found that a 1% (w/w) biochar application increased N_2O emissions after slurry application during 55 days, whereas a 3% biochar application reduced the N_2O emissions, however, none of them significantly (Bruun, Müller-Stöver, et al. 2011). In comparison, a Mexican incubation study examining co-application of manure and urea and waste water sewage, lasting 45 days, did not find a N_2O mitigating potential from biochar (Díaz-Rojas et al. 2014). In a German field trial, Dicke et al. (2015) found a reduction in N_2O emissions when applying biogas digestate and biochar to a soil as compared to digestate alone, although the reduction was not significant. A Danish field study with biochar added to cattle and pig slurry and the applied to soil (at a rate of approximately 1 Mg biochar ha^{-1}) found no mitigating effect on N_2O (Khanal 2011).

6.2.4 N_2O field emissions from application of the potential biochar parental material

Crop residues and livestock manure are common feedstocks used for biochar production. Both organic materials are often incorporated into the soil, leading to increases in N_2O emissions. For example, crop residue incorporation increases soil N_2O emissions by 40-50% compared to removing the residues from the field (Abalos et al. 2022). The stimulation of N_2O emissions caused by manure application can be sufficiently large to offset the benefits of manure for soil C sequestration (Zhou et al. 2017). Therefore, a simple strategy to reduce N_2O emissions from crop residues and from manure in the field would be removing the crop residues and not applying manure, thus eliminating the supply of N and C compounds, which would otherwise fuel soil microbial processes. However, crop residue removal and avoiding manure application cannot be recommended as beneficial management practices, because they provide other agroecosystem services, and because it would reduce the C inputs into agricultural soils. Turning these materials into biochar, which is subsequently applied to the field, has potential to reduce the associated N_2O emissions induced by the parental feedstock materials (mechanisms explained above), while avoiding the negative reduction in C inputs needed to increase soil C stocks.

For sewage sludge feedstock, there are N_2O emissions consideration in relation to storage, which need to be included in an overall analysis of the N_2O mitigation potential from producing and applying biochar from sewage sludge on agricultural soils (Willén et al. 2016).

6.3 Methane oxidation

Methane can be produced in water-logged anoxic soils by methanogenic archaea via methanogenesis (Conrad 2007), while well-aerated upland soils are frequently biological sinks for atmospheric CH_4 (Boone et al. 1993; Dunfield 2007). Biochar can increase (Zhang et al. 2010; Spokas & Bogner 2011), decrease (Feng et al. 2012; Reddy et al. 2014), or have no effect (Kammann et al. 2012) on CH_4 fluxes from soils. The main mechanisms by which biochar may affect CH_4 fluxes remain unclear, although several hypotheses have been proposed. Examples include sorption of CH_4 to biochar surfaces potentially reducing CH_4 emissions (Yaghoubi et al. 2014), and increased soil aeration after biochar addition, which may enhance diffusive CH_4 uptake (van Zwieten et al. 2010; Karhu et al. 2011). Conversely, the labile C fraction in biochar may serve as a substrate for methanogenesis in anaerobic environments, stimulating CH_4 production (Wang et al. 2012). Biochar has also been shown to promote methanotrophic CH_4 consumption in oxic/anoxic interfaces in anoxic environments, decreasing CH_4 emissions due to the “biofilter” function of CH_4 consumption (Feng et al. 2012; Reddy et al. 2014).

A recent meta-analysis synthesized the frequently contradicting information regarding biochar effects on CH_4 fluxes, revealing that biochar has the potential to generally mitigate CH_4 emissions from flooded soils (i.e. paddy fields), whereas addition of biochar to neutral and alkaline soils that do not have periods of flooding may have the potential to decrease the CH_4 sink strength of such soils (Jeffery et al. 2016). Denmark does not have paddy fields, and therefore the overall effect on CH_4 of biochar application is expected to be negative (i.e., decrease the capacity of agricultural soils to consume atmospheric CH_4). Although N_2O emissions from soils are far more important than the soil CH_4 sink function, and therefore the positive impact of biochar on soil C stocks and N_2O emissions overrides any negative effect on CH_4 fluxes, further research should try to find ways to improve the potentially detrimental consequences of biochar for CH_4 consumption under Danish conditions, or at least to document if such negative effects exists.

6.4 Conclusions

The H/C_{org} molar ratio should be reported for biochar batches since it is an important proxy for the biochar degradation rate, and furthermore, it can also be a proxy of biochars N_2O mitigation capacity. Further, it is important to include contents of total C (C_{tot}), organic C (C_{org}), H and O as well as analyses of DOC and carbohydrate content in the standard declaration of biochar. Long-term research on biochar degradation in soil is needed, including more knowledge on biochar field aging processes (Woolf et al. 2021; Wang et al. 2022). Despite overall international findings on N_2O mitigation from biochar soil application, there is a lack of evidence when it comes to Danish field experiments and thus, this could be prioritized in order to get more knowledge on whether an N_2O reduction can be assumed when applying biochar to fields in Denmark. On the other hand, there are only very few indications pointing at increased N_2O emissions after biochar application, which therefore could be neglected. This means that conversion of biomass C to biochar before field application may at least avoid the N_2O emission often associated with application of fresh biomass residues.

6.5 References

Abalos, D., Recous, S., Butterbach-Bahl, K., De Notaris, C., Rittl, T.F., Topp, C.F.E., Petersen, S.O., Hansen, S., Bleken, M.A., Rees, R.M., & Olesen, J.E. 2022. A review and meta-analysis of mitigation measures for nitrous oxide emissions from crop residues. Science of the Total Environment 828: 154388.

Alburquerque, J.A., Sánchez-Monedero, M.A., Roig, A., & Cayuela, M.L. 2015. High concentrations of polycyclic aromatic hydrocarbons (naphthalene, phenanthrene and pyrene) failed to explain biochar's capacity to reduce soil nitrous oxide emissions. Environmental Pollution 196: 72–77.

Ameloot, N., De Neve, S., Jegajeevagan, K., Yildiz, G., Buchan, D., Funkuin, Y.N., Prins, W., Bouckaert, L., & Sleutel, S. 2013. Short-term CO_2 and N_2O emissions and microbial properties of biochar amended sandy loam soils. Soil Biology and Biochemistry 57: 401–410.

Asibor, J.O., Clough, P.T., Nabavi, S.A., & Manovic, V. 2021. Assessment of optimal conditions for the performance of greenhouse gas removal methods. Journal of Environmental Management 294: 113039.

Bai, M., Wilske, B., Buegger, F., Bruun, E.W., Bach, M., Frede, H.G., & Breuer, L. 2014. Biodegradation measurements confirm the predictive value of the O/C-ratio for biochar recalcitrance. Journal of Plant Nutrition and Soil Science 177: 633–637.

Bamminger, C., Poll, C., & Marhan, S. 2018. Offsetting global warming-induced elevated greenhouse gas emissions from an arable soil by biochar application. Global Change Biology 24: e318–e334.

Boone, D.R., Whitman, W.B., & Rouvière, P. 1993. Diversity and Taxonomy of Methanogens. Methanogenesis. doi: 10.1007/978-1-4615-2391-8_2

Borchard, N., Schirrmann, M., Cayuela, M.L., Kammann, C., Wrage-Mönnig, N., Estavillo, J.M., Fuertes-Mendizábal, T., Sigua, G., Spokas, K., Ippolito, J.A., & Novak, J. 2019. Biochar, soil and land-use interactions that reduce nitrate leaching and N2O emissions: A meta-analysis. Science of the Total Environment 651: 2354–2364.

Bruun, E.W., Ambus, P., Egsgaard, H., & Hauggaard-Nielsen, H. 2012. Effects of slow and fast pyrolysis biochar on soil C and N turnover dynamics. Soil Biology and Biochemistry 46: 73–79.

Bruun, S., Clauson-Kaas, S., Bobul'ská, L., & Thomsen, I.K. 2014. Carbon dioxide emissions from biochar in soil: role of clay, microorganisms and carbonates. European Journal of Soil Science 65: 52–59.

Bruun, S., & EL-Zehery, T. 2012. Biochar effect on the mineralization of soil organic matter. Pesquisa Agropecuária Brasileira 47: 665–671.

Bruun, E.W., Hauggaard-Nielsen, H., Ibrahim, N., Egsgaard, H., Ambus, P., Jensen, P.A., & Dam-Johansen, K. 2011. Influence of fast pyrolysis temperature on biochar labile fraction and short-term carbon loss in a loamy soil. Biomass and Bioenergy 35: 1182–1189.

Bruun, E.W., Müller-Stöver, D., Ambus, P., & Hauggaard-Nielsen, H. 2011. Application of biochar to soil and N2O emissions: potential effects of blending fast-pyrolysis biochar with anaerobically digested slurry. European Journal of Soil Science 62: 581–589.

Bruun, E.W., Müller-Stöver, D., Pedersen, B.N., Hansen, L.V., & Petersen, C.T. 2021. Ash and biochar amendment of coarse sandy soil for growing crops under drought conditions. Soil Use and Management. doi: 10.1111/SUM.12783

Bruun, E.W., Petersen, C., Strobel, B.W., & Hauggaard-Nielsen, H. 2012. Nitrogen and Carbon Leaching in Repacked Sandy Soil with Added Fine Particulate Biochar. Soil Science Society of America Journal 76: 1142–1148.

Budai, A., Zimmerman, ; A R, Cowie, ; A L, Webber, ; J B W, Singh, ; B P, Glaser, ; B, Masiello, C.A., Andersson, D., Shields, F., Lehmann, J., Camps Arbestain, M., Williams, M., Sohi, S., & Joseph, S. 2013. Biochar Carbon Stability Test Method: An assessment of methods to determine biochar carbon stability. International Biochar Initiative

Butterbach-Bahl, K., Baggs, E.M., Dannenmann, M., Kiese, R., & Zechmeister-Boltenstern, S. 2013. Nitrous oxide emissions from soils: how well do we understand the processes and their controls? Philosophical Transactions of the Royal Society B: Biological Sciences 368: 20130122.

Case, S.D.C., McNamara, N.P., Reay, D.S., & Whitaker, J. 2012. The effect of biochar addition on N_2O and CO_2 emissions from a sandy loam soil - The role of soil aeration. Soil Biology and Biochemistry 51: 125–134.

Cayuela, M.L., Jeffery, S., & van Zwieten, L. 2015. The molar H/Corg ratio of biochar is a key factor in mitigating N2O emissions from soil. Agriculture, Ecosystems & Environment 202: 135–138.

Cayuela, M.L., Sánchez-Monedero, M.A., Roig, A., Hanley, K., Enders, A., & Lehmann, J. 2013. Biochar and denitrification in soils: when, how much and why does biochar reduce N_2O emissions? Scientific Reports 3: 1732.

Cayuela, M.L., van Zwieten, L., Singh, B.P., Jeffery, S., Roig, A., & Sánchez-Monedero, M.A. 2014. Biochar's role in mitigating soil nitrous oxide emissions: A review and meta-analysis. Agriculture, Ecosystems and Environment 191: 5–16.

Chagas, J., Figueiredo, C. de, & Ramos, M. 2022. Biochar increases soil carbon pools: Evidence from a global meta-analysis. Journal of Environmental Management 305:

Clough, T., Condron, L., Kammann, C., & Müller, C. 2013. A review of biochar and soil nitrogen dynamics. Agronomy 3: 275–293.

Conrad, R. 2007. Microbial Ecology of Methanogens and Methanotrophs. Advances in Agronomy 96: 1–63.

Cornelissen, G., Rutherford, D.W., Arp, H.P.H., Dörsch, P., Kelly, C.N., & Rostad, C.E. 2013. Sorption of pure N_2O to biochars and other organic and inorganic materials under anhydrous conditions. Environmental Science and Technology 47: 7704–7712.

Cross, A., & Sohi, S.P. 2013. A method for screening the relative long-term stability of biochar. GCB Bioenergy 5: 215–220.

Díaz-Rojas, M., Aguilar-Chávez, Á., Cárdenas-Aquino, M. del R., Ruíz-Valdiviezo, V.M., Hernández-Valdez, E., Luna-Guido, M., Olalde-Portugal, V., & Dendooven, L. 2014. Effects of wastewater sludge, urea and charcoal on greenhouse gas emissions in pots planted with wheat. Applied Soil Ecology 73: 19–25.

Dicke, C., Andert, J., Ammon, C., Kern, J., Meyer-Aurich, A., & Kaupenjohann, M. 2015. Effects of different biochars and digestate on N_2O fluxes under field conditions. Science of the Total Environment 524–525: 310–318.

Dunfield, P.F. 2007. The soil methane sink. In: Greenhouse Gas Sinks (Reay, D.S., Hewitt, C.N., Smith, K.A., & Grace, J., Eds), pp. 152–170. CABI Publishing, Wallingford, UK.

EBC, 2022. European Biochar Certificate - Guidelines for a Sustainable Production of Biochar, Version 10.1E of 10th January 2022. Arbaz, Switzerland.

Fang, Y., Singh, B., Singh, B.P., & Krull, E. 2014. Biochar carbon stability in four contrasting soils. European Journal of Soil Science 65: 60–71.

Feng, Y., Xu, Y., Yu, Y., Xie, Z., & Lin, X. 2012. Mechanisms of biochar decreasing methane emission from Chinese paddy soils. Soil Biology and Biochemistry 46: 80–88.

Fidel, R.B., Laird, D.A., & Parkin, T.B. 2017. Impact of Biochar Organic and Inorganic Carbon on Soil CO_2 and N_2O Emissions. Journal of Environmental Quality 46: 505–513.

Gelardi, D.L., & Parikh, S.J. 2021. Soils and beyond: optimizing sustainability opportunities for biochar. Sustainability 13, 10079.

Granli, T., & Bøckman, O.C. 1994. Nitrous oxide from agriculture. Norwegian Journal of Agricultural Sciences 12: 7–128.

Greenberg, I., Kaiser, M., Polifka, S., Wiedner, K., Glaser, B., & Ludwig, B. 2019. The effect of biochar with biogas digestate or mineral fertilizer on fertility, aggregation and organic carbon content of a sandy soil: Results of a temperate field experiment. Journal of Plant Nutrition and Soil Science 182: 824–835.

Haider, G., Steffens, D., Müller, C., & Kammann, C. 2017. Biochar may physically entrap nitrate during field aging or co-composting which become plant available under controlled conditions, in: 19th EGU General Assembly, EGU2017, proceedings from the conference held 23-28 April, 2017 in Vienna, Austria., p.15274.

Hale, S.E., Lehmann, J., Rutherford, D., Zimmerman, A.R., Bachmann, R.T., Shitumbanuma, V., O'Toole, A., Sundqvist, K.L., Arp, H.P.H., & Cornelissen, G. 2012. Quantifying the total and bioavailable polycyclic aromatic hydrocarbons and dioxins in biochars. Environmental Science and Technology 46: 2830–2838.

Hamer, U., Marschner, B., Brodowski, S., & Amelung, W. 2004. Interactive priming of black carbon and glucose mineralisation. Organic Geochemistry 35: 823–830.

Hansen, J.H., Hamelin, L., Taghizadeh-Toosi, A., Olesen, J.E., & Wenzel, H. 2020. Agricultural residues bioenergy potential that sustain soil carbon depends on energy conversion pathways. GCB Bioenergy 12: 1002–1013.

Hansen, V., Müller-Stöver, D., Ahrenfeldt, J., Holm, J.K., Henriksen, U.B., & Hauggaard-Nielsen, H. 2015. Gasification biochar as a valuable by-product for carbon sequestration and soil amendment. Biomass and Bioenergy 72: 300–308.

He, C., Manevski, K., Andersen, M.N., Hu, C., Dong, W., & Li, J. 2019. Abiotic mechanisms for biochar effects on soil N_2O emission. International Agrophysics 33: 537–546.

He, Y., Zhou, X., Jiang, L., Li, M., Du, Z., Zhou, G., Shao, J., Wang, X., Xu, Z., Hosseini Bai, S., Wallace, H., & Xu, C. 2017. Effects of biochar application on soil greenhouse gas fluxes: a meta-analysis. GCB Bioenergy 9: 743–755.

Horák, J., Kondrlová, E., Igaz, D., Šimanský, V., Felber, R., Lukac, M., Balashov, E. V., Buchkina, N.P., Rizhiya, E.Y., & Jankowski, M. 2017. Biochar and biochar with N-fertilizer affect soil N_2O emission in Haplic Luvisol. Biologia 2017 72:9 72: 995–1001.

Horák, J., Kotuš, T., Toková, L., Aydın, E., Igaz, D., & Šimanský, V. 2021. A sustainable approach for improving soil properties and reducing n_2o emissions is possible through initial and repeated biochar application. Agronomy 11: 582.

Hüppi, R., Felber, R., Neftel, A., Six, J., & Leifeld, J. 2015. Effect of biochar and liming on soil nitrous oxide emissions from a temperate maize cropping system. SOIL 1: 707–717.

Hüppi, R., Neftel, A., Lehmann, M., Krauss, M., Six, J., & Leifeld, J. 2016. N use efficiencies and N_2O emissions in two contrasting, biochar amended soils under winter wheat—cover crop—sorghum rotation. Environmental Research 11:.

IBI 2022. International Biochar Initiative. https://biochar-international.org/

IPCC 2019. 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories.

Ippolito, J.A., Cui, L., Kammann, C., Wrage-Mönnig, N., Estavillo, J.M., Fuertes-Mendizabal, T., Cayuela, M.L., Sigua, G., Novak, J., Spokas, K., & Borchard, N. 2020. Feedstock choice, pyrolysis temperature and type influence biochar characteristics: a comprehensive meta-data analysis review. Biochar 2: 421–438.

Jeffery, S., Verheijen, F.G.A., Kammann, C., & Abalos, D. 2016. Biochar effects on methane emissions from soils: A meta-analysis. Soil Biology and Biochemistry 101: 251–258.

Jensen, J.L. 2022. Tentative calculations of plant and manure C remaining in Danish soils (unpublished data).

Joseph, S., Cowie, A.L., Van Zwieten, L., Bolan, N., Budai, A., Buss, W., Cayuela, M.L., Graber, E.R., Ippolito, J.A., Kuzyakov, Y., Luo, Y., Ok, Y.S., Palansooriya, K.N., Shepherd, J., Stephens, S., Weng, Z., & Lehmann, J. 2021. How biochar works, and when it doesn't: A review of mechanisms controlling soil and plant responses to biochar. GCB Bioenergy 13: 1731–1764.

Kameyama, K., Miyamoto, T., Shiono, T., & Shinogi, Y. 2012. Infl uence of sugarcane bagasse-derived biochar application on nitrate leaching in calcaric dark red soil. Journal of Environmental Quality 41: 1131–1137.

Kammann, C., Ratering, S., Eckhard, C., & Müller, C. 2012. Biochar and Hydrochar Effects on Greenhouse Gas (Carbon Dioxide, Nitrous Oxide, and Methane) Fluxes from Soils. Journal of Environmental Quality 41: 1052–1066.

Karhu, K., Mattila, T., Bergström, I., & Regina, K. 2011. Biochar addition to agricultural soil increased CH4 uptake and water holding capacity – Results from a short-term pilot field study. Agriculture, Ecosystems & Environment 140: 309–313.

Khanal, G. 2011. Effects of biochar on soil microbial activity and greenhouse gas emissions. MSc Thesis, Aarhus University.

Kuzyakov, Y., Bogomolova, I., & Glaser, B. 2014. Biochar stability in soil: Decomposition during eight years and transformation as assessed by compound-specific 14C analysis. Soil Biology and Biochemistry 70: 229–236.

Kuzyakov, Y., Subbotina, I., Chen, H., Bogomolova, I., & Xu, X. 2009. Black carbon decomposition and incorporation into soil microbial biomass estimated by 14C labeling. Soil Biology and Biochemistry 41: 210–219.

Lehmann, J. 2007. Bio-energy in the black. Frontiers in Ecology and the Environment 5: 381–387.

Lehmann, J., Cowie, A., Masiello, C.A., Kammann, C., Woolf, D., Amonette, J.E., Cayuela, M.L., Camps-Arbestain, M., & Whitman, T. 2021. Biochar in climate change mitigation. Nature Geoscience 2021 14:12 14: 883–892.

Lehmann, J., Rillig, M.C., Thies, J., Masiello, C.A., Hockaday, W.C., & Crowley, D. 2011. Biochar effects on soil biota - A review. Soil Biology and Biochemistry 43: 1812–1836.

Leng, L., Huang, H., Li, H., Li, J., & Zhou, W. 2019. Biochar stability assessment methods: A review. Science of the Total Environment 647: 210–222.

Liang, B., Lehmann, J., Solomon, D., Kinyangi, J., Grossman, J., O'Neill, B., Skjemstad, J.O., Thies, J., Luizão, F.J., Petersen, J., & Neves, E.G. 2006. Black carbon increases cation exchange capacity in soils. Soil Science Society of America Journal 70: 1719.

Mandal, S., Sarkar, B., Bolan, N., Novak, J., Ok, Y.S., Van Zwieten, L., Singh, B.P., Kirkham, M.B., Choppala, G., Spokas, K., & Naidu, R. 2016. Designing advanced biochar products for maximizing greenhouse gas mitigation potential. Critical Reviews in Environmental Science and Technology 46: 1367–1401.

Mørkved, P.T., Dörsch, P., & Bakken, L.R. 2007. The N2O product ratio of nitrification and its dependence on long-term changes in soil pH. Soil Biology and Biochemistry 39: 2048–2057.

Morley, N., & Baggs, E.M. 2010. Carbon and oxygen controls on N_2O and N_2 production during nitrate reduction. Soil Biology and Biochemistry 42: 1864–1871.

Nielsen, O.-K. 2022. Indregning af Carbon Capture and Storage, Carbon Capture and Use og biokul i de nationale emissionsopgørelser. 23 s. – Notat nr. 2022|15.

Novak, J.M., & Johnson, M.G. 2019. Elemental and spectroscopic characterization of low-temperature (350°C) lignocellulosic- and manure-based designer biochars and their use as soil amendments. In: Biochar from Biomass and Waste, pp. 37–58. Elsevier.

Petersen, B.M., Knudsen, M.T., Hermansen, J.E., & Halberg, N. 2013. An approach to include soil carbon changes in life cycle assessments. Journal of Cleaner Production 52: 217–224.

Racek, J., Sevcik, J., Chorazy, T., Kucerik, J., & Hlavinek, P. 2020. Biochar – Recovery Material from Pyrolysis of Sewage Sludge: A Review. Waste and Biomass Valorization 11: 3677–3709.

Reddy, K.R., Yargicoglu, E.N., Yue, D., & Yaghoubi, P. 2014. Enhanced Microbial Methane Oxidation in Landfill Cover Soil Amended with Biochar. Journal of Geotechnical and Geoenvironmental Engineering 140: 04014047.

Rogovska, N., Laird, D., Cruse, R., Fleming, P., Parkin, T., & Meek, D. 2011. Impact of biochar on manure carbon stabilization and greenhouse gas emissions. Soil Science Society of America Journal 75: 871–879.

Saarnio, S., Heimonen, K., & Kettunen, R. 2013. Biochar addition indirectly affects N_2O emissions via soil moisture and plant N uptake. Soil Biology and Biochemistry 58: 99–106.

Simek, M., & Cooper, J.E. 2002. The influence of soil pH on denitrification: progress towards the understanding of this interaction over the last 50 years. European Journal of Soil Science 53: 345–354.

Singh, B.P., Cowie, A.L., & Smernik, R.J. 2012. Biochar carbon stability in a clayey soil as a function of feedstock and pyrolysis temperature. Environmental Science & Technology 46: 11770–11778.

Spokas, K.A. 2010. Review of the stability of biochar in soils: Predictability of O/C molar ratios. Carbon Management 1: 289–303.

Spokas, K.A., & Bogner, J.E. 2011. Limits and dynamics of methane oxidation in landfill cover soils. Waste Management 31: 823–832.

Sun, Z., Sänger, A., Rebensburg, P., Lentzsch, P., Wirth, S., Kaupenjohann, M., & Meyer-Aurich, A. 2017. Contrasting effects of biochar on N_2O emission and N uptake at different N fertilizer levels on a temperate sandy loam. Science of the Total Environment 578: 557–565.

Taghizadeh-Toosi, A., Christensen, B.T., Hutchings, N.J., Vejlin, J., Kätterer, T., Glendining, M., & Olesen, J.E. 2014. C-TOOL: A simple model for simulating whole-profile carbon storage in temperate agricultural soils. Ecological Modelling 292: 11–25.

Thers, H., Abalos, D., Dörsch, P., & Elsgaard, L. 2020. Nitrous oxide emissions from oilseed rape cultivation was unaffected by flash pyrolysis biochar of different type, rate and field ageing. Science of the Total Environment 724: 138140.

Thers, H., Djomo, S.N., Elsgaard, L., & Knudsen, M.T. 2019. Biochar potentially mitigates greenhouse gas emissions from cultivation of oilseed rape for biodiesel. Science of the Total Environment 671: 180–188.

Thomsen, T.P. 2021. Climate footprint analysis of straw pyrolysis & straw biogas: Assessment of the Danish climate crisis mitigation potential of two new straw management.

Verhoeven, E., Pereira, E., Decock, C., Suddick, E., Angst, T., & Six, J. 2017. Toward a Better Assessment of Biochar–Nitrous Oxide Mitigation Potential at the Field Scale. Journal of Environmental Quality 46: 237–246.

Wang, J., Chen, Z., Xiong, Z., Chen, C., Xu, X., Zhou, Q., & Kuzyakov, Y. 2015. Effects of biochar amendment on greenhouse gas emissions, net ecosystem carbon budget and properties of an acidic soil under intensive vegetable production. Soil Use and Management 31: 375–383.

Wang, H., Nan, Q., Waqas, M., & Wu, W. 2022. Stability of biochar in mineral soils: Assessment methods, influencing factors and potential problems. Science of the Total Environment 806: 150789.

Wang, J., Pan, X., Liu, Y., Zhang, X., & Xiong, Z. 2012. Effects of biochar amendment in two soils on greenhouse gas emissions and crop production. Plant and Soil 360: 287–298.

Wang, J., Xiong, Z., & Kuzyakov, Y. 2016. Biochar stability in soil: meta-analysis of decomposition and priming effects. GCB Bioenergy 8: 512–523.

Wang, Z., Zheng, H., Luo, Y., Deng, X., Herbert, S., & Xing, B. 2013. Characterization and influence of biochars on nitrous oxide emission from agricultural soil. Environmental Pollution 174: 289–296.

Weldon, S., Rasse, D.P., Budai, A., Tomic, O., & Dörsch, P. 2019. The effect of a biochar temperature series on denitrification: which biochar properties matter? Soil Biology and Biochemistry 135: 173–183.

Willén, A., Rodhe, L., Pell, M., & Jönsson, H. 2016. Nitrous oxide and methane emissions during storage of dewatered digested sewage sludge. Journal of Environmental Management 184: 560–568.

Woolf, D., Lehmann, J., Ogle, S., Kishimoto-Mo, A.W., McConkey, B., & Baldock, J. 2021. Greenhouse gas inventory model for biochar additions to soil. Environmental Science and Technology 55: 14795–14805.

Yaghoubi, P., Yargicoglu, E.N., & Reddy, K.R. 2014. Effects of biochar-amendment to landfill cover soil on microbial methane oxidation: Initial results.

Yuan, J.-H., Xu, R.-K., & Zhang, H. 2011. The forms of alkalis in the biochar produced from crop residues at different temperatures. Bioresource Technology 102: 3488–3497.

Zhang, A., Cui, L., Pan, G., Li, L., Hussain, Q., Zhang, X., Zheng, J., & Crowley, D. 2010. Effect of biochar amendment on yield and methane and nitrous oxide emissions from a rice paddy from Tai Lake plain, China. Agriculture, Ecosystems & Environment 139: 469–475.

Zhou, M., Zhu, B., Wang, S., Zhu, X., Vereecken, H., & Brüggemann, N. 2017. Stimulation of N_2O emission by manure application to agricultural soils may largely offset carbon benefits: a global meta-analysis. Global Change Biology 23: 4068–4083.

Zhu, K., Christel, W., Bruun, S., & Jensen, L.S. 2014. The different effects of applying fresh, composted or charred manure on soil N2O emissions. Soil Biology and Biochemistry 74: 61–69.

Zimmerman, A.R. 2010. Abiotic and microbial oxidation of laboratory-produced black carbon (biochar). Environmental Science & Technology 44: 1295–1301.

van Zwieten, L., Kimber, S., Morris, S., Chan, K.Y., Downie, A., Rust, J., Joseph, S., & Cowie, A. 2010. Effects of biochar from slow pyrolysis of papermill waste on agronomic performance and soil fertility. Plant and Soil 327: 235–246.

7 Nutrient composition of biochar and effects on nutrient availability and yields

Peter Sørensen and Diego Abalos (reviewer, Lars Elsgaard)

The concentrations of nutrients in biochars are influenced by the composition of the original organic input material (feedstock), by the pyrolysis temperature, and by other process parameters, such as the cooling strategy. Phosphorus (P) and potassium (K) are considered to remain in the material without significant losses in gasses, and as more organic material is lost by pyrolysis at increasing temperatures the concentration of stable nutrients, like P and K, increases with pyrolysis temperature (Bruun et al. 2017; Christel et al. 2015). For nitrogen (N) it is different, as N is lost in the gas, e.g., as N_2, NH_3 and HCN (Yuan et al. 2016). The N concentration in biochar therefore decreases with increasing pyrolysis temperature (Christel et al. 2015; Yuan et al. 2016) and the C/N ratio in biochar increases with pyrolysis temperature showing that the proportion between N and C loss by the pyrolysis increases with the temperature (Christel et al. 2015).

Concentrations of nutrients in biochars from different sources, produced mostly in Danish studies, are shown in Table 7.1. Data from China with sewage sludge were also included in the table as only limited data from Denmark were found for this feedstock. In many cases, the P concentration in biochars will be limiting how much biochar can be applied per hectare in accordance with the Danish regulation on maximal P application rates. Therefore the biochar application rate equivalent to 30 kg P/ha, which is the ceiling in most areas, is estimated in Table 7.1.

7.1 Nitrogen in biochars

The concentration of N in biochars is significantly influenced by pyrolysis temperature, with higher N losses (and thus lower N concentration in the biochar) at increasing temperatures (as also shown in Chapter 2, Figure 2.5). For the data compiled in Table 7.1, biochars derived from straw treated at 700°C or above only contain 3-6 kg N/tonne while biochars from pyrolysis at lower temperatures contain up to 29 kg N/tonne (Table 7.1). Similarly, Bruun et al. (2017) found that the N concentration in biochar from digestates treated at >600°C was only 3-9 kg N/tonne while biochar contained 15-19 kg N/tonne when treated at lower pyrolysis temperatures. The same picture was seen for biochar derived from sewage sludge where Yuan et al. (2016) measured 9 kg N/tonne when treated at 700°C, whereas the N concentrations were increasing to 15-61 kg N/tonne at decreasing pyrolysis temperatures. For a comparison, a common NPK fertiliser used in cereals contains about 20% N, 3% P and 10% K.

Table 7.1. Content of N, P and K in biochars derived from straw, manures and sewage sludge. Numbers at each biochar type indicate the pyrolysis/gasification temperature (°C). All values are presented on a dry matter basis. LT-CFB, low temperature circulating fluid bed gasification. NA, data not available.

Type	Nitrogen kg N/tonne	Phosphorous kg P/tonne	Potassium kg K/tonne	Tonne biochar per ha at 30 kg P/ha*	Reference
Straw					
Wheat straw LT-CFB 730	3	4	55	7.5	Li et al. (2017)
Wheat straw LT-CFB 730	3.7	4.2	57	7.1	Li et al. (2018)
Wheat straw slow pyrolysis 525	15	NA	NA	NA	Bruun et al. (2012)
Wheat straw fast pyrolysis 525	12	NA	NA	NA	Bruun et al. (2012)
Wheat straw, undefined temp (Swedish)	29	9.1	18	3.3	Parvage et al. (2013)
Wheat straw slow pyrolysis 725	5.3	1.2	35	25	Nissen et al. (2021)
Wheat straw flash pyrolysis 750	6.5	3.4	34	8.8	Nissen et al. (2021)
Wheat straw slow pyrolysis 300	24	3.4	27	8.8	Naeem et al. (2017)
Wheat straw slow pyrolysis 400	19	3.8	33	7.9	Naeem et al. (2017)
Wheat straw slow pyrolysis 500	19	4.2	41	7.1	Naeem et al. (2017)
Manures					
Poultry manure LT-CFB 730	8.3	57	91	0.53	Li et al. (2017)
Solid fraction digested slurry LT_CFB 730	NA	54	NA	0.56	Kuligowski et al. (2010)
Solid fraction pig slurry 400	NA	40	NA	0.75	Christel et al. (2016)
Solid fraction pig slurry 600	NA	54	NA	0.56	Christel et al. (2016)
Solid fraction pig slurry 550	15	71	NA	0.42	Zhu et al. (2014)
Solid fraction digested slurry 300	19	51	22	0.59	Bruun et al (2017)
Solid fraction digested slurry 450	18	53	24	0.57	Bruun et al (2017)
Solid fraction digested slurry 600	15	60	24	0.50	Bruun et al (2017)
Solid fraction digested slurry 750	9.1	66	27	0.45	Bruun et al (2017)
Solid fraction digested slurry 900	4.5	67	28	0.45	Bruun et al (2017)
Solid fraction digested slurry 1050	3.2	66	34	0.45	Bruun et al (2017)
Sewage sludge					
Straw + sewage sludge LT-CFB 730*	7.8	26	51	1.15	Li et al. (2017)
Straw + sewage sludge LT-CFB 730*	3.1	26	84	1.15	Li et al. (2017)
Sewage sludge 300	61	39	7	0.77	Yuan et al. (2016)
Sewage sludge 400	38	43	9	0.70	Yuan et al. (2016)
Sewage sludge 500	18	45	10	0.67	Yuan et al. (2016)
Sewage sludge 600	15	45	13	0.67	Yuan et al. (2016)
Sewage sludge 700	9	49	17	0.61	Yuan et al. (2016)

*Calculated amount of biochar that can be added to agricultural soil per ha without exceeding the P ceiling in Danish regulations on maximum P application rates (30 kg P/ha/year).

The availability of N in biochar is generally low and most of the N is associated with the stable C compounds. Bruun et al. (2012) found a decline in soil mineral N after application of straw-based biochars. Such decline could be due to adsorption of N by the biochar and/or due to microbial N immobilization. After application of flash-pyrolysis biochar from straw, Bruun et al. (2012) found microbial N immobilization, which could be related to the pool of readily available C being present in the biochar. After 65 days the net release of N was zero for slow-pyrolysis biochar and negative for flash-pyrolysis biochar (Bruun et al. 2012). Likewise, in a laboratory study with pyrolysed (550°C) solid fraction of pig slurry, Zhu et al. (2014) found no clear effect of biochar application on the release of mineral N after 100 days in soil.

The concentration of soluble N (ammonium-N + nitrate-N) is found to decrease with increasing pyrolysis temperature for biochar from sewage sludge (Yuan et al. 2016). However, the proportion of total N in soluble form increased with temperature due to a high loss of non-soluble N by increasing temperature and Yuan et al. (2016) found that 17% of total N was soluble N by the highest pyrolysis temperature (700ºC). Despite of this content of soluble N, Yuan et al. (2016) found no increase in leaching of inorganic N after application of biochars to soil in a leaching test.

In international studies, there are observations of both positive and negative effects of biochar on N availability (Hossain et al. 2020). We found relatively few Danish studies with focus on N availability, and therefore more studies of N availability in biochar from low temperature pyrolysis would be useful.

In conclusion, there is no indication of net N release within the first few months after biochar application to soil in Danish studies and the N in biochar is considered to be in a very stable form. Therefore, it is seemingly not relevant to set an N utilization percentage on N in biochars. However, we found limited information about N availability in biochar pyrolysed below 500ºC.

7.2 Availability of phosphorus in biochars

The amount of P in biochars is highly variable depending on the feedstock, as shown in Table 7.1. The availability of P in soil after application of biochar is influenced by different mechanisms: (1) biochar is a source of P and part of the biochar P is soluble, (2) biochar enhances the availability of soil P by influencing pH, complexation and metabolism in soil, (3) P can be adsorbed by biochar and thereby improve P retention in soil and affect P assimilation in plants (Yang et al. 2021).

A large number of both Danish and international studies have shown that soluble and plant-available P increase in soil after application of biochars (Li et al. 2017; Li et al. 2019; Yang et al. 2021). During the first days after application to soil, the availability of P is lower than in the feedstock applied directly to soil, but P availability from biochar increases over time (Bruun et al. 2017). The P availability is influenced by the type of feedstock (Li et al. 2017) and decreases at high pyrolysis temperatures (Bruun et al. 2017). The low initial P availability means that biochar are not so suitable as P starter fertilizers, but they are well suited as fertilizers to maintain an appropriate level of available P in the soil (Kuligowski et al. 2010) like when using organic manures.

In a Danish study comparing the availability of P in biochars with P in triple superphosphate fertilizer, Li et al. (2017) found a relative availability of 50% as evaluated 16 weeks after application of biochar derived from straw, shea nuts or poultry manure to three different soils. The relative availability of P in biochar from a mixture of sewage sludge and straw was lower and around 20%. The biochars studied by Li et al. (2017) all derived from gasification at ca. 730ºC. In another study, Bornø et al. (2018a,b) found that in slightly acidic soil, P availability was increased by adding two types of biochar, which was not the case in alkaline soil. They concluded that both biochar and soil properties as well as the P status of soil influences P bioavailability.

Whereas total total-P concentration generally increases with pyrolysis temperature, several studies have shown that bioavailable P decreases with increasing pyrolysis temperatures (Xu et al. 2016). In the investigation by Xu et al. (2016), lower pyrolysis temperature retained the P bioavailability of the feedstocks while more stable P fractions were increasingly formed when pyrolysis temperature increased. This is in line with the studies of Yang et al. (2020) and Yang et al. (2022) where increasing pyrolysis temperature from 550 to 700°C of two types of straw led to higher total P concentration, but decreased the P uptake of potatoes grown in loamy sand soil amended with the high-temperature biochars. Even though the high-temperature biochars induced a higher soil pH - into a range where soil P should be more plant-available - this could not outweigh the lower availability of endogenous P in the high-temperature biochars.

7.3 Availability of potassium in biochars

Table 7.1 shows that the K contents in biochar from straw and manures varied from 18-91 kg K/tonne, whereas the K content is lower in biochar derived from sewage sludge. However, when relating the K content to P content, the K/P ratio is much higher in biochar derived from straw than in biochar derived from manures, like it is in the feedstock. This means that biochar derived from manures, and especially from a separated fibre fraction, will only contribute with a small amount of K when applying a reasonable amount of P.

A study of availability of K in biochar applied to three different soils showed that 8 weeks after application, the availability of K in biochar from straw, shea nuts and poultry manure was 56-86% compared to a reference with mineral K fertilizer in the form of KCl (Li et al. 2018). For biochar derived from a mixture of sewage sludge and straw, K availability was slightly lower at 35-71%. A similar proportion was available one and eight weeks after the application (Li et al. 2018).

7.4 Application of N and P in biochars in relation to the Danish legislation

According to Danish legislation farmers have to take into account a given proportion of total N in organic fertilizers in their fertilizer planning and reporting. Since no studies with biochars have shown significant N release from biochars, it is not relevant to consider biochar N in the fertilizer planning and reporting. However, it is still uncertain how the biochar N will be released in the long-term.

In Denmark, farmers must register the amount of P applied with mineral and organic fertilizers. On most farms there is a maximum P application rate of 30 kg P/ha/yr calculated as an average for the whole farm, and termed the P ceiling. According to the knowledge about P availability in biochars described above, it is relevant to include also the input of P in biochars in the P ceiling calculation. This implies that the P application rate will often limit how much biochar can be applied, and it will often be difficult to combine the application of biochar with manures and other P-containing fertilizers. For biochars derived from digestates, separated slurry and from sewage sludge, it will only be possible to apply ca. 0.5 tonne biochar/ha/yr as an average without exceeding the P ceiling of 30 kg P/ha (Table 7.1). For biochar derived from straw, application rates can typically be in the range of 7-9 tonne/ha or even higher (Table 7.1), but here the K application rate becomes quite high, which may result in increased salinity and electrical conductivity (EC) as discussed in Chapter 4.

7.5 Additional mechanisms by which biochar affects crop yields

In addition to effects related to nutrient concentrations, biochar can modify crop yields through potential changes in other more uncertain drivers of yield. For example, low biochar rates can induce systemic resistance to pests caused by both soil-borne and foliar pathogens (Frenkel et al. 2017). This has been observed in strawberries, peppers and tomatoes for 15 different pathogens (fungi, oomycetes and nematodes) in 30 different pathosystems (i.e., plant/pathogen systems) (Elad et al. 2010; Meller Harel et al. 2012). However, the precise mechanism for this effect remains unclear.

Biochar tends to have a basic pH, and many field experiments show an increase in soil pH after biochar application in acidic soils (Jeffery et al. 2017 and Chapter 4). Accordingly, biochar can increase yield through a liming effect. This can be due to (i) higher soil nutrient availability and plant uptake mainly of P which is highly pH-dependent, but also N, Ca, Mg and Mo, and (ii) biochar-mediated reduction of available levels of some elements, which are toxic to plant growth, such as aluminium (Al^{3+}) and manganese (Mn^{2+}).

From a nutrient perspective, biochar can directly add nutrients present in its biomass as discussed above, but it can also increase the availability of soil mineral N by reducing N losses. This is because biochar can lower nitrate leaching (more details in Chapter 4), N_2O emissions (Chapter 6), and ammonia volatilization (Taghizadeh-Toosi et al. 2012). However, the precise mechanisms and the significance of these processes in a Danish agricultural context remain to be documented.

Another mechanism involves possible positive biotic interactions with arbuscular mycorrhizal (AM) fungi and biological N-fixers. AM fungi stimulate yields by enhancing plant uptake of N and P (Warnock et al. 2007). However, divergent effects of biochar on AM fungi have been reported and so far no firm conclusions on the effect of biochar on AM fungi can be reached (Chapter 5). Increasing biological N-fixation from the air in legumes can have positive effect on crop production directly in the legume and also indirectly in plant mixtures with legumes, due to increased availability of N in the soil (Rondon et al. 2007; Mia et al. 2014).

Biochar can also increase soil water retention, particularly in coarse-textured soils (Chapter 4), which are very common in Denmark. In turn, biochar can increase water availability for crops (see also Chapter 8, box 8.2). Additionally, biochar can increase the cation exchange capacity of soils (Chapter 4), promoting the retention of nutrients such as K, while reducing losses of P through leaching due to biochars capacity to absorb this nutrient on its surface (Beck et al. 2011; Slavich et al. 2013).

7.6 Negative effects of biochar on crop production

Some studies have reported yield reductions after biochar application (e.g., Hussain et al. 2017). The main processes behind these negative results are not well understood, but several mechanisms have been proposed and tested. Nitrogen immobilization is one of the better-characterized mechanisms (Bruun et al. 2012), although it is expected to last for only a short period of time during which the labile C fraction of biochar is released to the soil. Other hypothesized mechanisms are high sulfur content and salinity issues (Elseewi et al. 1978), the release of phytotoxic compounds, and reduced efficacy of pesticides (Jeffery et al. 2015). Issues related to harmful effects of the presence of the plant hormone ethylene (C_2H_4) in biochar have been reported (Spokas et al. 2010), but may be avoided by relatively simple post-production handling techniques, such as storage in the open (e.g., for 90 days) prior to soil amendment (Fulton et al. 2013).

7.7 Average effect of biochar on yield

The number of studies and meta-analyses investigating the effect of biochar on crop yield has grown sharply over the last years. Examples of mean effects of biochar on crop yield at a global scale are yield increases of 13% (Jeffery et al. 2017), 10% (Jeffery et al. 2011), 11% (Liu et al. 2013), 15% (Xu et al. 2021), 16% (Dai et al. 2020), and 25% (Bai et al. 2022). Due to the use of different methodologies and criteria to decide what studies are retained in the databases for the meta-analyses, the results from different studies can differ widely. For example, the effects on yield of biochar alone and biochar combined with chemical fertilizers were found to be 15% and 48%, respectively (Xu et al. 2021).

The generally positive effects of biochar on crop yield found in global meta-analyses should not be directly transferred to Danish field conditions, since these data often include tropical and sub-tropical regions, and also pot experiments. Indeed, recent research has shown that biochar has, on average, no effect on crop yield in temperate latitudes (Jeffery et al. 2017). Conversely, biochar promotes a 25% increase in yield in the tropics. This is because arable soils in the tropics often have low soil pH, low fertility, and low fertilizer inputs, whereas arable soils in temperate regions are more neutral in pH, have higher in fertility, and generally receive higher fertilizer inputs, limiting the potential yield benefits from biochar.

A number of field experiments conducted in Denmark support these findings. Sun et al. (2014) found that oat yields at Risø in 2011 showed no significant response to biochar. Total biomass of spring barley in 2012 at Risø had an 11% increase in response to biochar; however, for grain yields, the increase was only 6% and non-significant. Maize yields at Kalundborg in 2012 showed no significant response to biochar in any treatments except for the single dose of 50 Mg/ha given in 2012, which caused a 25% reduction in total biomass. This corroborated results from maize harvest at Kalundborg in 2011, where biochar applied at a rate of 50 Mg/ha caused biomass reductions of 22-24%. In Zealand, two rates of straw gasification biochar applied over three successive years to a field cropped with winter wheat and winter oilseed rape did not have any effect on crop grain yields (Hansen et al. 2017). Further, Thers et al. (2020) measured yields of oil-seed rape in field experiments with biochar of different type, rate and field ageing at a Danish sandy loam soil and found no statistically significant effects, although yields in treatments with biochar tended to be higher than in reference treatments without biochar.

The lack of effect of biochar application on crop yields in Denmark is consistent with other studies from the temperate region in the EU on sandy loam soils. For example, in a 4-year field experiment conducted in Norway, biochar did not alter crop yields of oat and barley during any of the four growing seasons (O'Toole et al. 2018). A 2-year field trial with maize, located in Merelbeke (Belgium), using biochar from a mixture of hard- and softwood, showed no effect on crop yield (Nelissen et al. 2015). In a 3-year field experiment conducted in Finland, with wheat, turnip rape, and faba bean, the grain yields and N uptake with biochar addition were not significantly different from the control in any year (Tammeorg et al. 2014).

7.8 Conclusions

The nitrogen content in biochars is very dependent on pyrolysis temperature as the gaseous N loss increases with temperature. International studies have shown both positive and negative effects on N availability, while in (the few) Danish studies only small effects of biochar on soil N availability have been observed. This implies that it is irrelevant to include N inputs from biochars in N fertilizer planning. The availability of P in biochar is variable, but in many cases the short-term availability is around 50% compared to mineral P fertilizers. In the longer term, P in biochar is expected to have a similar P availability as mineral fertilizers and manures. This implies that the P content in biochars often will set the limit for how much biochar can be applied without exceeding the Danish P application ceilings. The availability of K in biochar is somewhat lower than in mineral fertilizer in the first year, but in the longer term K in biochar is expected to become available to plants. Overall, the empirical evidence collected in Denmark and in regions with similar pedoclimatic conditions indicate that biochar is unlikely to have a significant effect on crop yield. Perhaps the regions with an acidic sandy soil, such as those of Southern Jutland, are the most likely to have a yield benefit after biochar application (see also box 8.2 for potential use of biochar for subsoil improvement in such soils).

7.9 References

Bornø ML, Eduah JO, Muller-Stover DS, Liu F, 2018a. Effect of different biochars on phosphorus (P) dynamics in the rhizosphere of *Zea mays* L. (maize). Plant Soil 431, 257-272.

Bornø ML, Muller-Stover, DS, Liu FL 2018b. Contrasting effects of biochar on phosphorus dynamics and bioavailability in different soil types. Sci Tot Environ 627, 963-974.

Bruun EW, Ambus P, Egsgaard H, Hauggaard-Nielsen H, 2012. Effects of slow and fast pyrolysis biochar on soil C and N turnover dynamics. Soil Biol Biochem 46, 73-79.

Bruun S, Harmer SL, Bekiaris G, Christel W, Zuin L, Hu Y, Jensen LS, Lombi E, 2017. The effect of different pyrolysis temperatures on the speciation and availability in soil of P in biochar produced from the solid fraction of manure. Chemosphere 169, 377-386.

Christel W, Bruun S, Magid J, Kwapinski W, Jensen LS, 2015. Pig slurry acidification, separation technology and thermal conversion affect phosphorus availability in soil amended with the derived solid fractions, chars or ashes. Plant Soil 401, 93-107.

Dai Y, Zheng H, Jiang Z, Xing B, 2020. Combined effects of biochar properties and soil conditions on plant growth: a meta-analysis. Sci Tot Environ 713, 136635.

Elad Y, David DR, Harel YM, Borenshtein M, Ben Kalifa H, Silber A, Graber ER, 2010. Induction of systemic resistance in plants by biochar, a soil-applied carbon sequestering agent. Phytopathology 100, 913-921.

Elseewi AA, Bingham FT, Page AL, 1978. Availability of sulfur in fly ash to plants. J Environ Qual 7, 69-73.

Frenkel O, Jaiswal AK, Elad Y, Lew B, Kammann C, Graber ER, 2017. The effect of biochar on plant diseases: what should we learn while designing biochar substrate? J Environ Eng Landscape Manage 25, 105-113.

Fulton W, Gray M, Prahl F. et al. 2013. A simple technique to eliminate ethylene emissions from biochar amendment in agriculture. Agron. Sustain. Dev. 33, 469–474.

Hansen V, Mueller-Stoever D, Imparato V, Krogh PH, Jensen LS, Dolmer A, Hauggaard-Nielsen H, 2017. The effects of straw or straw-derived gasification biochar applications on soil quality and crop productivity: a farm case study. J Environ Manage 186, 88-95.

Hossain MZ, Bahar MM, Sarkar B, Donne SW, Ok YS, Palansooriya KN, Kirkham MB, Chowdhury S, Bolan N, 2020. Biochar and its importance on nutrient dynamics in soil and plant. Biochar 2, 379-420.

Hussain M, Farooq M, Nawaz A, Al-Sadi AM, Solaiman ZM, Alghamdi SS, Ammara U, Ok YS, Siddique KH, 2017. Biochar for crop production: potential benefits and risks. J Soil Sed 17, 685-716.

Jeffery S, Abalos D, Prodana M, Bastos AC, van Groenigen JW, Hungate BA, Verheijen F, 2017. Biochar boosts tropical but not temperate crop yields. Environ Res Lett 12, 053001.

Jeffery S, Abalos D, Spokas KA, Verheijen FGA, 2015. Biochar effects on crop yields. In: Biochar for Environmental Management, 2nd ed. (Lehmann J, Joseph S, Eds.), pp. 301– 325. London, UK: Earthscan.

Jeffery S, Verheijen FGA, van der Velde M, Bastos AC, 2011. A quantitative review of the effects of biochar application to soils on crop productivity using meta-analysis. Agric Ecosys Environ 144, 175-187.

Kuligowski K, Poulsen TG, Rubaek GH, Sorensen P, 2010. Plant-availability to barley of phosphorus in ash from thermally treated animal manure in comparison to other manure based materials and commercial fertilizer. Europ J Agron 33, 293-303.

Li FY, Liang XQ, Niyungeko C, Sun T, Liu F, Arai Y, 2019. Effects of biochar amendments on soil phosphorus transformation in agricultural soils. Advances in Agronomy 158, 131-172.

Li XX, Rubaek GH, Muller-Stove, DS, Thomsen TP, Ahrenfeldt J, Sorensen P, 2017. Plant availability of phosphorus in five gasification biochars. Front Sustain Food Syst 1, 2.

Li X, Rubaek GH, Sorensen P, 2018. Availability of potassium in biomass combustion ashes and gasification biochars after application to soils with variable pH and clay content. Arch Agron Soil Sci 64, 1119-1130.

Liu X et al. 2013. Biochar's effect on crop productivity and the dependence on experimental conditions—a meta-analysis of literature data. Plant Soil 373, 583-594.

Meller Harel Y, Elad Y, Rav-David D, Borenstein M, Shulchani R, Lew B, et al. 2012. Biochar mediates systemic response of strawberry to foliar fungal pathogens. Plant Soil 357: 245-257.

Mia S, van Groenigen JW, van de Voorde T FJ, Oram NJ, Bezemer TM, Mommer L, Jeffery S, 2014. Biochar application rate affects biological nitrogen fixation in red clover conditional on potassium availability. Agric Ecosyst Environ 191, 83-91.

Naeem MA, Khalid M, Aon M, Abbas G, Tahir M, Amjad M, Murtaza B, Yang AZ, Akhtar SS, 2017. Effect of wheat and rice straw biochar produced at different temperatures on maize growth and nutrient dynamics of a calcareous soil. Arch Agron Soil Sci 63, 2048-2061.

Nelissen V, Ruysschaert G, Manka'Abusi D, D'Hose T, De Beuf K, Al-Barri B, Cornelis W, Boeckx P, 2015. Impact of a woody biochar on properties of a sandy loam soil and spring barley during a two-year field experiment. Europ J Agron 62, 65-78.

Nissen R, Khanal G, Elsgaard L, 2021. Microbial ecotoxicity of biochars in agricultural soil and interactions with linear alkylbenzene sulfonates. Agronomy 11, 828.

O'Toole A, Moni C, Weldon S, Schols A, Carnol M, Bosman B, Rasse DP, 2018. Miscanthus biochar had limited effects on soil physical properties, microbial biomass, and grain yield in a four-year field experiment in Norway. Agriculture 8, 171.

Parvage MM, Ulen B, Eriksson J, Strock J, Kirchmann H, 2013. Phosphorus availability in soils amended with wheat residue char. Biol Fertil Soils 49, 245-250.

Rondon MA, Lehmann J, Ramırez J, Hurtado M, 2007. B,ological nitrogen fixation by common beans (*Phaseolus vulgaris* L.) increases with bio-char additions. Biol Fertil Soils 43, 699-708.

Spokas KA, Baker JM, Reicosky DC, 2010. Ethylene: potential key for biochar amendment impacts. Plant Soil 333, 443–452.

Sun Z, Bruun EW, Arthur E, et al. 2014. Effect of biochar on aerobic processes, enzyme activity, and crop yields in two sandy loam soils. Biol Fertil Soils 50,1087-1097.

Taghizadeh-Toosi A, Clough TJ, Sherlock RR, Condron LM, 2012. Biochar adsorbed ammonia is bioavailable. Plant Soil 350, 57-69.

Tammeorg P, Simojoki A, Mäkelä P, Stoddard FL, Alakukku L, Helenius J, 2014. Biochar application to a fertile sandy clay loam in boreal conditions: effects on soil properties and yield formation of wheat, turnip rape and faba bean. Plant Soil 374: 89-107.

Thers H, Abalos D, Dörsch P, Elsgaard L, 2020. Nitrous oxide emissions from oilseed rape cultivation were unaffected by flash pyrolysis biochar of different type, rate and field ageing. Sci Tot Environ 724, 138140.

Warnock DD, Lehmann J, Kuyper TW, Rilling MC, 2007. Mycorrhizal responses to biochar in soil – concepts and mechanisms. Plant Soil 300, 9-20.

Xu G, Zhang Y, Shao H, Sun J, 2016. Pyrolysis temperature affects phosphorus transformation in biochar: Chemical fractionation and 31P NMR analysis. Sci Tot Environ 569-570, 65-72.

Xu H, Cai A, Wu D, Liang G, Xiao J, Xu M, Colinet G, Zhang W, 2021. Effects of biochar application on crop productivity, soil carbon sequestration, and global warming potential controlled by biochar C:N ratio and soil pH: A global meta-analysis. Soil Tillage Res 213, 105125.

Yang L, Wu YC, Wang YC, An WQ, Jin J, Sun K, Wang XK, 2021. Effects of biochar addition on the abundance, speciation, availability, and leaching loss of soil phosphorus. Sci Tot Environ 758, 143657.

Yang Q, Ravnskov S, Andersen MN, 2020. Nutrient uptake and growth of potato: Arbuscular mycorrhiza symbiosis interacts with quality and quantity of amended biochars. J Plant Nutr Soil Sci 183, 220-232.

Yang Q, Ravnskov S, Pullens JWM Andersen MN, 2022. Interactions between biochar, arbuscular mycorrhizal fungi and photosynthetic processes in potato (*Solanum tuberosum* L.). Sci Tot Environ 816, 151649.

Yuan HR, Lu T, Wang YZ, Chen Y, Le, TZ, 2016. Sewage sludge biochar: Nutrient composition and its effect on the leaching of soil nutrients. Geoderma 267, 17-23.

Zhu K, Christel W, Bruun S, Jensen L.S, 2014. The different effects of applying fresh, composted or charred manure on soil N_2O emissions. Soil Biol Biochem 74, 61-69.

8 Summary, conclusions and knowledge gaps

Lars Elsgaard, Anders Peter S. Adamsen, Henrik Bjarne Møller, Uffe Jørgensen, Esben Øster Mortensen, Emmanuel Arthur, Mathias N. Andersen, Anne Winding, Diego Abalos, Henrik Thers, Peter Sørensen (reviewer, Jørgen E. Olesen)

8.1 General considerations on need for agricultural biochar research

International agronomic biochar research has increased almost exponentially since 2010, when less than 100 publications were available (Schmidt et al. 2021). As of April 2022, more than 23.000 scientific papers are available on Web of Science for the search term biochar. The research interest in Denmark has followed a similar trend, although based on a modest number of studies and publications (Figure 8.1).

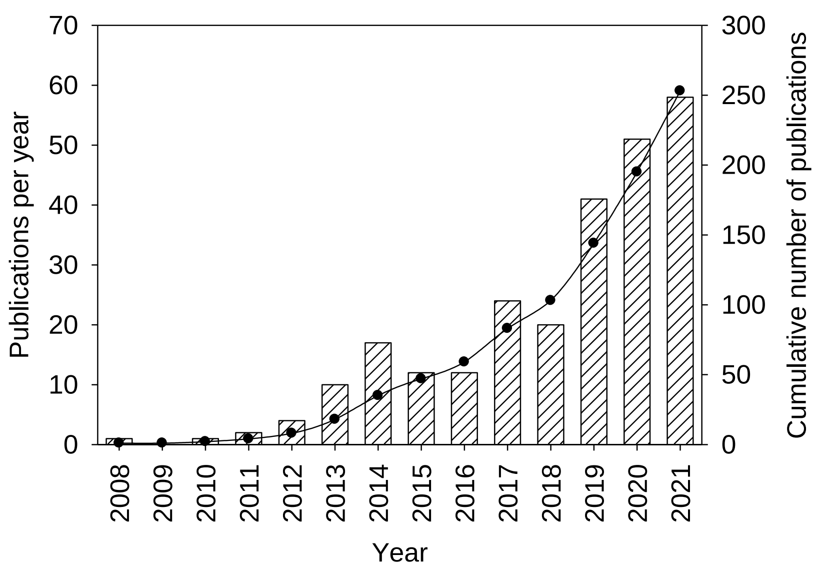

Figure 8.1. Time line of annual publications on biochar with reference to Denmark (black circles) as well as the cumulative numbers of publications (hatched bars) as retrieved from Web of Science (April 2022).

Biochar currently attracts attention mainly as a negative carbon emission technology in countries that are committed to ambitious climate goals, such as Denmark, with a target of 70% reduction in greenhouse gas (GHG) emissions by 2030. As this target is based on net GHG reductions, it may be realized partly by initiatives that offset CO_2 emissions through carbon sequestration, where pyrolysis of biomass is suggested to be an important element (Klimarådet 2020). Therefore, as stated by Nissen at al. (2021) large-scale application of biochar to agricultural land could become a megatrend. This is further supported by updated EU rules on fertilizing products, which will apply from 16 July 2022, making biochar available on the market for soil amendment across the EU. However, the rapid increase in the interest in biochar during few years means that empirical documentation and mechanistic understanding lag behind when it comes to evaluating agronomical and environmental long-term effects of biochar. This is aggravated by the fact that many of the earlier studies (i) were performed under laboratory conditions, and (ii) represent studies of typically less than 1 year, and (iii) performed with different biochars that were often not fully characterized, especially regarding pyrolysis conditions. It is important, therefore, that long-term research is initiated and prioritized based on relevant types of biochar expected to become available for Danish agroecosystems. Such research is needed in relation to documentation of the pyrogenic carbon capture and storage as well as the persistent effects of biochar on ecosystem services, including potentially adverse effects of biochar in agricultural soil.

Whereas many earlier studies were performed with single types of biochar and/or single types of soil and plant systems, it is encouraged to focus more systematically on investigating varying soil and biochar properties to increase the mechanistic understanding of biochar effects (Cai et al. 2021). Studies performed under controlled laboratory conditions still have merit, as they allow to vary single factors (e.g., soil temperature or biochar rates) at very detailed conditions while other factors are kept constant. This may allow for insight into how single factors control biochar effects in soil, e.g., in relation to the stability of biochar C. However, long-term field-based research should increasingly be encouraged, which integrates biochar effects in realistic agroecosystems and allows for studies of long-term effects.

Jeffery et al. (2015) discussed how to advance biochar research with emphasis on proper experimental designs, taking into account positive and negative controls, sufficient replication, and effects of biochar ageing in soil, which will change the biochar properties and effects in the soil ecosystem. Indeed, based on characteristics of earlier studies, which to some extent were pioneering, several recommendations have been made for stronger biochar research in relation to soil agroecosystems (Jeffery et al. 2015; Cai et al. 2021). Notably, biochar studies should preferably add to a mechanistic understanding of biochar effects, rather than being merely descriptive. Mechanistic understanding may allow for extrapolation of findings from one system to another and allow to predict under which conditions certain effects of biochar may be expected to appear or not (Cai et al. 2021), for example in relation to the N_2O mitigating effects of biochar.

As repeatedly highlighted in the preceding chapters, biochar properties resulting from different feedstock and thermal conversion conditions (e.g., pyrolysis temperature and residence time) are variable. Hence, 'biochar' is a common term that covers a range of compounds with different properties. This makes it intrinsically difficult to generalize the effects of biochar in agricultural soils, which moreover is also modulated by edaphic factors, climate, and agricultural management. This means that publications on biochar effects should be accompanied by a characterization of the applied biochar and its production conditions, including information on the feedstock. For example, as discussed in Chapter 6, the ratio of hydrogen to organic C (H/C_{org} ratio) should be reported in studies aiming at the use of biochar for long-term carbon sequestration. But also, properties, such as pH, surface area, and cation exchange capacity should be reported. Likewise, soil properties should be characterized since the eventual effects of biochar result from the interaction between biochar and the given soil conditions.

Studies of biochar added to soil need to consider both positive and negative effects in relation to agronomical and environmental impact (Chapter 5). As an example, biochar may provide a beneficial liming effect and increase pH in acidic soils, whereas such an effect in pH neutral or slightly alkaline soils may facilitate emission of nitrogen as ammonia (NH_3). In particular, negative effects of biochar application due to contents of potentially toxic elements (PTEs) need to be addressed. Harmful biochar constituents, such as volatile organic compounds (VOCs), polycyclic aromatic hydrocarbons (PAHs), organic contaminants as dioxin and PFAS, and heavy metals need to be regulated to prevent accumulation in the soil. Standard criteria for properties of biochars that may safely be spread on agricultural fields have been suggested, e.g., by the European Biochar Certificate (Chapter 1). However, these quality criteria do not include biological endpoints of ecosystem functioning, which makes it timely to strengthen the research emphasis also on ecotoxicological tests with biochar-amended soils both for the short- and long term.

In the present report, background and current knowledge have been synthesized in relation to the use of biochar in agricultural soils primarily under Danish conditions and based on major streams of available feedstock. This has indicated a number of knowledge gaps, which are listed below. The focus on the use of biochar in agriculture has been on soil amendment, which holds the potential for increasing soil C sequestration, but alternative uses of biochar in agroecosystems, e.g., as a feed additive in animal husbandry can also be envisaged (see Box 8.1).

8.2 Summary and research needs in relation to feedstock selection and balances of energy and greenhouse gas emissions by pyrolysis

The analysis of feedstock and technologies for large-scale production of biochar under Danish conditions in the present knowledge synthesis focuses on sewage sludge, straw and degassed fibers from biogas plants. In an analysis of the climate impact of such biochar production and use in agriculture, it is important to include the alternative (reference) use of the biomass and determine the long-term carbon storage of the biochar scenario as well as the reference scenario, in order to assess whether pyrolysis to biochar can play a crucial role in the overall effect. For example, energy production and energy consumption play a major role in the overall greenhouse gas balance. Calculations of the long-term carbon storage of biochar can be based on a model in which the molar ratio between hydrogen and carbon (H/C_{org}) is an important parameter that characterizes the intrinsic long-term stability of the biochar carbon. A low ratio is desirable if a high stability of carbon is to be achieved. Indeed, a threshold value of 0.7 is proposed in the European Biochar Certificate (EBC 2022), which means that pyrogenic products with an H/C_{org} ratio higher than 0.7 should not be characterized as biochar.

Chapter 2 presents an analysis of the effects of pyrolysis based on sewage sludge, straw and degassed fibers.

In the case of sewage sludge, it is estimated that 260 kg CO_2eq will be stored after 100 years by treatment of 1 tonne of sewage sludge dry matter. This should be compared with 118 kg CO_2eq stored by application of sewage sludge without pyrolysis, i.e., corresponding to a net effect of 142 kg CO_2eq. In the reference situation without pyrolysis, the wastewater sludge will be stored and dispersed, which will give rise to emissions of greenhouse gases in the form of methane and nitrous oxide corresponding to a total of 506 kg CO_2eq. In addition, there could be an energy surplus in the process that could have a positive effect of approx. 365 kg CO_2eq by substitution of natural gas after deduction of consumption for process electricity.

In the case of straw, 427 kg of CO_2eq will be stored after 100 years by treatment of 1 tonne of dry matter, which should be compared with 46 kg CO_2eq by mulching of straw, i.e., corresponding to a net effect of 381 kg CO_2eq. In addition, there could be an energy surplus in the process that could have a positive effect of approx. 596 kg CO_2eq by substitution of natural gas after deduction of consumption for process electricity.

In the case of biogas fibers, 425 kg of CO_2eq will be stored after 100 years by treatment of 1 tonne of dry matter in contrast to 132 kg CO_2eq by application of biogas fibers without pyrolysis, i.e., corresponding to a net effect of 293 kg CO_2eq. In the reference situation without pyrolysis, biogas fibers will be stored and dispersed, which will give rise to greenhouse gas emissions in the form of methane and nitrous oxide corresponding to a total of 197 kg CO_2eq. In addition, there could be an energy surplus in the process that could have a positive effect of approx. 373 kg CO_2eq by substitution of natural gas after deduction of consumption for process electricity.

All the above calculations are based on a number of assumptions, some of which are subject to considerable uncertainty (as described in Chapter 1 and 2), and a number of knowledge gaps and research needs have been identified that should be pursued to allow for more accurate calculations. There is need for:

- Data from commercial scale on how the choice of technology and operating parameters (temperature and time) affects the biochar properties, energy balances etc.
- Better estimation on effects of pyrolysis on GHG emission reductions by avoided emissions from storage and application of degassed fibers and sewage sludge.
- Documentation and estimates on the nitrogen turnover in the process and potentials for avoided gaseous losses of nitrous oxides and ammonia during storage and application of digestate fibers.
- Better understanding on how separation technology for digestate and pyrolysis affect the overall environmental impact.
- Studies of how biochar pyrolysis technology directly and indirectly influences the environmental impact under various land use scenarios.

- Improve correlations between pyrolysis process design, biomass type, biochar characteristics and effect on both carbon persistence and effects on GHG emissions in various scenarios of biochar use.

8.3 Summary and research needs in relation to biomass potentials

The biomass potentials for straw, biogas fibers and sewage sludge for pyrolysis have been analysed based on estimates of the current and the future resources (Chapter 3). Straw is the largest resource but has a number of competing uses today and potentially even more in the future. The currently unused resource of straw (grain + rapeseed + grass seed straw) is estimated at approx. 1.99 million tonne (Mt) dry matter (Table 8.1), taking into account that a certain proportion of farmers want to keep the straw as a source of organic matter input for the benefit of soil structure and fertility. However, in addition to the unused resource, pyrolysis may potentially compete for the amount of straw currently used for energy purposes (1.38 Mt dry matter), if such a transition is favoured by societal and economical drivers i.e., bringing the potential to 3.37 Mt dry matter. In future scenarios, there are opportunities to choose cereals with a higher amount of straw if farmers can see a market for increased straw sales. It will also be possible to collect a larger proportion of the straw that is currently left in the field. On the other hand, the general decrease in agricultural area, wishes for more nature areas, rewetting of organic soils, etc. will decrease the grain crop production area, In optimized scenarios for 2030, where these possibilities and limitations are included, total straw resources for bioenergy and biorefining of 3.09-3.85 Mt dry matter have been estimated.

Table 8.1. Summary table of estimated availability of straw (grain, rapeseed and seed grass straw), biogas fibre and sewage sludge for pyrolysis currently and in a 2030 scenario. Data are presented in Mt (million tonne) dry matter (DM) per year. See Chapter 3 for details.

Biomass type	Resource today (Mt DM)	Potential resource in 2030 (Mt DM)
Straw	1.99[a] - 3.37[b]	3.09 - 3.85
Biogas fibre	0.46	1.14 - 1.71
Sewage sludge	0.08 - 0.09	0.10 - 0.11

[a] Straw resource not utilized today
[b] Straw resource unutilized + resource utilized for energy purposes

For biogas fibers, only the current biogas use of livestock manure (+25% addition of other raw materials) is considered, whereby possible use of straw for biogas is excluded to avoid double counting. A potential availability of biogas fibers of 0.46 Mt dry matter has been calculated, as approx. 25% of livestock manure is assumed to be utilized for biogas in 2022. In scenarios for 2030 with an increased degree of utilization for biogas and when ± 20% change in livestock production is included, potentials of 1.14-1.71 Mt biogas fibers for pyrolysis are estimated.

The available amounts of sewage sludge are somewhat uncertain, as various inventories have shown approx. 30% difference in available quantity. In this synthesis, it was estimated that there is 0.08-0.09 Mt dry matter, which is currently applied to agricultural land, and which could be pyrolysed before application. In addition, an amount of approx. 0.02 Mt sludge, which today is incinerated, could possibly be pyrolysed, whereby undesirable organic constituents would expectedly be decomposed. Thus, a future resource of 0.10-0.11 Mt dry matter sludge can be estimated, as no significant change in sludge volumes is expected over time. However, it must be ensured that the contents of heavy metals will not compromise the quality for agricultural use.

If future potentials are to be realized, it requires research efforts, e.g., about opportunities for increased straw production and collection without compromising soil quality and crop yield, as assumptions about this are based on older studies and have not been demonstrated on a practical scale. It may also be important to investigate whether all straw fractions (straw, leaves, husks and spikes) as well as different types of seed grass straw (approx. 0.4 Mt dry matter) can be used for pyrolysis and give a uniform quality of the biochar product. Finally, the effects of different technology integration on the yield of energy, carbon and nutrients for recycling should be analysed; it can be integration of green biorefining (extraction of, e.g., protein) with biogas and pyrolysis. It will probably also be important to assess the presence of heavy metals in raw materials for pyrolysis and the limits thereof for use for agricultural purposes.

Further research needs identified in relation to the biomass resource for biochar are related to:

- Energy and material system analysis for the biosector in order to depict the most optimal resource use of the biomass for either energy, material, or carbon sequestration.
- Assessment of both current and near-future competitive use scenarios for various biomass types
- Effects of crop species and variety on straw quality for pyrolysis, as well as influences of harvest time, and weather conditions during crop ripening, which has influence on combustion quality.
- Improved assessment of biomass potential on local to national scale.
- Assessment of expected development in national crop production following national, regional and global scale shifts in food demands and climate.

Box 8.1: Biochar as feed additive (Søren Krogh Jensen, Dept. of Animal Science, AU)

In addition to the use of biochar for soil amendment, as discussed in the present knowledge synthesis, there are possibilities for different agricultural uses of biochar in relation to, e.g., animal husbandry. It was reported in 2018, that about 90% of the traded biochar in Germany, Austria and Switzerland was used in animal husbandry, mainly as feed additive (Kammann et al. 2018). However, this topic has been almost neglected in Danish biochar research.

Activated carbon, which is a common feed additive, has undergone steam activation to increase the internal porosity and enhance the absorption of potentially toxic elements (PTEs). However, unlike activated carbons, biochars also have surface functional groups, such as carboxyl, hydroxyl and phenolic groups. These functional groups enable additional sorption mechanisms over and beyond absorption, such as precipitation of metals and compounds to form insoluble compounds on the biochar surface (Li et al. 2017). This mechanism makes biochars particularly suited to the absorption of heavy metals, such as cadmium, lead and mercury, but the mechanisms will also work for, e.g., microbially produced toxins. Indeed, when used as feed additive, biochar may bind and immobilize PTEs and also contribute to reduce GHG emissions by changing conditions and nutrient adsorption in animal gut and excreta (Mia et al. 2017, Kammann et al. 2017). Moreover, the interactions of biochar surfaces with PTEs can reduce their bio-accumulation and magnification with significant impacts on animal health and quality of products (Man et al. 2021). It should be noted, though, that sorption mechanisms for biochars depend on both the target compound and specific biochar properties, and it may require post-production modification to achieve the benefits (Mia et al. 2017).

So far, and for future use in Danish agroecosystems, there is still a need for testing functionality of biochar and characterize their binding properties. This should include in vitro testing mimicking the animal gut environment. Bioassays must be conducted to examine the effect of organic compounds in biochar (e.g., polycyclic aromatic hydrocarbons, phenolics, and polyphenolics) on animal health (Wang et al. 2019). Furthermore, biochar should be tested in dose response experiments with animals, where also animal health and well-being are monitored. The effect of biochar feeding on GHG emissions from the animals (Huhtanen et al. 2019) and from animal excreta (Sommer et al. 2004) should be examined by chamber techniques. Finally, the quality and safety of animal products (milk, meat, and egg) need to be examined through both chemical and sensoric analysis in order to find the optimal supplementation rate of biochar to animal feed.

8.4 Summary and research needs in relation to soil physical and chemical effects of biochar

The physical properties of the soil reflect how the soil components (solid particles, liquid and gas) relate to each other. This has important impact on how well the soil can support plant growth and contribute to ecosystem services, such as nutrient retention. It is therefore important to document and understand how biochar changes and interacts with soil physical properties. Some of the soil physical parameters that are generally affected by biochar are (i) the volume weight (bulk density), which describes how tightly packed the solid particles are in a given volume and (ii) the retention of water in the soil, which reflects how large or small the pores of the soil are and how they are distributed in the soil.

In Chapter 4, the impact of biochar on soil physical properties was assessed using a meta-analysis of 31 relevant articles. All soil types were represented, both sandy and clay soils as well as intermediate topsoil types. The starting materials to produce biochar included straw, manure and sewage sludge. The application rate to the soil ranged from 0.25 to 10% by weight with an average of 2.5%. Across soil and biochar type, biochar input resulted in an average decrease of 9.7% in volume weight and an increase of approx. 35% in the plant available water volume (largest for sandy soils).

The ability of the soil to form stable aggregates is an important measure for soil quality, since stable soil aggregates contributes to a stable soil structure, which allows movement of gases, water, and nutrients. Biochar addition resulted in a mean increase in aggregate stability of 59%, but with a large dispersion around the mean due to few studies.

The soil's saturated hydraulic conductivity is essential for soil properties such as infiltration, drainage and nutrient transport. The effect of biochar application ranged from large increase to slight reduction in conductivity depending on both soil type and biochar type and properties. Studies have also shown that biochar usually does not affect the rewetting ability of soils, which may be reduced after drought.

The influence of biochar on soil chemical properties was assessed by reviewing the available literature on biochar from straw, manure and sewage sludge. Soil pH and cation exchange capacity are two crucial properties that affect the availability of both nutrients and water for crop uptake. When plant material is heated during the pyrolysis process in the presence of alkali ions, basic carbonates are formed, depending on both the pyrolysis conditions and the composition of the starting material. Thus, the pyrolysis temperature generally increases the formation of carbonates and thus the biochar’s ability to increase soil pH. As straw often contains alkali ions in larger amounts than wood, straw biochar is often found to have a greater ability to increase soil pH, which is also confirmed by experiments where direct comparison has been made. Biochars ability to increase soil pH can be characterized, and since biochar can to a certain extent replace agricultural lime, it will have a value both economically and via avoided CO_2 emission from added lime.

The functional groups formed during pyrolysis can provide biochar with cation exchange properties. Thus, biochar can contribute to soil cation exchange capacity (CEC). The literature indicates very different values for the CEC of biochar. However, it is rarely in line with humus and thus does not have an order of magnitude that can profoundly increase soil CEC except on sandy soils, where up to approx. 30% increase has been reported with biochar amendment.

Knowledge gaps and research needs identified in relation to the effects of biochar on soil physical and chemical properties include the need for:

- Knowledge on long-term effects under field conditions in Danish soil types, including studies with manure and sewage sludge biochar, which are particularly scarce.
- Understanding the effects of biochar on soil aggregation, particularly for coarse-textured soils.
- Assessment of the feasibility of designer-biochars with, e.g., high CEC or the ability to catalyse the oxidation of nitrous oxide.
- Evaluation of salinity effects after application of biochar and methods to remove K from biochar before soil application.
- Better understanding of the longevity of the liming effect of biochar in different Danish soil types.
- Studies of interactions between biochar and soil mineral N in relation to N leaching losses.
- Studies related to the role of hydrophobic/hydrophilic properties of biochar for soil physical effects.

8.5 Summary and research needs in relation to effects of biochar on soil biology

Biochars can affect soil biology differently and in both negative and positive ways. The chemical and physical properties of the biochar are of high importance, and these properties relates to the feedstock and the pyrolysis conditions including temperature, residence time during pyrolysis and cooling processes. These parameters are also crucial for concentrations of contaminants in biochars, such as PAHs and heavy metals, which affect soil biology. Also, the amount and particle size of biochar added, the soil texture, structure and organic matter and nutrient content are important factors to consider. The physical and chemical modifications, which biochar impose on the soil, might increase the water holding capacity, change pH (Chapter 4) and provide habitable pore spaces for soil organisms.

The literature reporting effects of biochar on soil biology repeatedly presents the same conclusion: the effects on soil biology are dependent the specific biochar properties and rates. This includes also the biochar content of easily degradable organic carbon (a minor pool, typically around 3% by weight), the size and pore structure, and the adsorption capacity of nutrients, minerals and organic contaminants. Hence, different batches of feedstock (e.g., straw, biogas digestate and sewage sludge) may result in biochars with different properties and with different resulting effects on soil biology (Brtnicky et al. 2021; Lehmann et al. 2011).

Furthermore, the effects of biochar on soil biology are strongly dependent on the site-specific soil properties including physical and chemical parameters. The combined role of biochar properties and soil properties makes it difficult to reach a general conclusion from the numerous primary publications on the effects of biochar on soil biology. It also highlights the need for standardized characterization of biochars at least in relation to chemical and physical characterization, but preferably also in relation to biological endpoints, such as effects on soil microorganisms which play an important role in organic matter and nutrient cycling in soil.

International studies on biochar effects on soil fauna have shown both positive and negative effects on earthworms and in a review Brtnicky et al. (2021) indeed concluded that the effects of biochar on earthworms are contradictory. In a Danish three-year field experiment with straw-based biochar applied at a total rate of either 3 tonne/ha or 16 tonne/ha it was found that earthworm abundance was unaffected compared both to straw-amended soil and control treatment.

Studies on soil microorganisms have shown that biochar may change microbial communities, but it is often uncertain (i) whether this has positive or negative impact on soil quality, (ii) whether the changes are larger or comparable to normal agricultural practices, and (iii) how long-lasting the changes are. The effects of biochar on the soil microbiome should be separated into short-term and long-term effects, but so far there is limited data on long-term effects. Stimulatory effects may be due, factors such as (i) a pool of easily degradable organic matter released from the biochar right after introduction into soil, (ii) amelioration of acidic soil pH or (iii) improved conditions for microbial activity in the soil due to better aeration. However, both short-term and longer-term negative effects could be caused by polycyclic aromatic hydrocarbons (PAHs), volatile organic compounds (VOCs), or heavy metals, which are known potential contaminants in biochars.

In addition to effects on living soil organisms, biochar may also affect the activity of extracellular enzymes in the soil, which are important for turn-over of organic matter. Interaction with biochar could stabilize the pool of extracellular enzymes, but their activity could be dependent on the orientation of the active sites in the biochar-enzyme complexes and also increase organic matter turn-over.

Due to the theoretical risk of leaching of biochar to aquatic environments the effects on aquatic organisms are also studied in the literature and several harmful effects have been reported. However, these results often relate to effects of biochar added directly to water and the effect of biochar after interaction with soil particles prior to leaching has not been thoroughly studied.

As biochar currently attracts attention as a negative carbon emission technology for large-scale application, there is a need for regulations and biochar quality criteria to secure environmentally safe application in agricultural soils. So far, biochar quality criteria, such as those suggested by the European Biochar Certificate (EBC 2022) mainly concerns physical and chemical properties of biochar. It is therefore timely to strengthen the research emphasis on soil biological tests with biochar-amended soils to better understand both short-term and long-term effects in the agricultural soil ecosystem.

Knowledge gaps and research needs in relation to the effects of biochar on soil biology include focus on:

- Long-term effects on soil living organisms after biochar amendment under field conditions.
- Understanding how biochar affects the microbial processes involved in nutrient transformations, such as microbial N cycling.
- Possibility for use of fast-responding microbial indicators for screening of biochar ecotoxicity.
- Effects of biochar feedstock classes (e.g., sewage sludges) in relation to environmental effects.
- Effects of biochar on microbial cell-to-cell communication and interactions among trophic levels.
- Potential adaptation of microbial communities to decomposition of recalcitrant biochar C.
- Interactions between biochar and mineralization of native soil organic carbon.
- Effects of biochar on activity of extracellular enzyme activity in the soil ecosystem.
- Effects of biochar post-treatments on soil biology in relation to soil type and management.
- Consequences of supplying various ratios of inert biochar versus fresh labile organic material (e.g., plant residues) for the soil microbial community as well as higher trophic fauna levels.
- Potential leaching of biochar constituents to aquatic ecosystems.

8.6 Summary and research needs in relation to soil carbon sequestration and greenhouse gas emissions

As reviewed in Chapter 6, several studies conclude a carbon storage potential by incorporating biochar into agricultural soil. This is because biochar degrades more slowly in the soil than the original biomass used for production of the biochar. However, the reported degradation rates of biochar in soil varies widely depending on both biochar properties and the soil agroecosystem, i.e., related to such factors as soil texture, cropping systems and temperature. Also, despite the general long-term stability of biochar, it is recognized that a minor (but variable) proportion of the biochar C represents relatively labile compounds. For simplicity and modelling, biochar C is often considered a as two-pool system with one pool described as easily decomposable (labile) and the other pool described as stable. A recent meta-study found the average distribution between the two pools to be 3% labile C and 97% stable C (Chapter 6). However, this can vary and must therefore be known for specific biochars to estimate the carbon storage potential.

The international literature describes several emerging methods for estimating the long-term stability of biochar in soil, including the preliminary IPCC approach, where the biochar production (pyrolysis) temperature is used to predict the biochar stability (Chapter 6). However, this must be considered as a very crude approach, since it does not consider the intrinsic biochar properties. Extended research has documented that the degradation rate of specific biochars can be more adequately estimated from the molar H to organic C ratio (H/C_{org} ratio) in the biochar (as also discussed in Chapter 1). Thus, a comprehensive approach for estimating biochar stability based on the H/C_{org} ratio has been suggested by Woolf et al. (2021) and is currently the most elaborate methodology, although there are substantial uncertainties involved (e.g., in relation to characterizing the effects of soil temperature on biochar decomposition). Furthermore, the metric used to characterize long-term stability of biochar in soil is the so-called F_{perm} factor, denoting the amount of the original biochar C that remains in soil after 100 years. Clearly, in order to derive such a metric, severe extrapolation is needed, since many studies only report biochar mineralization profiles for one to few years, and often under conditions that deviate from realistic field conditions, e.g., studies performed under controlled laboratory conditions. Furthermore, based on a recent Danish study (Thers et al. 2019), it was indicated that to evaluate the importance of biochar C sequestration for climate change mitigation, a metric like F_{perm} may not be fully sufficient. Instead, the concept of avoided atmospheric CO_2 emissions (Petersen et al. 2013) may be developed for biochar C sequestration. This was tentatively done by Thers et al. (2019) for biochar produced from rape seed straw, but such an approach needs to be consolidated for biochar for other feedstocks of relevance for Danish agriculture and integrated in life-cycle assessments.

Another research need, though not explicitly treated in Chapter 6, relates to the development of dynamic process-oriented models to simulate the profile of biochar mineralization under Danish agricultural conditions. In the international scientific literature, there are dynamic models for carbon conversion in soil, where an adaptation to biochar is described for the RothC model (Pulcher et al. 2022). This model, however, is not calibrated to Danish conditions and is not used in Denmark's National Inventory Report on greenhouse gas emissions (Nielsen et al. 2021), which relies on soil C modelling by the simple dynamic process-oriented simulation model C-TOOL (Taghizadeh-Toosi et al. 2014). Pulcher et al. (2022) included biochar in the RothC model by adding two new biochar pools to the model - one labile (4%) and one stable pool (96%). In addition, they included the effect of biochar on the turnover of the intrinsic soil organic carbon (SOC) pool, i.e., accounting for the so-called priming effect. For Danish conditions there is a research need in relation to potential integration of biochar in the C-TOOL model where it needs be tested, e.g., whether biochar should be distributed in the existing three carbon pools in C-TOOL or whether additional pools for biochar should be added. In addition, it must be determined whether soil properties such as clay content, C/N ratio and temperature affect the decomposition of biochar differently than the decomposition of the fresh sources of carbon input (e.g., straw and manure) that are included in the C-TOOL model (Jensen, 2022).

In relation to nitrous oxide N_2O, several international meta-studies have reported that biochar may significantly reduce the emission of this powerful greenhouse gas from agricultural soils, e.g., by up to 38% on average. However, considering only field studies (and not laboratory results) the effect is lower, and also it has been reported that the interpretation of the effects depends on applied methods of data analyses. Furthermore, it is uncertain whether the effect of biochar persists over annual time spans (i.e., beyond one or few years after biochar application). There are only few Danish field studies where the effects of biochar on

N_2O emission have been measured, and these studies have not been able to confirm a reduction in N_2O emission after biochar application (Chapter 6). Thus, while it is the empirically well-documented by meta-studies that biochar can mitigate N_2O emission from cultivated soils, individual field studies may or may not be able to document an effect for specific combinations of biochar, soil and climatic conditions (Thers et al. 2020).

This discrepancy reflects the current knowledge gap in relation to understanding the precise mechanisms for the interaction between biochar and the biogeochemical N cycling, which could result in lower N_2O emissions (Ameloot et al. 2016). Increased soil pH after biochar amendment (Chapter 4) can be one important driver of biochar-mediated effects on N_2O emissions (Obia et al. 2015), but no single mechanism stands out as preeminent, and the strength of different possible mechanisms is so far unclear. This lack of mechanistic understanding of biochar effects makes it difficult to predict the best combination of biochar and agroecosystems to enhance the mitigation of N_2O emissions. Future studies should explore how biochar changes the underlying microbial communities that are contributing to the N transformation processes in soil, resulting in N_2O emissions. This should comprise analyses of the changes in microbial community composition, including changes in the abundance and gene expression of ammonia oxidizing and denitrifying microorganisms, respectively, which are involved in oxidative transformations of ammonia (NH_3) and reductive nitrate transformation of nitrate (NO_3^-) with concomitant production of N_2O.

Finally, it should be considered that Danish experiments and the international literature typically do not report an increase in nitrous oxide emissions from the soil after biochar addition. This is in contrast to the addition of the fresh organic biomass (such as plant residues and animal slurry), which often is reported to result in increased N_2O emissions. Therefore, transformation of the biomass to biochar before field application holds a potential for emission reductions in a systems perspective.

Methane emissions from mineral agricultural soils are generally negligible, whereas oxidation of atmospheric methane is common, i.e., contributing to removal of CH_4 from the atmosphere. A meta-analysis found that the effect of biochar on methane oxidation could be negative, but so far there is a lack of data support a conclusion on the effects of biochar on methane oxidation in Danish agricultural soils. Some interactions in relation to N fertilization and improved soil aeration by biochar may occur, but further studies are needed to evaluate positive or negative effects of biochar for methane balances in a Danish agricultural context.

Research needs and knowledge gaps identified in relation to the effects of biochar on carbon sequestration and greenhouse gas emissions include:

- Need for long-term data on biochar degradation in soil, including more knowledge on biochar field aging processes.
- Validation of relationship between H/C_{org} ratios and biochar decomposition for relevant Danish feedstocks and soil conditions.
- Studies of interactions between microbial mineralization of biochar and native soil organic matter.
- Mechanistic understanding of the soil, climatic and management conditions promoting N_2O reductions after biochar application.
- Integration of biochar in soil organic carbon models, including effects of environmental drivers of biochar decomposition.
- Assessment of the effect of biochar on methane oxidation in N fertilized cropping systems.
- Possible effects of biochar post-process treatment, dosing, amendment technology etc. on carbon persistence and direct and indirect GHG emissions from soil ecosystems.
- Characterization of the effets of soil temperature on biochar decomposition.

8.7 Summary and research needs in relation to nutrient composition of biochar and effects on nutrient availability and yields

Biochars are often found to hold the potential to enhance ecosystem services in agricultural soils, such as water holding capacity, aggregate stability, and nutrient availability. However, it is difficult to generalize the strength of these effects, because biochar properties depend on feedstocks and pyrolysis conditions. Furthermore, the effect of individual biochars is modulated by edaphic factors, climate, and agricultural management. A common finding in international studies is that biochar is more likely to stimulate ecosystem services in deprived soils than in fertile soils and well-managed cropping systems. This means that biochar generally plays a negligible role as nutrient source in Danish agricultural soils, which are well managed in terms of fertilization and return of organic residues. Indeed, a yield increase cannot be expected if the crop growth/production is already close to the potential level (i.e., as defined by crop/variety characteristics in combination with climate), which is the case for most Danish agricultural soils. However, on coarse sandy soils the overall soil fertility may be low, mainly due to a low water holding capacity and mechanical resistance to root growth in the subsoil. Here, soil improvement by biochar may hold a potential to stimulate crop yields (Ahmed et al. 2020; see also Box 8.2). Nevertheless, despite the absence of general effects on crop yields, the content of nitrogen (N), phosphorus (P) and potassium (K) in biochars still needs to be considered in relation to the interaction with mineral and organic fertilizers.

Box 8.2: Biochar for improvement of coarse sandy subsoils (Dorette Müller-Stöver, Esben Wilson Bruun and Carsten Tilbæk Petersen, Department of Plant and Environmental Sciences, KU)

Coarse sandy soils account for a large proportion of the agricultural land in the north-western part of Europe (Ahmed et al. 2020). In Denmark, they constitute about 24% of the classified land (Madsen et al. 1992). The quality of the topsoil layer is based on its high content of organic matter, but overall soil fertility is low, mainly due to a very low water holding capacity and mechanical resistance to root growth in the subsoil. Non-irrigated fields often experience temporal drought, and poor nutrient utilization by crops leads to large amounts of nitrate being leached in periods of excess rainfall. More irregular precipitation patterns in the future combined with higher temperatures and evaporative demands will make sustainable crop production on this soil type even more difficult.

Some attempts to improve subsoil characteristics such as mechanical subsoil loosening (Munkholm et al. 2003) or incorporation of large amounts of organic material have shown positive effects, but they generally only lasted for a few years. More recently, it has been shown that amending coarse sandy subsoil with fine-grained biochar in appropriate amounts can markedly increase the plant-available water capacity (AWC) and improve both root and shoot growth (Ahmed et al. 2020; Bruun et al. 2014; Petersen et al. 2016; Hansen et al. 2016; Bruun et al. 2022). The increase in AWC is mainly the effect of fine biochar particles forming interstitial soil pores, thereby converting drainable pores into water-retaining pores. This implies that small differences in the particle-size distribution of the added biochar have a large impact on the AWC (Petersen et al. 2016).

In addition to the effect on AWC, biochar amendment has also been shown to decrease dry bulk density and soil compressibility (Rogovska et al. 2014; Petersen et al. 2016; see also Chapter 4 in this knowledge synthesis), which additionally explains some of the positive effects on root development and grain yields reported (Bruun et al. 2014, Hansen et al. 2016, Ahmed et al. 2020). Thus, the addition of large amounts of fine-grained biochar to coarse sandy subsoils may be able to initiate a long-term transformation of the amended soils, improving crop resource utilization through better water and nutrient uptake from greater soil depths, mitigating climate change by sequestering biochar carbon, and increasing yields on infertile sandy soils in a future climate with more frequent drought periods. However, implementation in practice still requires field experiments that confirm the effects seen under laboratory and semi-field conditions.

During production of biochar by pyrolysis, phosphorus and potassium can be expected to remain in the material without significant losses in gases, whereas nitrogen is lost in the gas in the form of, e.g., N_2, NH_3 and HCN. The N concentration in biochar therefore decreases with increasing pyrolysis temperature and the C/N ratio in biochar increases with the pyrolysis temperature. For example, biochar derived from straw treated at >700°C contained only 3-6 kg N/tonne, while biochar from pyrolysis of straw at lower temperatures had up to 29 kg N/tonne (Table 7.1). Furthermore, the bioavailability of N in biochar is generally low as most of the

N is bound to stable C compounds. It is concluded that no net release of N has been found within the first few months after application of biochar to soil in Danish studies, and N in biochars are considered to be in a very stable form. Therefore, it is not relevant to set an N utilization rate of N in biochar. However, this knowledge synthesis found limited information on N availability in biochar pyrolysed at temperatures below 500°C. Therefore, more studies of N-availability in biochar from especially pyrolysis at lower temperatures would be desirable.

In international studies, there are observations of both positive and negative effects of biochar on N availability in the soil (i.e., availability of N from sources other than biochar). However, there are only few Danish studies focusing on N-availability as compared for soils with and without biochar. Biochars may physically adsorb/release both ammonium (NH_4^+) and nitrate (NO_3^-), but it needs to be better understood how biochar interacts with added mineral and organic N amendments. Nitrogen response curves, where crop yields are measured at a gradient of added N rates, should be performed in soils with and without added biochar, to test if biochars could lower the N rate for optimal yields.

In some studies, biochar has been documented to delay nitrate leaching from the root zone, presumably by still elusive mechanisms of physical nitrate capture to internal biochar porosities (Kammann et al. 2018). This nitrate seems to be only partly detectable with standard methods (Haider et al. 2016), but such nitrate capture could contribute to reduce the nitrate availability for microbial denitrifiers and thereby reduce N_2O emissions. Under Danish conditions, there is a lack of studies on the interaction between biochar and nitrate leaching, which should be encouraged both in relation the potential effects on the aquatic environment (less nitrate leaching), the atmospheric environment (less N_2O emission) and the cropping system (more fertilizer N retained in the root zone).

The amount of P in biochar is dependent on the feedstock. The availability of P in the soil after biochar amendment is affected by various mechanisms, such as: (i) biochar is a source of P, (ii) biochar affecting the availability of soil P, e.g., by altering pH and (iii) P adsorption to biochar. Many Danish and international studies have shown that soluble and plant-accessible P increases in the soil after the addition of biochar. However, it is generally found that P-availability in biochar decreases with increasing pyrolysis temperatures. A variable part of the P in biochars is immediately available and a part is less available, which means that biochar is not a suitable starter P fertilizer.

In relation to Danish fertilizer regulations, the P concentration in biochar will often be limiting for how much biochar can be added per hectare. It was shown (Chapter 7) that for biochars derived from digestates, separated slurry and from sewage sludge, it will only be possible to apply approx. 0.5 tonne biochar/ha/yr as an average without exceeding the P ceiling of 30 kg P/ha. Thus, for large scale application of biochar from such feedstocks further research should be prioritized in order to develop production pathways that minimize the resulting P content in the biochars. For biochar produced from straw as feedstock, calculations according to the P ceiling indicate that typically 7-9 tonne biochar/ha/yr (but up to 25 tonne biochar/ha/yr) can be applied without exceeding the P ceiling of 30 kg P/ha (see Table 7.1).

Potassium concentrations in biochar can be high depending on the feedstock. In biochar from straw and manure, 18-91 kg K/tonne has been reported, whereas the K content is lower in biochar originating from sewage sludge. The ratio between K and P is much higher in biochar derived from straw than in biochar derived from livestock manure, just as it is in the feedstock.

Research needs and knowledge gaps in relation to nutrient composition of biochar and effects on nutrient availability and yields include the following:

- Experiments investigating potential yield benefits of biochar through improvements in soil water retention, root development and nutrient availability, such as phosphorous.
- Understanding the soil and climatic conditions in which biochar may increase crop production, and the specific crops that are more responsive to biochar application.
- Finding optimum biochar application rates that avoid potentially negative effects on yield.
- Measurements and understanding of the effects of biochar on mitigating nitrate leaching in Danish agricultural soils, including assessment of short- versus long-term effects.
- More knowledge about the N availability in biochars from different temperature regimes, including relatively low-temperature pyrolysis when relevant.
- Knowledge on systemic benefits on national and regional scale of nutrient separation and increased P-concentration and mobility.
- Systematic investigations on the potential of biochar to partly replace mineral P fertilizer
- Investigations of the risk of P loss in surface water run-off after surface application of biochars
- Investigations on the fertilizer effects of mixtures of biochar and organic materials.

8.8 References

Ahmed F, Arthur E, Liu H, Andersen MN, 2020. New rootsnap sensor reveals the ameliorating effect of biochar on in situ root growth dynamics of maize in sandy soil. Frontiers in Plant Science 11, 949.

Ameloot N, Maenhout P, De Neve S, Sleutel S, 2016. Biochar-induced N_2O emission reductions after field incorporation in a loam soil. Geoderma 267, 10-16.

Brtnicky M, Datta R, Holatko J, Bielska L, Gusiatin ZM, Kucerik J, Hammerschmiedt T, Danish S, Radziemska M, Mravcova L, Fahad S, Kintl A, Sudoma M, Ahmed N, Pecina V, 2021. A critical review of the possible adverse effects of biochar in the soil environment. Sci Tot Environ 20, 796, 148756.

Bruun EW, Müller-Stöver D, Pedersen BN, Hansen LV, Petersen CT, 2022. Ash and biochar amendment of coarse sandy soil for growing crops under drought conditions. Soil Use and Management 38, 1280-1292.

Bruun EW, Petersen CT, Hansen E, Holm JK, Hauggaard-Nielsen H, 2014. Biochar amendment to coarse sandy subsoil improves root growth and increases water retention. Soil Use and Management, 30, 109-118.

Cai Y, Ok YS, Lehmann J, Chang S, 2021. Recommendations for stronger biochar research in soil biology and fertility. Biology and Fertility of Soils 57, 333–336

EBC, 2022. European Biochar Certificate - Guidelines for a Sustainable Production of Biochar, Version 10.1E of 10th January 2022. Arbaz, Switzerland.

Haider G, Steffens D, Müller C, Kammann C, 2016. Standard extraction methods may underestimate nitrate stocks captured by field-aged biochar, Journal of Environmental Quality 45, 1196–1204.

Hansen V, Hauggaard-Nielsen, Petersen CT, Mikkelsen TN, Müller-Stöver D, 2016. Effects of gasification biochar on plant-available water capacity and plant growth in two contrasting soil types. Soil and Tillage Research 161, 1-9.

Huhtanen P, Ramin M, Hristov AN, 2019. Enteric methane emission can be reliably measured by the Green Feed monitoring unit. Livestock Sci 222, 31–40.

Jeffery S, Bezemer TM, Cornelissen G, Kuyper TW, Lehmann J, Mommer L, Sohi SP, van de Voorde TFJ, Wardle DA, van Groenigen JW, 2015. The way forward in biochar research: targeting trade-offs between the potential wins. GCB Bioenergy 7, 1–13

Jensen JL, Thers H, Elsgaard L, 2022. Afklaring om videns- og ressourcebehov ved at integrere biokul i C-TOOL modellen til brug for emissionsopgørelser. Rådgivningsnotat fra DCA – National Center for Fødevarer og Jordbrug (17.05.2022).

Kammann C, Ippolito J, Hagemann N et al. 2018. Biochar as a tool to reduce the agricultural greenhouse-gas burden – knowns, unknowns and future research needs. J Environ Eng Landscape Manage 25, 114-139.

Klimarådet 2020. Known paths and new tracks to 70 per cent reduction - Direction and measures for the next 10 years climate action in Denmark. Klimarådet, March 2020. Availlble at https://klimaraadet.dk/en/system/files_force/downloads/english_version_-_known_paths_and_new_tracks_to_70_per_cent_1.pdf?download=1.

Lehmann, J., Rillig, M.C., Thies, J., Masiello, C.A., Hockaday, W.C., & Crowley, D. 2011. Biochar effects on soil biota - A review. Soil Biology and Biochemistry 43: 1812–1836.

Li H, Dong X, da Silva EB, de Oliveira LM, et al. 2017. Mechanisms of metal sorption by biochars: Biochar characteristics and modifications. Chemosphere 178, 466-478.

Madsen HB (1992). Atlas over Danmark - Atlas of Denmark: Den danske jordklassificering. Det Kongelige Danske Geografiske Selskab, CA Reitzel.

Man KY, Chow K L, Man YB et al. 2021. Use of biochar as feed supplements for animal farming. Crit Rev Environ Sci Technol 51, 187-217.

Mia S, Dijkstra FA, Singh B, 2017. Long-term aging of biochar: a molecular understanding with agricultural and environmental implications. Adv Agron 141, 1-51.

Munkholm LJ, Schjønning P, Sørensen H, 2003. Jordpakning og mekanisk løsning på grovsandet jord. Danmarks Jordbrugs Forskning.

Nielsen OK, Plejdrup MS, Winther M et al. 2021. Denmark's National Inventory Report 2021. Emission Inventories 1990-2019 - Submitted under the United Nations Framework Convention on Climate Change and the Kyoto Protocol. Aarhus University, DCE – Danish Centre for Environment and Energy, Scientific Report No. 437, 944 pp.

Nissen R, Khanal G, Elsgaard L, 2021. Microbial ecotoxicity of biochars in agricultural soil and interactions with linear alkylbenzene sulfonates. Agronomy 11, 828.

Obia A, Cornelissen G, Mulder J, Dörsch P, 2015. Effect of soil pH increase by biochar on NO, N2O and N2 production during denitrification in acid soils. PLoS ONE 10, e0138781

Petersen BM, Knudsen MT, Hermansen JE, Halberg N, 2013. An approach to include soil carbon changes in life cycle assessments. J Clean Prod 52, 217-224

Petersen CT, Hansen E, Larsen HH, Hansen LV, Ahrenfeldt J, Hauggaard-Nielsen H, 2016. Pore-size distribution and compressibility of coarse sandy subsoil with added biochar. European Journal of Soil Science 67, 726-736.

Pulcher R, Balugani E, Ventura M, Greggio N, Marazza D, 2022. Inclusion of biochar in a C-dynamics model based on observations from an 8-year field experiment. SOIL 8, 199–211.

Rogovska N, Laird DA, Rathke SJ, Karlen DL, 2014. Biochar impact on Midwestern Mollisols and maize nutrient availability. Geoderma 230, 340-347.

Schmidt, H.-P., Kammann, C., Hagemann, N., Leifeld, J., Bucheli, T. D., Sánchez Monedero, M. A., & Cayuela, M. L. (2021). Biochar in agriculture – A systematic review of 26 global meta-analyses. GCB Bioenergy, 13, 1708– 1730.

Sommer SG, McGinn SM, Hao X et al (2004). Techniques for measuring gas emissions from a composting stockpile of cattle manure. Atmos Environ 38, 4643–4652

Taghizadeh-Toosi A, Christensen BT, Hutchings NJ, Vejlin J, Kätterer T, Glendining M, Olesen JE, 2014. C-TOOL: A simple model for simulating whole-profile carbon storage in temperate agricultural soils. Ecological Modelling 92, 11-25,

Thers, H., Abalos, D., Dörsch, P., & Elsgaard, L. 2020. Nitrous oxide emissions from oilseed rape cultivation was unaffected by flash pyrolysis biochar of different type, rate and field ageing. Science of the Total Environment 724: 138140.

Thers, H., Djomo, S.N., Elsgaard, L., & Knudsen, M.T. 2019. Biochar potentially mitigates greenhouse gas emissions from cultivation of oilseed rape for biodiesel. Science of the Total Environment 671: 180–188.

Wang J, Odinga ES, Zhang W et al. 2019. Polyaromatic hydrocarbons in biochars and human health risks of food crops grown in biochar-amended soils: A synthesis study. Environ Int 130, 104899.w

Woolf, D., Lehmann, J., Ogle, S., Kishimoto-Mo, A.W., McConkey, B., & Baldock, J. 2021. Greenhouse gas inventory model for biochar additions to soil. Environmental Science and Technology 55: 14795–14805.

9 Dansk sammendrag, konklusioner og videnshuller

Lars Elsgaard, Anders Peter S. Adamsen, Henrik Bjarne Møller, Uffe Jørgensen, Esben Øster Mortensen, Emmanuel Arthur, Mathias N. Andersen, Anne Winding, Diego Abalos, Henrik Thers, Peter Sørensen (based on Chapter 8, reviewed by Jørgen E. Olesen)

9.1 Behov for forskning i biochar til jordbrugsformål

International forskning i anvendelse af biokul er steget markant siden 2010, hvor mindre end 100 videnskabelige publikationer var tilgængelige. Siden udgangen af april 2022 findes mere end 23.000 videnskabelige artikler tilgængelige på Web of Science for søgeordet "biochar" eller biokul som det ofte oversættes til på dansk. Forskningsinteressen i Danmark har fulgt en lignende tendens, dog baseret på et beskedent antal undersøgelser og publikationer (se Figur 8.1).

Biokul tiltrækker sig i øjeblikket primært opmærksomhed som en teknologi der kan bidrage til negativ CO_2-udledning i lande med ambitiøse klimamål, såsom Danmark, med et mål om 70% reduktion i drivhusgasemissioner inden 2030. Da dette mål er baseret på nettoreduktioner af drivhusgasser, kan det delvis realiseres ved initiativer, der kompenserer for CO_2-udledning gennem kulstofbinding, hvor pyrolyse af biomasse er blevet foreslået som et vigtigt element (Klimarådet 2020). Derfor, kan storstilet anvendelse af biokul til landbrugsjord blive en trend i de kommende år. Dette understøttes yderligere af nye EU-regler om gødningsprodukter, som vil gælde fra den 16. juli 2022, og som gør biokul tilgængeligt på markedet i EU med henblik på jordforbedring.

Den hurtige stigning i interessen for biokul betyder imidlertid, at empirisk dokumentation og mekanistisk forståelse halter bagefter, når det kommer til at evaluere langsigtede agronomiske og miljømæssige virkninger af biokul. Dette forværres af, at mange af de tidligere undersøgelser (i) blev udført under laboratorieforhold og (ii) repræsenterer undersøgelser af en varighed på mindre end 1 år og (iii) blev udført med forskellige former for biokul, der ofte var mangelfuldt karakteriserede med hensyn til indholdsstoffer og produktionsforhold. Det er derfor vigtigt, at langsigtet forskning igangsættes og prioriteres ud fra de relevante typer biokul, der forventes at blive tilgængelige for danske landbrugsjorde. Der er behov for en sådan forskning for at dokumentere den pyrogene kulstoflagring samt undersøge biokuls virkning på økosystemtjenester, herunder potentielt negative virkninger af biokul i landbrugsjord på både kort og langt sigt.

Mens mange tidligere undersøgelser blev udført med enkelte typer biokul og/eller enkelte jord- og plantesystemer, opfordres der nu til at fokusere mere systematisk på studier af effekten af forskellige typer af biokul og forskellige jorde for at øge den mekanistiske forståelse af biokuls effekter i jordmiljøet (Cai et al. 2021). Undersøgelser udført under kontrollerede laboratorieforhold har stadig værdi, da de gør det muligt at variere enkeltfaktorer (fx. jordtemperatur) mens andre faktorer holdes konstante. Dette kan give mulighed for indsigt i, hvordan enkeltfaktorer styrer effekterne af biokul i jord, f.eks. i forhold til stabiliteten af kulstoffet (C) i biokul. Men der bør dog i stigende grad tilskyndes til langsigtede undersøgelser under realistiske markforhold, som integrerer effekten og skæbnen af biokul i agro-økosystemet og giver mulighed for undersøgelser af langtidsvirkninger.

Jeffery et al. (2015) diskuterede, hvordan man fremmer forskningen i biokul med vægt på korrekt eksperimentelt design under hensyntagen til positive og negative kontroller, tilstrækkelig replikation og betydningen af biokuls ældning (biochar aging) i jorden og deraf følgende ændringer i de fysisk/kemiske egenskaber. Baseret på erfaringer fra tidligere undersøgelser er der fremsat en række anbefalinger til stærkere biokul-forskning i jordbrugssystemer (Jeffery et al. 2015; Cai et al. 2021). Især bør undersøgelserne bidrage til en mekanistisk forståelse af biokuls effekter snarere end blot at være beskrivende. Mekanistisk forståelse kan give en bedre mulighed for ekstrapolering af resultater fra et system til et andet og gøre det muligt at forudsige under hvilke betingelser bestemte ønskede virkninger af biokul kan forventes at forekomme eller ej (Cai et al. 2021).

Som det gentagne gange fremhæves i denne vidensyntese afhænger biokuls egenskaber i høj grad af hvilke biomasser (feedstock) og termiske konverteringsbetingelser (fx pyrolysetemperatur og opholdstid) der anvendes ved produktionen. Derfor er 'biokul' en samlet betegnelse, der dækker over produkter, der kan have vidt forskellige egenskaber. Dette gør det vanskeligt at generalisere virkningerne af biokul i landbrugsjord, som desuden også påvirkes af andre faktorer, som klima og dyrkningspraksis. Det betyder, at publikationer om biokuls effekter altid skal ledsages af en grundig karakterisering af det anvendte biokul og dets produktionsbetingelser, herunder oplysninger om biomassen (feedstock), der anvendes til pyrolyse. For eksempel er forholdet mellem hydrogen (H) og organisk C (H/C_{org} forholdet) meget afgørende for stabiliteten af biokul C, efter det er udbragt på jorden. Men også egenskaber som pH, overfladeareal og kationbytningskapacitet (CEC) er vigtige for biokuls fysiske egenskaber og hvordan biokul påvirker de biologiske processer i jorden. Ligeledes bør jordegenskaber karakteriseres, da de eventuelle virkninger af biokul skyldes samspillet mellem biokul og de givne jordbundsforhold.

Undersøgelser af biokul udbragt på landbrugsjord skal tage højde for både positive og negative virkninger i forhold til agronomiske og miljømæssige forhold. Især skal der tages hensyn til mulige negative virkninger af biokul som følge af indholdet af potentielt skadelige indholdsstoffer i biokul, såsom flygtige organiske forbindelser (VOC'er), polycykliske aromatiske kulbrinter (PAH'er), og andre forureninger som dioxin, PFAS og tungmetaller, der skal undgås og reguleres for at forhindre ophobning i jorden. Standardkriterier for egenskaber og indholdsstoffer i biokul, der sikkert kan spredes på landbrugsarealer, er foreslået i European Biochar Certificate (EBC), der er en frivillig certificerings-ordning (EBC, 2022). Disse kvalitetskriterier omfatter imidlertid ikke biologiske kriterier, hvilket gør det påkrævet at styrke forskningen i forhold til økotoksikologiske virkninger af biokul i jord både på kort og lang sigt.

Denne vidensyntese sammenfatter baggrund for og aktuel viden om anvendelsen af biokul i landbrugsjord, med fokus på danske forhold og baseret på pyrolyse af af relevente biomasser (feedsocks) som halm, afgassede fibre fra biogasanlæg og spildevandsslam. Arbejdet med videnssyntesen har endvidere afdækket en række videnshuller, der også er beskrevet i rapporten.

9.2 Sammendrag og forskningsbehov i forhold til råmaterialevalg, energibalancer og drivhusgasemissioner ved pyrolyse (Kapitel 1 og 2)

Analysen af råmaterialer (biomasser) og teknologier til industriel produktion af biokul under danske forhold har i den nuværende vidensyntese fokuseret på spildevandsslam, halm og afgassede fibre fra biogasanlæg. I en analyse af klimapåvirkningen af produktion og anvendelse af biokul i landbruget er det vigtigt at inddrage den alternative (reference) anvendelse af biomassen og bestemme den langsigtede kulstoflagring af biokul-scenariet samt reference-scenariet for at vurdere, om pyrolyse til biokul kan spille en afgørende rolle i den samlede effekt. Energiproduktion og energiforbrug spiller ligeledes en vigtig rolle i den samlede drivhusgasbalance. Beregninger af den langsigtede kulstoflagring af biokul kan baseres på en model, hvor det molære forhold mellem brint og kulstof (H/C_{org}) er en vigtig parameter, der karakteriserer den langsigtede kulstofstabilitet. Et lavt forhold er ønskeligt, hvis der skal opnås en høj kulstofstabilitet. Der foreslås en tærskelværdi på 0,7 i European Biochar Certificate (EBC, 2022), hvilket betyder, at pyrogene produkter med et H/C_{org} forhold højere end 0,7 ikke bør karakteriseres som biokul.

For spildevandsslam anslås det, at 260 kg CO_2-ækv vil blive lagret efter 100 år ved behandling af 1 ton spildevandsslam (tørstof). Dette skal sammenholdes med 118 kg CO_2-ækv lagret ved anvendelse af spildevandsslam uden pyrolyse, dvs. svarende til en nettoeffekt på 142 kg CO_2-ækv. I reference-situationen uden pyrolyse vil spildevandsslammet blive opbevaret og spredt på marken, hvilket vil give anledning til udledning af drivhusgasser i form af metan og lattergas svarende til i alt 506 kg CO_2-ækv. Derudover kan der være et energioverskud i processen, der bidrager med en positiv effekt på ca. 365 kg CO_2ækv ved substitution af naturgas efter fradrag af forbrug af el til processen.

For halm vil der efter 100 år blive lagret 427 kg CO_2-ækv efter behandling af 1 ton tørstof, hvilket skal sammenholdes med 46 kg CO_2-ækv ved nedmuldning af halm, svarende til en nettoeffekt på 381 kg CO_2-ækv. Derudover vil der være et energioverskud i pyrolyseprocessen, der kunne have en positiv effekt på ca. 596 kg CO_2-ækv ved substitution af naturgas efter fradrag af forbrug af el til processen.

For biogasfibre vil der efter 100 år blive lagret 425 kg CO_2-ækv i modsætning til 132 kg CO_2-ækv ved anvendelse af biogasfibre uden pyrolyse, svarende til en nettoeffekt på 293 kg CO_2-ækv. I referencesituationen uden pyrolyse vil biogasfibre blive lagret og udspredt, hvilket vil give anledning til drivhusgasemissioner i form af metan og lattergas svarende til i alt 197 kg CO_2-ækv. Derudover kan der være et energioverskud i processen, der kan have en positiv effekt på ca. 373 kg CO_2-ækv ved substitution af naturgas efter fradrag af forbrug af el til processen.

Alle ovenstående beregninger er baseret på en række forudsætninger, hvoraf nogle er behæftet med betydelige usikkerheder (som beskrevet i Kapitel 1 og 2), og der er identificeret en række videnshuller og forskningsbehov for at kunne foretage mere nøjagtige beregninger. Der er bl.a. behov for:

- Data fra kommerciel skala om hvordan valg af teknologi og driftsparametre (temperatur og opholdstid) påvirker egenskaberne ved biokul, energibalancer mv.
- Bedre estimering af virkningerne af pyrolyse på reduktioner af drivhusgasemissioner fra opbevaring og anvendelse af afgassede fibre og spildevandsslam i reference-scenarierne.
- Dokumentation og skøn over kvælstofomsætningen i processen og potentialer for at undgå gasformige tab af lattergas og ammoniak under lagring og udbringning af biogasfibre.
- Bedre forståelse af, hvordan separationsteknologi til afgasset biomasse og pyrolyse af biogasfibre påvirker den samlede miljøpåvirkning.
- Forbedret viden om sammenhæng mellem pyrolyseprocesdesign, biomassetype og biokul-karakteristika i forhold til stabiliteten af kulstof og effekter på drivhusgasemissioner.

9.3 Sammenfatning og forskningsbehov i forhold til biomassepotentialer (Kapitel 3)

Biomassepotentialerne for halm, biogasfibre og spildevandsslam til pyrolyse er analyseret ud fra estimater af de nuværende og de fremtidige ressourcer (Kapitel 3). Halm er den største ressource, men har en række konkurrerende anvendelser i dag og potentielt endnu flere i fremtiden. Den p.t. uudnyttede ressource af halm (korn + raps + græsfrøhalm) er estimeret til ca. 1,99 millioner tons (Mt) tørstof, idet der tages højde for, at en vis andel af landmændene ønsker at beholde halmen som kilde til organisk stof til gavn for jordens struktur og frugtbarhed. Ud over den uudnyttede ressource kan pyrolyse potentielt konkurrere om den mængde halm, der i øjeblikket bruges til energiformål (1,38 Mt tørstof). Hvis en sådan omlægning bliver favoriseret af samfundsmæssige og økonomiske incitamenter øges potentialet for halm til pyrolyse derved til 3,37 Mt tørstof. I fremtidige scenarier er der muligheder for at vælge kornarter og -sorter med en højere mængde halm, hvis landmændene kan se et marked for øget salg af halm. Det vil også være muligt at indsamle en større del af det halm, der i øjeblikket bliver tilbage på marken. På den anden side vil det generelle fald i landbrugsarealet, ønsker om flere naturarealer, vådlægning af organiske jorder mv. mindske produktionsarealet for kornafgrøder. I optimerede scenarier for 2030, hvor disse muligheder og begrænsninger indgår, estimeres de samlede halmressourcer til bioenergi og bioraffinering til 3,09-3,85 Mt tørstof.

For biogasfibre tages der kun hensyn til den nuværende biogasanvendelse af husdyrgødning (+25 % tilførsel af andre råvarer), hvorved eventuel anvendelse af halm til biogas udelukkes for at undgå dobbelttælling.

Der er beregnet en potentiel tilgængelighed af biogasfibre på 0,46 Mt tørstof, da ca. 25 % af husdyrgødningen forudsættes udnyttet til biogas i 2022. I scenarier for 2030 med øget udnyttelsesgrad for biogas, hvor ± 20 % ændring i husdyrproduktionen medregnes, er der anslået potentialer på 1,14-1,71 Mt biogasfibre til pyrolyse.

De tilgængelige mængder af spildevandsslam er usikre, da forskellige opgørelser har vist ca. 30% forskel i tilgængelig mængde. I denne syntese er estimeret, at der er 0,08-0,09 Mt tørstof, som i dag udbringes på landbrugsjord, og som kunne pyrolyseres før udbringning. Hertil kommer ca. 0,02 Mt slam, som i dag forbrændes, som muligvis i stedet kan pyrolyseres, hvorved uønskede organiske bestanddele forventes at blive nedbrudt. Der kan således estimeres en fremtidig ressource på 0,10-0,11 Mt tørstof slam, da der ikke forventes nogen væsentlig ændring i slammængderne over tid. Det skal dog sikres, at indholdet af tungmetaller ikke vil kompromittere kvaliteten til af biokul til jordbrugsformål.

Hvis fremtidige potentialer skal realiseres, kræver det en forskningsindsats, fx om muligheder for øget halmproduktion og indsamling uden at gå på kompromis med jordkvalitet og afgrødeudbytte, da antagelser herom er baseret på ældre undersøgelser og ikke er afprøvet i praktisk målestok. Det kan også være vigtigt at undersøge, om alle halmfraktioner (halm, blade, aks og avner) samt forskellige typer frøgræshalm (ca. 0,4 Mt tørstof) kan anvendes til pyrolyse og give en ensartet kvalitet af biokullet. Endelig bør virkningerne af forskellig teknologi-integration på udbyttet af energi, kulstof og næringsstoffer til genanvendelse analyseres; det kan være integration af grøn bioraffinering (udvinding af fx protein) med biogas og pyrolyse. Det vil formentlig også være vigtigt at vurdere tilstedeværelsen af tungmetaller i råvarer til pyrolyse og grænserne herfor i forbindelse med brug af biokul til landbrugsformål.

Yderligere forskningsbehov identificeret i forhold til biomasseressourcen for biokul er relateret til:

- Energi- og systemanalyse for biosektoren for at beskrive den mest optimale ressourceanvendelse af biomassen til enten energi, materialer eller til kulstofbinding.
- Vurdering af både nuværende og nær-fremtidige konkurrencemæssige anvendelsesscenarier for forskellige biomassetyper.
- Effekter af afgrødearter og -sorter på halmkvaliteten til pyrolyse, samt påvirkninger af høsttidspunkt og vejrforhold under afgrødemodning, hvilket vides at have indflydelse på forbrændingskvalitet.
- Forbedret vurdering af biomassepotentiale på lokal til national skala.
- Vurdering af forventet udvikling i national afgrødeproduktion efter nationale, regionale og globale ændringer i fødevarebehov og klima.

9.4 Sammenfatning og forskningsbehov i forhold til jordfysiske og -kemiske effekter af biokul (Kapitel 4)

Jordens fysiske egenskaber afspejler, hvordan jordkomponenterne (faste partikler, væske og gas) forholder sig til hinanden. Dette har en vigtig indflydelse på, hvor godt jorden kan understøtte plantevækst og bidrage til økosystemtjenester, såsom tilbageholdelse af næringsstoffer og gode afdræningsforhold. Det er derfor vigtigt at dokumentere og forstå, hvordan biokul påvirker jordens fysiske egenskaber. Nogle af de jordfysiske parametre, som generelt påvirkes af biokul, er (i) volumenvægten, som beskriver, hvor tæt pakkede de faste partikler er i et givent volumen og (ii) tilbageholdelsen af vand i jorden, som afspejler, hvor store/små porerne i jorden er, og hvordan de er fordelt i jorden.

I Kapitel 4 beskrives biokuls indvirkning på jordens fysiske egenskaber vurderet ved hjælp af en meta-analyse af 31 relevante artikler. Alle jordtyper var repræsenteret, både sand- og lerjord samt jordtyper med intermediær tekstur i pløjelaget. Udgangsmaterialerne til produktion af biokul omfattede halm, husdyrgødning og spildevandsslam. Tilførslen til jorden varierede fra 0,25 til 10 vægt-% med et gennemsnit på 2,5 %. På tværs af jord og type af biokul resulterede biokul-tilførsel i et gennemsnitligt fald på 9,7 % i volumenvægt og en stigning på ca. 35 % i det plantetilgængelige vandindhold (størst for sandjorde).

Jordens evne til at danne stabile aggregater er et vigtigt mål for jordkvaliteten, da stabile jordaggregater bidrager til en stabil jordstruktur, som muliggør optimal bevægelse af gasser, vand og næringsstoffer. Biokul-tilførsel resulterede i en gennemsnitlig stigning i aggregatstabiliteten på 59 %, men med en stor spredning omkring middelværdien på grund af få undersøgelser.

Jordens mættede hydrauliske ledningsevne er afgørende for jordens egenskaber fx i forhold til infiltration, dræning og transport af næringsstoffer. Effekten af biokul-tilførsel varierede fra stor stigning til en lille reduktion i ledningsevnen afhængig af både jordtype og typen af biokul. Undersøgelser har også vist, at biokul normalt ikke påvirker jordens opfugtnings-egenskaber, som kan være reducerede efter tørke.

Biokuls indflydelse på jordens kemiske egenskaber blev vurderet ved at gennemgå den tilgængelige litteratur om biokul fra halm, gødning og spildevandsslam. Jordens pH og CEC er to afgørende egenskaber, som påvirker tilgængeligheden af både næringsstoffer og vand til afgrøderne. Når plantemateriale opvarmes under pyrolyseprocessen under tilstedeværelse af alkali-ioner, dannes basiske karbonater, afhængigt af både pyrolyse-betingelserne og sammensætningen af biomassen der pyrolyseres (feedstock). Højere pyrolysetemperatur øger generelt dannelsen af karbonater og dermed biokullets evne til at øge jordens pH. Da halm ofte indeholder alkali-ioner i større mængder end træ, har halm-biokul ofte en større evne til at øge jordens pH. Dette bekræftes af forsøg, hvor der er foretaget en direkte sammenligninga af pH i forskellige typer af biokul. Biokuls evne til at øge jordens pH kan bestemmes, og da biokul således i et vist omfang kan erstatte jordbrugskalk, vil dette potentielt have værdi både økonomisk og via undgået CO_2-udledning fra tilsat kalk ($CaCO_3$).

De funktionelle grupper dannet under pyrolysen kan resultere i biokul med gode CEC egenskaber. Litteraturen angiver dog meget forskellige værdier for CEC i biokul. Og CEC for biokul er sjældent så højt som for naturligt organisk stof og har således ikke en størrelsesorden, der kan øge jordens CEC betydeligt undtagen på sandjord, hvor op til ca. 30 % forøgelse er rapporteret ved tilførsel af biokul.

Videnshuller og forskningsbehov, der identificeret i forhold til biokuls effekt på jordens fysiske og kemiske egenskaber omfatter behov for:

- Langvarige markstudier under danske forhold, herunder undersøgelser med biokul fra gylle og spildevandsslam, som p.t. er fåtallige.
- Undersøgelser af biokuls betydning for jordaggregering, især for jord med grov tekstur.
- Undersøgelser af, hvordan designer-biokul kan fremstilles med fx høj CEC eller evnen til at katalysere omsætning af lattergas til frit kvælstof.
- Undersøgelser af salinitet efter tilførsel af biokul og metoder til at fjerne K fra biokul før udbringning.
- Undersøgelser af varigheden af biokuls kalkningseffekt på forskellige danske jordtyper.

9.5 Sammenfatning og forskningsbehov i forhold til biokuls virkninger på jordbiologi (Kapitel 5)

Biokul kan påvirke jordbiologien forskelligt og på både negative og positive måder. Biokullets kemiske og fysiske egenskaber er af stor betydning, og disse egenskaber bestemmes i høj grad af råmaterialet (biomassen) og pyrolyseforholdene, herunder temperatur, opholdstid under pyrolyse og køleprocesser. Disse parametre er også afgørende for koncentrationerne af forurenende stoffer i biokul, såsom PAH'er og tungmetaller. Også mængden og partikelstørrelsen af tilsat biokul, jordens tekstur, struktur og organisk materiale og næringsindhold er vigtige faktorer. De fysiske og kemiske ændringer, som biokul bevirker i jorden, kan øge vandholdningskapaciteten, ændre pH (Kapitel 4) og skabe porerum til mikroorganismer.

Litteraturen, der beskriver effekter af biokul på jordbiologi, bidrager ofte til den samme konklusion, nemlig at virkningerne på jordbiologi er afhængige af de specifikke egenskaber ved den pågældende type biokul og den mængde, der er udbragt. Forskelligheder i biokul består blandt andet i andelen af let nedbrydeligt kulstof (en mindre pulje, typisk omkring 3 vægt-%), partikkelstørrelsen, porestrukturen og adsorptionskapaciteten i forhold til næringsstoffer andre molekyler i jorden. Derfor kan forskellige typer og endog partier af råmaterialer resultere i biokul med forskellige egenskaber og med forskellige virkninger på jordbiologien.

Desuden er biokullets virkninger på jordbundens biologi stærkt afhængige af de stedsspecifikke jordbundsegenskaber, herunder fysiske og kemiske parametre. Den kombinerede effekt af biokul-egenskaber og jordegenskaber gør det vanskeligt at nå en generel konklusion fra de mange primære publikationer om biokuls indvirkning på jordbiologi. Det understreger også behovet for standardiseret karakterisering af biokul i det mindste i forhold til kemisk og fysisk karakterisering, men helst også i forhold til biologiske funktioner i jorden, fx påvirkningen af mikroorganismer, der spiller en vigtig rolle i kredsløbet af organisk materiale og næringsstoffer.

Internationale undersøgelser af effekten af biokul på jordfaunaen har vist både positive og negative virkninger på regnorme. Et dansk treårigt feltforsøg med halmbaseret biokul anvendt i en samlet mængde på enten 3 ton/ha eller 16 ton/ha viste, at regnormens forekomst var upåvirket i forhold til både kontroller med tilført halm og kontroller uden tilført biomasse.

Undersøgelser af mikroorganismer har vist, at biokul kan ændre mikrobielle samfund i jorden, men det er ofte usikkert, (i) om dette har positiv eller negativ indvirkning på jordkvaliteten, (ii) om ændringerne er større eller sammenlignelige med normale påvirkningen fra landbrugsdrift, og (iii) hvor langvarige ændringerne er. Biokullets virkninger på jordens mikrobiologi bør opdeles i kort- og langsigtede virkninger, men indtil videre er der begrænsede data om langtidseffekter.

Stimulerende effekter af biokul kan skyldes faktorer som (i) en pulje af let nedbrydeligt organisk materiale frigivet fra biokulet lige efter udbringning på jorden, (ii) modvirkning af forsuring ved at øge pH eller (iii) forbedrede betingelser for mikrobiel aktivitet i jorden på grund af bedre porestrukturer og adgang for ilt. Negative virkninger, både kortsigtede og langsigte, kan være forårsaget af polycykliske aromatiske kulbrinter (PAH'er), flygtige organiske forbindelser (VOC'er) eller tungmetaller, som er kendte potentielle forurenende stoffer i biokul.

Ud over virkninger på levende organismer kan biokul også påvirke aktiviteten af ekstracellulære enzymer i jorden, som er vigtige for omsætning af organisk materiale. Interaktion med biokul kan muligvis stabilisere puljen af ekstracellulære enzymer, men enzymernes aktivitet er afhængig af orienteringen af de aktive sites, hvilket kan betyde at aktiviteten nedsættes ved dannelse af biokul-enzymkomplekser.

På grund af den teoretiske risiko for udvaskning af biokul til vandmiljøer er virkningerne på vandlevende organismer også undersøgt i litteraturen, og der er i den forbindelse også rapporteret om skadelige virkninger. Disse resultater vedrører imidlertid ofte virkninger af biokul tilsat direkte til vand, og effekten af biokul efter interaktion med jordpartikler før udvaskning er ikke blevet grundigt undersøgt.

Da biokul i tiltrækker opmærksomhed som en anvendt teknologi til negativ kulstofemissions i stor skala, er der behov for regler og kvalitetskriterier for biokul for at sikre miljømæssigt sikker anvendelse i landbrugsjord. Indtil videre vedrører kvalitetskriterier som foreslås af EBC (2022) hovedsageligt biokuls fysiske og kemiske egenskaber. Det er derfor rettidigt at styrke forskningsindsatsen der vedrører jordbiologiske test for bedre at forstå både kortsigtede og langsigtede virkninger af biokul i landbrugsjordens økosystem.

Videnshuller og forskningsbehov i forhold til effekter af biokul på jordbiologien omfatter:

- Langsigtede virkninger på jordlevende organismer efter udbringning af biokul under danske markforhold.
- Forståelse af, hvordan biokul påvirker de mikrobielle processer, der er involveret i omsætning af næringsstoffer, fx i det mikrobielle N kredsløb.
- Mulighed for anvendelse af hurtigt reagerende mikrobielle indikatorer til screening af økotoksikologiske effekter af biokul.
- Betydning af feedstock for biokul (fx spildevandsslam) i forhold til miljøpåvirkninger.
- Virkninger af biokul på mikrobiel celle-til-celle kommunikation og interaktioner mellem trofiske niveauer.
- Potentiel tilpasning af mikrobielle samfund til nedbrydning af relativt stabilt biokul C.
- Interaktioner mellem biokul og mineralisering af naturligt organisk kulstof i jorden.
- Virkninger af biokul på aktiviteten af ekstracellulær enzymaktivitet i jordens økosystem.
- Konsekvenser af at tilføre kulstof som inert biokul versus tilførsel frisk organisk materiale (fx planterester) som er fødegrundlag for jordens mikrobielle samfund samt højere trofiske niveauer.
- Potentiel udvaskning af biokul og indholdsstoffer fra biokul til akvatiske økosystemer.

9.6 Sammenfatning og forskningsbehov i forbindelse med kulstofbinding i jorden og drivhusgasemissioner (Kapitel 6)

Som gennemgået i Kapitel 6 konkluderer flere undersøgelser et stort potentiale for lagring af kulstof ved at inkorporere biokul i landbrugsjord. Det skyldes, at biokul nedbrydes langsommere i jorden end den oprindelige biomasse, der blev brugt til produktion af biokullet. De rapporterede nedbrydnings-hastigheder for biokul i jord varierer imidlertid meget afhængigt af både biokullets egenskaber og den jord, hvor biokullet udbringes, dvs. relateret til faktorer som jordstruktur, afgrødesystemer og temperatur. På trods af biokullets generelle langsigtede stabilitet erkendes det også, at en mindre (men variabel) andel af biokul C udgøres af relativt let omsættelige forbindelser. Som en forenkling betragtes biokul C ofte som et to-pulje-system, hvor den ene pulje beskrives som let nedbrydelig (labil) og den anden pulje beskrives som stabil. En nylig meta-undersøgelse viste, at den gennemsnitlige fordeling mellem de to puljer var 3% for labilt C og 97% for stabilt C. Dette forhold kan dog variere og skal derfor bestemmes for specifikke typer af biokul for at estimere potentialet for kulstoflagring.

Den internationale litteratur beskriver flere nye metoder til estimering af biokullets langsigtede stabilitet i jorden, herunder den foreløbige IPCC-tilgang, hvor pyrolyse temperaturen bruges til at forudsige stabiliteten.

Dette skal dog betragtes som en meget foreløbig og forenklet tilgang, da den ikke tager hensyn til de iboende egenskaber i biokullet. Yderligere forskning har dokumenteret, at nedbrydningshastigheden for specifikke typer af biokul kan estimeres mere hensigtsmæssigt ud fra det molære H til organiske C-forhold (H/C_{org} ratio) i biokullet. Således er en grundigere tilgang til estimering af biokul-stabilitet baseret på H/C_{org} forholdet blevet foreslået af Woolf et al. (2021) og er i øjeblikket den mest detaljerede metode, selvom der også her er betydelige usikkerheder involveret (fx i forhold til karakterisering af virkningerne af jordtemperatur på nedbrydning af biokul). Den målestok, der bruges til at karakterisere langsigtet stabilitet af biokul i jord, er den såkaldte F_{perm} faktor, der angiver mængden af det oprindelige biokul C, der forbliver i jorden efter 100 år. For at udlede denne værdi er der behov for omfattende ekstrapolering af data, da mange undersøgelser kun rapporterer resultater om biokul mineralisering i forsøg af få års varighed (eller mindre) og ofte under forhold, der afviger fra realistiske markforhold, fx undersøgelser udført i laboratoriet. Dette medføre i sig selv en usikkerhed i hvor godt vi kan estimere stabiliteten af biokul på langt sigt. På baggrund af en nylig dansk undersøgelse (Thers et al. 2019) blev det desuden angivet, at for at evaluere betydningen af biokul C-binding for afbødning af klimaændringer, er en målestok som F_{perm} muligvis ikke fuldt tilstrækkelig. Der er derfor behov for bedre metoder til at bestemme klimaeffekten af biokul.

I forhold til lattergas (N_2O) har flere internationale meta-studier rapporteret, at biokul kan reducere udledningen af denne kraftige drivhusgas fra landbrugsjord betydeligt. Nogle data analyser har estimeret en gennemsnitlig reduktion på 38%, men tages kun resultater fra feltstudier i betragtning (og altså ikke laboratoriestudier), så er effekten mindre, og det er også blevet rapporteret, at konklusionerne afhænger af de anvendte metoder til data analyse. Desuden er det usikkert, om effekten af biokul på N_2O emission fortsætter over flere år efter udbringning, og flere resultater tyder på at effekten er størst i det første år. Der er kun få danske feltstudier, hvor biokuls effekt på N_2O emission er blevet målt, og disse undersøgelser har ikke kunnet bekræfte en reduktion i N_2O emissionen. Så mens det er empirisk veldokumenteret ved meta-studier, at biokul generelt kan modvirke N_2O emission fra dyrkede jorder, kan feltstudier ikke altid dokumentere en sådan effekt for specifikke kombinationer af biokul, jord og klimatiske forhold (Thers et al. 2020). Denne uoverensstemmelse afspejler det nuværende videnshul i forhold til at forstå de præcise mekanismer, der styrer interaktionen mellem biokul og det biogeokemiske N kredsløb, hvor lattergassen dannes og omsættes. Øget pH i jorden efter udbringning af biokul kan medvirke til at nedsætte N_2O-emissionen, men ingen enkelt mekanisme ser ud til at kunne forklare effekten. Manglen på mekanistisk forståelse af effekten af biokul på N_2O emissionen gør det vanskeligt at forudsige under hvilke forhold vi kan forvente en gavnlig effekt og hvorfor effekten ikke altid optræder. Fremtidige undersøgelser bør undersøge, hvordan biokul ændrer de underliggende mikrobielle samfund, der bidrager til de processer i N kredsløbet, der styrer dannelsen og omsætningen af N_2O.

Endelig skal det tages i betragtning, at danske forsøg og den internationale litteratur typisk ikke rapporterer om en stigning i lattergasudledningen fra jorden efter tilførsel af biokul. Dette står i modsætning til tilførsel af den friske organiske biomasse (såsom planterester og gylle), som ofte medfører øgede N_2O emissioner.

Derfor har omdannelse af biomassen til biokul inden udbringning på marken et indirekte potentiale i forhold til at nedsætte N_2O emissionerne i et systemperspektiv.

Metan emissioner fra mineralsk landbrugsjord er generelt ubetydelige, mens oxidation af atmosfærisk metan er almindelig, og bidrager til fjernelse af CH_4 fra atmosfæren. En meta-analyse viste, at effekten af biokul på metan-oxidation kunne være negativ, men indtil videre mangler der data, der understøtter en konklusion om biokuls effekt på metan-oxidation i danske landbrugsjorde.

Forskningsbehov og videnshuller, der er identificeret i forbindelse med biokuls virkninger på kulstofbinding og drivhusgasemissioner, omfatter:

- Behov for langsigtede data om nedbrydning af biokul i jord, herunder mere viden om biokullets aldringsprocesser.
- Validering af forholdet mellem H/C_{org} ratio og biokul nedbrydning for relevante danske biomasser (feedstock) og jordbundsforhold.
- Undersøgelser af interaktioner mellem mikrobiel omsætning af biokul og omsætning af naturligt organisk materiale i jorden.
- Mekanistisk forståelse af jord-, klima- og dyrkningsmæssige forhold, der fremmer N_2O-reduktioner efter tilførsel af biokul.
- Integration af biokul i modeller for omsætning og opbygning af organisk kulstof i jord.
- Vurdering af biokuls virkning på metan-oxidation i dyrkede jorder med N tilførsel.
- Mulige virkninger af post-modificering af biokul på C stabilitet i jord, samt og direkte og indirekte drivhusgasemissioner.
- Karakterisering af jordtemperaturens effekt på nedbrydning og stabilitet af biokul.

9.7 Sammenfatning og forskningsbehov i relation til næringsstof-sammensætning af biokul og effekter på næringsstof-tilgængelighed og udbytter (Kapitel 7)

Biokul har egenskaber, der potentielt kan forbedre dyrkningsegenskaberne af landbrugsjord, fx ved forbedret vandholdende evne, bedre aggregatstabilitet og øget tilgængelighed af næringsstoffer. Det er dog vanskeligt at generalisere betydningen af disse effekter, fordi biokuls egenskaber afhænger af både biomassen (feedstock) og pyrolyseforhold. Endvidere er effekten af disse faktorer afhængige af jordtypen, klimaforhold og jordens næringsstofstatus. I flere internationale undersøgelser er der gennemsnitligt fundet en signifikant positiv effekt af biokul på jordens dyrkningsegenskaber og på udbytter, men på tempererede jorde har man oftest ikke kunnet påvise positive effekter på afgrødeudbytter. Biokul spiller således generelt en lille rolle for udbyttet i danske landbrugsjorde, der oftest er kalkede (dvs. ikke forsurede) og har et højt indhold af næringsstoffer. Der er således ikke påvist signifikante udbyttestigninger efter tilførsel af biokul under danske markforhold. Der kan ikke forventes en udbyttestigning, hvis afgrødevæksten/produktionen allerede er tæt på det potentielt mulige niveau, hvilket er tilfældet for de fleste danske landbrugsjorde. Enkelte forsøg på lille skala tyder dog på, at der på grovsandede jorde kan opnås højere udbytter ved anvendelse af biokul.

Biokul indeholder både kvælstof (N), fosfor (P) og kalium (K), og en del af indholdet af P og K er plante-tilgængeligt på kort sigt, mens den resterende del må forventes at blive plantetilgængeligt på længere sigt. Under produktion af biokul ved pyrolyse forbliver fosfor og kalium i materialet uden væsentlige tab i gasser, hvorimod en betydelig del af kvælstof går tabt i gassen i form af fx N_2, NH_3 og HCN, og tabet af kvælstof stiger med stigende pyrolysetemperaturer. N-koncentrationen i biokul falder derfor med stigende pyrolysetemperatur og C/N-forholdet i biokul stiger med pyrolysetemperatur. For eksempel indeholdt biokul fra halm behandlet ved >700°C kun 3-6 kg N/ton, mens biokul fra pyrolyse af halm ved lavere temperaturer kan indeholde op til 29 kg N/ton (vist i Tabel 7.1). Biotilgængeligheden af N i biokul er generelt lav, da det meste af N er bundet til stabile C-forbindelser. Det konkluderes, at der i danske undersøgelser ikke er fundet nettofrigivelse af N inden for de første måneder efter udbringning af biokul til jord, og N i biokul anses for at være i en meget stabil form. Derfor er det ikke relevant at fastsætte en N-udnyttelsesgrad på N i biokul. I vidensyntesen blev der imidlertid kun fundet begrænset information om N-tilgængelighed i biokul pyrolyseret ved temperaturer under 500°C. Derfor vil flere undersøgelser af N-tilgængelighed i biokul fra især pyrolyse ved lavere temperaturer være ønskelige. Det er dog usikkert, hvorvidt pyrolyse ved temperaturer på meget under 500°C vil blive anvendt i praksis på grund af risiko for lavere stabilitet af kulstoffet i det producerede biokul.

I internationale undersøgelser er der observationer af både positive og negative effekter af biokul på N-tilgængeligheden i jorden (dvs. tilgængeligheden af N fra både biokul og andre kilder end biokul). Der er dog kun få danske undersøgelser, der har fokuseret på N-tilgængelighed efter tilførsel af biokul. Biokul kan fysisk adsorbere/frigive både ammonium (NH_4^+) og nitrat (NO_3^-), men forståelsen af disse processer i biokul er mangelfuld.

På grovsandede jorder kan den samlede jordfrugtbarhed være lav, hovedsageligt på grund af en lav vandholdende kapacitet og mekanisk modstand mod rodvækst i undergrunden. Her kan jordforbedring med biokul have et potentiale til at stimulere afgrødeudbyttet.

I nogle undersøgelser er det dokumenteret, at biokul forsinker nitratudvaskningen fra rodzonen, formentlig ved stadig ukendte mekanismer, hvorved nitrat fastholdes i de små porer i biokul. Dette nitrat synes kun delvist at kunne påvises med standardmetoder, men en sådan tilbageholdelse af nitrat kan bidrage til at reducere N tilgængeligheden for denitrificerende organismer og derved reducere N_2O-emissioner. Under danske forhold mangler der undersøgelser af samspillet mellem biokul og nitratudvaskning, der potentielt kan have effekter på vandmiljøet (mindre nitratudvaskning), drivhusgas emission (mindre N_2O-emission) og N forsyningen (mere gødnings-N tilbageholdes i rodzonen). Det er også usikkert hvor store mængder biokul der skal tilføres for eventuelt at opnå en målbar effekt, og om dette vil medføre en overskridelse af fosforlofterne, som følge af P indholdet i biokul.

Mængden af P i biokul afhænger af råmaterialet (biomassen), der pyrolyseres. Tilgængeligheden af P i jorden efter tilførsel af biokul påvirkes af forskellige mekanismer, såsom: (i) biokul som en kilde til P, (ii) biokuls påvirkning af tilgængeligheden af jord-P, fx ved at ændre pH og (iii) P adsorption til biokul. Mange danske og internationale undersøgelser har vist, at mængden af opløseligt og plantetilgængeligt P stiger i jorden efter tilsætning af biokul. Det er dog generelt konstateret, at P-tilgængeligheden i biokul falder med stigende pyrolysetemperaturer. En variabel del af P i biokul er umiddelbart tilgængelig, men forsøg har vist, at en del af fosforet i biokul først bliver tilgængeligt hen igennem vækstperioden. Det betyder, at biokul normalt ikke vil være velegnet som startgødning i afgrøder, der har brug for en hurtig forsyning med P i starten af vækstperioden, men biokul kan være velegnet til at opretholde niveauet af tilgængeligt P i jorden (P-tallet).

I forhold til danske gødningsbestemmelser vil P-koncentrationen i biokul ofte være begrænsende for, hvor meget biokul, der kan tilføres per hektar. For biokul produceret af fiberfraktion (fra afgasset eller rå gylle) og fra spildevandsslam vil det kun være muligt at anvende ca. 0,5 ton biokul/ha/år som gennemsnit uden overskridelse af P-loftet på 30 kg P/ha (se Tabel 7.1). Det betyder, at biokul fra disse kilder kun kan anvendes i små mængder per ha, eller at der skal arbejdes på at fraseparere fosfor før pyrolysen. For biokul produceret fra halm vil der typisk kunne anvendes 7-9 ton biokul/ha/år (og op til 25 ton biokul/ha/år) uden overskridelse af P-loftet på 30 kg P/ha ud fra de undersøgelser, der er sammenstillet i Tabel 7.1.

Ved udbringning af biokul på arealer med stor overfladeafstrømning af vand, kan der potentielt ske tab af fosfor i afstrømmende vand via biokul partikler med en relativ høj P koncentration. Der er imidlertid ikke fundet litteratur, der kan belyse denne problematik.

Kalium koncentrationen i biokul kan være høj afhængigt af råmaterialet (feedstock). I biokul fra halm og gylle er der rapporteret 18-91 kg K/ton, hvorimod K-indholdet er lavere i biokul, der stammer fra spildevandsslam. Forholdet mellem K og P er meget højere i biokul fra halm end i biokul fra husdyrgødning, ligesom det er i råvaren. Hovedparten af K i biokul kan forventes at være plantetilgængeligt umiddelbart efter tilførslen, men andelen af tilgængeligt K varierer dog.

Forskningsbehov og videnshuller i forhold til næringsstof-sammensætning af biokul og effekter på næringsstof-tilgængelighed og afgrødeudbytte omfatter:

- Eksperimenter, der undersøger potentielle udbyttefordele ved biokul gennem forbedringer i jordens vandretention, rodudvikling og tilgængelighed af næringsstoffer, såsom fosfor.
- Bedre forståelse for under hvilke jordbunds- og klimatiske forhold, tilførsel af biokul kan øge afgrødeproduktionen, herunder viden om forskellige afgrøders respons.
- Identifikation af optimale mængder af biokul tilførsel under hensyntagen til indholdet af næringsstoffer.
- Målinger og forståelse af biokuls effekter på reduktion af nitratudvaskning i danske landbrugsjorde, herunder vurdering af kort- versus langsigtede effekter.
- Mere viden om N-tilgængeligheden i biokul ved forskellige pyrolyse temperatur, herunder pyrolyse ved relativt lav temperatur, når/hvis det er relevant.
- Viden om systemiske fordele på national og regional skala af næringsstofseparation og øget P-koncentration og mobilitet.
- Systematiske undersøgelser af biokuls potentiale til delvist at erstatte mineralsk P-gødning. Specielt undersøgelser hvordan den utilgængelige del af P i biokul evt bliver frigivet på længere sigt.
- Undersøgelser af risikoen for P tab til vandmiljøet ved overfladeafstrømning efter overfladeudbringning af biokul uden indarbejdning i jord.
- Undersøgelser af gødningseffekter af blandinger af biokul og organiske materialer.

9.8 References

Cai Y, Ok YS, Lehmann J, Chang S, 2021. Recommendations for stronger biochar research in soil biology and fertility. Biology and Fertility of Soils 57, 333–336

EBC, 2022. European Biochar Certificate - Guidelines for a Sustainable Production of Biochar, Version 10.1E of 10th January 2022. Arbaz, Switzerland.

Jeffery S, Bezemer TM, Cornelissen G, Kuyper TW, Lehmann J, Mommer L, Sohi SP, van de Voorde TFJ, Wardle DA, van Groenigen JW, 2015. The way forward in biochar research: targeting trade-offs between the potential wins. GCB Bioenergy 7, 1–13

Klimarådet 2020. Known paths and new tracks to 70 per cent reduction - Direction and measures for the next 10 years climate action in Denmark. Klimarådet, March 2020. Available at https://klimaraadet.dk/en/system/files_force/downloads/english_version_-_known_paths_and_new_tracks_to_70_per_cent_1.pdf?download=1

Thers, H., Abalos, D., Dörsch, P., & Elsgaard, L. 2020. Nitrous oxide emissions from oilseed rape cultivation was unaffected by flash pyrolysis biochar of different type, rate and field ageing. Science of the Total Environment 724: 138140.

Thers, H., Djomo, S.N., Elsgaard, L., & Knudsen, M.T. 2019. Biochar potentially mitigates greenhouse gas emissions from cultivation of oilseed rape for biodiesel. Science of the Total Environment 671: 180–188.

Woolf, D., Lehmann, J., Ogle, S., Kishimoto-Mo, A.W., McConkey, B., & Baldock, J. 2021. Greenhouse gas inventory model for biochar additions to soil. Environmental Science and Technology 55: 14795–14805.

Knowledge synthesis on biochar in Danish agriculture

- Economic assessment of biochar production and use (part 2)

Advisory report from DCA – Danish Centre for Food and Agriculture

Contents

1 Economic assessment of biochar production and use

Katarina Elofsson and Addisu Anteneh Dilnessa (reviewer: Berit Hasler).

1.1 Dansk sammendrag

I denne delrapport undersøges økonomiske aspekter af produktionen af biokul med henblik på at anvende biokul på landbrugsjorden. Som en del af denne undersøgelse, vurderes betingelserne for at biokul kan være en omkostningseffektiv foranstaltning til at afbøde klimaændringer. For at vurdere omkostnings-effektiviteten, foretages en sammenligning af omkostningerne ved produktion og anvendelse af biokul med CO_2-prisen, da dette er den omkostning der anvendes som incitament for CO_2-udledere til at reducerer udledningerne.

Den økonomiske analyse er baseret på et litteraturstudie af relevante videnskabelige publikationer fra Danmark og sammenlignelige lande i den tempererede klimazone samt offentligt tilgængelige rapporter, når det skønnes relevant. Desuden er inddraget information om energsideprodukter og næringsstofindhold fra kapitel 2 og 7 i første delrapport i vidensyntesen, i en analyse af mulig salgspris og værdi af biokul.

I den første del af kapitlet undersøges investerings- drifts- og råvareomkostningerne ved produktion af biokul for hurtige og langsomme pyrolyseprocesser. Gennemgangen viser flere studier hvor råmaterialerne halm og spildevandsslam blev analyseret, mens der ikke er fundet studier der kan benyttes til vudering af omkostninger ved produktion af biokul udfra afgassede fibre fra biogasanlæg. Dernæst estimeres potentielle indtægter ved biokulproduktionen relateret til værdien af næringsstoffer og energibiprodukterne for de tre råmaterialer, der er i fokus i nærværende rapport, dvs. halm, spildevandsslam og afgassede fibre fra biogasanlæg. Dette skøn er ikke direkte sammenligneligt med omkostningsoverslagene fra den første del af den økonomiske analyse fordi de gennemgåede undersøgelser har brugt andre datakilder til at beregne inputomkostningerne til f.eks. råmaterialer og energi i produktionen. Forskellige tilgange i litteraturen til spørgsmålet om størrelse og lokalisering af biokulanlæg, samt de omkostninger der påløber landmændene til håndteringen af biokul, diskuteres. Efterfølgende sammenlignes studiernes konklusioner i forhold til omkostnings-effektiviteten af biokul som et instrument til reduktion af drivhusgasser, samt konklusionerne vedrørende udformningen af reguleringsinstrumenterne.

Litteraturstudiet tyder på, at der er stordriftsfordele i biokulproduktion. Sådanne stordriftsfordele skal afvejes i forhold til større transportomkostninger, som følge af at det område, hvor råvarerne tages fra, er større. For at styrke og afprøve disse konklusioner, diskuteres den økonomisk optimale anlægsstørrelse i en dansk kontekst.

Biokul fremstillet på basis af spildevandsslam og afgassede fibre fra biogasanlæg har et relativt højt potentiale for at være økonomisk rentabelt for producenterne, da disse råvarer kan anses som affald. Det betyder, at prisen på disse råvarer kan være nul eller endda negativ. Transportomkostningerne kan stadig være betydelige. Det er sandsynligt, at større mængder spildevandsslam vil kunne være til rådighed inden for en mindre afstand, især i større byområder. Det kan begrunde at placere store biokulanlæg i nærheden af større byområder hvor der er tilgang til spildevandsslam. Biokulproduktion fra halm er ikke så attraktivt på grund af det den høje pris på råvaren, og det lave gødningsindhold i slutproduktet der kan medføre en lavere pris på slutproduktet.

Biokul kan potentielt være en omkostningseffektiv foranstaltning, der bidrager til opfyldelse af klimamålene. Dette er især tilfældet, hvis råvareomkostningerne er lave eller negative, som det kan forekomme for spildevandsslam og afgassede fibre fra biogasanlæg, samt også hvis det er økonomisk relevant at bygge store produktionsanlæg. Anlæg, der anvender halm som råmateriale, er også potentielt omkostningseffektive, da CO_2-reduktionsomkostningerne svarer til, eller er lavere end den CO_2-pris, som Energistyrelsen anbefaler for 2022 (83 EUR per ton CO_2). Dette tyder på, at biokul allerede på kort sigt kan være omkostningseffektivt. Alle de gennemgåede undersøgelser beskriver, at CO_2-reduktionsomkostningerne for halm, er konkurrencedygtige i forhold til den langsigtede CO_2-pris der er beregnet og foreslået af Det Økonomiske Råd for at opfylde 2030-målene. Dette gælder især for større produktionsanlæg.

Selvom tilskud til brugen af biokul på landbrugsjord kan forventes at tilskynde til øget brug af biokul, så kendes det støtteniveau, der er nødvendigt for at opnå en given optagelse af praksis, ikke. I betragtning af at teknologien er umoden, kan det begrundes at etablere økonomiske incitamenter rettet mod produktion samt læring og erfaringsudveksling blandt landmænd, såfremt brug af biokul ønskes fremmet.

Vidensyntesen har afdækket flere videnshuller. For det første er der ikke fundet nogen sammenligninger af omkostningseffektivitet mellem biokul som et kulstoftabs-begrænsende virkemiddel og andre virkemidler til biologisk eller teknisk kulstofbinding. Sådanne undersøgelser vil være værdifulde i betragtning af at biokul giver langvarige effekter på grund af kulstofbinding. For det andet er der ikke fundet studier, der undersøger den økonomisk optimale lokalisering af biokulanlæg, som tager hensyn til den rumlige fordeling af råmaterialeforsyning samt potentialet for stordriftsfordele ved størrelse på anlæggene. Vidensyntesen indikerer, at muligheden for at gøre brug af forskellige typer råmateriale i et givent anlæg kan påvirke den optimale placering. For det tredje er landmændenes villighed til at acceptere anvendelse af biokul på deres marker og de faktorer, der påvirker denne beslutning, ikke blevet undersøgt. For det fjerde, selvom nogle studier diskuterer mulighederne for økonomisk at støtte produktion og anvendelse af biokul, så er designet af effektive reguleringsinstrumenter ikke blevet undersøgt. Dette forskningsspørgsmål skal undersøges under hensyntagen til de potentielle synergier og konflikter mellem politiske instrumenter rettet mod biokul, biobrændstoffer og fossile brændstoffer, og overensstemmelser eller konflikter med reguleringer og politikker, der er rettet mod andre metoder til biologisk og teknisk kulstofbinding bør overvejes. Endelig, og relateret til formålet med denne rapport, blev der ikke fundet undersøgelser, der specifikt undersøger omkostningerne og indtægterne forbundet med at bruge afgassede fibre fra biogasanlæg som råmateriale.

1.2 Costs and revenues of biochar production

Biochar has achieved increasing attention as a potentially cost-effective measure to combat climate change. So far, relatively few studies have examined the costs and revenues that accrue to producers of biochar. In this section, we will review and compare these studies with respect to the fixed and operating costs associated with the production, the costs for feedstock, and the potential revenues for nutrients in biochar and the energy side streams. Based thereon, the resulting net costs per unit of biochar produced can be calculated.

The studies found vary with respect the the technologies applied and feedstock used. McCarl et al. (2009) makes a thorough cost-benefit analysis of biochar with the study being applied in the USA. They consider both fast and slow pyrolysis[1] with maize residues as the feedstock, taking into account transportation costs, nutrient replacement, and the private and social value of biochar and energy side streams. Shackley et al. (2011) extends on this approach by comparing biochar produced at differently large facilities in the UK using slow pyrolysis, considering nine different types of feedstock, including straw and sewage sludge. They further summarize results reported in a couple of earlier studies, including Bridgwater (2002) and Bridgewater et al. (2009), which is a UK based study investigating fast pyrolysis plants of different size, and Masek (2010), which provides costs for a slow pyrolysis plant in Japan. Barry et al. (2019) report costs for a slow pyrolysis plant using sewage sludge as feedstock in a US context, while EA Energianalyse (2020), which is a Danish consultancy report, examines costs and revenues for slow pyrolysis production with straw as feedstock, using Danish data. The amount and detail of information provided varies considerably across studies, and the technical, environmental, and agronomic data can differ across studies as well as differ from the data reported in the foregoing chapters in Part 1 of the Knoweldge synthesis. This affects the studies' conclusions on costs per unit of input and output, and the cost-effectiveness of biochar as a greenhouse gas mitigation option.

1.2.1 Investment and operating costs

In the above mentioned studies, ten observations were found in the literature that combine information on capital investment costs and plant size in terms of annual feedstock inputs. These observations are shown in Figure 1.1. As can be seen, investment costs are increasing in plant size, and the figure indicates the presence of economies of scale. None of the studies indicated that there would be any difference in investment costs between fast and slow pyrolysis plants.

[1] For definitions and description of fast and slow pyrolysis see chapter 2 in Part 1 of the Knowledge synthesis. The fast pyrolysis yields relatively less biochar and larger energy side streams, compared to the slow pyrolysis, see also chapter 2.

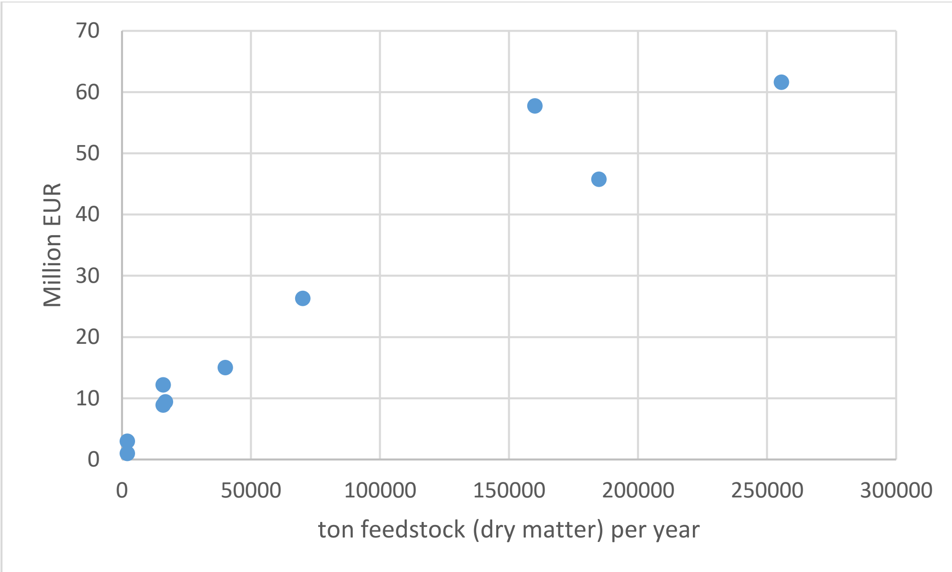

Figure 1.1. Plant investment costs in Million EUR in 2021 year value in relation to annual feedstock input (dry matter) for the plant in question.
Sources: McCarl et al. (2009), Shackley at al. (2011), Barry et al. (2019), Ea Energianalyse (2020).
Note: All observations providing information on combinations of feedstock inputs and capital costs are included. Data from Bridgwater (2002) and Masek et al. (2010) were obtained from Shackley et al. (2011). For Ea Energianalyse (2020), the project denoted Facility 5, which provides estimated future costs of a more efficient facility, is considered. Costs in national currencies are converted to 2021 year value using the CPI for each country in question using the OECD database, then converted to EUR using the average exchange rate for 2021 according to the European Central Bank.

Likewise, ten observations were found that combine annual operating costs per ton of feedstock and plant size in terms of total feedstock input, see Figure 1.2. For slow pyrolysis, operating costs seem to be decreasing in plant size, indicating economies of scale in operations. For fast pyrolysis, observations are fewer and more scattered. Moreover, the observations in Figure 1.2 do not reveal whether fast and slow pyrolysis plants differ in operating costs.

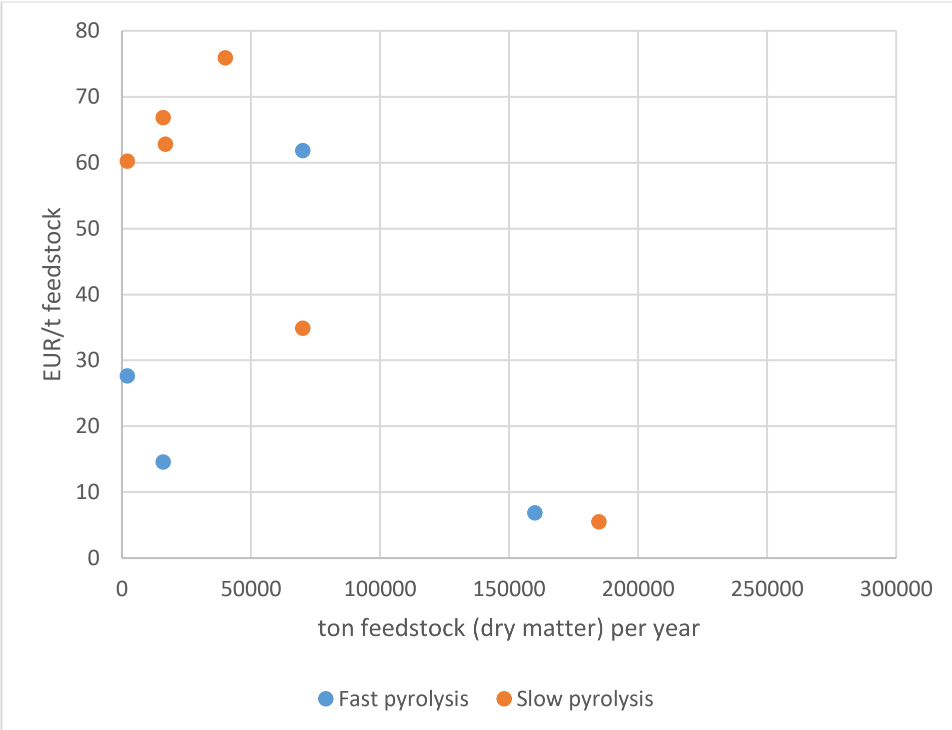

Figure 1.2. Operating costs in EUR per ton feedstock (dry matter), in 2021 year value. Blue dots refer to fast pyrolysis, red dots to slow pyrolysis.
Sources: McCarl et al. (2009), Shackley at al. (2011), Barry et al. (2019), Ea Energianalyse (2020).
Note: All observations providing information on combinations of feedstock inputs and operating costs are included. Data from Bridgwater (2002) was obtained from Shackley et al. (2011). For Ea Energianalyse (2020), facility 5, which provides estimated future costs of a more efficient facility, is considered. Costs in national currencies are converted to 2021 year value using the CPI for each country in question from the OECD database, and converting to EUR using the average exchange rate for 2021 according to European Central Bank.

Next, the information in Figure 1.1 and 1.2 was combined in order to calculate the annual production costs per ton of feedstock. To this end, capital costs were converted to annual costs assuming a 20 year life time, similarly as used in McCarl et al. (2009) and Shackley et al. (2011). Moreover we use a 3.5% discount rate as recommended by the Ministry of Finance (2021) to be used in the evaluation of public projects. The plot in Figure 1.3 reveals that the costs per unit of feedstock are decreasing in the scale of production, with plants using 150,000 – 200,000 tonnes feedstock per year have a unit cost that can be less than half of that for plants using 50,000 – 100,000 tonnes of feedstock per year. No clear difference was found between slow and fast pyrolysis plants.

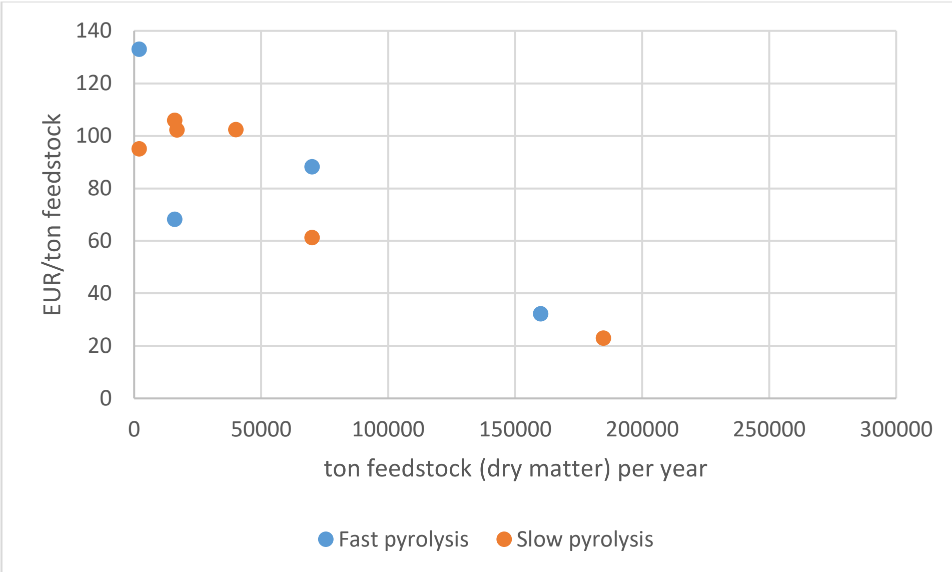

Figure 1.3. Annual costs in EUR per ton feedstock (dry matter), in 2021 year value. Blue dots refer to fast pyrolysis, red dots to slow pyrolysis. Life time of plant is assumed to be 20 years, and the discount rate is set to 3.5%.

1.2.2 Feedstock costs

Data on feedstock costs for any of the feedstocks straw, sewage sludge, and biogas digestate, was only found in five studies, one relating to maize residues, three to cereal straw, and one to sludge, see Figure 1.4. For maize and straw, the cost is calculated slightly differently across the studies. The cost for maize residues as feedstock in McCarl et al. (2009) was calculated including harvesting, transportation, and storage costs, and also considered the costs at the farm for the nutrients foregone as well as the decreased costs of tillage. Considering straw as feedstock, Shackley et al. (2011) calculate the cost assuming that the biochar will be returned to the same field. Most of the phosphorus and potassium, but less than half of the nitrogen, would then be retained. They further account for the additional cost of baling the straw, which would be incurred if the straw is to be sold for other purposes than biochar. The assumption of nutrients being returned to the same field is a possible reason for the lower cost in this study. EA Energianalyse (2020) make use of data on straw price from the Danish Energy Agency's socio-economic fuel price assumptions (Energistyrelsen, 2022). In the calculation of straw costs, Teichmann (2015) takes into account the harvesting, baling, and in-field loading of the bales, all contributing positively to costs. For sewage sludge as feedstock, considered in Shackley et al. (2011), the cost is calculated as the savings that accrue from not having to pay the gate fee for delivering the sludge to the landfill. Hence, there is a negative cost in this case. Cost estimates for biogas digestate as feedstock have not been found in studies on biochar production. However, Herbes et al. (2020) report that in some German regions, biogas digestate can be sold at favorable prices to neighboring farming business, whereas in others, the biogas plant has to pay a disposal fee or incur considerable transport costs to truck the digestate over long distances. In the latter cases, this is consistent with a negative price.

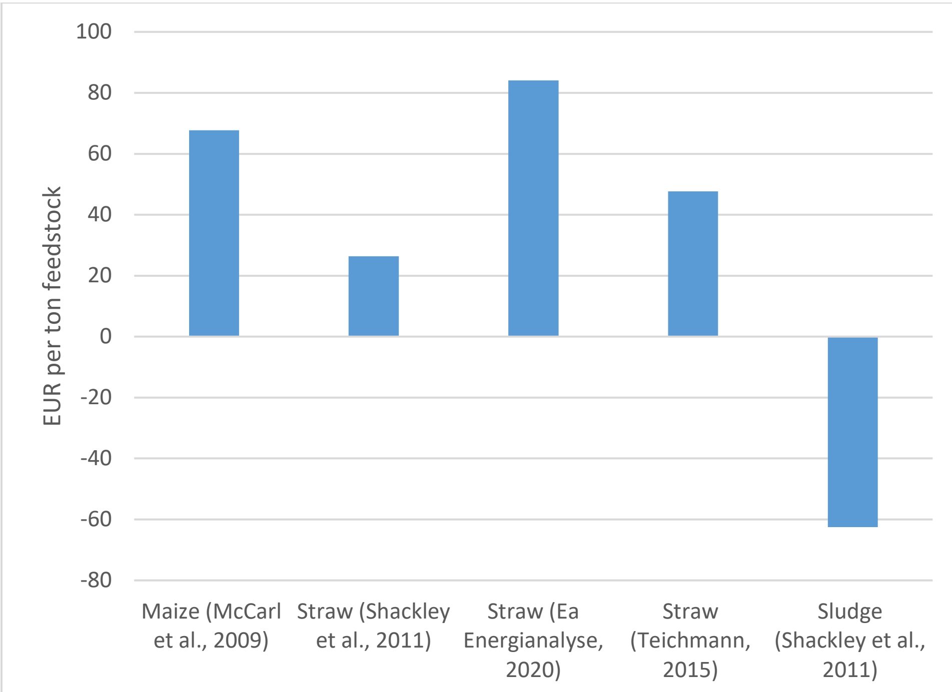

Figure 1.4. Feedstock cost in EUR per ton feedstock (dry matter), in 2021 year value for biochar produced with slow pyrolysis.
Note: Costs in national currencies are converted to 2021 year value using the CPI for each country in question from the OECD database, and converting to EUR using the average exchange rate for 2021 according to the European Central Bank. It should be noted that Danish feedstock costs could differ from the costs reported above.

1.2.3 Net production costs after consideration of feedstock purchases

Next, the net cost per unit of biochar, taking into account feedstock costs but excluding the value of energy side streams, is calculated using data presented in Figure 1.3 and 1.4 above, in combination with the reported yield of biochar, measured as the tonnes of biochar obtained from one ton of feedstock, in the same studies. This exercise suggests that the cost per ton of biochar produced from straw could range from 239 EUR per ton to 444 EUR ton, see Figure 1.5. The lowest estimated cost when using straw as feedstock is obtained from Shackley et al. (2011), and the low level is likely to be due to the comparatively low cost for straw applied in that study. The highest cost estimate with straw as feedstock is obtained from EA Energianalyse (2020). With sewage sludge as feedstock, the cost per unit of biochar would be negative for a large facility processing 184,800 tons of feedstock (dry matter) annually. For a medium-sized plant, processing 16,000 ton of sludge annually, the cost per ton of biochar would still be lower than for any of the straw-based facilities studied. The small scale plant processing 2,000 tons of feedstock annually, using sewage sludge, would have a cost per unit of biochar which is similar to the straw-based facilities, and in particular, it is similar to the cost for the same-sized straw-based facility in the same study.

In addition to the results presented in Figure 1.5, it was possible to calculate a corresponding cost for biochar produced with fast pyrolysis based on the data provided in McCarl et al. (2009). This exercise showed that the cost per ton of biochar would amount to 3465 EUR. The high cost per ton of biochar in this case is due to the low assumed biochar yield under fast pyrolysis (0.045 ton biochar per ton of feedstock) in the study in question, in combination with the total production costs here being attributed to biochar even though energy side streams could be economically more important in this case.[2]

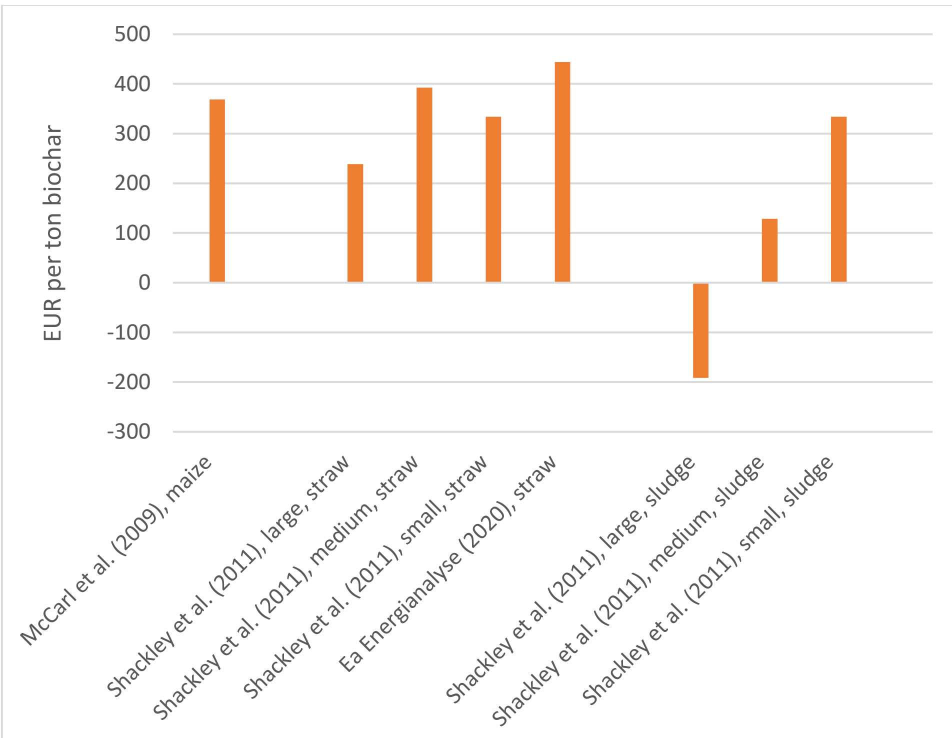

Figure 1.5. Costs per ton of biochar, including feedstock costs, but excluding the value of nutrients and energy side streams. Estimates are grouped by feedstock. For the three right-most observations, a large facility processes 184,800 tons of feedstock annually, a medium-sized plant processes 16,000 ton of sludge annually, and a small scale plant processes 2,000 ton of sludge annually. The figure displays costs using slow pyrolysis.

[2] The revenues from energy side streams are considered in the study when the cost-effectiveness of biochar as a climate change mitigation measure is evaluated, see Section 1.4.1.

1.2.4 Potential revenues for biochar nutrients and energy side streams

Farmers could be willing to pay a positive price for biochar depending on the nutrient content, and on the impacts on yields due to the improvement of soil structure and quality. In the following we abstract from yield effects due to the lack on evidence of an impact under conditions similar to those in Denmark, see Chapter 7 in part 1 of the Knowledge synthesis. Moreover, the benefits associated with biochar's carbon sequestration do not accrue to the Individual farmer, but is a positive externality that benefits society at large. This benefit is examined in sections 1.4 and 1.5 below.

To examine the potential value of biochar for farmers, we base our assumptions on nutrient contents on the data provided in Chapter 7 in part 1 of the Knowledge synthesis. We calculate the average of nutrients in biochar produced from straw, manure, and sewage sludge from Table 7.1. For nitrogen in biochar, the bioavailability could however be lower than this, and even be zero, see Chapter 7. We therefore consider the possibility that the nitrogen available could range from 0 kg per ton of biochar up to the average value of the data in Table 7.1. The phosphorus and potassium content is assumed to be given by the calculated averages. The economic value of nutrients is reflected in fertilizer prices. We obtained nutrient fertilizer prices from SEGES Farmtal Online for the 2nd quarter 2021 (SEGES, 2022). The resulting nutrient content, and the nutrient value for biochar produced with different feedstock, can be found in Table 1.1. As can be seen from the right column in Table 1.1, the nutrient value in biochar produced from straw is between 31 and 45% of that for biochar produced from biogas digestate, and between 38 and 48% of that for biochar produced from sewage sludge.

Table 1.1. Average nutrient content per ton of biochar, price of fertilizer nutrients in EUR/kg, and nutrient value of biochar in EUR/ton.

	N, kg/ton biochar	P, kg/ton biochar	K, kg/ton biochar	Nutrient value, EUR/ton biochar
Straw	0 – 13.7	4.2	37.5	39.37 – 51.24
Biogas digestate	0 – 11.5	58.1	35.7	125.18 – 135.19
Sewage sludge	0 – 21.7	39.0	27.3	86.92 –105.80

Note: The nutrient content is obtained from averaging values in Table 7.1. Manure content is assumed to equal biogas digestate content. The prices are obtained from SEGES Farmtal Online (SEGES, 2022), using data for the 2nd quarter 2021, where prices were 0.87 EUR/kg N, 1.62 EUR/kg P, and 0.87 EUR/kg K.

It must be emphasized that the values calculated in Table 1.1 should be interpreted with care. First, nutrient values also affect the price of feedstock: a high price tends to increase feedstock costs, in particular for straw because the farmers will demand a higher price for straw due to the nutrients that they forgo. The feedstock costs reported in the literature reviewed in section 1.3.4 can be the result of other values being placed on the nutrients. Second, there have been large recent changes in fossil fuel prices and therefore also in fertilizer prices. For example, SEGES report prices for the 2nd quarter of 2022 being 169%, 33%, and 31% higher for nitrogen, phosphorus, and potassium, respectively, compared to the above used price levels for the same quarter in 2021 (SEGES, 2022).

Energy outputs per ton of biochar produced was calculated using data from Chapter 2, in Part 1 of the Knowledge synthesis. It can be noted that energy outputs from sewage sludge is uncertain, see Chapter 2. Earlier economic studies have valued energy outputs assuming they can be substituted for electricity (McCarl et al., 2009; Shackley et al., 2011; Teichmann, 2015), Here, the value was calculated assuming that energy side streams can be substituted for natural gas, following the approach in Chapter 2.[3] The resulting value of the energy side streams, related to the different feedstocks, can be found in Table 1.2, suggesting that the value ranges between 51 and 59 EUR per ton biochar. Also in this case, the reader should be cautious in the use of these numbers, which may not be consistent with numbers used in reviewed studies to calculate investment, operating, and feedstock costs, and where there have been large recent changes in the fossil fuel prices.

Table 1.2. Energy output in GJ per ton of biochar, and value of of energy output, for different feedstock in EUR/ton biochar. Biochar is produced with slow pyrolysis.

	Energy output, GJ/ton	Value of energy output, EUR/ton
Straw	34.22	58.86
Biogas digestate	20.04	51.01
Sewage sludge	17.20	56.24

Note: Energy output, and tons of biochar per ton of feedstock, are obtained from Table 2.4 in Part 1 of the Knowledge synthesis (for biogas digestate), Table 2.8 (for straw), and Figure 2.17 (for sewage sludge). Natural gas prices are obtained from Energistyrelsen (2022), Table 2, and refers to prices for 2020 in 2021 year value, and was 3.94 EUR/GJ. This was multiplied by 1.66 to arrive at prices in 2021 year value. The price Increase between 2020 and 2021 is obtained using the relative prices in the second half of 2020 and 2021, obtained from Danmark statistics (www.dst.dk/en/Statistik/emner/miljoe-og-energi/energiforbrug-og-energipriser/el-og-naturgaspriser).

1.2.5 Net costs after consideration of revenues for nutrients and energy side streams

Finally, the net cost per ton biochar can be calculated for different feedstock based on the above presented data on investment and operating costs, the value of the nutrient content in biochar, and the value of the energy side streams. To this end, we use the average costs from Figure 1.5, and assume that the corresponding net cost for biochar produced from biogas digestate equals that produced from sewage sludge. We then deduct the nutrient value in Table 1.1, which could serve as a proxy for the potential market price of biochar, and the value of the energy side streams in Table 1.2. The resulting net cost is then 242 to 254 EUR/ton biochar from straw, (-96) to (-86) EUR/ton for biochar from biogas digestate, and (-72) to (-48) EUR/ton for biochar from sewage sludge. These numbers should be interpreted with care. A reasonable interpretation is that this results further supports the likelihood that biochar production using biogas digestate residues or sewage sludge has a larger chance of being economically viable compared to biochar produced from straw.

When calculating cost-effectiveness of biochar as a greenhouse gas mitigation tool in section 1.5, we make use of the net costs from the respective studies, which consider also the revenues for biochar and energy

[3] The potential costs for upgrading to the energy side streams to higher quality is not considered here.

side streams, valued with different methods and data. Thus, we do not make use of the calculations in section 1.2.4 and 1.2.5 (which were based on results from chapter 2 and 7 in Part 1 of the Knowledge synthesis) because this would imply that different data were used for the price of nutrients and energy side streams when calculating costs and revenues from production, respectively.

1.2.6 Localization and size of facilities

The localization of plants is important from an economic and climate emission perspective because it affects the need for transports. In particular, feedstock is bulky and the associated transports are therefore costly. The relative importance of feedstock hauling costs, and the costs for hauling biochar back to the fields, depends on the difference in volume of feedstock and biochar. Biochar is much less bulky than the feedstock, and the costs of transporting the final product is therefore smaller than for tranporting the feedstock. The transportation costs also depends on farmers' willingness to apply biochar on their fields. Farmers' attitudes to biochar then matter: if many farmers are reluctant to apply biochar, the biochar transport distance could increase. As shown above, one can expect economies of scale in production, favoring large plants. However, large plants require a large feedstock uptake area and hence large costs for feedstock transports. So if feedstock transportation costs are large, this will favor smaller plants. There is thus a trade-off between scale advantages and transportation costs.

Studies which investigate the economically optimal localization of biochar production plants have not been found. However, transportation costs are addressed in several studies. McCarl et al. (2009) consider the spatial dimension, and hence transportation costs, in a simplified manner by considering maize residue volume per hectar of maize, and assuming that 20% of the land around the plant is allocated to maize production. This allows them to derive a formula for the average hauling distance, and hence hauling costs. Transport distances to haul back biochar to the field are then assumed to be the same, but costs are lower due to the lower volume. Shackley et al. (2011) apply a similar approach, but do not report the details of the calculations. Evaluating plants of varying size[4], and hence considering economies of scale versus transports, Teichmann (2015) acknowledges the varying transport distance (and hence costs) associated with different plant size. She finds that the difference in transportation implies that the GHG mitigation potential is highest for small-scale pyrolysis units and lowest for large-scale pyrolysis units. Transport distances in Teichmann (2015) are calculated using a formula where the distance is determined by national total land area and the number of pyrolysis plants. Barry et al. (2019) assume a fixed transport distance, and EA Energianalyse (2020) assume a fixed transport cost per unit of volume.

[4] Using plant investment and operating costs from Shackley et al. (2011).

1.3 Costs of biochar use at farm level

In the following, we discuss the costs of managing biochar at the farm level.[5] These costs including those for the hauling of biochar to the farm, which depend on the distance and the volume. Transportation costs are calculated in McCarl et al. (2009), specific to their particular application. Shackley et al. (2011) provide a ballpark estimate of hauling and spreading costs, taking into account the capital costs for a spreader and labour costs for hauling, loading, wetting, transporting to field, and application. Based thereon, they judge that the management costs would amount to 7 EUR per ton of biochar. Teichmann (2015) follows a similar procedure for calculating on-farm costs for management of biochar.[6] For the calculation she includes costs for unloading and storing[7] the biochar at the farm, equipment and labor needed for spreading, and fuel. The results in Teichmann (2015) then show a net cost for the farmer equal to 21 EUR per ton biochar with biochar storage on the farm, and 12 EUR per ton of biochar with biochar storage at the biochar plant[8]. EA Energianalyse report a cost for spreading biochar equal to 2.30 EUR per ton biochar, while costs for labor, transports and storage are summary figures including also activities carried out by the biochar plant operator, and therefore costs that would accrue at farm level cannot be separated.

1.4 Cost-effectiveness and carbon offsetting

In this section, we first review the method for calculating the carbon impact in economic studies on biochar, which varies considerably. We then either use the reported cost per ton of biochar (when directly available in the study), or calculate the cost per ton of biochar using data provided in the study in question. The two are then used to calculate cost-effectiveness ratios. Those are compared to carbon prices recommended by Danish agencies and the CO_2 price in the EU Emission Trading System (EU ETS). The outcome is presented in section 1.4.1. The next section 1.4.2 discusses the use of policy instruments to incentivize biochar production and use.

1.4.1 Cost-effectiveness of biochar as a climate change mitigation measure

The cost-effectiveness of biochar as a greenhouse gas mitigation measure is determined by the impacts on greenhouse gas emissions, and the costs of biochar production and use. In particular the costs per unit of CO_2-eq mitigation should be compared to the same cost for other mitigation measures in order to

[5] It can be noted that the costs that are associated with the provision of feedstock are not included below, as these costs should be captured by the straw feedstock price, discussed in section 1.3.2. In addition, the benefits associated with the nutrient contents in biochar are reported in section 1.3.4.
[6] The data is obtained from Teichmann (2015) Tables 32-35. We report the gross costs for biochar used at the farm, to avoid double counting, given that feedstock costs reported in section 1.3.2. should capture the net costs to the farmer for providing the feedstock.
[7] Storage at farm level is assumed relevant in Teichmann (2015) in the case that biochar production occurs at small-scale facilities.
[8] Averaging over different fuel price paths.

determine whether biochar can be part of the cost-effective portfolio of measures in the Danish climate policy.

In the reviewed studies, the reported carbon impacts of biochar can be categorized into (i) the offsetting of fossil fuels, which results from the production of energy side streams, (ii) emissions saved and/or increased as a consequence of changes in the need for fossil fuels in the production, transports and consumption of biochar, and (iii) the impact on carbon sequestration (McCarl et al., 2009). McCarl et al. (2009) credit the total carbon content in biochar as a sequestration gain, concluding that this gain would equal 0.122 CO_2-eq per ton feedstock for fast pyrolysis and 0.963 ton CO_2-eq per ton feedstock for slow pyrolysis. Thus, sequestration is almost 8 times larger per ton feedstock for slow pyrolysis. This is then added together with all other carbon impacts relevant to (i) and (ii), resulting in a net carbon effect equal 0.823 CO_2-eq per ton feedstock for fast pyrolysis and 1.113 ton CO_2-eq per ton feedstock for slow pyrolysis.[9] After this operation, the carbon impact of biochar produced with slow pyrolysis is thus only 35% larger than that of fast pyrolysis. Using the study's assumptions on a 75% carbon content of biochar[10], and biochar yields equal to 0.045 and 0.350 ton biochar per ton of feedstock, we obtain a corresponding effect of 24.39 ton CO_2-eq per ton of biochar in the case of fast pyrolysis, and 4.24 ton CO_2-eq per ton of biochar in the case of slow pyrolysis. The high effect per ton of biochar In the case of fast pyrolysis is due to the low production of biochar per ton of feedstock in combination with the high production of energy sidestreams, and the fact that the study assumes that the energy side streams will displace electricity produced with coal. The latter implies that the displacement of CO_2-eqs will be relatively high.

Similar to McCarl et al. (2009), Teichmann (2015) counts in carbon impacts across all the three categories (i) – (iii), finding that CO_2-eq removals range from 0.648 to 1.002 ton CO_2 per ton feedstock in the case of straw, with the removal being dependent on plant size and assumptions about the type of fossil fuel replaced by the energy side streams. Given assumptions in the same study of a 35% biochar yield and 60% carbon content of the biochar, we obtain a range of 3.09 to 4.77 ton CO_2-eq per ton of biochar.

EA Energi analyse (2020) first calculate the CO_2-eq removal effect when all impacts (i) – (iii) are counted in. In addition, they argue that due to the presence of an existing green premium for the use of biofuels in the Danish transport sector, there could be a reason not to count in the associated carbon impact, although the cost savings that accrue because of the sales of the pyrolysis oil, including the premium, are still accounted for. In the first case, they arrive at an estimate of 117 EUR per ton of biochar, and in the second case 65 EUR per ton of biochar.

[9] McCarl et al. (2010) specifically take into account the following factors when calculating the carbon effect: feedstock collection on farm, transports of feedstock and biochar, replacement of nutrients in the field, saved fuel tillage, operation of pyrolysis, reduction of nutrients used on the farm, credits for displacement of coal electricity, sequestration lost due to residue removal, and sequestration gain from biochar, see table 19.7 in the study in question.
[10] It can be noted that Shackley et al.'s estimate differs from estimates provided in chapter 2, in Part 1 of the Knoweldge synthesis.

Shackley at al. (2011), on the other hand, account only for the carbon content in the biochar, and assume that 68% of the carbon remains in the soil after 100 years. As mentioned above, they also assume that biochar is returned to the same field from which the feedstock was extracted. Given the study's assumption of 60% carbon content in biochar produced from straw, and 44% when produced from sewage sludge, the CO_2 removal would equal 1.49 and 0.92 ton CO_2-eq per ton of biochar from straw and sewage sludge, respectively.

The above studies illustrate the difficulties to provide a clear conclusion regarding the cost-effectiveness of biochar as a carbon mitigation measure: it is not evident where to draw the system boundaries. In particular, it is not evident whether changes in CO_2-eq emissions resulting from changes in the use of fossil fuels should be counted in when there are other policy instruments targeting fossil fuels and biobased fuels. Also the reviewed studies vary in the underlying assumption of f.x carbon content of the biochar and hence the calculated sequestration. Given the scope of this report, the cost-effectiveness will here be evaluated using the data from the above mentioned studies on production and operation costs, and costs for feedstock purchases, reported in section 1.2.1 to 1.2.3. Hence, we do not include the estimated revenues for nutrients and energy side streams, reported in section 1.3.4, which are not directly comparable to the studies in question.

At the bottom of the table we provide, for comparison, the Energy Agency's suggested CO_2 price to be used in socioeconomic evaluations for the year 2022, the Economic Council's modeled estimate of the CO_2 price needed to meet 2030 carbon targets, and the average carbon credit price in the EU ETS in the year 2021. These carbon prices serve as a baseline, against which the cost-effectiveness of biochar can be compared. Results in the table suggest that biochar can potentially be a cost-effective mitigation tool. This is in particular the case if feedstock costs are low or negative, such as could occur for sewage sludge and biogas digestate, and it is economically relevant to build large-scale production plants: for example, results in Shackley et al. (2011) indicate a negative cost equal to -208 EUR per ton CO_2-eq removal. However, plants using straw as feedstock are also potentially cost-effective: McCarl et al. (2009), Teichmann (2015) and EA Energianalyse (2020) all report CO_2 mitigation costs that are similar to or below the carbon price recommended by Energistyrelsen (2022) for 2022, suggesting that biochar could be cost-effective already in the short term. Moreover, the reviewed studies report CO_2 mitigation costs for straw and maize residues that are competitive in relation to the longer term carbon price suggested by the Economic Council for meeting 2030 targets (Det Økonomiske råd, 2020), at least for larger production plants. Notably, this applies also for the case with fast pyrolysis studied in McCarl et al. (2009), where the carbon effects relating to the displacement of fossil fuels are important for the outcome. Biochar is less likely to be cost-efficient in comparison with EU ETS prices in 2021 that averaged 30 EUR per tonne CO_2. However, recent prices in EU ETS have reached above 100 EUR per ton CO_2. Thus, in the future, biochar could be cost-efficient also in comparison to this market. It should still be acknowledged that the cost-effectiveness of biochar depends on the spatial configuration of feedstock availability in the neighbourhood of the plant.

Table 1.3. CO_2-eq removal per ton biochar, unit net cost of biochar production, and unit cost per ton of CO_2-eq mitigation. Costs are expressed in EUR in 2021 year value.

	CO_2-eq removal per ton biochar	Cost (EUR per ton biochar)	Cost (EUR per ton of CO_2-eq removal)
McCarl et al. (2009), fast pyrolysis, maize residues	24.39	3465	142
McCarl et al. (2009), slow pyrolysis, maize residues	4.24	369	87
Teichmann (2015), slow pyrolysis, straw	3.09 – 4.77	132 – 252	28 – 81
Shackley at al. (2011), slow pyrolysis, straw	1.49	238 – 392	159 – 263
Shackley at al. (2011), slow pyrolysis, sludge	0.92	(-192) – 334	(-208) – 363
EA Energi analyse (2020), slow pyrolysis, incl. pyrolysis oil CO_2 impacts			117
EA Energianalyse (2020), slow pyrolysis, excl. pyrolysis oil CO_2 impacts, including green premium			65
Energistyrelsen (2022), 2022 CO_2 price			83
Det Økonomiske råd (2020), CO_2 price to meet 2030 targets			161
Average carbon offset price EU ETS 2021 (ICAP, 2022)			40

Note: CO_2-eq removal per ton biochar is calculated using information in the respective studies on CO_2-eq net impacts of biochar, compared to the reference use of the feedstock. Costs per unit of biochar is calculated using information in the studies on carbon content of the biochar, and information in section 1.3.3, considering investment, operating and feedstock costs. It should be noted that revenues for nutrient content and energy products are not included in the above figures. For Teichmann (2015) costs per unit of biochar is obtained as an average from Tables 37-38 in the report. For the first five rows, the cost per ton of CO_2-eq removal is obtained by dividing the number in the second column by that in the first. The two last rows are included for comparison of cost-effectiveness against other measures.

1.4.2 Policy instruments and carbon offsets

The studies discussed above, which examine the costs, revenues, and climate benefits of biochar do not address the suitable design of policy instruments. However, a few other studies discuss the potential for different policy instruments to incentivize biochar production and use. Chiaramonti and Panoutsou (2019) examine the potential of including biochar use in the Rural Development Program (RDP) in Italy, suggesting that biochar could contribute to Priority 4 aims to restore, preserve and enhance agricultural and forest ecosystems, and Priority 5 aims to promote the efficient use of resources and support the transition to a low-carbon economy. This could motivate its inclusion in the RDP. Alternatively, biochar production and use could be linked to the EU ETS allowance system, allowing the sequestered CO_2-eq to serve as an offset against emissions covered by this scheme. Moreover, policies promoting the use of the energy side streams as biofuels, e.g. blend-in quotas, could serve to promote biochar production. Their results suggest that a modest support under the RDP would be needed for biochar in combination with compost (140–70 €/ha and year), whereas a higher support would be necessary for the use of pure biochar (250–190 €/ha and year). They also discuss the potential for combining such support with compensation for offsetting under the ETS, and biofuel premiums.

Honegger et al. (2021) argue that characteristics of carbon storage technologies, including biochar production, use and sequestration, suggest a need for up-front capital and long-term operating funding, in combination with a differentiated support based on the permanence of carbon storage, and conclude that this should influence policy instrument design. They note that in the longer term, biochar ought to be supported based on carbon mitigation results.

1.5 Business model for the three scenarios

A viable business model requires that the business is profitable, or at least is not running a deficit. This applies to biochar producers as well as farmers. From society's perspective the issue of viable business models should be considered jointly with the issue of the socioeconomic benefits generated by biochar as a carbon mitigation tool, as these climate benefits are the main motivation for policy interventions. In the longer term, policy interventions should thus focus on the carbon impact. The size of potential subsidy to biochar use should then equal the marginal benefit due to the carbon mitigation effect. It can be foreseen that this marginal benefit, reflected in the carbon price, will increase over time as climate policies become successively more expensive. For example, Energistyrelsen (2021) projects that the carbon price should increase by 16% until 2030, and by 70% until 2040, compared to the level in 2022. Hence, biochar will successively become a more cost-effective mitigation option, in particular in the light of the fact that the costs of other mitigation options, such as reductions in fossil fuel use, can be expected to increase over time as the use of those are further reduced as a consequence of policies.

In the short term, support to the establishment of biochar production plants, and support to farmers for applying biochar on their fields can be necessary to reach a level of production and use which is sufficient for benefitting from economies of scale, and for learning from experience to successively find more cost-effective methods for both production and application on field. Such support is probably only motivated over a limited time period. Some guidance regarding the level of support needed for producers and land owners to be willing to be involved could be obtained from the above reported costs and revenues.

1.6 Concluding remarks

The review of the literature suggests the presence of economies of scale in biochar production. Such economies of scale must be traded off against the larger transport costs when the feedstock uptake area is larger. The outcome of such a trade-off needs to be evaluated in a Danish context to reach relevant conclusions. Such an evaluation must build on an estimation of the feedstock availability in different parts of Denmark, and the consequential feedstock transportation costs. Further, the review suggest that biochar produced from sewage sludge and biogas digestate have a higher potential for being economically viable for the producers, as these feedstocks could be associated with a low or even negative price for the biochar producer. Also in this case transport costs can be considerable. It seems likely that at least in larger urban areas, larger quantities of sewage sludge could be available within a smaller distance, which could favour locating large biochar plants in the neighbourhood of such areas. However, much of the sewage sludge In Denmark Is currently applied directly to the field, and to evaluate the availability in different parts of Denmark it is necessary to further study the situation in different locations. Biochar production from straw is further disadvantaged by the low nutrient content in the final product, implying that its value in agricultural production is lower.

Biochar can potentially be a cost-effective measure contributing to climate targets. Although support to the use of biochar on farmland can be expected to increase its use, the level of support needed to achieve a given uptake of the practice is not known. Given the immaturity of the technology, economic incentives targeting biochar production as well as learning and sharing experience among farmers could potentially be motivated. In addition to directly supporting biochar production and use, the introduction of a CO_2 tax on fossil fuels would strengthen the incentives for biochar production by raising the willingness-to-pay, and hence prices, on the energy side streams.

All conclusions of the economic feasibility of biochar, and the relevance of biochar as a climate policy measure, depend on costs and prices. During the last year, markets have been unstable, and energy and fertilizer prices have increased considerably. High energy and fertilizer prices positively affect both the costs of biochar production, and the potential prices that can be obtained for the outputs. The net effect on the profitablity of biochar production, and the cost-effectiveness of biochar for the purpose of carbon storage, cannot be determined from the reviewed studies, but would require a separate analysis.

1.7 Knowledge gaps

This literature review shows several knowledge gaps. First, we did not find any comparisons between biochar as a carbon mitigation tool, and other measures for biological or technical carbon sequestration such as for example increased incorporation of crop residues in the soil, and carbon capture and storage (CCS). Such studies should be valuable given the importance of permanence and certainty of such sequestration. Second, we have not found any studies examining the economically optimal localization of biochar plants, considering the spatial configurations of feedstock provision and the potential for scale economies in biochar plant size. It can be foreseen that the possibility to make use of several different types of feedstock in a given plant could affect the optimal choice of localization. This would need to be examined in the Danish context, taking into account feasible locations for biochar production. Moreover, to examine the economic viability of biochar production, it would be valuable to have further knowledge on the price paid, or compensation obtained, by farmers that currently apply sewage sludge or biogas digestate on their fields as this is important for the prices that would be paid by the biochar plant operator. This data could be directly collected from the suppliers and receipients in question. Third, farmers' willingness-to-accept application of biochar on their fields, and the factors that affect this decision, has not been investigated. Further knowledge on this could be obtained by carrying out surveys on the willingness-to-accept biochar, which would inform on the necessary compensation to farmers. Such a study could also shed light on different farmers' preference for both supplying straw as feedstock, and for using bicohar on the field. Fourth, although some studies discuss the possibilities to financially support biochar production and use, the design of efficient policy instruments has not been investigated. This research question needs to be explored considering the potential synergies and conflicts between policy instruments targeting biochar, biofuels, and fossil fuels, and needs to consider the consistency with policies targeting other methods for biological and technical carbon sequestration. Finally, and related to the purpose of this report, no studies were found that examine specifically the costs and revenues associated with using biogas digestate as feedstock, or to the externality costs associated with hazardous substances and their potential presence in biochar.

1.8 References

Barry D, Barbiero C, Briens C & Berruti F. 2019. Pyrolysis as an economical and ecological treatment option for municipal sewage sludge. Biomass and Bioenergy, 122, 472-480.

Bridgwater A. 2002. The future for biomass pyrolysis and gasification: status, opportunities and policies for Europe. ALTERNER Programme, European Commission, contract 4.1030/5/01-009/2001 (2002).

Bridgwater A. et al. 2009. Life cycle and techno-economic assessment of the northeast biomass to liquids project. 31.

Chiaramonti D & Panoutsou C. 2019. Policy measures for sustainable sunflower cropping in EU-MED marginal lands amended by biochar: case study in Tuscany, Italy. Biomass and Bioenergy, 126, 199-210.

Ea Energianalyse. 2020. SkyClean. Pyrolyse af halm og nedmuldning af biokul som klimavirkemiddel. As available 02-06-2022 at https://www.ea-energianalyse.dk/da/publikationer/skyclean-pyrolyse-af-halm-og-nedmuldning-af-biokul-som-klimavirkemiddel/.

Energistyrelsen. 2022. Samfundsøkonomiske beregningsforudsætninger for energipriser og emissioner.

Herbes C, Roth U, Wulf S & Dahlin J. 2020. Economic assessment of different biogas digestate processing technologies: A scenario-based analysis. Journal of Cleaner Production, 255, 120282.

Honegger M, Poralla M, Michaelowa A & Ahonen H M 2021. Who is paying for carbon dioxide removal? Designing policy instruments for mobilizing negative emissions technologies. Frontiers in climate, 3, 672996.

ICAP. 2022. International Carbon Action Partnership (ICAP) – ETS Prices. Retrieved on 29 August 2022 from https://icapcarbonaction.com/en/etsprices.

McCarl B A, Peacocke C, Chrisman R, Kung C C & Sands RD. 2009. Economics of biochar production, utilization and greenhouse gas offsets. In Lehmann J, Joseph S (Eds).: Biochar for environmental management. Earthscan, London (pp. 341-357).

Ministry of Finance (Finansministeriet). 2021. Dokumentationsnotat – den samfundsøkonomiske diskonteringsrente. 7. januar 2021. As available 2022-06-09 at https://fm.dk/media/18371/dokumentationsnotat-for-den-samfundsoekonomiske-diskonteringsrente_7-januar-2021.pdf.

SEGES. 2022. Farmtal Online. Gødning, prognosepriser. As available 2022-06-22 at https://farmtalonline.dlbr.dk/Grid/uiGrid.aspx?Farmtal=230&ViewType=View&Start=01-07-2020&Slut=01-04-2023.

Shackley S, Hammond J, Gaunt J & Ibarrola R. 2011. The feasibility and costs of biochar deployment in the UK. Carbon Management, 2(3), 335-356.

Teichmann I. 2015. An economic assessment of soil carbon sequestration with biochar in Germany.

Økonomiske råd. 2020. Økonomi og Miljø 2020: Kapitel I: Dansk klimapolitik frem mod 2030.

About DCA

DCA - Danish Centre for Food and Agriculture is the entrance to research in food and agriculture at Aarhus University (AU).

The Centre comprises AU departments with food and agricultural science activities. These are primarily Department of Agroecology, Department of Animal Science, Department of Food Science, Centre for Quantitative Genetics and Genomics, and parts of Department of Engineering.

DCA has a Centre Unit, which supports and coordinates DCA activities in relation to research based policy support, industrial and sector collaboration, international collaboration, and communication.

Research results from DCA
Research results are published in international scientific journals, and they are available at the university publication database (pure.au.dk).

DCA reports
DCA also publishes a report series, which primarily communicates policy support tasks from DCA to the Ministry of Food and Environment of Denmark. Further publications include reports that communicates knowledge from research activities. The reports may be downloaded free of charge at the DCA website: dca.au.dk.

Newsletters
A Danish and English DCA newsletter communicate knowledge within agricultural and food research, including research results, advice, education, events and other activities. You can register for the free newsletter at dca.au.dk.

SUMMARY

During the last years the application of biochar to arable land has been suggested as a method to improve soil fertility and plant growth, reduce greenhouse gas emissions and sequester carbon for centuries. The rapid increase in the interest in biochar means that empirical documentation and mechanistic understanding need to be assessed continuously. In the present report, background and current knowledge have been synthesized in relation to the use of biochar in agricultural soils primarily under Danish conditions and based on major streams of available feedstock. The report also features an economical assessment and points to areas with knowledge gaps where better documentation and research is needed in relation to the pyrogenic carbon capture and storage as well as the persistent effects of biochar on soil ecosystem services.