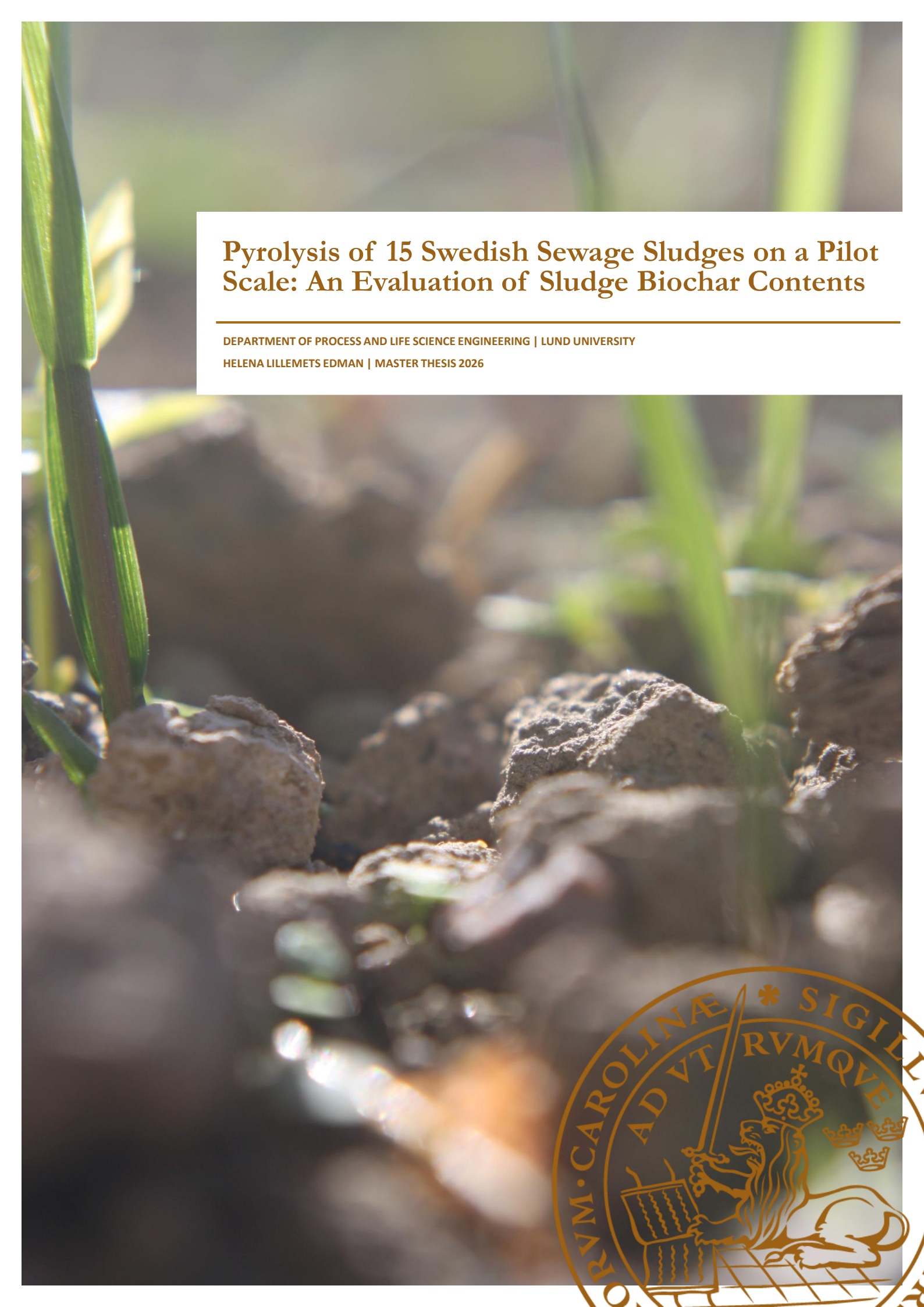




Pyrolysis of 15 Swedish Sewage Sludges on a Pilot Scale: An Evaluation of Sludge Biochar Contents

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Picture on front page: Harry's soil. Photo by Helena Lillemets Edman.

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Preface

This thesis was written in collaboration with VA SYD in their project Testbed Ellinge, using data from their newly built pilot plant at Ellinge WWTP, Eslöv.

I appreciate the helpful guidance I have received from my supervisors Åsa Davidsson and David Gustavsson during this thesis. I want to give a special thanks to David for giving me opportunities to meet the people involved in Testbed Ellinge as well as to present my work to a wider audience, and a special thanks to Åsa for her kind support and encouragement. I would also like to thank my examiner Per Falås for his interest in this thesis. Lastly, I am thankful for my supporting boyfriend, family and friends, on whom I have leaned many times during this final semester of my engineering degree.

Populärvetenskaplig sammanfattning

Pyrolys av avloppsslam: Tungmetaller både avskiljs och anrikas i slambiol

Genom att bryta ner organiska mikroföroreningar är pyrolys av avloppsslam ett fördelaktigt sätt att återföra fosfor till åkermark utan att bidra till ackumuleringen av PFAS i marken, men var tar tungmetallerna vägen när slammet hettas upp?

Som en grundläggande byggsten i DNA är fosfor en helt avgörande förutsättning för liv, både i människor och de grödor vi äter. Då vi inte kan ersätta fosfor som gödselmedel är det viktigt att vi säkrar en långsiktigt hållbar fosforanvändning. Mineraliska fosforreserver finns främst i Marocko och Kina och en fosforanvändning baserad på utvinnandet av dessa medför två problem: 1) De mineraliska fosforförråden är ändliga och kommer ta slut och 2) beroendet av import gör Europa geopolitiskt sårbart.

Det är därför nödvändigt att vi återcirkulerar den fosfor som redan är bruten och det är gynnsamt om Sverige blir mer självförsörjande i processen. Den fosfor som en gröda tar upp ansamlas, efter konsumtion, i avloppsvattnet och slutligen i avloppsslammet. Idag sprids ca 60% av avloppsslammet i Sverige på åkermark som ett organiskt gödselmedel med goda skörderesultat. I slammet ansamlas dock även organiska mikroföroreningar (som PFAS) och tungmetaller, vilka kan leda till en ackumulering av PFAS i mark och dagmask. Hur kan vi säkra den nödvändiga återföringen av fosfor till åkermark utan att sprida föroreningar? En lösning kan vara att hetta upp slammet.

Vid upphettning utan syre, kallat pyrolys, kan organiska mikroföroreningar brytas ner, och vissa tungmetaller kan även avskiljas från avloppsslammet. Samtidigt stannar fosfor kvar i produkten slambiol. Detta har visats i många studier, men det saknas kunskap om hur slambehandling och olika reningsprocesser påverkar slambiolens innehåll. Därför startades projektet Testbädd Ellinge där avloppsslam från femton olika svenska avloppsreningsverk har pyrolyserats vid 600–800 °C i en pilotanläggning. I detta examensarbete undersöktes hur förbehandlingen och innehållet av dessa slam, samt pyrolystemperatur, påverkade slambiolens innehåll med fokus på tungmetaller och näringsämnen.

Resultaten bekräftade att kadmium och kvicksilver kan avskiljas från slambiol och att en pyrolystemperatur på 650–750 °C kan vara optimal för att maximera kadmiumavskiljning och minimera kväveförluster. Ökade mängder av krom och nickel i slambiol tydde på att byggnadsmaterialet i pyrolysugnen potentiellt eroderar. Utöver effekten av eventuell erosion ökade koncentrationerna av de mer stabila tungmetallerna som följd av förlusten av organiskt material under upphettningen, vilket ledde till att biokolen i lägre grad mötte lagkrav gällande tungmetallinnehåll jämfört med slammen. Denna anrikande effekt blev större i rötade slam än orötade och skillnader i tungmetallkoncentrationer mellan slam från olika reningsverk, som följd av t.ex. olika reningsmetoder eller tillskott från industriellt avloppsvatten, bidrog till halterna i slambiol. Trots högre halter av vissa tungmetaller har studier visat att pyrolys gör tungmetallerna mindre biologiskt tillgängliga. I kombination med minskade halter av kadmium och kvicksilver, samt nedbrytningen av organiska mikroföroreningar, utgör slambiol därför ett fortsatt intressant alternativ till slamspridning.

Summary

With limited phosphorus supplies there is a growing need of recovering phosphorus from sewage sludge. Potentially reduced risks of heavy metal and organic micropollutant contamination makes sludge biochar, a product of pyrolysed sewage sludge, a promising alternative as an organic fertilizer. The effect of pyrolysis temperature, treatment methods and sludge characteristics on sludge biochar contents were examined by pyrolysing sewage sludges from fifteen different WWTPs at pyrolysis temperatures between 600-800 °C in a pilot-scale, continuous screw, slow pyrolysis unit. Mass flows of nutrients and heavy metals were determined.

Sludge biochar contents of mercury could be reduced to trace amounts at the lowest pyrolysis temperature while cadmium and nitrogen concentrations decreased with increasing pyrolysis temperature. A pyrolysis temperature of 650-750 °C could be optimal depending on if nitrogen retention or cadmium removal is prioritised. The concentrations of other heavy metals increased during pyrolysis due to a high retention and loss of mainly organic matter, causing more of the biochars to exceed limit values compared to the sludges. The likelihood of a biochar meeting legal requirements regarding heavy metal content increased if the sludge was undigested and if the initial heavy metal content of the sludge was low, highlighting the importance of upstream work. Considering the removal of cadmium and mercury, as well as the immobilisation of other heavy metals and the removal of organic micropollutants, pyrolysis of sewage sludge can on the whole provide environmental advantages to sludge spreading. An accumulation of chromium and nickel in the biochar could be caused by erosion of the machinery. The water scrubber had a low potential of removing cadmium and mercury from the flue gas.

Sammanfattning

Med begränsade fosforförråd finns ett ökande behov av att återföra fosfor från avloppsslam till åkermark. En potentiellt minskad risk för spridning av tungmetaller och organiska mikroföroreningar gör slambiol, en produkt av pyrolyserat avloppsslam, till ett lovande alternativ som organiskt gödselmedel. Effekten av slamegenskaper och driftparametrar på slambiolinnehåll undersöktes genom pyrolys av avloppsslam från femton olika avloppsreningsverk vid pyrolystemperaturer mellan 600–800 °C i en pyrolysugn med kontinuerliga skruvmatare. Massflöden av näringsämnen och tungmetaller bestämdes.

Redan vid den lägsta pyrolystemperaturen reducerades kvicksilverinnehållet i slambiol till odetekterbara värden medan koncentrationerna av kadmium och kväve minskade med ökande pyrolystemperatur. En pyrolystemperatur på 650–750 °C kan därmed vara optimal beroende på om kväveretention eller kadmiumavskiljning prioriteras. Koncentrationerna av andra tungmetaller ökade under pyrolys till följd av höga retentionsgrader och förlust av främst organiskt material, vilket ledde till att biokolen överskred tillåtna värden av tungmetaller i högre grad än slammen. Sannolikheten att ett slambiol mötte lagkraven gällande tungmetallinnehåll ökade om slammet var orötat och om det initiala tungmetallinnehållet i avloppsvattnet var lågt, vilket visar på vikten av uppströmsarbete. Avskiljningen av kadmium och kvicksilver, samt immobiliseringen av andra tungmetaller och nedbrytningen av organiska mikroföroreningar, tyder på att pyrolys av avloppsslam sammantaget kan ge miljömässiga fördelar över slamspridning. Tillskott av krom och nickel kan ha orsakats av erosion av maskindelar. Vattenskrubben hade en låg potential att rena rökgaserna från kadmium och kvicksilver.

List of Abbreviations

AD = Anaerobically digested

BC = Sludge biochar

CEPT = Chemically enhanced primary treatment

DC = Dryer condensate

DS = Dry solids

DSS = Dewatered sewage sludge

EBPR = Enhanced biological phosphorus removal

PAH = Polycyclic aromatic hydrocarbons

PBDE = Polybrominated diphenyl ethers

PFAS = Per- and polyfluoroalkyl substances

SG = Scrubbed flue gas

SW = Scrubber water

UD = Undigested sewage sludge

VS = Volatile solids

VSR = Volatile solids reduction

WWTP = Wastewater treatment plant

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1 Introduction

Phosphorus was added to the list of critical raw materials for the EU in 2014 (European Commission, 2014) because of our dependence of it as a fertilizer, its limited supply and the limited mining possibilities in Europe (European Commission, 2014). With an increased need of phosphorus recovery and self-sufficiency, sewage sludge has been identified as an alternative organic phosphorus fertilizer due to its high phosphorus content and local availability, giving benefits such as improved soil structure and increased crop yields (Börjesson & Kätterer, 2018). The interest in utilizing sewage sludge as a fertilizer has increased over the last few years, as the fraction of Swedish sewage sludge that was spread on farmland increased from 35% in 2017 to 60% in 2024 (Naturvårdsverket, 2025).

Despite growing interest, the use of sewage sludge in agriculture has been coupled with concerns regarding the spreading of heavy metals and organic micropollutants, such as pharmaceuticals and per- and polyfluoroalkyl substances (PFAS). Although long-term field trials indicate no to limited crop uptake (Hörsing *et al.*, 2014; Andersson, 2015; Österås *et al.*, 2015; Zaragüeta *et al.*, 2021) and limited soil accumulation (Hörsing *et al.*, 2014; Österås *et al.*, 2015; Börjesson & Kätterer, 2018) of heavy metals and most organic micropollutants, sludge spreading has been linked to the accumulation of PFAS in earth worms and crop residues (Österås *et al.*, 2015; Kärrman *et al.*, 2024). Considering this and the current revision of the Sewage Sludge Directive which may lead to stricter limit values of contaminants in sewage sludge (Egle *et al.*, 2023), alternatives to sludge spreading may become necessary to evaluate.

Thermal treatment technologies such as pyrolysis (heating in the absence of oxygen) and incineration can be used to destroy PFAS (Thoma *et al.*, 2022) and other micropollutants (Zielińska & Oleszczuk, 2015; Hoffman *et al.*, 2016; Moško *et al.*, 2021) in sewage sludge, while preserving phosphorus (Yuan *et al.*, 2016; Vali *et al.*, 2023). The interest for thermal treatment of sewage sludge is increasing in Sweden, with the first Swedish full-scale pyrolysis plant soon being taken into operation outside Stockholm (Lazic, 2025) and the first Swedish full-scale mono-incineration plant being planned to be constructed in Umeå (Vakin, 2023). Compared to incineration, pyrolysis is advantageous in that it can sequester substantial amounts of carbon present in the sewage sludge, storing it in the product called biochar, or *sludge biochar*, potentially up to millions of years (Sanei *et al.*, 2024). Besides the lower climate impact, the physical and chemical properties of sludge biochar can provide agronomic benefits when used as a fertilizer, such as improved soil structure and water holding capacity (Hossain *et al.*, 2020) as well as decreased nutrient and heavy metal leaching (Méndez *et al.*, 2012; Ji *et al.*, 2022), while recycling phosphorus.

While studies have investigated the effect of pyrolysis temperature on sludge biochar contents and mass flows (Kistler & Widmer, 1987; Méndez *et al.*, 2013; Yuan *et al.*, 2015), these studies generally only examine one or a few types of sewage sludges and almost always on a laboratory scale. There is therefore a lack of knowledge regarding how and why sludge biochar contents vary and how the variability is affected by wastewater treatment methods, sludge pretreatment and sludge characteristics. There is furthermore a lacking experience of operation of pyrolysis units in pilot and full scale. In examining sludge biochars produced at the pilot-scale pyrolysis plant in Testbed Ellinge, with sludges from fifteen Swedish wastewater treatment plants (WWTPs), this thesis addresses these research gaps.

1.1 Aim

The aim of this thesis was to study the variability of the chemical characteristics of biochar produced from sewage sludge, and how these depend on sludge characteristics and operating conditions. More specifically, the analysis will be focused on the fate of heavy metals and plant nutrients and how these are affected by:

- Pyrolysis temperature
- Wastewater treatment methods and characteristics of influent wastewater
- Sludge treatment, specifically anaerobic digestion

2 Background

2.1 The Spreading of Sewage Sludge on Farmland

As already mentioned in the introduction, the fraction of sewage sludge that is spread on farmland in Sweden has increased over the last few years and reached 60% of the total sludge production in 2024 (Naturvårdsverket, 2025). This section describes the legislative framework as well as the environmental effects of spreading sewage sludge on farmland.

2.1.1 Legislation and Certification

The spreading of sewage sludge on farmland in Sweden is regulated by the EU in the Council Directive 86/278/EEC of 12 June 1986 on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture, or the *Sewage Sludge Directive*, with maximum permitted heavy metal concentrations in sewage sludge that is spread. This has been adopted with even stricter limit values into Swedish law in 20 § of the statute about the handling of chemicals (SFS 1998:944) and in the ordinance on the usage of sewage sludge in agriculture by the Swedish Environmental Protection Agency (SNFS 1994:2). The laws specify the maximum permitted concentrations of heavy metals in sewage sludge that is spread (SFS 1998:944), as well as maximum dosages of nutrients and heavy metals per hectare (ha) (SNFS 1994:2).

In addition to the legislative requirements, about 50% of the sewage sludge produced in Sweden in 2024 was Revaq certified (Revaq, 2025). The Revaq certification entails upstream work by the WWTPs to reduce heavy metal and micropollutant concentrations in the wastewater and the resulting sewage sludge, sludge hygienisation to avoid the spreading of salmonella and traceability of the sludge spreading (Revaq, 2026). Besides this, the Revaq certification has long-term goals regarding the accumulation rate of heavy metals in the soil caused by sludge spreading, with a maximum of 0.20% per year (Revaq, 2026). This ensures that it takes at least more than 500 years for the heavy metal contents in the soil to double. These maximum heavy metal dosages are shown in Table 2.1 as maximum metal/P ratios, calculated with an average dosage of 22 kg P/(ha*year), which is the highest permitted phosphorus dosage for soils with medium to high phosphorus contents when applying sewage sludge (SNFS 1994:2).

Table 2.1. Maximum permitted concentrations of heavy metals in sewage sludge to be spread on farmland according to the Sewage Sludge Directive (86/278/EEC), Swedish law (SFS 1998:944) and the long-term goals of the Revaq certification (Revaq, 2026). DS = Dry solids.

Heavy metal	86/278/EEC mg/kg DS	SFS 1998:944 mg/kg DS	Revaq mg/kg P
Cd	20-40	2	21.4
Cr	-	100	1 818
Cu	1 000-1 750	600	13 636
Hg	16-25	2.5	10.5
Ni	300-400	50	1 136
Pb	750-1 200	100	1 136
Zn	2 500-4 000	800	27 273

Table 2.1. also shows the maximum permitted concentrations of heavy metals in sewage sludge that is spread on farmland according to EU legislation (86/278/EEC) and Swedish legislation (SFS 1998:944). Exceptions to the long-term goal of an accumulation rate of a maximum of 0.20% per year are lead and cadmium, as lead concentrations are more heavily regulated by legislation and the long-term goal for cadmium is a maximum Cd/P ratio equivalent to the average concentration in mineral fertilizers in addition to the average Cd/P ratio in blackwater (7+14=21 mg Cd/kg P).

2.1.2 Effects of using Sewage Sludge as a Fertilizer

The nutrient content in Swedish municipal sewage sludge has rather consistently contained about 2.7% phosphorus and 4.8% nitrogen since the year 2000 (Statistiska Centralbyrån, 2025). The concentrations of the heavy metals cadmium, copper, chromium, mercury, nickel, lead and zink in municipal sewage sludge that is spread on farmland in Sweden varies from year to year but have generally decreased since the year 2000 (Statistiska Centralbyrån, 2025), as seen in Figure 2.1 with a few heavy metals as examples.

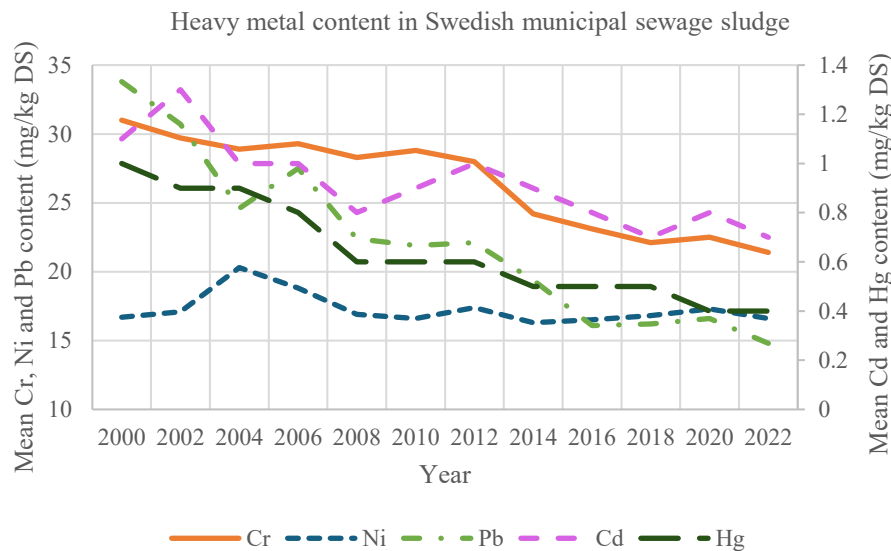


Figure 2.1. Mean heavy metal content in Swedish municipal sewage sludge (Statistiska Centralbyrån, 2025). DS = Dry solids.

Nutrients

Since 1981, field trials have been conducted on two sites in Scania, Sweden, to study the long-term effect of applying dewatered and anaerobically digested sewage sludge on crop growth and soil conditions. The trials have shown that sewage sludge improves soil structure and quality by increasing soil carbon content and lowering soil bulk density (Börjesson & Kätterer, 2018). Despite relatively high phosphorus contents in the sludges applied to the soils (3-4% of the dry solids (DS) compared to the general average of 2.7%), only about 2% of the applied phosphorus was soluble and plant available after 30 years of application (Börjesson & Kätterer, 2018). Similarly, the nitrogen availability in the sludge was low with a nitrogen use efficiency of 3-8% (Börjesson & Kätterer, 2018). Despite this, fertilization with sewage sludge increased yields when plant nutrients were limited. Börjesson and Kätterer (2018) suggested that the increased yields were more so a consequence of the sludge's contribution to an improved soil structure, promoting plant growth, than of an increased soil nutrient content, considering the limited nutrient availability in the sludge.

Heavy Metals

The long-term application of sewage sludge did not increase plant uptake of heavy metals grown in the field trials in Scania (Andersson, 2015), despite moderate accumulation of copper, zinc and mercury in the soil (Börjesson & Kätterer, 2018). With the exception of these three metals, no significant increase in soil concentration was observed for other heavy metals when fertilizing with sewage sludge. Limited crop uptake and soil accumulation of heavy metals have been detected in long-term field trials conducted in northern Spain as a result of sludge spreading, although not exceeding legislative limit concentrations for food or soil (Zaragüeta *et al.*, 2021). Furthermore, Börjesson *et al.* (2014) showed that the addition of sewage sludge did not affect the microbial community in the soils, and thus that the metal concentrations in the sludge were not harmful to microbes. Juhansun *et al.* (2021) similarly concluded that no long-term effects of sludge application could be seen on the microbial communities in the soils.

Organic Micropollutants

Despite the presence of organic micropollutants, such as pharmaceuticals and per- and polyfluoroalkyl substances (PFAS), in sewage sludge (Statistiska Centralbyrån, 2025; Magnér *et al.*, 2016), most micropollutants are undetectable in soils fertilized with sludge (Hörsing *et al.*, 2014; Österås *et al.*, 2015). More easily degradable pollutants, like certain pharmaceuticals, can be degraded in the soil (Magnér *et al.*, 2016). Meanwhile, sludge spreading can cause persistent chemicals like PFAS and perfluorooctanesulfonic acid (PFOS) to accumulate in soil and microbiota (e.g. earthworms) (Österås *et al.*, 2015; Kärrman *et al.*, 2024). Regarding crop uptake, different crops are more or less susceptible to micropollutants. No traces of micropollutants have been found in wheat grown on plots exposed to over 40 years of sludge spreading (Österås *et al.*, 2015), while nonyl- and octylphenol were detected in sugar beets grown on the same plots (Hörsing *et al.*, 2014). The concentrations were, however, low enough not to pose health risks by consumption. All things considered, the research suggests that the long-term spreading of sewage sludge on farmland does not pose health risks to humans or soil ecosystems, with regards to micropollutants (Österås *et al.*, 2015).

Climatic Impact

The spreading of sewage sludge on farmland requires sludge storage, which can produce considerable greenhouse gas emissions in the form of methane and nitrous oxide (Majumder *et al.*, 2014; Willén *et al.*, 2017). Sludge spreading can also increase nitrous oxide emissions from farmland after land application (Willén *et al.*, 2016).

Anaerobic digestion of sewage sludge can recover considerable amounts of energy as methane rich biogas (Khawer *et al.*, 2022), reducing greenhouse gas emissions from sludge storage and land application.

2.2 Pyrolysis of Sewage Sludge

Pyrolysis is defined as the heating of organic materials in the absence of oxygen and has recently gained more attention as a sewage sludge treatment alternative due to advantages concerning nutrient recovery, energy efficiency (Cao & Pawłowski, 2012), soil amendment potential and the decomposition of organic micropollutants (Pelagalli *et al.*, 2024).

2.2.1 Pyrolysis Products

Products of pyrolysis can be biochar, pyrolysis gas or oil and the fractions of these can vary depending on feedstock characteristics and operating parameters (Pelagalli *et al.*, 2024).

The biochar fraction is usually described as the biochar yield and defined as the ratio between dry solids (DS) in the biochar and DS in the sludge. The biochar yield decreases with increasing pyrolysis temperature due to increasing loss of mainly organic matter (Kistler & Widmer, 1987; Hossain *et al.*, 2011; Yuan *et al.*, 2013) and is typically around 60% at 500 °C and around 50% at 700 °C for anaerobically digested sludges (Hossain *et al.*, 2011; Zhu *et al.*, 2022; Vali *et al.*, 2023), while usually closer to 30-40% for undigested sludges (Vali *et al.*, 2023). The type of sewage sludge (e.g. primary, secondary, mixed) also affects the biochar yield, which is usually lower (35-40%) in primary sludges at 650-700 °C (Gascó *et al.*, 2005; Zhu *et al.*, 2022). In connection with decreasing biochar yields, Yuan *et al.* (2013) observed an increase in pyrolysis gas and oil yield with increasing temperatures, while Khanmohammadi *et al.* (2015) observed a significant increase in the gas yield but only a slight increase in the oil yield with increasing temperatures.

Pyrolysis gas and oil can be energy rich due to the contents of hydrocarbons (Hossain *et al.*, 2009). In full-scale pyrolysis systems, pyrolysis gas can be recirculated back into the system as fuel and, depending on sludge characteristics (Hossain *et al.*, 2009) and operating conditions (Karaca *et al.*, 2018), sustain an energy-neutral process (Hossain *et al.*, 2009).

2.2.2 Operating Conditions

Slow, Fast and Flash Pyrolysis

Pyrolysis can be divided into three categories (slow, fast and flash pyrolysis) depending on residence times and heating rates (Table 2.2).

Table 2.2. Typical operating conditions of three types of pyrolysis.

Operating Parameter	Unit	Slow Pyrolysis	Fast Pyrolysis	Flash Pyrolysis
Heating Rate	°C s ⁻¹	0.1-1	10-200	>200
Residence time		5-30 minutes	0.1-0.3 s	<1 s
Pyrolysis temperature	°C	300-700	400-600	750-1000

The different kinds of pyrolysis can be used to obtain desired products, where slow pyrolysis produces a higher amount of biochar while fast and flash pyrolysis give higher oil and gas yields (Pelagalli *et al.*, 2024).

Operating Parameters

The pyrolysis temperature has been identified as the most important operating parameter during slow pyrolysis, affecting e.g. biochar yield, heavy metal concentrations and pH (Agrafioti *et al.*, 2013; Hossain *et al.*, 2013), while residence times generally have little effect on biochar characteristics (Agrafioti *et al.*, 2013).

Reactor Types

There are different types of pyrolysis reactors, such as fixed/fluidised bed, rotary kiln and auger/screw reactors, providing different advantages. Fixed bed reactors are suitable for batch processes and are usually used on a laboratory scale, while rotary kiln and screw reactors can be used in continuous processes and are more common in full-scale pyrolysis units (Pelagalli *et al.*, 2024).

2.3 Sludge Biochar

Biochar produced from the pyrolysis of sewage sludge is usually called *sludge biochar*. Sludge biochar has been identified as an advantageous organic fertilizer alternative to

sewage sludge, with the potential to improve soil fertility and increase crop yields (Sousa & Figueiredo, 2016; Radwaniecka *et al.*, 2025) both through the recycling of nutrients but also due to altered physical and chemical characteristics gained during pyrolysis (Paz-Ferreiro *et al.*, 2018; Pelagalli *et al.*, 2024). Several studies have been made on the behaviour and fate of heavy metals and nutrients during pyrolysis of sewage sludge on a laboratory scale and some have been made on their effectiveness as fertilizers. This section gives an overview of the current knowledge in the field.

2.3.1 Physical and Chemical Biochar Properties

Carbon Content

The carbon content of sludge biochar can vary significantly depending on sludge characteristics but generally decreases slightly with pyrolysis temperature due to losses of volatile organic matter (Kistler & Widmer, 1987; Hossain *et al.*, 2011; Yuan *et al.*, 2013). Adhikari *et al.* (2019) observed a decline in sludge biochar carbon content from 49.9% to 46.0% of the DS between 400-600 °C. Other studies have observed sludge biochar carbon contents around 20% of the DS with a slight decrease between 500 °C and 600-700 °C (Hossain *et al.*, 2011; Jin *et al.*, 2016). There is a limited number of studies investigating the effect of sludge characteristics (e.g. primary or mixed sludges) and sludge pretreatment on carbon content.

Ash Content

Sludge biochar generally has a higher ash content than biochars produced from other feedstocks (e.g. forestry or crop residues) (Ji *et al.*, 2022), and it increases with pyrolysis temperature (Hossain *et al.*, 2011). Biochar ash content typically ranges between 55-70% of the DS at 500 °C (Hossain *et al.*, 2011; Adhikari *et al.*, 2019) and between 60-80% at 700 °C (Yuan *et al.*, 2015; Adhikari *et al.*, 2019).

Specific Surface Area, Porosity and CEC

Sludge biochar has a high specific surface area and porosity, which can improve soil structure by decreasing soil bulk density and improving soil water holding capacity (Hossain *et al.*, 2020). Despite this, pore clogging due to the high ash content in sewage sludge contributes to a lower surface area and porosity in sludge biochar compared to biochars produced from feedstocks with a higher lignin content, such as forestry residues (Ji *et al.*, 2022). The high ash content also contributes to a higher cation exchange capacity (CEC) in sludge biochar compared to biochars from other feedstocks as the negatively charged mineral compounds in ash (such as PO_4^{3-}) promote cationic adsorption (Ji *et al.*, 2022).

pH

Regarding pH, many studies have found that the pH of biochars increase with increasing pyrolysis temperature (Kistler & Widmer, 1987; Hossain *et al.*, 2011) and can range from acidic to alkaline. This could be due to the loss of organic acids during pyrolysis (Yuan *et al.*, 2015). Biochar can thus be used to amend or prevent soil acidification (Dai *et al.*, 2017).

2.3.2 Nutrients

Phosphorus

Close to 100% of phosphorus can be retained in the biochar during pyrolysis at 450°C (Yuan *et al.*, 2016), 600 °C (Adhikari *et al.*, 2019), 700 °C (Zhu *et al.*, 2022) and 900°C

(Vali *et al.*, 2023). Vali *et al.* (2023), however, noticed a total phosphorus loss up to 20% in undigested biochar pyrolysed at 700-900°C and suggested it may be attributed to the treatment of the wastewater as well as the pretreatment of the sludge. As a result of the high retention of phosphorus and the loss of volatile matter during pyrolysis, the total phosphorus concentration typically increases in the biochar compared to the sludge (Hossain *et al.*, 2011; Yuan *et al.*, 2013; Yuan *et al.*, 2016), allowing sludge biochar to increase phosphorus contents in soils (Yuan *et al.*, 2016). Despite the increase in total phosphorus concentration, the fraction of bioavailable phosphorus in sludge biochar typically decreases with higher pyrolysis temperatures, as phosphorus is converted into more mineralized, insoluble forms such as apatite (Xu *et al.*, 2016; Zhu *et al.*, 2022). This could be disadvantageous from an agronomic point of view short term, but Wang *et al.* (2013) have suggested that biochar containing less bioavailable phosphorus could be used as a slow-release fertilizer and thereby sustain a more stable phosphorus supply over time.

There are also studies indicating that soils amended with sludge biochar have more bioavailable phosphorus, leading to a higher plant uptake of phosphorus and higher yields (Lehmann *et al.*, 2003). One explanation for this is that the high surface area of the biochar provides favourable environments for microbes, leading to increased microbial growth and enzymatic activity (Hossain *et al.*, 2020). This can promote the growth of phosphorus solubilizing microbes and thereby increase phosphorus availability (Zhou *et al.*, 2020). Moreover, the high porosity and surface area of the biochar in combination with an increased carbon content increases the cation exchange capacity (CEC) of the soil, which generally allows for increased nutrient retention and thereby decreases nutrient leaching from the soil (Yuan *et al.*, 2016; Hossain *et al.*, 2020).

Nitrogen

The nitrogen content in sludge biochar decreases with increasing pyrolysis temperature (Hossain *et al.*, 2011; Agrafioti *et al.*, 2013), likely due to the loss of volatile nitrogen compounds (Bagreev, Bandosz & Locke, 2001). Hossain *et al.* (2011) noticed a decline in nitrogen content at temperatures over 300 °C and Figueiredo *et al.* (2018) similarly identified a pyrolysis temperature of 300 °C as optimal when only considering nutrient availability. Compared with incinerated sewage sludge ash that only contained trace amounts of nitrogen, Kleemann *et al.* (2017) observed that biochar produced from flash-pyrolysed sewage sludge at 850°C still contained significant concentrations of nitrogen, despite a decrease in nitrogen content.

Potassium

Potassium, similarly to phosphorus, is mostly retained in biochar produced from pyrolysed sewage sludge and therefore increases in concentration with increasing pyrolysis temperature (Hossain *et al.*, 2011), despite having a relatively low boiling point around 760°C. Yuan *et al.* (2015) could see that 65.0% and 83.3% of potassium was retained in the biochar after pyrolysis at 300 °C and 700 °C, respectively, indicating that a fraction of potassium was volatilized as well. The bioavailability of potassium in biochar can increase during pyrolysis (Lehmann *et al.*, 2003) and the application of sludge biochar on farmland typically leads to increased plant uptake of potassium (Hossain *et al.*, 2020).

2.3.3 Heavy Metals

Mass Flows

Kistler and Widmer (1987) observed increased concentrations of copper, chromium, nickel, zink and lead in sludge biochar compared with the sewage sludge it was derived from,

correlated with increasing pyrolysis temperature, and many studies have since found the same (Méndez *et al.*, 2013; Yuan *et al.*, 2015; Jin *et al.*, 2016). This is likely a consequence of a reduced overall biochar mass due to the evaporation of mainly organic matter coupled with high boiling points of the heavy metals, preventing them from evaporating and leading to an enrichment of heavy metals in relation to organic material (Yuan *et al.*, 2015). Contrary to previous studies, Han *et al.* (2017) observed that lead could evaporate at pyrolysis temperatures of 850°C, lowering the lead concentrations in the biochar produced from sewage sludge. Vali *et al.* (2023) also noticed a reduction in zink concentrations after pyrolysis of sewage sludge. Wang and Tomita (2003) explain this by suggesting that the presence of carbon and lime assist the vaporization of both lead and zink. There are also some studies suggesting that chromium can evaporate at sufficiently high pyrolysis temperatures. Yuan *et al.* (2015), for example, observed a 30% reduction in chromium content during pyrolysis of sewage sludge at 700 °C while Hossain *et al.* (2011) saw a 50% reduction in chromium content at 700 °C.

Regarding the more volatile heavy metals, Kistler and Widmer (1987) found that cadmium and mercury tended to evaporate at temperatures higher than 600 °C and 350°C respectively, which has been observed in subsequent studies as well (Méndez *et al.*, 2013; Yuan *et al.*, 2015; Jin *et al.*, 2016). This means that pyrolysis can effectively reduce the concentrations of cadmium and mercury in the resulting biochar compared to the sludge it originated from.

Bioavailability

Despite increased concentrations of certain heavy metals in the biochar (Kistler & Widmer, 1987), the bioavailability of these metals decrease during pyrolysis. Kistler and Widmer (1987) observed a reduction in solubility of heavy metals in the biochar compared to the sludge with rising pyrolysis temperatures, as a result of increasing pH values. Using other methods, He *et al.* (2010) showed that the bioavailability of cadmium, copper, lead and zink decreased with increasing pyrolysis temperature, while Hossain *et al.* (2011) showed the same for cadmium, copper, zink, manganese and iron. Jin *et al.* (2016) showed that, with increasing temperature, the fractions of the biounavailable or potentially biounavailable residual and oxidizable forms of heavy metals increased, while the bioavailable fractions decreased. Li *et al.* (2021) found this as well and made an ecological risk assessment taking toxicity and total concentration into consideration, where they found that sewage sludge posed a higher ecological risk than sludge biochar regarding the metals cadmium, chromium, copper, nickel, lead and zink, especially at temperatures higher than 600 °C. This suggests that biochar may be a more environmentally safe fertilizer than sewage sludge considering the bioavailability of heavy metals, despite higher concentrations in the biochar compared to the sludge.

Effects on Soil Properties

When applied to farmland, the increased metal content in sludge biochars compared to sludge does not imply heavy metal accumulation in soils and poses low risks for soil contamination (De Figueiredo *et al.*, 2019). Soils amended with sludge biochar also have a lower risk of heavy metal leaching compared to soils amended with sewage sludge, due to the immobilization of heavy metals (Méndez *et al.*, 2012; Kumar *et al.*, 2018). This is an effect of the high porosity and surface area of sludge biochar, as well as their high ash content and CEC compared with biochars produced from other feedstocks (e.g. forestry residues or crop residues), promoting adsorption of heavy metals (Ji *et al.*, 2022). For this reason, Ji *et al.* (2022) suggested that sludge biochars could be especially suited for the

remediation of heavy metal polluted soils, as well as of soils polluted with cationic organic pollutants, and can thereby improve the environmental state of the soil. Long-term trials are, however, needed to further ascertain the effects of sludge biochar on soil properties.

2.3.4 Organic Micropollutants

Pyrolysis can effectively remove organic micropollutants such as PFAS (Thoma *et al.*, 2022), polycyclic aromatic hydrocarbons (PAHs) (Zielińska & Oleszczuk, 2015a; Moško *et al.*, 2021), endocrine-disruptors (Hoffman *et al.*, 2016; Moško *et al.*, 2021) and pharmaceuticals (Moško *et al.*, 2021) from sewage sludge, producing biochars with lower ecotoxicity. The removal efficiency typically increases with pyrolysis temperature, and temperatures over 600 °C has been suggested as sufficiently high to remove most compounds beyond detection limits (Moško *et al.*, 2021). While concentrations of most PAHs decrease during pyrolysis, the concentrations of certain PAHs, like naphthalene, can increase (Zielińska & Oleszczuk, 2015a). This can be the effect of higher pyrolysis temperatures (Zielińska & Oleszczuk, 2015a) and pyrolysis reactor types (De la Rosa *et al.*, 2019).

2.3.5 Wastewater Treatment and Sludge Pretreatment

Some studies have investigated the influence of sludge characteristics on biochar properties regarding sludge types (e.g. primary, secondary), wastewater treatment methods and sludge treatment.

Sludge Types

Zhu *et al.* (2022) investigated the phosphorus content and bioavailability of sludge biochar originating from different treatment steps of the same WWTP. They concluded that biochar originating from dewatered cake sludge (anaerobically digested and dewatered mixed sludge) was especially suitable as a phosphorus fertilizer compared with biochar from primary (moisture content of about 92%), secondary (moisture content of about 92%) and digested sludge (not dewatered) due to higher levels of available phosphorus. They also concluded that pyrolysis temperature affected the chemical forms of phosphorus present in the biochar and that sludge biochar from dewatered cake sludge pyrolysed at 500 °C could be used as a fertilizer directly due to it containing higher amounts of bioavailable phosphorus, while biochar from the same sludge pyrolysed at 700 °C could be used as a slow-release fertilizer due to higher amounts of inorganic and apatite phosphorus.

Chemically Enhanced Primary Treatment

Lee *et al.* (2026) studied the influence of chemically enhanced primary treatment (CEPT) on sludge biochar characteristics. Sludges that had undergone CEPT had higher Cl^- contents due to the dosing of coagulants and this led to a higher degree of heavy metal removal during pyrolysis as Cl^- contributed to the volatilisation of heavy metals. However, they also found that the stability of heavy metals was lower in biochars produced from sewage sludges that had undergone CEPT compared to ones that had not, due to a lower initial stability in the sludge. This was especially notable at temperatures above 800 °C, suggesting that pyrolysis temperature can be a way of optimizing biochar characteristics when the sludge has undergone CEPT.

Anaerobic Digestion

Zielinska *et al.* (2015b) studied biochars produced from sewage sludges from four different WWTPs in Poland and found that both the carbon content of the sludges and the pyrolysis temperature affected the properties of the biochars. More recently, Vali *et al.* (2023) compared the pyrolysis of two sewage sludges from Sweden, one of which had been

anaerobically digested and the other one not, in order to investigate the effect of pretreatment on the biochar characteristics. They found that the digested sludge produced a higher biochar yield than the undigested, likely due to a higher ash content leading to a lower relative loss of organic matter in the pyrolysis.

2.3.6 Pilot Scale

In recent years, a few studies have been made on the pyrolysis of sewage sludge on a pilot scale, none however, with the same focus as this study. Salimbeni *et al.* (2023) and Bianca *et al.* (2026) used pilot-scale, continuous screw, slow pyrolysis ovens with a feeding rate of 3 kg/h to study the properties of biochars produced at 450 °C and 550 °C respectively. These studies did not examine the influence of pyrolysis temperature or sludge characteristics on biochar properties.

2.3 Testbed Ellinge

Testbed Ellinge is a project with the aim of exploring the possibility of using pyrolysis as a sludge treatment alternative in Sweden (Sweden Water Research, 2026). The project includes a study at the newly built pilot plant at Ellinge WWTP in Eslöv, Sweden, where municipal sewage sludge from at least 30 different Swedish WWTPs will be, or has already been, dried and pyrolysed. The project started in 2021 and is expected to continue until 2027. The project is financed by five Swedish municipal water utility companies (VA SYD, Gryaab, Kungsbacka kommun, Lidköping miljö och teknik and VIVAB), several research organisations (the Swedish Innovation Agency, Vinnova, the Interreg Baltic Sea Region, Sweden Water Research and Svenskt Vatten Utveckling) as well as by the water and wastewater organisations providing sewage sludge for analysis.

The project provides an opportunity to identify operational challenges and beneficial operating conditions connected to the pyrolysis of sewage sludge, which can be applied to the optimization of future full-scale plants. Moreover, the project will explore what factors influence the variability of the physical and chemical properties of sludge biochar, including sludge origin, treatment methods and operational conditions. The application of the sludge biochar as a fertilizer will also be evaluated by examining the plant uptake of phosphorus and heavy metals in short-term pot and frame cultivation trials. Lastly, possible markets for sludge biochar will be investigated, considering the agricultural and environmental value of sludge biochar, possible business models and legislation.

3 Materials and Methods

3.1 Sewage Sludge Characteristics

Dewatered municipal sewage sludge from fifteen different WWTPs in Sweden were pyrolysed at temperatures ranging from 600-800 °C. Among the sludges, varying wastewater treatment methods as well as sludge treatment methods were used, allowing for the analysis of the effect of these treatment methods on biochar characteristics (Table 3.1).

Table 3.1. Wastewater characteristics and treatment methods concerning the dewatered sewage sludge, as well as abbreviations of WWTPs that will be used from this point. Irregular metal dosing or dosing <1 g/m³ is considered “Low” and marked in yellow. PE = Population equivalent (1 PE = 70 g BOD7/day), EBPR = Enhanced biological phosphorus removal, AD = Anaerobic digestion.

WWTP	Abbreviation	Load	Industrial load	P precipitation		EBPR	Anaerobic digestion	Volatile solids reduction	Number of sludge samples
		P.E.	% of BOD load	Al	Fe			%	
Lidköping WWTP	Lidköping	40 000	13	Yes	No	No	No		1
Lycksele WWTP	Lycksele	10 000	Yes, unknown	No	No	No	No		1
Margretelund WWTP	Margretelund	20 000	Yes, unknown	Yes	Yes	No	No		2
Ormanäs WWTP	Ormanäs	10 000	No	Yes	No	No	No		1
Duvbacken		70 000	10	No	No	Yes	Yes	41	1
Ellinge WWTP	Ellinge	80 000	72	No	Yes	No	Yes	Yes, unknown	5
Henriksdal WWTP	Henriksdal	940 000	6	No	Yes	No	Yes	48	1
Himmerfjärdsverket		230 000	13	Low	Yes	No	Yes	69	1
Hässelholm WWTP	Hässelholm	20 000	10	No	Yes	No	Yes	50	1
Klagshamn WWTP	Klagshamn	80 000	4	No	Yes	No	Yes	Yes, unknown	2
Kungsängsverket		170 000	8	Low	Yes	No	Yes	50	1
Käppalaverket		510 000	13	No	Yes	No	Yes	58	1
Skebäckverket		120 000	9	Low	Yes	No	Yes	56	1
Uddebo		60 000	5	Yes	No	No	Yes	44	1
Öresundsverket		180 000	14	No	No	Yes	Yes	Yes, unknown	1

Eleven of the sludges had been anaerobically digested and the remaining four were undigested. Another parameter of interest is the use of aluminium and iron containing coagulants in the wastewater treatment, for example as a part of chemical phosphorus removal. Two of the WWTPs, Duvbacken and Öresundsverket, use enhanced biological phosphorus removal (EBPR), while chemical phosphorus removal had been used for the remaining sludges. Industrial load in the influent wastewater is another parameter potentially affecting biochar characteristics, varying between 0% and 72% among the analysed sludges. Table 3.1 also shows the number of sludge samples taken from each WWTP.

3.1.1 Analysis Methods

Sludge samples were taken in connection with the pyrolysis of the sludge and were tested externally by Eurofins. Five samples were taken of each sludge batch (4x1000 mL, 1x400 mL) evenly distributed across one of approximately seven plastic bags, each containing approximately 700 kg, that was delivered to the plant from each WWTP. Two sets of

analysis methods were used when determining the heavy metal content in the dewatered sludge, one by a Swedish laboratory and the other by a Finnish laboratory. The major difference between the sets of analysis methods was that the Swedish laboratory used inductively coupled plasma atomic emission spectroscopy (ICP-AES) in the analysis of most heavy metals, while the Finnish laboratory used inductively coupled plasma mass spectrometry (ICP-MS). Regarding mercury, the Swedish laboratory used aqua regia with cold vapour atomic fluorescence spectroscopy, while the Finnish laboratory used ICP-MS. A comprehensive list of methods and standards used for the characterisation of the dewatered sludge is found in Table A1 of Appendix A.

The ability of the two methods to detect heavy metals were compared. Slightly higher levels of mercury and lead were detected in the dewatered sludge using ICP-MS, while slightly higher levels of cadmium, zink, copper and iron were detected using ICP-AES, but these differences were not significant. Significantly higher levels of chromium and nickel could, however, be detected using ICP-MS ($p < 0.05$) (Figure 3.1). For this reason, the results from the measurement set using ICP-MS were used in the analysis of all metals in the dewatered sludge, to assess the most conservative scenario regarding heavy metal concentrations.

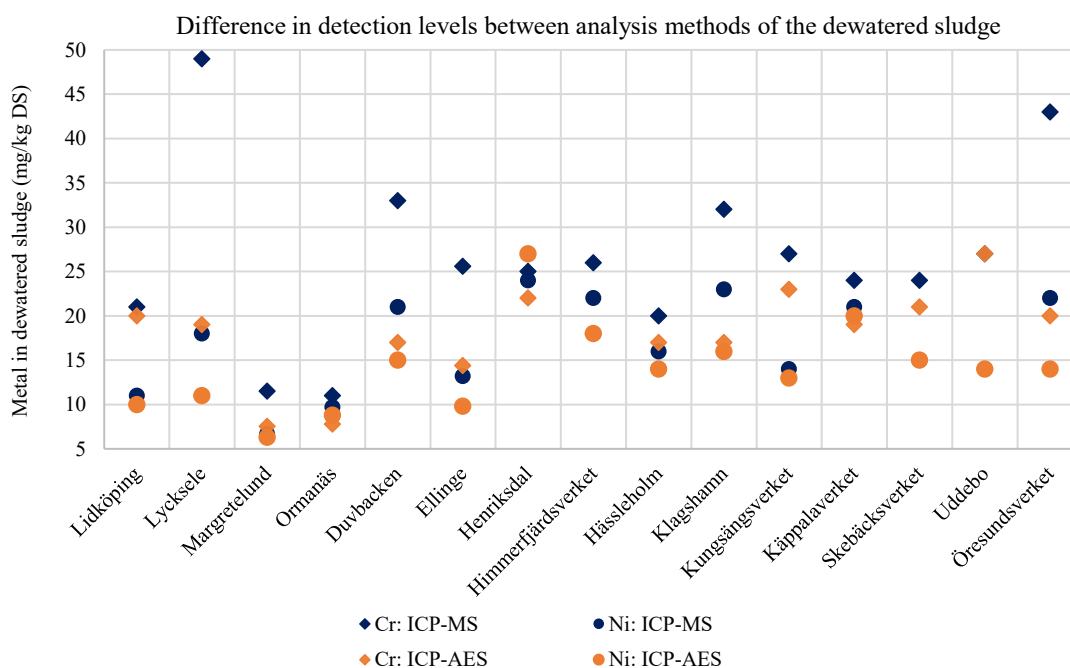


Figure 3.1. Comparison between the detection levels of two measurement methods for heavy metals, inductively coupled plasma mass spectrometry (ICP-MS) and inductively coupled plasma atomic emission spectroscopy (ICP-AES). DS = Dry solids.

3.2 The Pyrolysis Process

The pilot plant was of the model HECLA Setores 300 HTE by Aquagreen ApS, Denmark, including a steam dryer and a continuous screw, slow pyrolysis unit, as well as a condenser and a water scrubber (Figure 3.2). A schematic overview of the pilot plant can be seen in Figure 3.3.

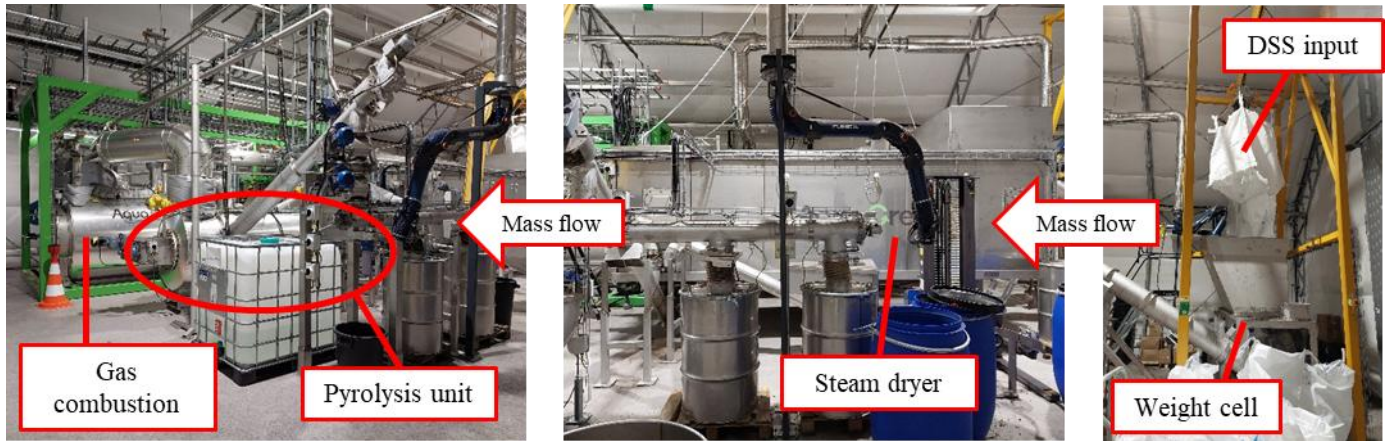


Figure 3.2. The pilot plant (Photos by Helena Lillemets Edman). DSS = Dewatered sewage sludge.

Schematic overview of the pilot plant

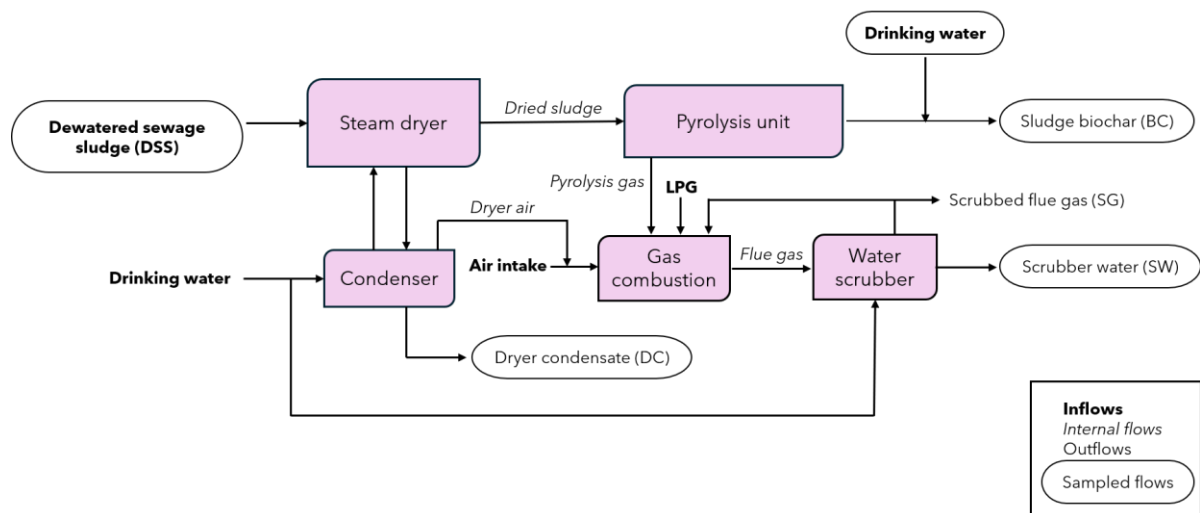


Figure 3.3. Schematic overview of units and mass flows at the pilot plant at Ellinge WWTP.

The dewatered sludge was delivered to Ellinge WWTP in plastic bags (1 m³ each) containing around 700 kg sludge. The dewatered sludge was continuously fed into the steam dryer through screws at a rate of 80 kg/h and was dried at 120-180 °C with a residence time of 120-180 minutes, depending on the moisture content of the sludge. The dryer air was fed to the combustion chamber. The dried sludge was then fed into the pyrolysis oven through another continuous screw where it was pyrolysed at 600-800 °C for a duration of 20 minutes. The set pyrolysis temperature was reached at least during some point of the residence time, but not throughout the entire oven (Figure 3.4).

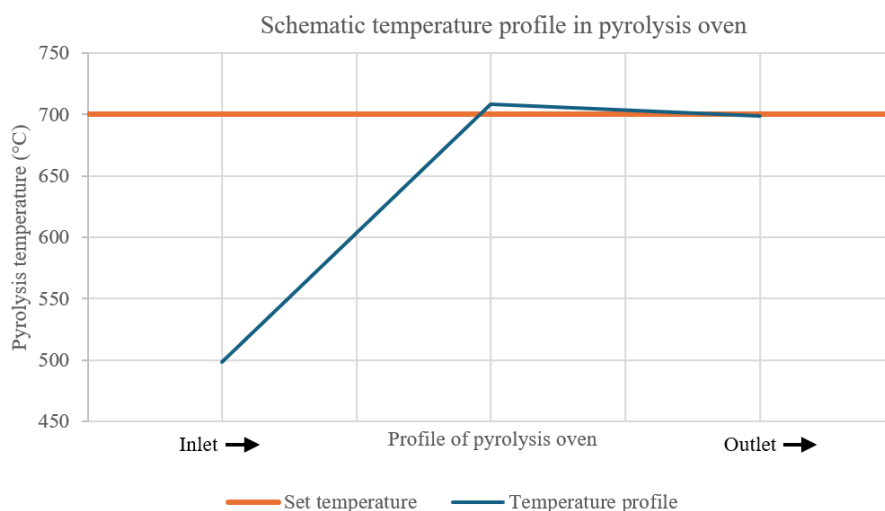


Figure 3.4. Conceptual temperature profile of the pyrolysis oven.

After exiting the pyrolysis oven, the sludge biochar was soaked with around 10 weight-% drinking water to reduce the risk of ignition. The pyrolysis gas was led to a combustion chamber powered by the produced pyrolysis gas, with a pilot flame powered by liquified petroleum gas (LPG). The resulting flue gas was led to a water scrubber for some particle removal before letting the flue gas out through the chimney.

3.3 Biochar Characterisation and Determination of Mass Flows

3.3.1 Analysis Methods

The influent sludge was continuously weighed before entering the steam dryer. After exiting the process, a certain amount of outgoing sludge biochar, produced at a stable temperature profile in the pyrolysis oven, was weighed and this amount was defined as one batch. The amount of sludge that produced the biochar batch could be calculated using residence times and feed velocities from the process. Five samples were taken of each biochar batch as well as of the dryer condensate and the scrubber water produced in each batch respectively (Figure 3.3). Five samples of the drinking water used to soak the biochar was taken once for each sludge.

Unlike the dewatered sludge, only one set of analysis methods was used to examine the sludge biochar, the dryer condensate, the scrubber water and the drinking water (Tables A2-3 in Appendix A). The heavy metal concentrations in the sludge biochar were determined using ICP-MS, as well as most of the heavy metal concentrations in the dryer condensate and scrubber water, except for mercury, which was detected using atomic fluorescence spectroscopy.

3.3.2 Calculations

Retention, i.e. the amount of a substance that was retained in the biochar during the pilot treatment, and mass flows were determined in order to study the variability of the sludge biochar contents, as well as the fate of compounds.

Biochar Yield and Retention

The biochar yield was defined as the amount of dry solids (DS) left in the biochar after the treatment (DS_{BC}) in relation to the amount of DS in the dewatered sludge (DS_{DSS}), expressed in %, as seen in equation (1):

$$\text{Biochar yield} = \frac{DS_{BC}}{DS_{DSS}} \times 100 . \quad (1)$$

The retention of a compound, i , was similarly defined as the amount of the compound that was left in the biochar after the treatment in relation to the amount present in the dewatered sludge, expressed in %, as shown in equation (2):

$$\text{Retention}_i = \frac{\text{Conc}_{i,BC} \times DS_{BC}}{\text{Conc}_{i,DSS} \times DS_{DSS}} \times 100 , \quad (2)$$

where $\text{Conc}_{i,BC}$ and $\text{Conc}_{i,DSS}$ are the concentrations of the compound in the biochar and the dewatered sludge respectively and DS_{BC} and DS_{DSS} are the amounts of DS in the biochar and sludge respectively.

To avoid any potential contamination from the drinking water (that was added to the biochar after the treatment) influencing the results, compounds present in the drinking water were subtracted from the biochar content. It was assumed that all moisture in the sludge biochar was attributed to the added drinking water. The amount of drinking water added to each batch, and consequently the amount of compounds to be subtracted from each batch, could thus be calculated using the measured moisture content in the biochar.

Including Anaerobic Digestion

When accounting for losses of volatile solids (VS) during anaerobic digestion, the total biochar yield was calculated as the amount of DS left in the biochar (DS_{BC}) in relation to the total DS in the raw sludge (pre-anaerobic digestion) using equation (3):

$$\text{Biochar yield}_{\text{total}} = \frac{DS_{BC}}{DS_{DSS} + VS_{\text{lost}}} \times 100 , \quad (3)$$

where the DS in the dewatered sludge (DS_{DSS}) and the VS lost during digestion (VS_{lost}) constituted the total DS in the raw sludge.

Estimations of the VS lost during anaerobic digestion, VS_{lost} , were made using values of volatile solids reduction (VSR) from Table 3.1 according to equation (4):

$$VS_{\text{lost}} = \text{Total VS} - VS_{\text{left}} = \frac{VS_{\text{left}}}{100 - \text{VSR}} \times \text{VSR} , \quad (4)$$

where VS_{left} is the amount of VS left in the dewatered sludge.

The total retention of carbon in the biochar, when including losses during anaerobic digestion, was estimated using equation (5):

$$\text{Retention}_{\text{total C}} = \frac{C_{BC}}{C_{DSS} + (\text{Conc}_{C,VS} \times VS_{\text{lost}})} \times 100 , \quad (5)$$

where the carbon mass in the biochar (C_{BC}) was divided by the estimated carbon mass in the raw sewage sludge (pre-anaerobic digestion), comprised of the carbon in the DSS (C_{DSS}) and the estimated amount of carbon in the VS lost during digestion. The carbon concentration of the VS ($\text{Conc}_{C,VS}$) was assumed to be 55%, based on an average carbon content of 56%±2% in the VS of the fifteen examined sludges.

Mass Balances

For the mass balances calculations, measurements were made of the dewatered sewage sludge (DSS), sludge biochar (BC), dryer condensate (DC) and scrubber water (SW). The mass in the scrubbed flue gas (SG) was calculated as the remaining mass after subtracting the other outgoing masses from the dewatered sludge, as shown in equation (6):

$$SG = DSS - (BC + DC + SW) . \quad (6)$$

The mass flows were presented in relation to the mass in the dewatered sludge, which was considered as 100% of the mass in the system. Because the scrubbed flue gas was calculated, and the outflows were assumed to be equal to the inflow, the total outflows sometimes appear to be exactly 100% and can sometimes appear to be negative, when the sum of the outputs are larger than the mass of the dewatered sludge. A schematic overview of the mass flows is illustrated in Figure 3.5.

Schematic overview of mass flows

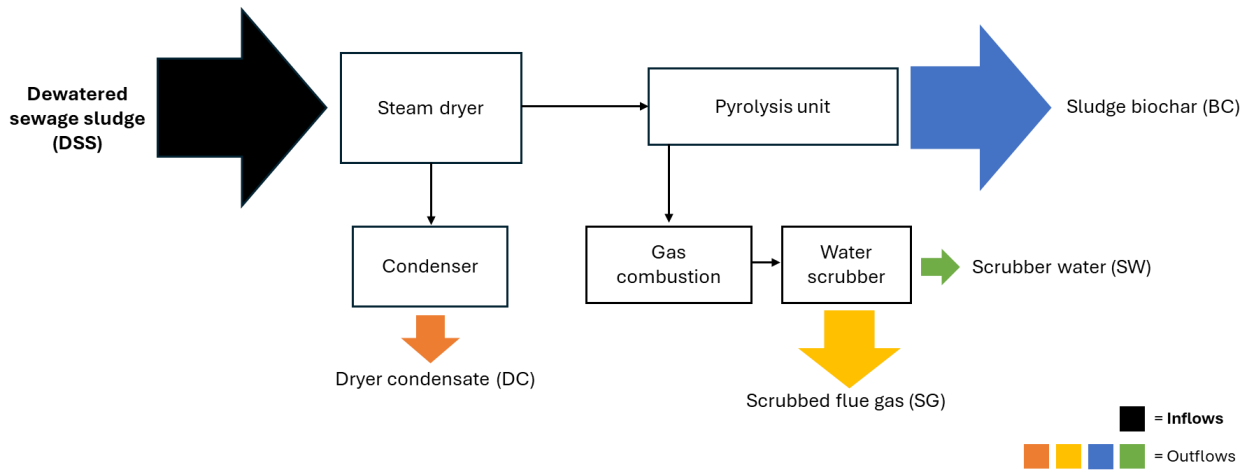


Figure 3.5. Schematic overview of mass flows.

When including biogas produced from anaerobic digestion as a mass flow, the following equation was used to calculate the amount of carbon in the biogas, C_{biogas} :

$$C_{\text{biogas}} = VS_{\text{lost}} \times \text{Conc}_{C,VS} , \quad (7)$$

where VS_{lost} is the estimated amount of VS lost during digestion from equation (4) and $\text{Conc}_{C,VS}$ is the carbon content of the lost VS, which was estimated to 55%. Figure 3.6 shows a schematic overview of the mass flows including biogas produced from anaerobic digestion.

Schematic overview of mass flows: Including anaerobic digestion

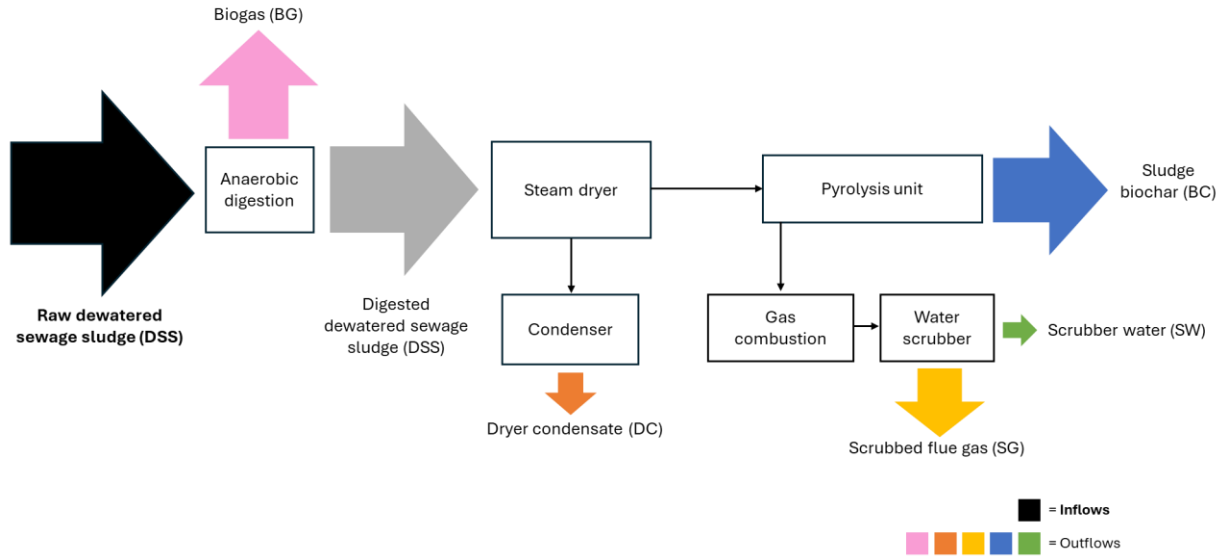


Figure 3.6. Schematic overview of mass flows, including anaerobic digestion.

Mass balances were only made for sludges where samples of DC and SW had been taken. In the mass balance of carbon, the TOC in the dryer condensate and scrubber water were assumed to represent total carbon. No mass balance was made for potassium as potassium was not measured in the dryer condensate or scrubber water.

Uncertainties in Determining the Dry Solids in the Sludge

As mentioned above, the sludge biochar batches were weighed after exiting the process and the weight of the sludge that had produced a certain biochar batch was calculated based on residence times and feeding rates. As the process was in pilot scale, there could, however, be risks of uneven flows due to build ups in the sludge transport of the system, leading to inaccurate approximations of the actual weight of the influent sludge of a batch. An alternative way of determining the DS in the sludge is to calculate it using the sludge ash content and an assumed sludge ash mass, as in equation (8).

$$\text{Dry solids (DS) in sludge (kg)} = \frac{\text{Ash in sludge batch (kg)}}{\text{Sludge ash content (\% of DS)}} \times 100 \quad (8)$$

This can be done assuming the influent ash is only negligibly affected by the process and consequently that the mass of ash in the sludge is equivalent to the mass of ash in the biochar, which is known.

Assumption: Ash in sludge batch (kg) = Ash in biochar batch(kg)

The method with which the ash content was determined in this study was by combustion at 550 °C, which is lower than the pyrolysis temperatures used in the process. It could therefore be possible that additional compounds would evaporate in the pyrolysis and thereby cause a reduction in the ash content of the biochar, compared with the ash content of the sludge. As was mentioned in the background, studies have shown that cadmium typically evaporates at temperatures above 600 °C, and is an example of a compound that could impact the ash content depending on the temperature. Moreover, it has been seen that mass loss of biomass ash can occur during pyrolysis due to the decarbonation of carbonates, especially at temperatures of 600-900°C (Vassilev *et al.*, 2022). Most ash compounds,

however, are not affected by temperatures up to 900°C and it could therefore be reasonable to assume that these potential losses are negligible and that the assumption above is valid.

A way of estimating the accuracy of these methods in determining the DS in the sludge was to calculate the retention of heavy metals with high boiling points (> 500 °C), such as chromium, copper, iron and nickel, using both methods. These metals are likely predominantly unaffected by the pyrolysis temperature and their masses would therefore be expected to remain relatively constant throughout the process. In other words, if close to 100% of these metals was retained in the biochar it could be interpreted as an indication that the method used was reliable. As can be seen in Figure 3.7, on average around 100% of the copper was, as expected, retained in the biochar for both methods.

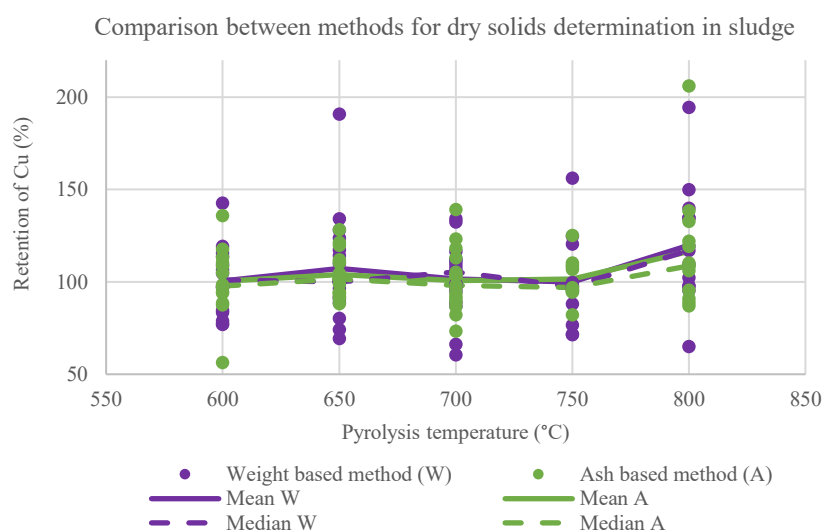


Figure 3.7. Retention of copper in the biochar, calculated using two methods for determining the dry solids (DS) in the dewatered sludge.

The retention of the remaining metals was higher than 100% (Table 3.2), and the implications thereof are discussed further in the Results and Discussion.

Table 3.2. Statistical analysis of two methods for determining the dry solids in the dewatered sludge.

Metal	Method	Mean retention rate (%)	Standard deviation	Variance
Cr	Weight based	130	46	2 100
	Ash based	130	43	1 900
Cu	Weight based	100	26	680
	Ash based	100	25	600
Fe	Weight based	120	36	1 300
	Ash based	120	27	750
Ni	Weight based	130	41	1 700
	Ash based	130	31	950

No significant differences were detected between the methods for any of the metals. This indicates that deviations from a retention of 100% is not dependent on the method for determining the DS content in the sludge, but is rather the result of other factors. It was therefore concluded that the methods were of equivalent accuracy. Considering this, the

only difference between the methods was that the ash based method consistently had a lower variance than the weight based method (Table 3.2) and the ash based method in equation (8) was therefore used in the calculations in this study.

3.3.3 Excluded Data

Some of the early batches of the study were excluded from the analyses due to operating difficulties, such as clogging and leakages. Batches lacking data on sludge or biochar properties due to process failures were also excluded. The remaining batches are shown in Table 3.3, along with the number of replicates and analysed mass flows for each sludge.

Table 3.3. Pyrolysis temperatures analysed for each sludge, where number of batches are shown and excluded batches due to process failures are highlighted in red. DSS = Dewatered sewage sludge, BC = Sludge biochar, DC = Dryer condensate, SW = Scrubber water.

WWTP	Pyrolysis temperature					Analysed mass flows	
	600 °C	650 °C	700 °C	750 °C	800 °C	DSS and BC	DC and SW
Lidköping	0	2	2	1	0	Yes	No
Lycksele	1	1	1	0	1	Yes	Yes
Margretelund	1	2	2	1	1	Yes	Yes
Ormanäs	1	1	1	1	1	Yes	Yes
Duvbacken	1	1	1	1	1	Yes	Yes
Ellinge	1	1	1	0	1	Yes	No
Henriksdal	1	1	1	1	1	Yes	Yes
Himmerfjärdsverket	1	0	1	1	1	Yes	Yes
Hässleholm	1	1	1	1	1	Yes	Yes
Klagshamn	2	1	2	1	1	Yes	No
Kungsängsverket	1	1	1	1	0	Yes	Yes
Käppalaverket	1	1	2	1	1	Yes	Yes
Skebäckverket	1	1	1	0	1	Yes	Yes
Uddebo	1	1	0	0	1	Yes	Yes
Öresundsverket	1	1	1	1	1	Yes	Yes

3.4 Statistical Analysis

In order to determine significant differences between sets of batches, for example regarding sludge origin or pyrolysis temperature, one-way analyses of variance (ANOVA) were performed with a significance level of 0.05.

4 Results and Discussion

The following chapter presents the contents of both sludges and biochars analysed in the study and discusses causes and implications thereof. Please see the schematic overviews of the mass flows in Figures 3.5-6 when reading the mass flow charts.

4.1 General Biochar Properties

A comprehensive table of the results regarding general sludge and biochar properties is presented in Table B1 in Appendix B.

4.1.1 Biochar Yield and Carbon

The biochar yield was significantly higher ($p < 0.05$) for anaerobically digested sludges than for undigested (Figure 4.1).

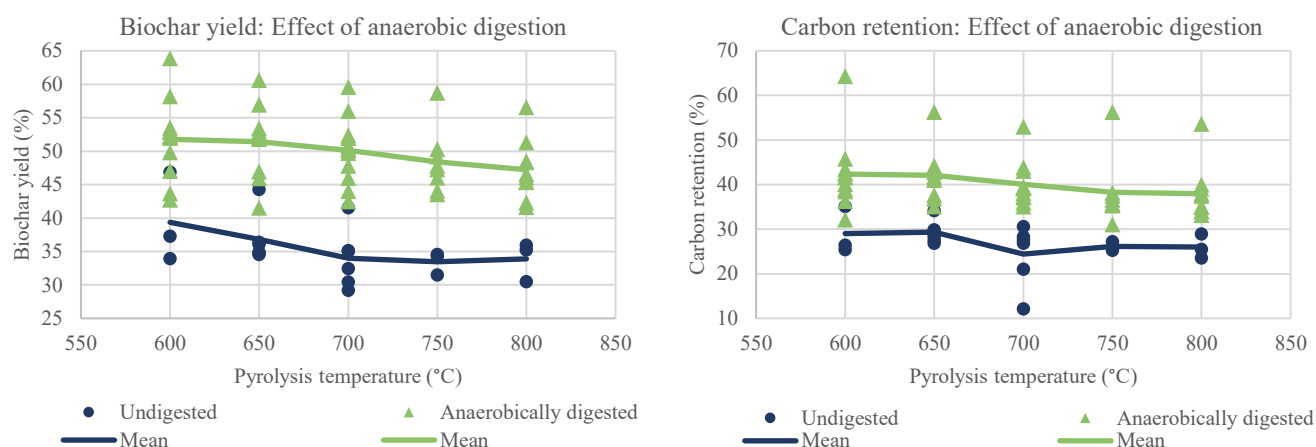


Figure 4.1. Biochar yield and carbon retention.

The anaerobically digested sludges produced an average biochar yield of about 50%, compared to an average of about 35% in the undigested sludges. Similar biochar yields have been observed in the pyrolysis of digested (Hossain *et al.*, 2011; Vali *et al.*, 2023) and undigested (Vali *et al.*, 2023) sewage sludges. The same trend can be seen in the retention of carbon, which were significantly higher for digested sludges than for undigested (Figure 4.1). This is likely due to a higher content of easily degradable carbon compounds in undigested sludges compared to digested sludges, which have already lost a substantial part of these compounds as methane and carbon dioxide during digestion. When these compounds are lost during pyrolysis instead, this leads to a greater relative mass loss in the undigested sludges compared to the digested, resulting in a lower biochar yield. This is supported by the lower carbon content in the anaerobically digested sludges (Figure 4.2).

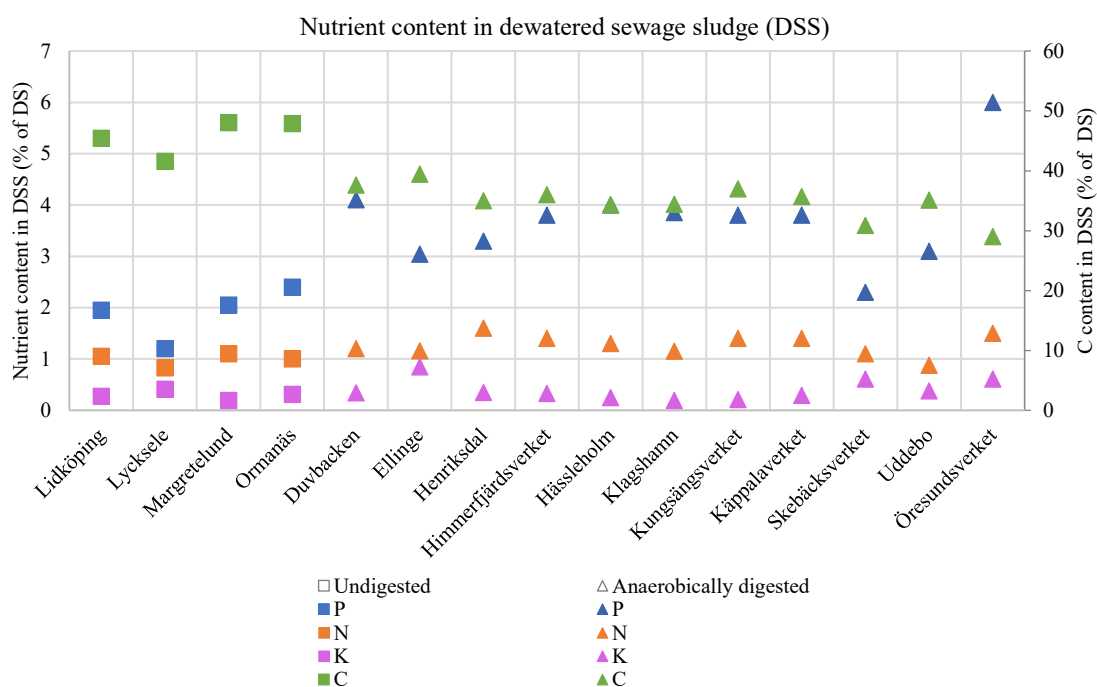


Figure 4.2. Mean plant nutrient contents in the dewatered sewage sludge (DSS), distinguishing between anaerobically digested and undigested sludges. Statistical information in Table B2 in Appendix B.

Furthermore, the biochar yield and carbon retention decrease somewhat with increasing temperature, which can be explained by further degradation of mainly organic matter at higher pyrolysis temperatures (Hossain *et al.*, 2011).

Despite a higher biochar yield and carbon retention in the anaerobically digested sludges, their carbon content was significantly lower compared to the undigested ($p < 0.05$) (Figure 4.3).

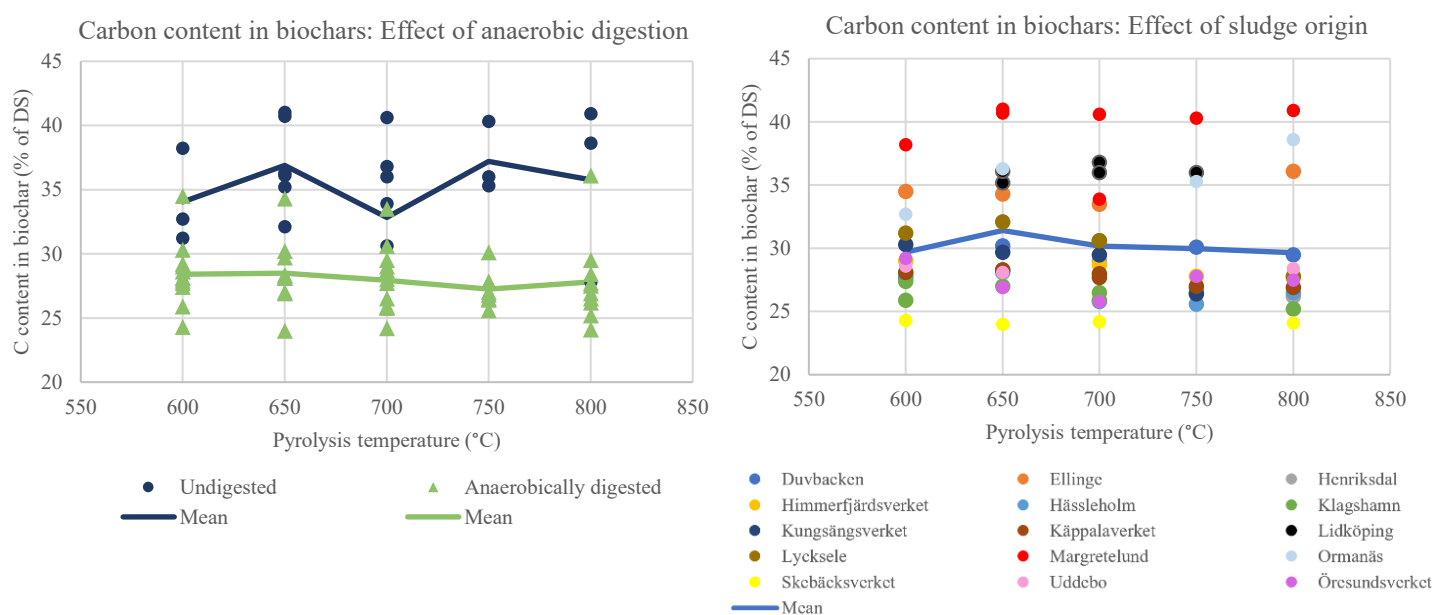


Figure 4.3. Carbon content in biochars, distinguishing between biochars from anaerobically digested and undigested sludges, as well as between biochars originating from different WWTPs.

This suggests that despite a higher loss rate, the excess content of carbon compounds in the undigested sludges still contributes to a higher final carbon concentration in the biochar.

Regarding the carbon that was lost during the process, the vast majority entered the scrubbed flue gas (yellow in Figure 4.4).

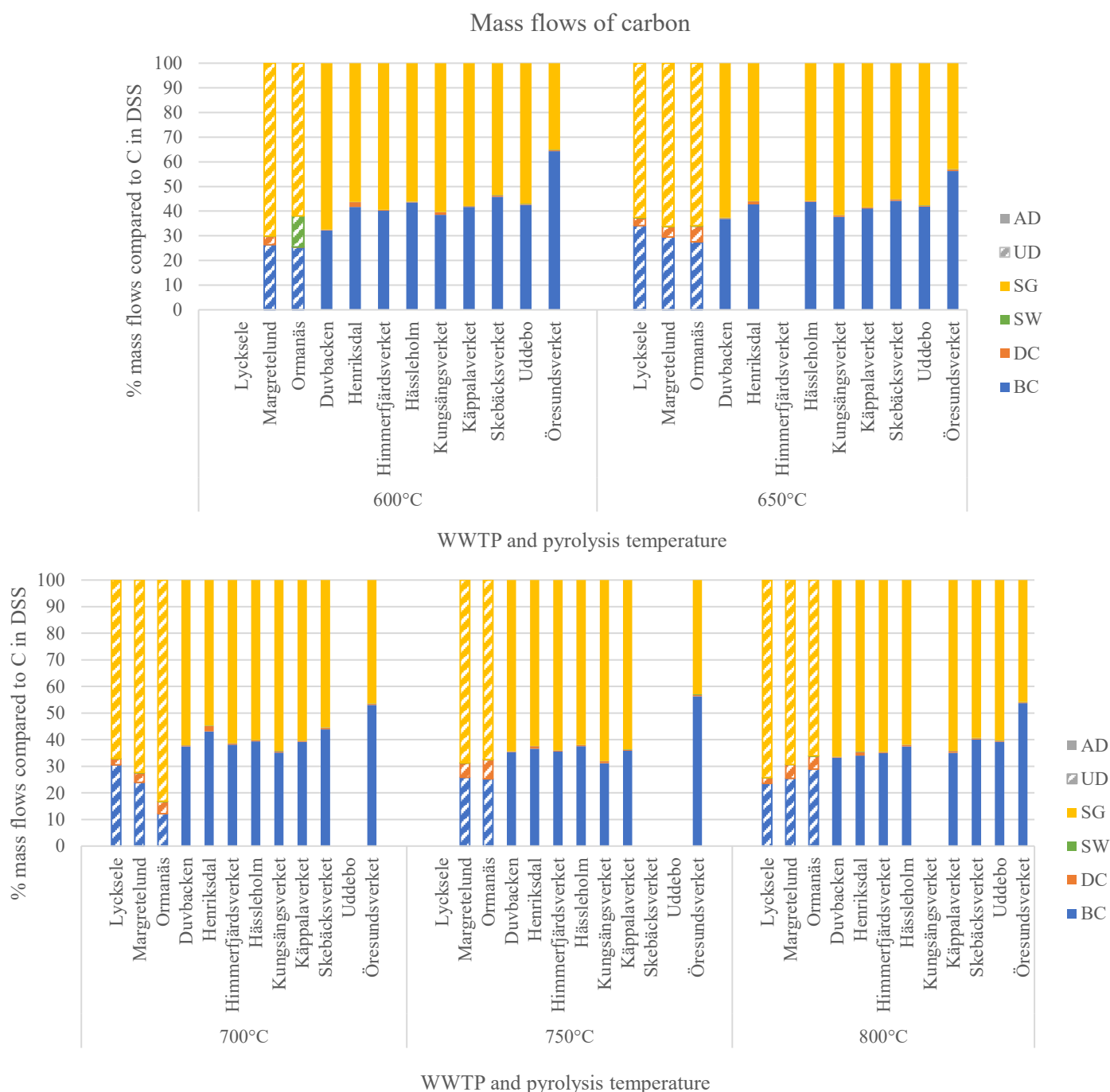


Figure 4.4. Mass flows of carbon, presented in relation to the contents of the dewatered sewage sludge (DSS), representing 100%. AD = Anaerobically digested, UD = Undigested, SG = Scrubber flue gas, SW = Scrubber water, DC = Dryer condensate, BC = Sludge biochar.

This could potentially indicate that it evaporated during pyrolysis rather than during the drying but it should be noted that the gas from the steam dryer was fed to the combustion chamber, making it difficult to draw conclusions about the origin of compounds in the scrubber water. A small fraction of the carbon from the undigested sludges could be found in the dryer condensate. This is likely easily degradable carbon compounds, more likely to evaporate at lower temperatures. As these compounds are likely degraded during digestion, it can explain why this mass flow was only detected in the undigested sludges.

It should be noted that the reason the outflows correspond to exactly 100% of the inflow in the mass balances above is that the contents of the scrubbed flue gas were calculated assuming equal inflows and outflows.

Including Biogas from Anaerobic Digestion as a Mass Flow

As carbon was converted into biogas during anaerobic digestion, it can be useful to analyse the implications of this mass flow on biochar and carbon balances. Figure 4.5 shows a conceptual illustration of the biochar yields from anaerobically digested and undigested sludges, where the biochar yield of digested sludges is lower than that of undigested when considering total losses of VS.

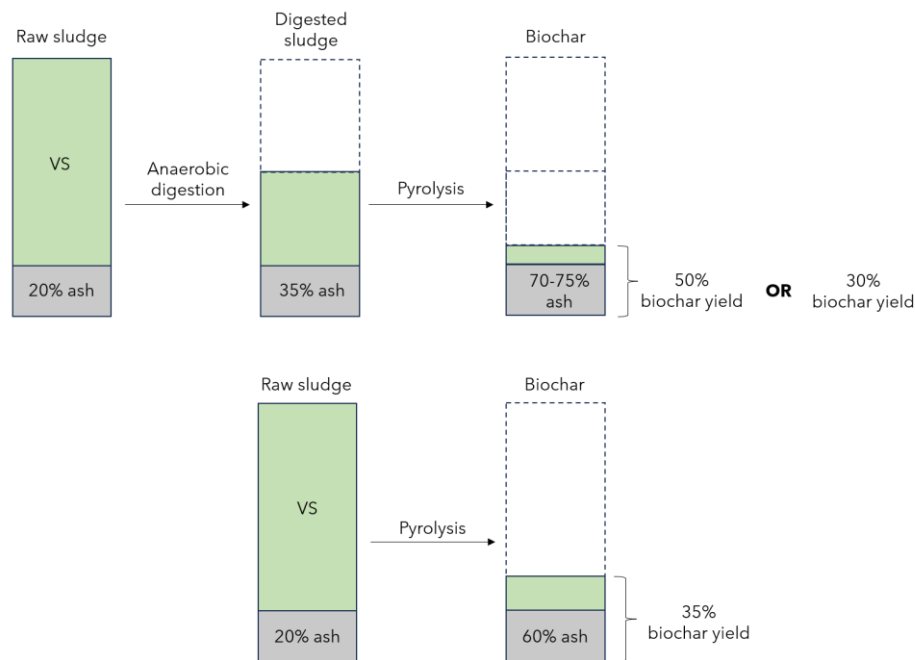


Figure 4.5. Conceptual illustration of biochar yields from anaerobically digested and undigested sludges.

An estimation of the percentage carbon loss by anaerobic digestion is presented in Figure 4.6, where total mass flows of carbon have been calculated from raw sludge (pre-anaerobic digestion), based on volatile solids reduction and an assumed biogas carbon content of 55%, according to equation (7).

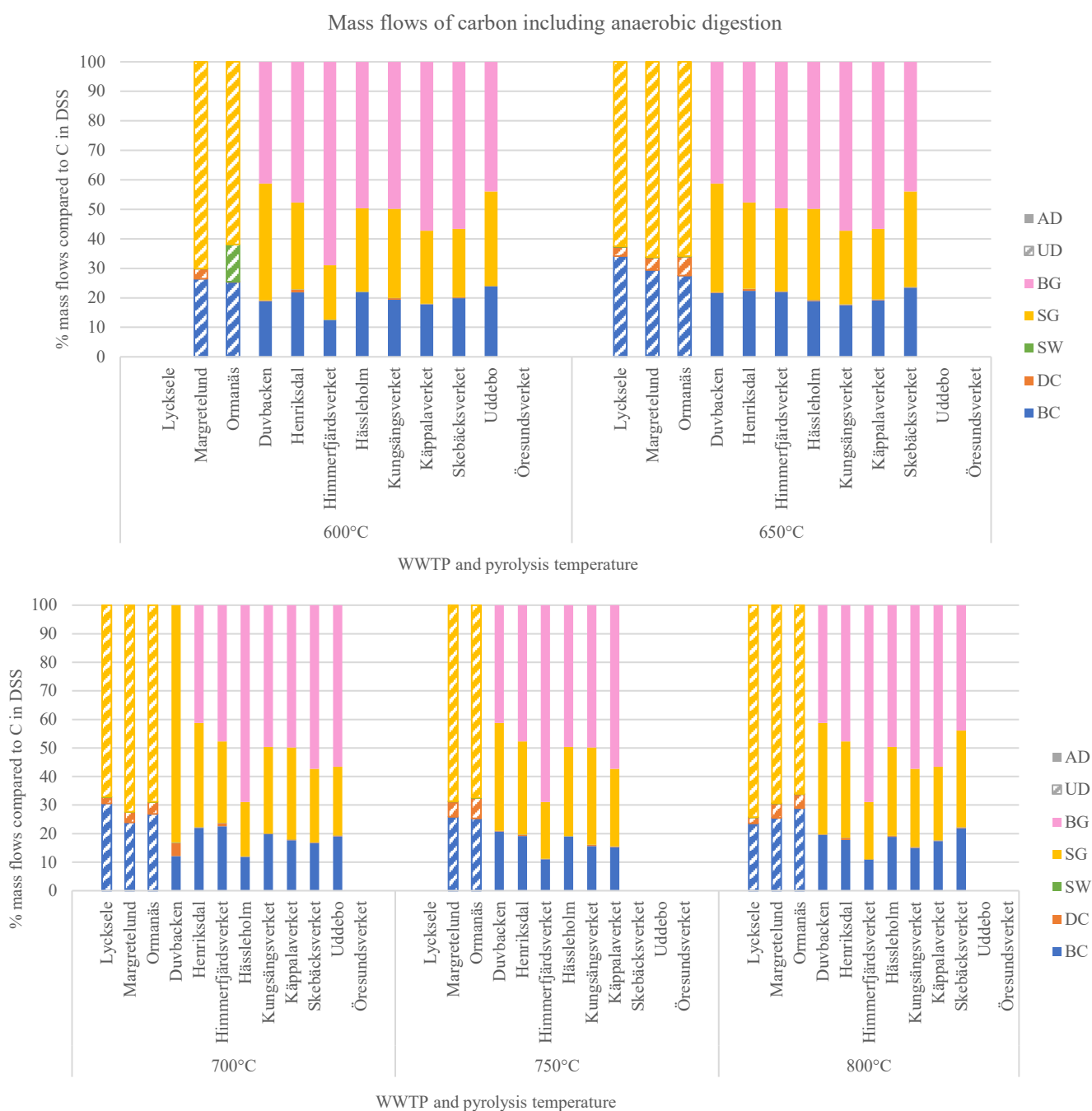


Figure 4.6. Mass flows of carbon, including biogas, presented in relation to the contents of the dewatered sewage sludge (DSS), representing 100%. AD = Anaerobically digested, UD = Undigested, BG = Biogas, SG = Scrubber flue gas, SW = Scrubber water, DC = Dryer condensate, BC = Sludge biochar.

In the anaerobically digested sludges, approximately 40-70% of the initial sludge carbon content was converted into biogas during digestion (pink in Figure 4.6). When including this mass flow, the total biochar yields and carbon retention, counted from raw sludge, were significantly lower in biochars produced from anaerobically digested sludges than in

biochars produced from undigested sludges (Figure 4.7), due to carbon losses during digestion.

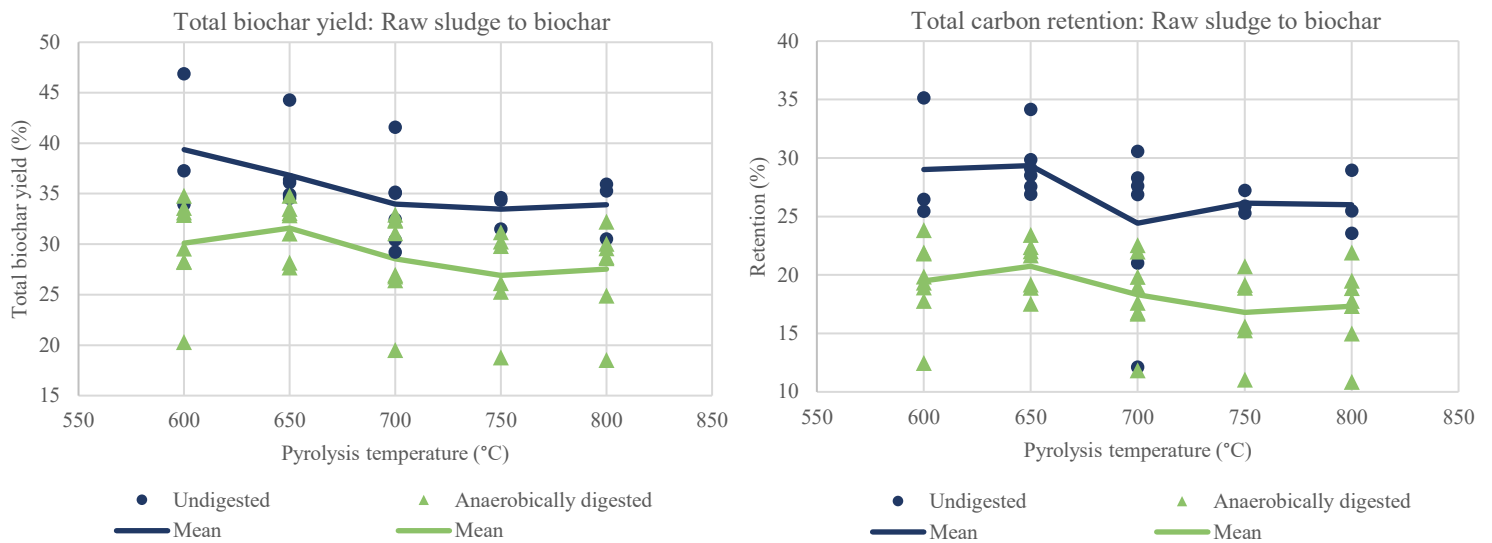


Figure 4.7. Total biochar yield and carbon retention, when analysing raw sludges.

This suggests that the pyrolysis of undigested sludge can provide a higher degree of carbon sequestration (higher carbon retention) and a higher percentage production of pyrolysis gas compared to anaerobically digested sludge, which, on the other hand, allows for biogas production. As pyrolysis gas can be recirculated back into the system as fuel (Hossain, Strezov & Nelson, 2009), undigested sludges may therefore be more likely to produce sufficient amounts of pyrolysis gas to be able to sustain the pyrolysis process. More extensive energy analyses comparing the energy efficiency of pyrolysis of anaerobically digested and undigested sludges, considering pyrolysis gas as a potential fuel and the energy consumption of drying, are needed to make further conclusions about the energy potential of possible sludge treatment combinations.

4.1.2 Ash content

The biochar ash content increased significantly with higher pyrolysis temperatures ($p < 0.05$) and biochars from anaerobically digested sludges had significantly higher ash contents than biochars from undigested sludges (on average 70-77% and 57-66% respectively), as seen in Figure 4.8. This is likely an effect of the lower carbon content in the biochars from digested sludges (Figure 4.3).

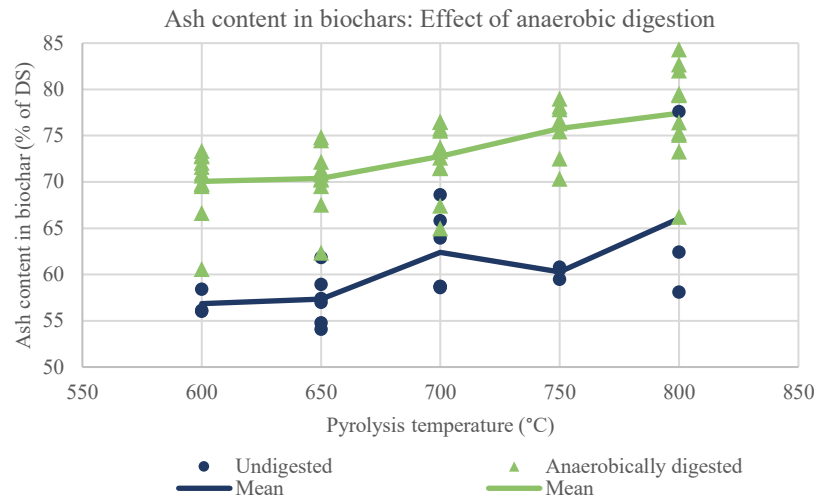


Figure 4.8. Ash content in biochars, distinguishing between biochars from anaerobically digested and undigested sludges.

4.1.3 pH

The anaerobically digested sludges had significantly higher pH values than the undigested ($p < 0.05$), with an average of 8.1 and 6.8 respectively (Figure 4.9).

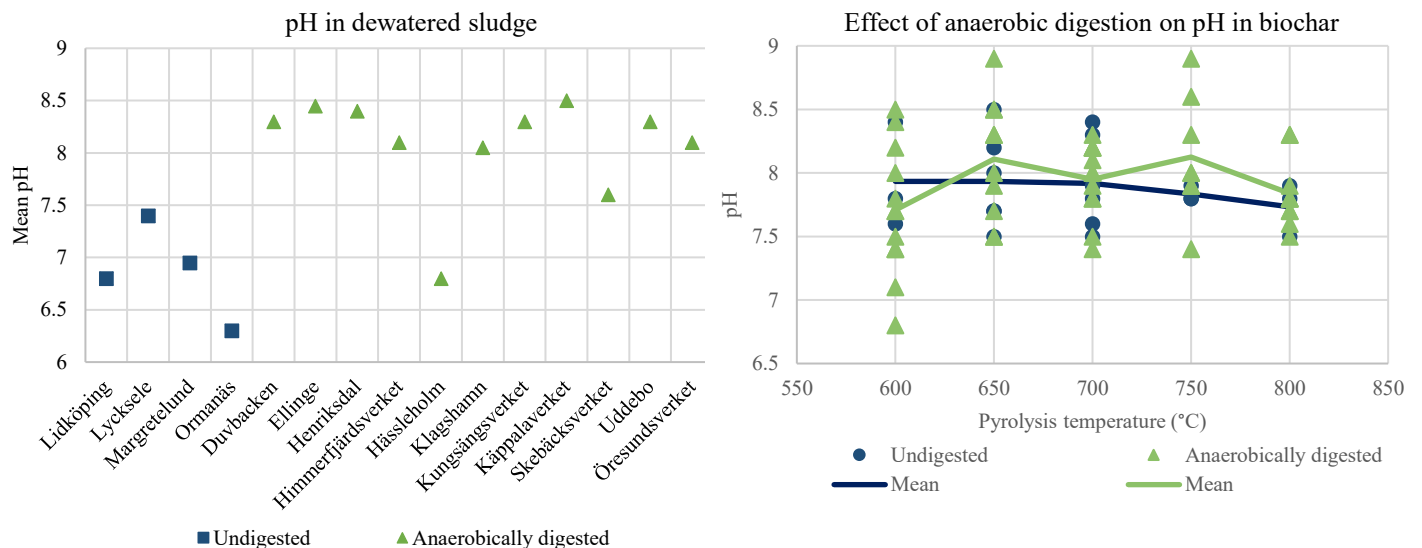


Figure 4.9. Mean pH values in dewatered sludge (left) and pH in biochars (right), distinguishing between anaerobically digested and undigested sludges and biochars. Standard deviations are presented in Table B1 in Appendix B.

Biochars from both digested and undigested sludges stabilised around a pH of 7.9, meaning that the pH of the undigested sludges increased significantly during pyrolysis while the pH of the digested sludges decreased slightly. Previous studies have found that higher pyrolysis temperatures (>500 °C) can produce alkaline biochars (Hossain *et al.*, 2011; Yuan *et al.*, 2015), partly due to the loss of organic acids (Yuan *et al.*, 2015).

4.2 Nutrients

A comprehensive table of the results regarding nutrients is presented in Table B2 in Appendix B.

4.2.1 Phosphorus

On average 100% of the phosphorus in the sludges was retained in the biochars, regardless of pyrolysis temperature and anaerobic digestion of the sludge (Figure 4.10).

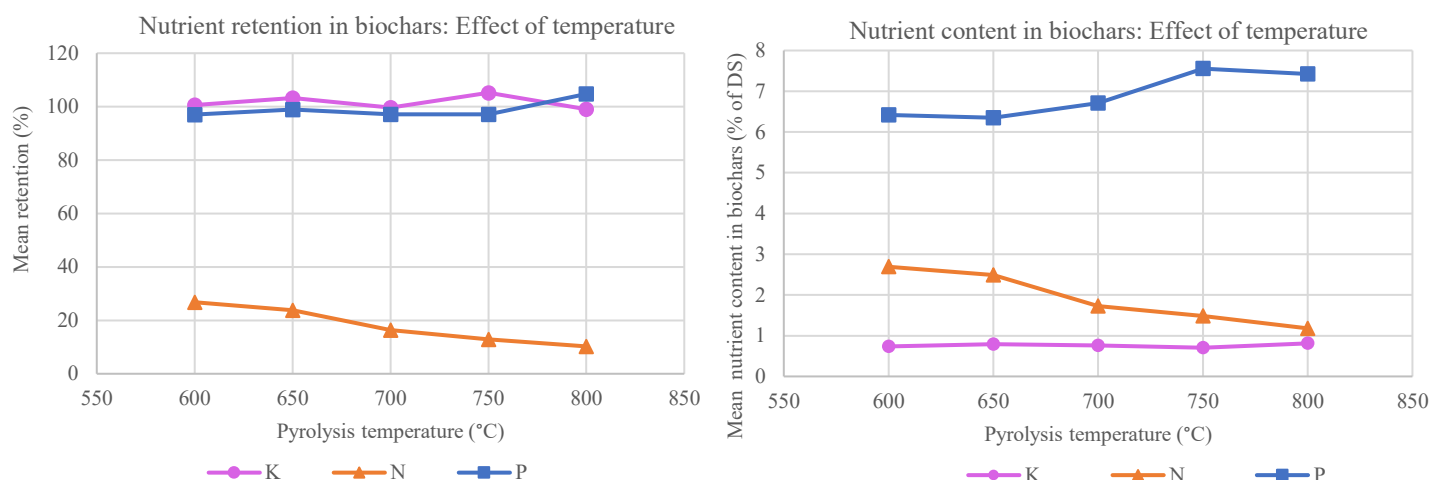


Figure 4.10. Mean nutrient retention (left) and contents (right) in the biochar. Statistical information in Table B5 in Appendix B.

Previous studies have also found that phosphorus is preserved during pyrolysis (Adhikari *et al.*, 2019; Zhu *et al.*, 2022; Vali *et al.*, 2023), indicating that biochar produced from pyrolysed sewage sludge has potential as a phosphorus fertilizer. The preservation of phosphorus in combination with a decreasing biochar yield with increasing pyrolysis temperature led to higher phosphorus concentrations in the biochars compared with the sludges (Figure 4.2), from around 2-4% in the sludges to 6-8% in the biochars. A slight increase in phosphorus concentration was detected with increasing pyrolysis temperatures, especially at 750-800 °C. Phosphorus was not detected in the dryer condensate or scrubber water, suggesting that any removed phosphorus was released in the scrubber flue gas (Appendix C).

There was a high variability in the phosphorus content of sludges from different WWTPs leading to a correspondingly high variability in the phosphorus concentrations in the biochars, with concentrations ranging between 30-100 g/kg DS (Figure 4.11).

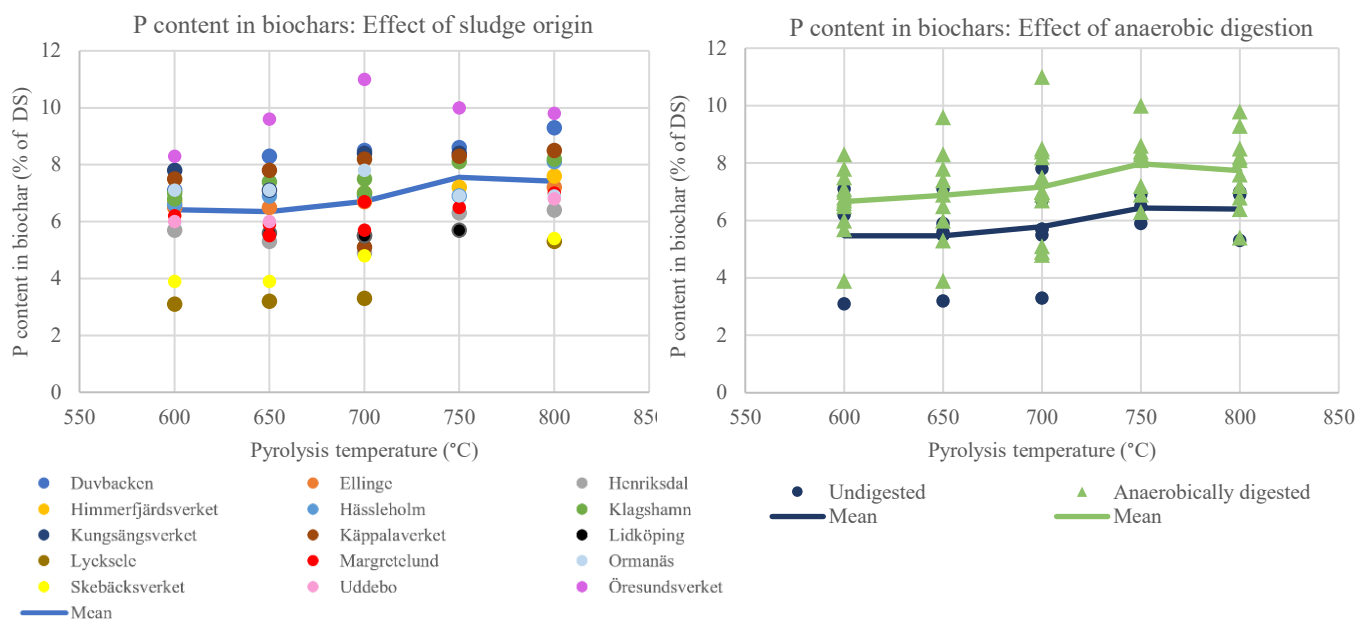


Figure 4.11. Phosphorus concentrations in biochars produced at different pyrolysis temperatures distinguishing between different WWTPs (left) and anaerobic digestion (right).

The sludge from Lycksele WWTP had an especially low phosphorus content of 1.2 % of the DS (Figure 4.2), leading to a correspondingly low phosphorus content in the biochar. Meanwhile, the sludge from Öresundsverket had a remarkably high phosphorus content of about 60 000 mg/kg DS, leading to a high biochar content. The cause of the elevated phosphorus content is uncertain, but it could be attributed to a heterogeneity in the sludge, causing an unrepresentative sample. Another parameter affecting the phosphorus concentration in the sludges and biochars was anaerobic digestion, as anaerobically digested sludges and biochars had significantly higher phosphorus concentrations than undigested (Figures 4.2 and 4.11). This is likely due to the higher carbon content in the undigested sludges and biochars, diluting other compounds to a higher extent.

4.2.2 Nitrogen

Nitrogen was increasingly volatilised with increasing pyrolysis temperature, independently of sludge origin or treatment methods (Figure 4.10). The average retention was 20-30% at 600 °C and decreased to about 10% at 800 °C. Previous studies have also observed increasing nitrogen volatilisation at temperatures higher than 300 °C (Hossain *et al.*, 2011; Yuan *et al.*, 2013), which can be explained by the decomposition of organic nitrogen and the loss of volatile nitrogen compounds (Bagreev, Bandosz & Locke, 2001; Wei *et al.*, 2015). In connection with this, a decline in nitrogen concentration in the biochars was observed independently of sludge origin or sludge treatment (Figure 4.10). The nitrogen concentrations decreased from around 2.5-3% of DS at 600 °C to slightly over 1% of DS at 800 °C, and this is likely a consequence of the nitrogen compounds being more volatile than the remaining carbon compounds, leading to a higher loss rate of nitrogen than carbon with rising temperatures.

Unlike most other elements, a substantial fraction of the nitrogen in the sludge was found in the dryer condensate, indicating that some nitrogen was volatilized during the drying (Figure 4.12). Most of the nitrogen that was not retained in the biochar, however, ended up in the scrubber flue gas, suggesting that most of the nitrogen was volatilized during pyrolysis.

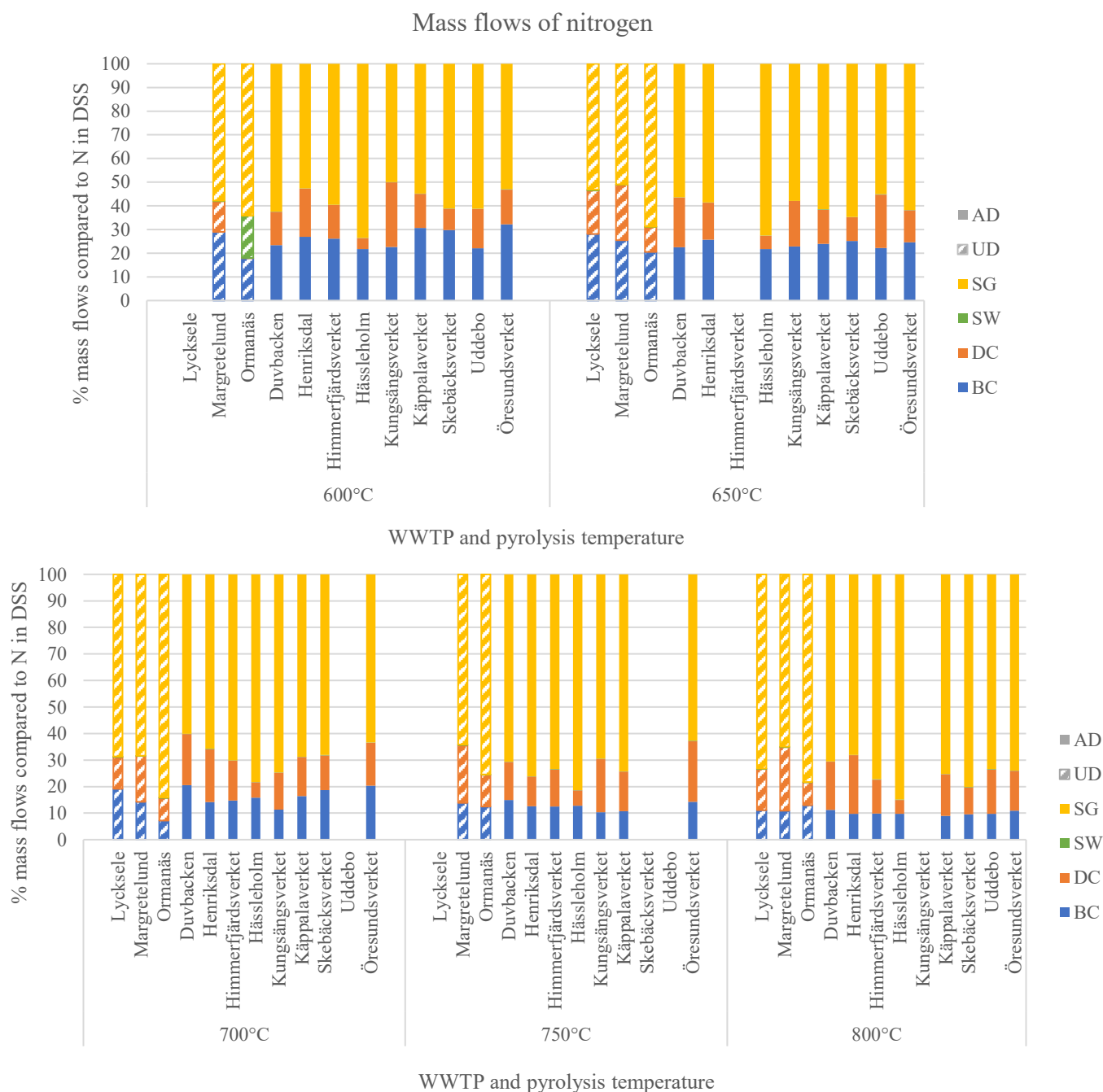


Figure 4.12. Mass flows of nitrogen, presented in relation to the contents of the dewatered sewage sludge (DSS), representing 100%. AD = Anaerobic digestion, UD = Undigested, SG = Scrubber flue gas, SW = Scrubber water, DC = Dryer condensate, BC = Sludge biochar.

4.2.3 Potassium

On average, around 100% of the potassium was retained in the biochars independently of pyrolysis temperature, sludge origin or sludge treatment (Figure 4.10). This suggests that the potassium in the biochar was bound to mineral compounds, prohibiting it from evaporating even when reaching its boiling point of around 760 °C (Hossain *et al.*, 2011).

Potassium concentrations in the biochars were significantly affected by potassium concentrations in the sludges, giving distinguishable differences between biochars from different WWTPs (Figure 4.13).

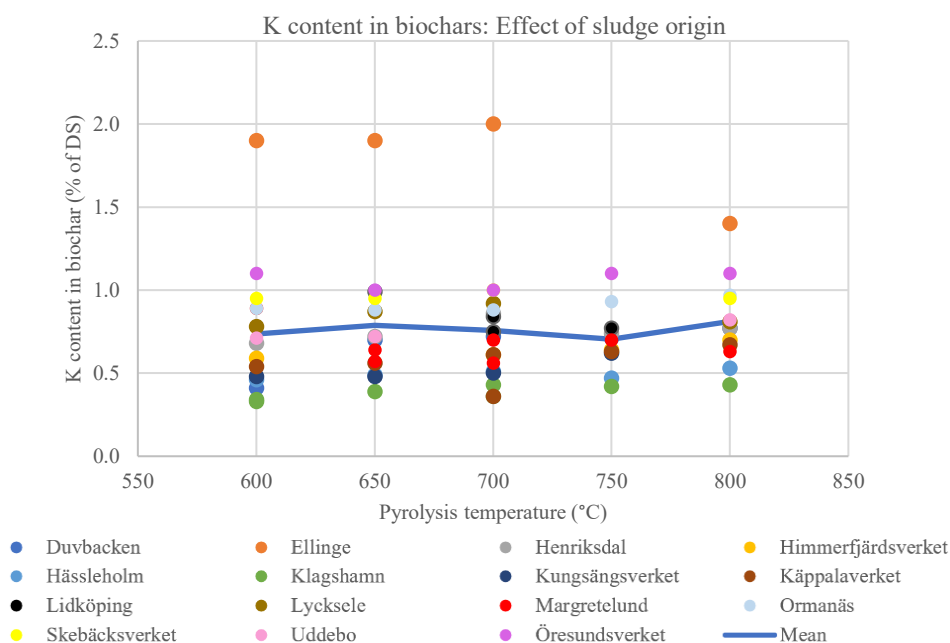


Figure 4.13. Potassium content in biochars from different WWTPs.

Notably, while biochars from most WWTPs had potassium concentrations ranging from 0.5-1 % of DS, the biochars from Ellinge WWTP reached potassium concentrations of 2 % of DS, significantly higher than the mean ($p < 0.05$). This is likely due to the high potassium content in the sludge from Ellinge WWTP, compared to sludges from other WWTPs (Figure 4.2), possibly as a result of the high fraction of industrial wastewater (72% of BOD) in the wastewater load (Table 3.1), mainly originating from food industry. No significant effect of treatment methods on potassium content was detected.

4.3 Heavy Metals

A comprehensive table of the results regarding the heavy metals is presented in Table B3 in Appendix B.

4.3.1 Cadmium and Mercury

Limit Values

Higher pyrolysis temperatures led to a higher volatilisation of cadmium, with an average retention in the biochars of around 50% at 600 °C to an average of about 6% at 800 °C (Table B3 in Appendix B). Increased cadmium volatilisation with increasing temperature has been observed in several other studies as well (Kistler & Widmer, 1987; Méndez *et al.*, 2013; Yuan *et al.*, 2015; Jin *et al.*, 2016). As a result, both the cadmium concentrations and

the Cd/P ratios in the biochars decreased with higher pyrolysis temperatures, from an average of around 0.8 mg/kg DS and 13 mg/kg P respectively at 600 °C to around 0.08 mg/kg DS and 1 mg/kg P respectively at 800 °C (Figure 4.14).

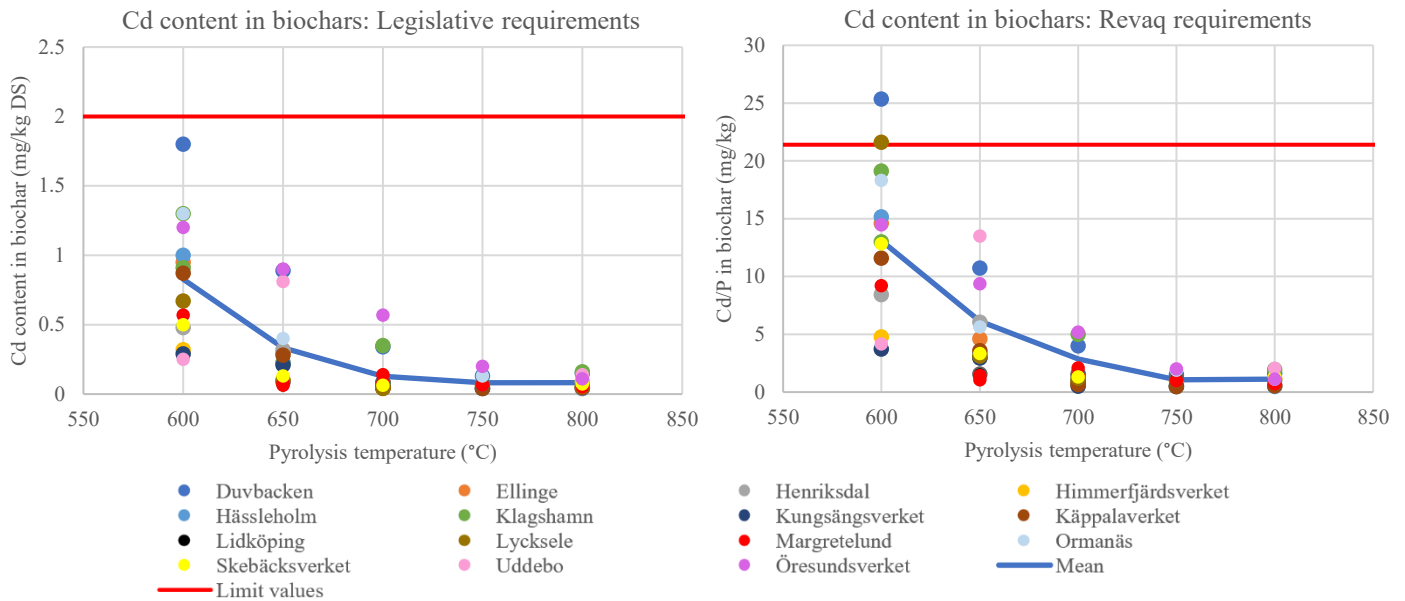


Figure 4.14. Cadmium content in biochars, compared to legislative requirements (left) and requirements by Revaq (right).

Consequently, all biochars had cadmium concentrations below the legislative limit of 2 mg/kg DS (SFS 1998:944) and below the Revaq limit of 21 mg/kg P at pyrolysis temperatures of 650 °C and higher. Considering that only 53% of the sludges had sufficiently low Cd/P ratios to be permitted to be used as fertilizers by Revaq (Figure 4.15), the biochars were more suited as fertilisers than the sludges when only considering cadmium content.

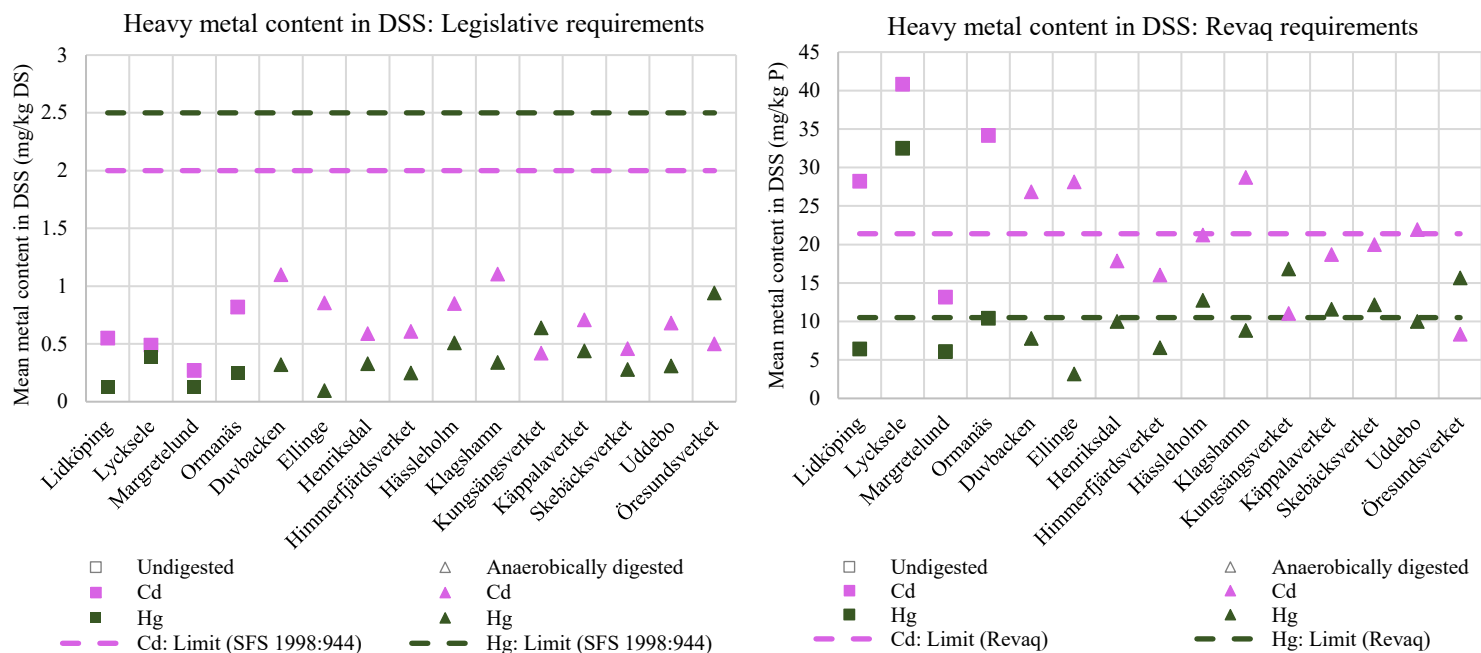


Figure 4.15. Mean heavy metal contents in dewatered sewage sludge (DSS) as concentrations (left) and metal/P ratios (right), including limit values (SFS 1998:944 and Revaq's long-term goal) and distinguishing between anaerobically digested and undigested sludges. Statistical information in Tables B3-4 in Appendix B.

No clear effects of treatment methods on cadmium concentrations in the biochar were found, but sludges with high cadmium concentrations generally produced biochars with high concentrations.

No detectable levels of mercury were found in the biochars (<0.080 mg/kg DS), regardless of pyrolysis temperature, indicating that even the lowest temperature of 600°C was sufficient in removing trace amounts of mercury in the biochars (Appendix B). This means that all biochars would be allowed to be used as fertilizers both by legislative standards (2.5 mg/kg DS) and Revaq's long-term goal (10.5 mg Hg/kg P), when only considering mercury. This is, similarly to cadmium, an improvement compared to the sludges, of which only 47% reached Revaq's long-term goal (Figure 4.15).

Mass Flows

When analysing mass flows, the majority of the volatilised cadmium ultimately ended up in the flue gas (yellow and green in Figure 4.16).

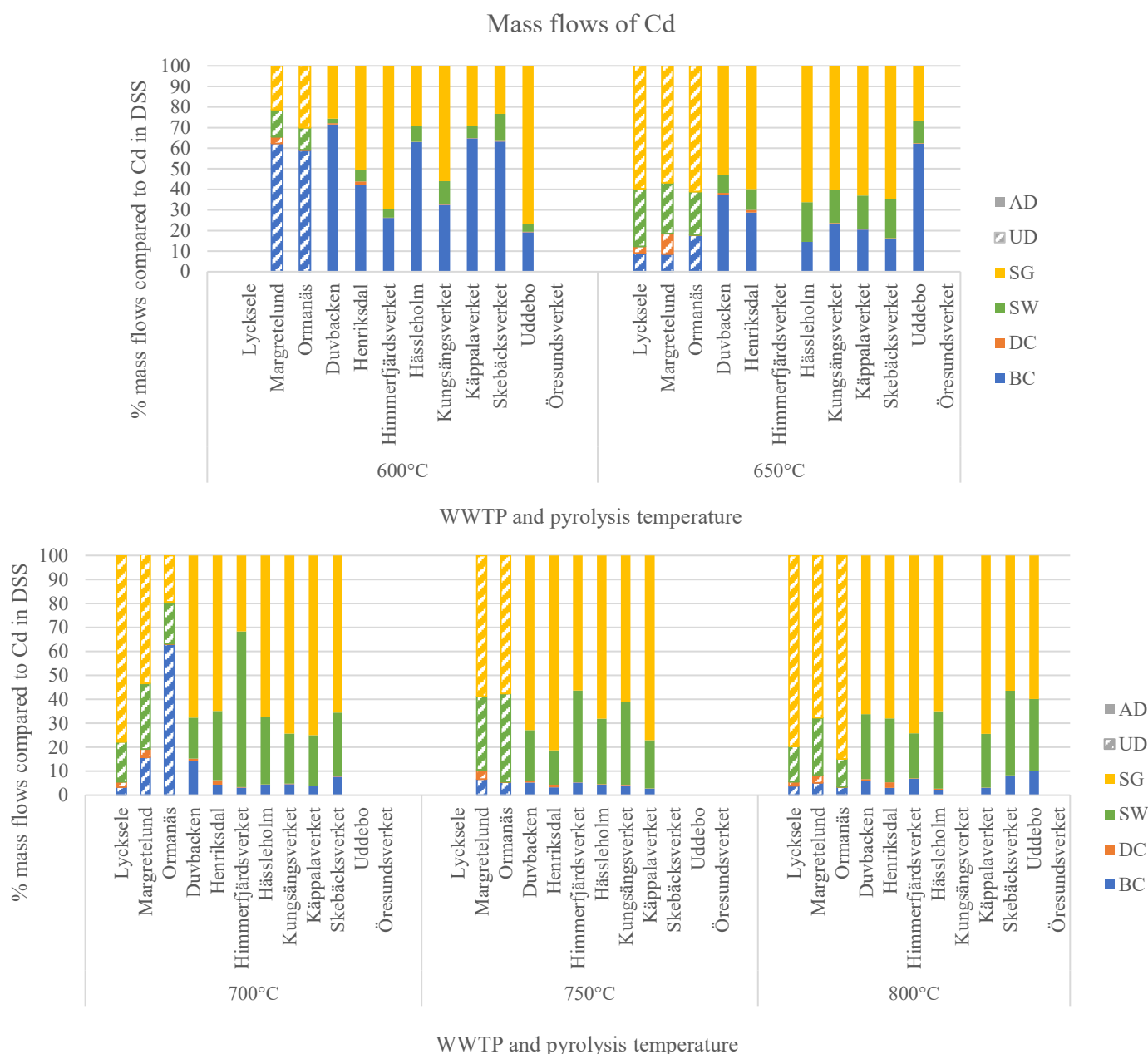


Figure 4.16. Mass flows of cadmium, presented in relation to the contents of the dewatered sewage sludge (DSS), representing 100%. AD = Anaerobic digestion, UD = Undigested, SG = Scrubber flue gas, SW = Scrubber water, DC = Dryer condensate, BC = Sludge biochar.

At 600 °C, a small fraction of the cadmium content in the flue gas was removed by the water scrubber and transferred to the scrubber water (green in Figure 4.16). At 650 °C, this fraction increased to about 25% and at higher temperatures about one third of the cadmium in the flue gas was removed by the scrubber. This indicates that the water scrubber had a low potential of removing cadmium from the flue gas.

Only a small fraction of cadmium was detected in the dryer condensate, potentially indicating that the removed cadmium mainly evaporated during pyrolysis rather than during the drying. It should, however, be noted that the gas from the steam dryer was fed to the

combustion chamber, making it difficult to draw conclusions about the origin of compounds in the scrubber water.

Of the outgoing mass flows, detectable amounts of mercury were only found in the scrubber water (Figure 4.17).

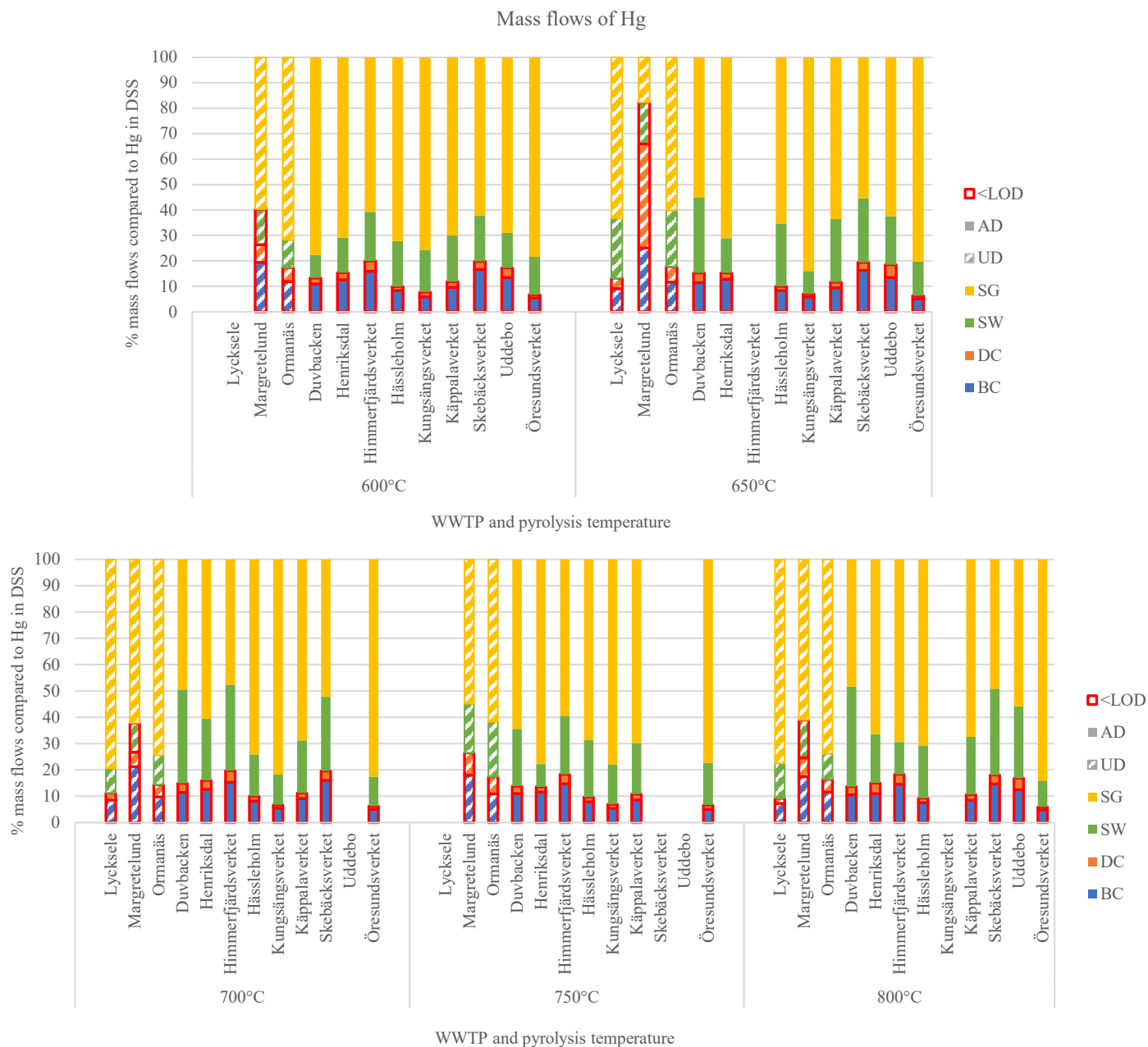


Figure 4.17. Mass flows of mercury, presented in relation to the contents of the dewatered sewage sludge (DSS), representing 100%. LOD = Limit of detection, AD = Anaerobically digested, UD = Undigested, SG = Scrubbed flue gas, SW = Scrubber water, DC = Dryer condensate, BC = Sludge biochar.

The mercury contents of the biochars (as mentioned above) and dryer condensate were below detection limits in all samples (<0.080 mg/kg DS and <0.00010 mg/L DS respectively) and as the contents of the scrubbed flue gas were calculated using these, these three mass flows could not be determined beyond detection limits. The only certain mass

flow was the scrubber water, which on average constituted around 20% of the influent mercury, with no significant effect of increased pyrolysis temperature. This means that on average only between 20-30% of the evaporated mercury was stripped from the flue gas, while the rest passed the scrubber untreated, depending on the contents of the biochars and dryer condensate (possibly ranging between 0% to detection limits). This suggests that the water scrubber had a low potential of removing both cadmium and mercury from the flue gas.

Since no detectable amounts of mercury were found in the dryer condensate this could potentially indicate that the mercury removal was mainly achieved during pyrolysis. This is supported by the fact that the steam dryer could only reach a maximum temperature of 180 °C while mercury typically starts to evaporate at pyrolysis temperatures of 350 °C (Kistler & Widmer, 1987).

4.3.2 Lead, Zinc and Copper

Retention

Unlike the more volatile metals, copper, lead and zinc were mostly retained in the biochars, independently of pyrolysis temperature (Figure 4.18).

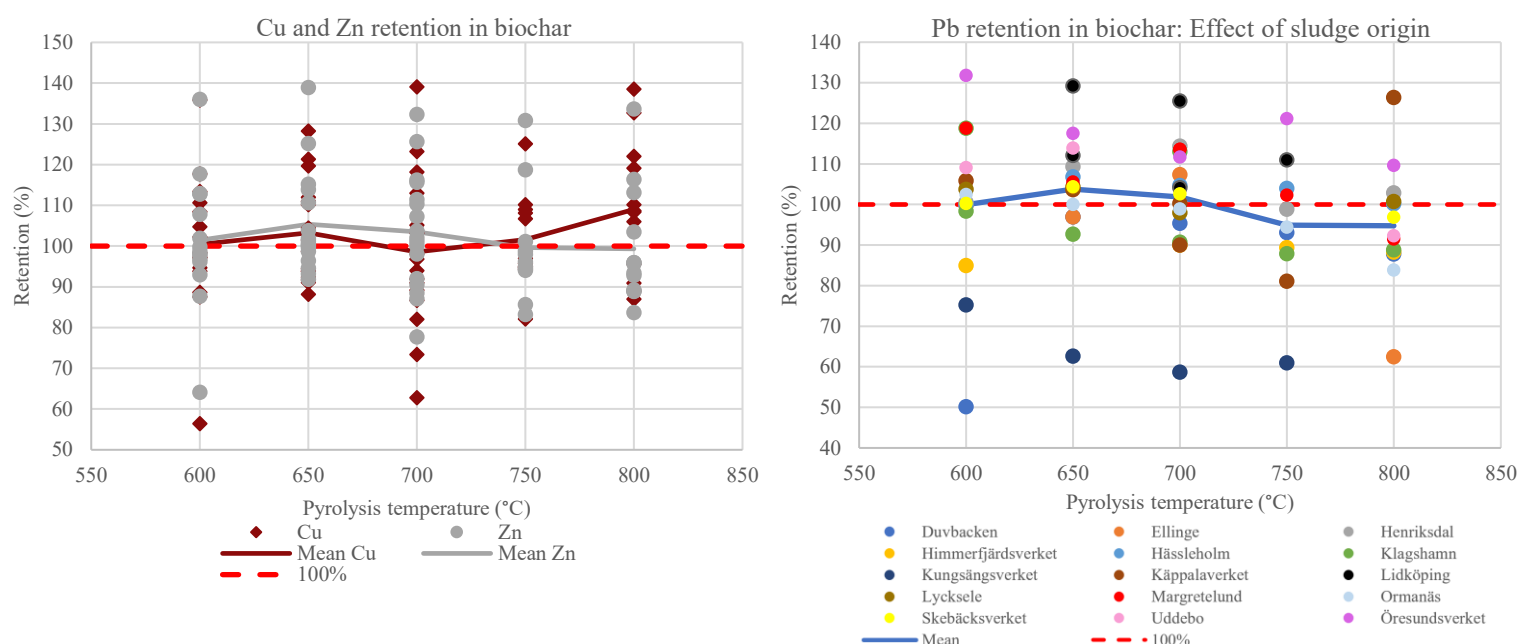


Figure 4.18. Retention of copper and zinc (left) and lead (right).

This was expected given their high boiling points and is consistent with the results from previous studies (Kistler & Widmer, 1987; Méndez *et al.*, 2013; Yuan *et al.*, 2015; Jin *et al.*, 2016). The average lead retention decreased slightly at temperatures over 700 °C (Figure 4.18), possibly indicating an increased volatilisation of lead (Wang & Tomita, 2003; Han *et al.*, 2017). A notable exception was the biochars produced from Kungsängsverket, which had significantly lower lead yields than the other biochars even at low temperatures, with a mean of 64%. As the lead concentration of 15 mg/kg DS in the sludge from Kungsängsverket was comparable to those of the other sludges in this study, this can likely not be explained by a measurement error and could therefore possibly suggest that the specific chemical composition in the sludge from Kungsängsverket makes lead more prone to evaporate. As this behaviour differs from previous findings about lead

which only suggest evaporation potential at pyrolysis temperatures of 850 °C (Han *et al.*, 2017), repeated measurements should be made before any conclusions are drawn.

Limit Values

As seen in Figures 4.19-20, the concentrations of copper, lead and zink were significantly higher in the biochars than in the sludges ($p < 0.05$), due to the loss of organic matter during pyrolysis coupled with the metals being retained in the biochar (Kistler & Widmer, 1987; Méndez *et al.*, 2013; Yuan *et al.*, 2015; Jin *et al.*, 2016).

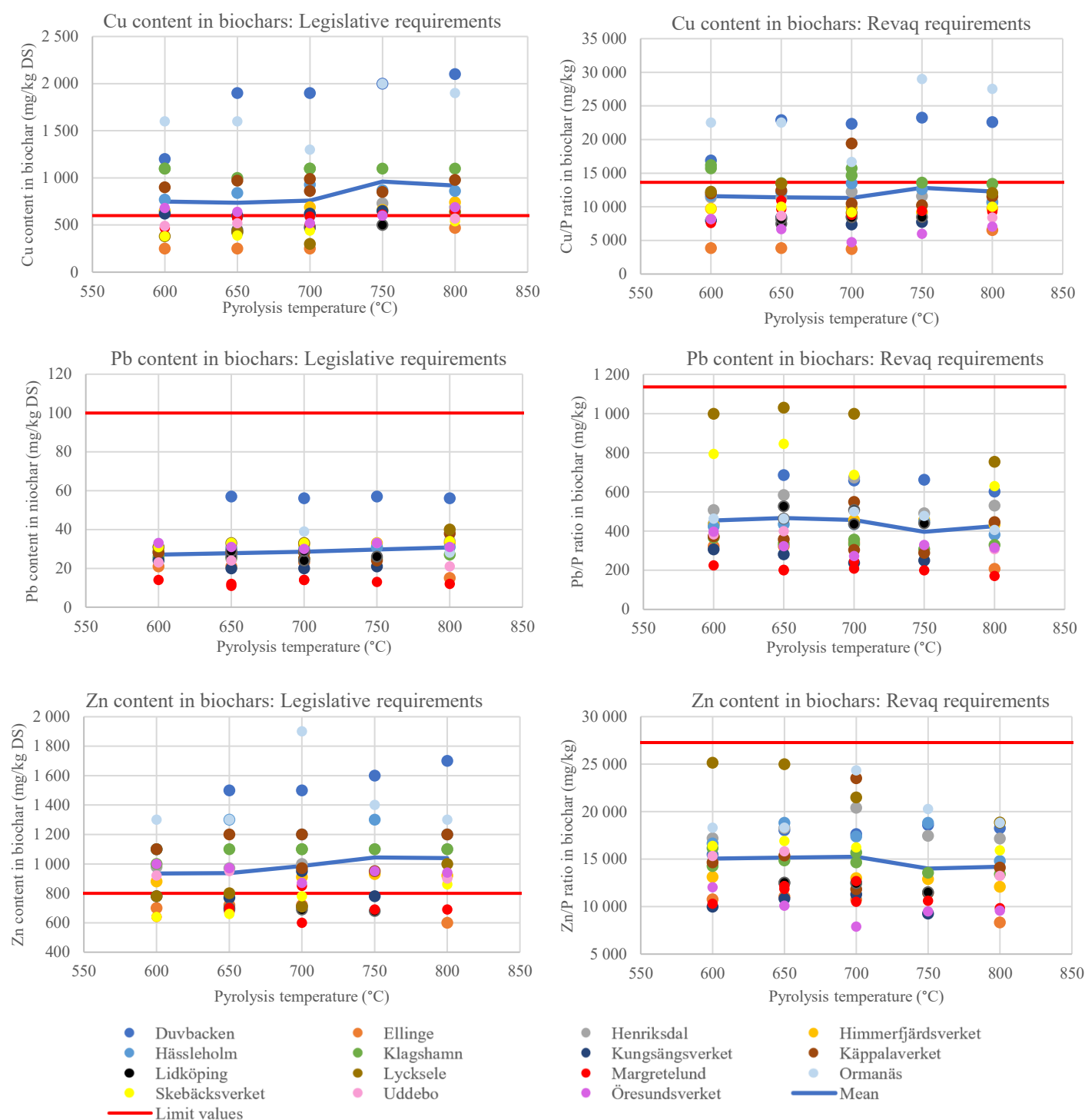


Figure 4.19. Copper, lead and zink contents in biochars, after legislative requirements (left) and requirements by Revaq (right).

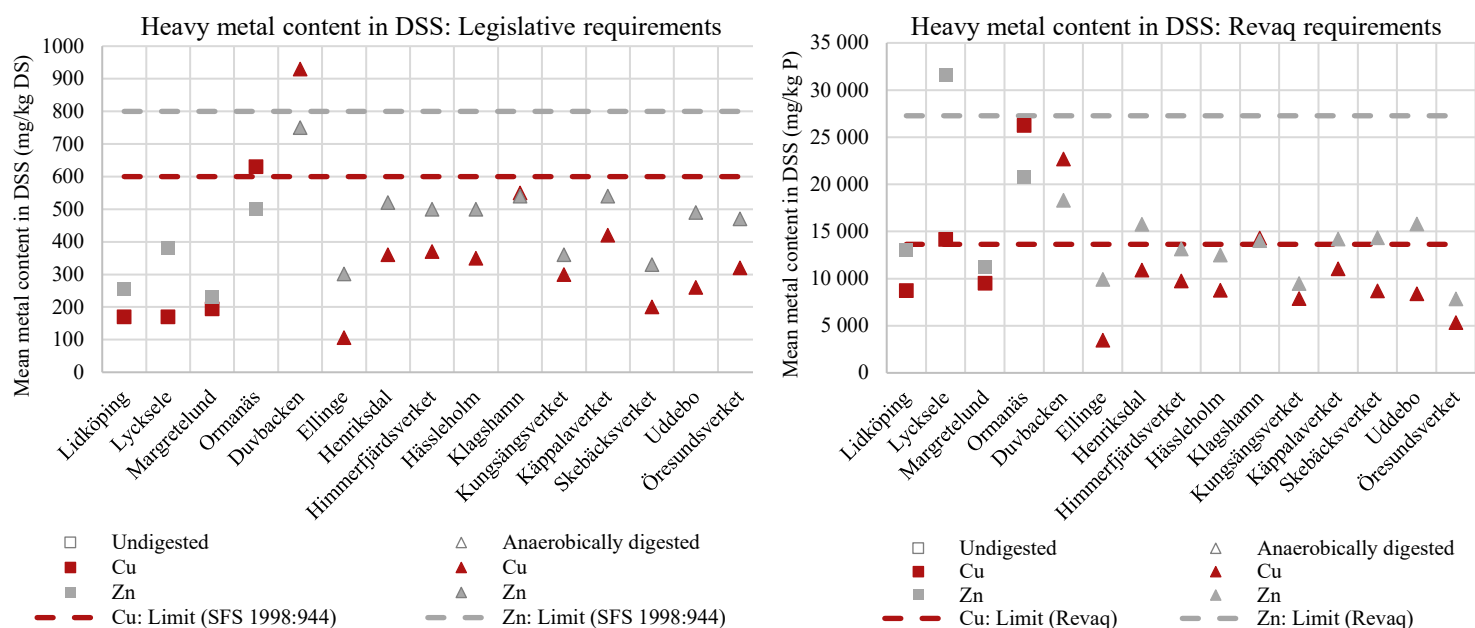


Figure 4.20. Mean heavy metal contents in dewatered sewage sludge (DSS) as concentrations (left) and metal/P ratios (right), including limit values (SFS 1998:944 and Revaq's long-term goal) and distinguishing between anaerobically digested and undigested sludges. Statistical information in Tables B3-4 in Appendix B.

Despite increased concentrations of lead, all biochars had lead concentrations below both the legislative limit and the long-term goal of Revaq, regardless of pyrolysis temperature. The biochars from Lycksele WWTP and Skebäckverket stand out with the highest Pb/P ratios. This is more so a reflection of their relatively low sludge phosphorus concentrations rather than their lead concentrations, which are close to average at about 35 mg/kg DS. For zink and copper, most of the sludges met the legislative requirements and the Revaq requirements, although several sludges had too high Cu/P ratios to be spread on farmland (Figure 4.20). Given the high retention of both the metals and phosphorus, the metal/P ratios were relatively unchanged in the biochars compared to the sludges they were derived from. Consequently, most biochars had acceptable metal/P ratios of copper and zink, while exceeding the legislative limits due to increased concentrations (Figure 4.19).

Causes for Heavy Metal Variability in Sludge and Biochar

The sludges from Duvbacken and Ormanäs WWTP, as well as their respective biochars, had especially high concentrations of copper and zink (Figures 4.19-20). The biochars from Duvbacken also had significantly higher lead concentrations than the other biochars. Duvbacken has historically had elevated levels of copper, lead and zink (Gästrike Vatten, 2026). Industrial sources have for instance been identified as a potential cause (Andersson, 2012) as well as copper material in the water pipe system (Bjarne & Kull, 2016). The copper concentration of about 600 mg/kg DS in the sludge from Ormanäs WWTP is unusually high compared to levels from the last few years according to environmental reports from that WWTP, which are usually around 300 mg/kg DS (Mittskåne 2023, 2024, 2025), making copper in the pipe systems an unlikely cause for the elevated levels.

Biochars from anaerobically digested sludges had significantly higher lead and zink concentrations than biochars from undigested sludges ($p < 0.05$), due to a lower carbon content. The same was seen for copper when disregarding the biochars from Ormanäs

WWTP, which had unusually high levels of copper compared to other undigested sludges (Figure 4.20).

Mass Flows

As was said above, on average 100% of the copper, lead and zink was retained in the biochar. No metals were detected in the dryer condensate, indicating that the drying did not affect the metal contents of the biochars (Appendix C). Only trace amounts of the metals were detected in the scrubber water, suggesting that metals that may have evaporated during pyrolysis were mainly unaffected by the scrubber and ended up in the scrubber flue gas. It should, however, be noted that the contents of the scrubber flue gas were calculated and not measured and that yields below 100% may therefore not be indicative of metals evaporating, but could simply be the result of a heterogeneity in the sludge and biochar causing deviations in individual batches.

4.3.3 Chromium and Nickel

Retention

Contrary to the expectation of a retention around 100% of the (Kistler & Widmer, 1987; Méndez *et al.*, 2013; Yuan *et al.*, 2015; Jin *et al.*, 2016), chromium and nickel accumulated in the biochars with mean retention rates around 125%, regardless of pyrolysis temperature (Figure 4.21).

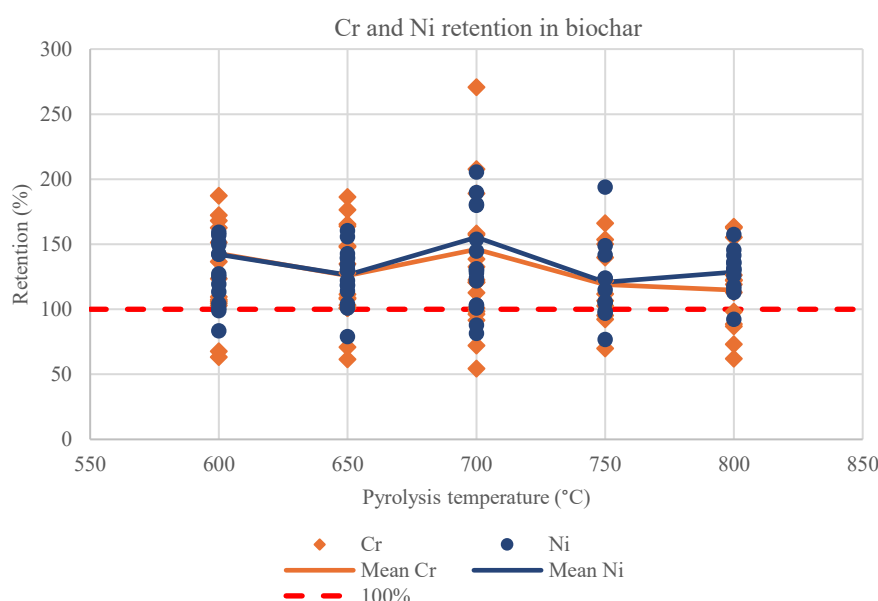


Figure 4.21. Retention of chromium and nickel in the biochar.

Given that the retention of other metals with high boiling points (such as copper, lead and even zink) averaged around 100%, it is unlikely that the accumulation of chromium and nickel would be the result of an inaccurate analysis method. It also seems unlikely that the accumulation would be the result of a heterogeneity in the biochar, in which case, high retention values would be deviations rather than averages given the large data set. As the screws in the steam dryer, as well as the screws connecting the different units in the system, were made by stainless steel containing chromium and nickel (253MA) and the pyrolysis unit was made of Kanthal steel containing chromium, the accumulation could be indicative of erosion of the machinery, potentially caused by friction between screws and outer walls

aided by high operating temperatures (Adamiec, 2009; Shong *et al.*, 2014; Buss *et al.*, 2016). The high ash content in the sludge may also have contributed to friction and erosion.

A few batches with retentions between 350-500% (Ormanäs 700 °C and a replicate of Klagshamn 600 °C) are not visible in Figure 4.21 but explain the particularly high averages at 600 °C and 700 °C and could be the result of the heterogenous nature of the sludge or particularly impactful erosion events.

A similar accumulation of chromium, nickel and iron was observed by Buss *et al.* (2016) during pyrolysis of crops grown on contaminated land in a continuous screw pyrolysis unit made by the same type of stainless steel (253MA, 21% Cr and 11% Ni), making erosion of the machinery a likely cause of the accumulation. As a solution, Buss *et al.* (2016) proposed that a rotary drum pyrolysis unit may cause less friction and could thereby be a more suitable option in this regard. However, further studies on this phenomenon are needed to confirm this.

Limit Values

Both the retention of the original metal content and the additional metal input, possibly from the pyrolysis reactor, in combination with the loss of organic matter during pyrolysis led to higher concentrations of chromium and nickel in the biochars compared to the sludges (Figures 4.22-23).

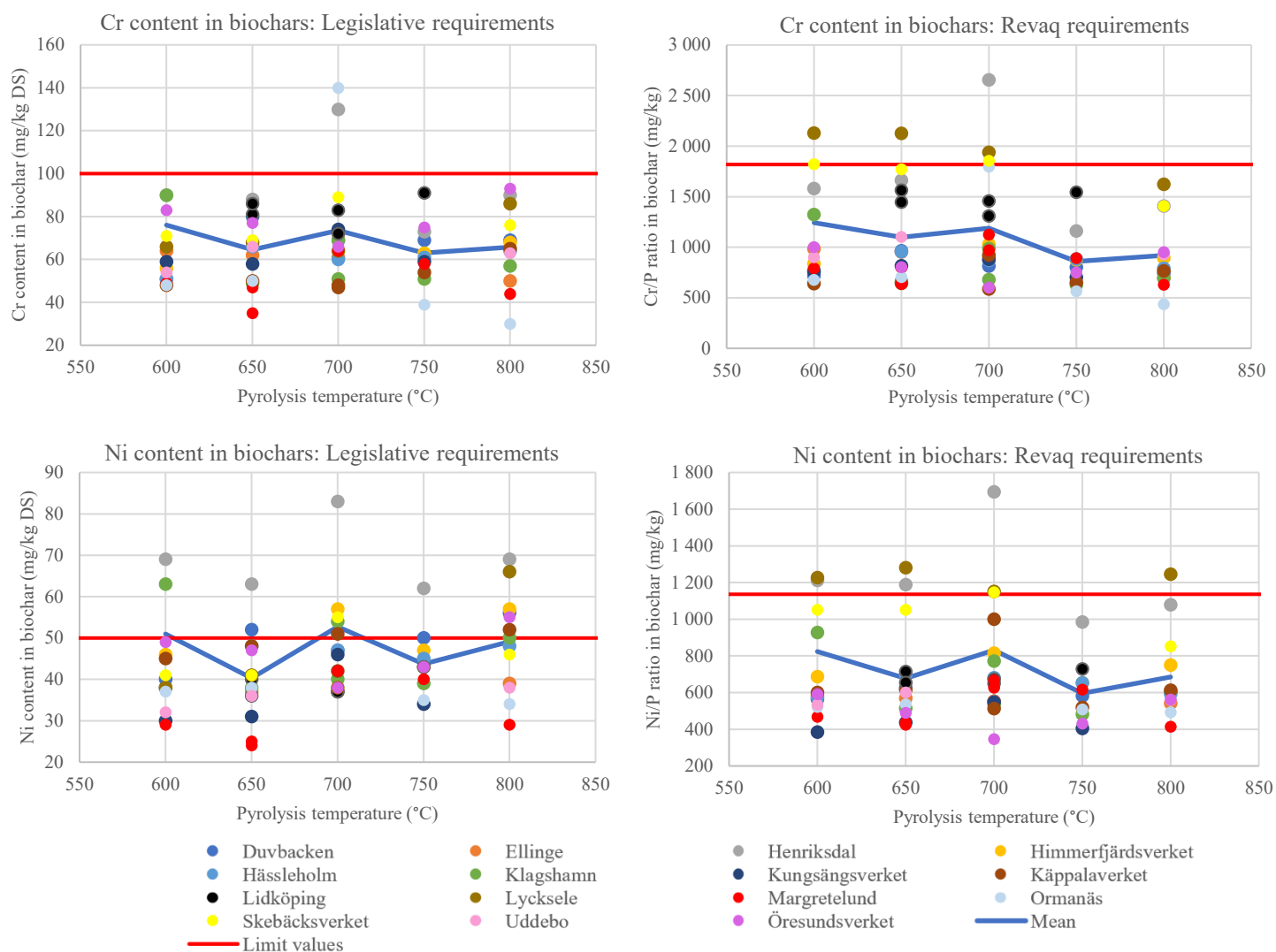


Figure 4.22. Mean heavy metal contents in dewatered sludge after legislative requirements (left) and requirements by Revaq (right), distinguishing between anaerobically digested and undigested sludges.

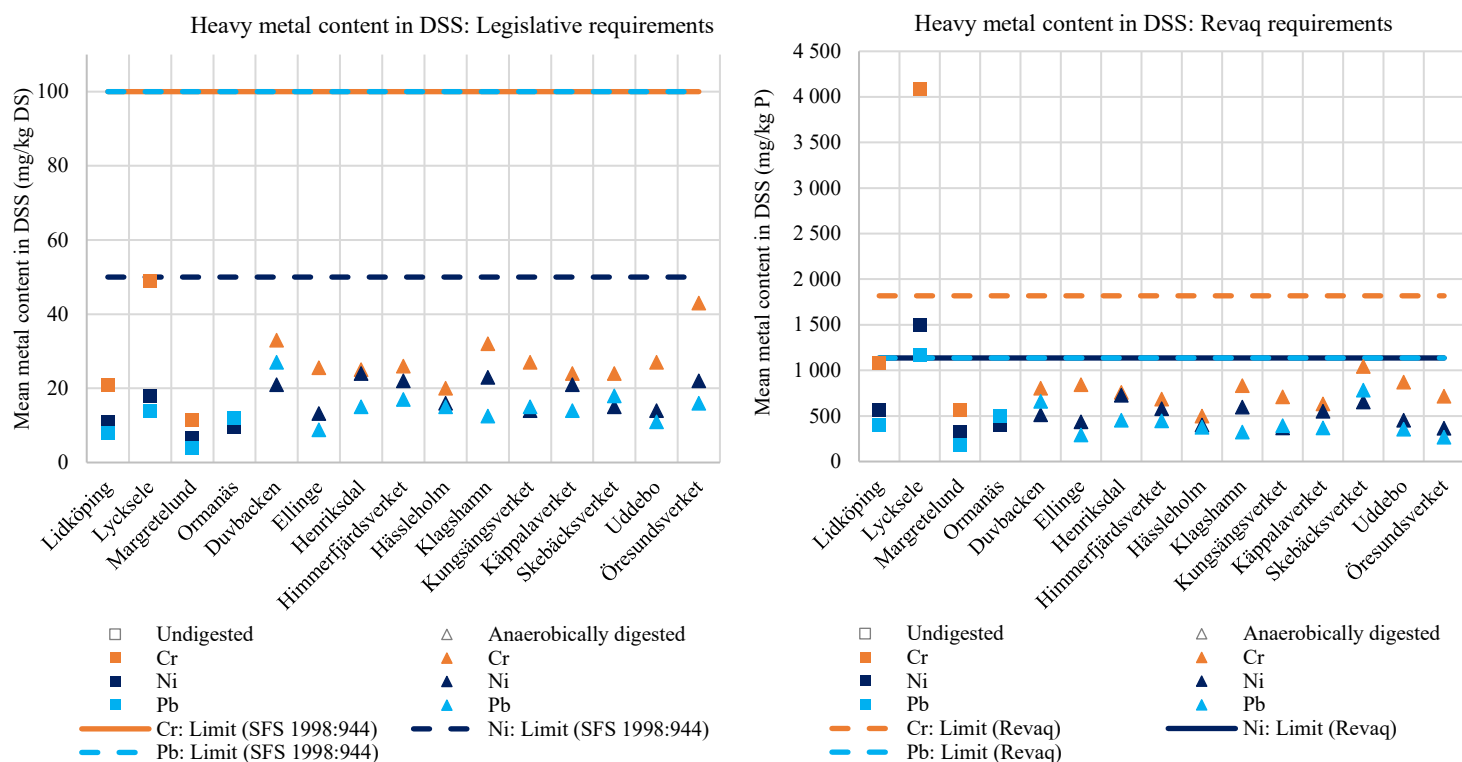


Figure 4.23. Mean heavy metal contents in dewatered sewage sludge (DSS) as concentrations (left) and metal/P ratios (right), including limit values (SFS 1998:944 and Revaq's long-term goal) and distinguishing between anaerobically digested and undigested sludges. Statistical information in Tables B3-4 in Appendix B.

The sludges generally met the legislative requirements as well as Revaq's long-term goal regarding chromium, with a few exceptions. Biochars derived from sludges with low phosphorus content (Figure 4.2) had higher metal/P ratios (Figure 4.22), contributing to a few of them failing to meet Revaq's requirement of a maximum of 1818 mg Cr/kg P. More of the biochars failed to meet the requirements regarding nickel, where the average concentration was just below the legislative limit (50 mg/kg DS) and where, similarly to chromium, low sludge phosphorus concentrations contributed to high Ni/P ratios, causing biochars from some sludges to exceed the Revaq limit of 1 136 mg Ni/kg P.

A couple of batches (Ormanäs 700 °C and a replicate of Klagshamn 600 °C) are not visible in Figure 4.22 but contribute to the elevated means at 600 °C and 700 °C due to particularly high values (Tables B3-4 in Appendix B). These elevated values could be the result of particularly impactful erosion events or a heterogeneity in the sludge.

The metal/P ratios were expected to be relatively unchanged in the biochars compared to the sludges due to the high retention of both the metals and phosphorus, but due to the additional metal input during the pyrolysis the ratios were elevated in the biochars (Figures 4.22-23). It should be noted that more of the sludges may have met the requirements without this addition, emphasizing the importance of investigating the cause of this phenomenon and possible solutions.

4.3.4 Aluminium and Iron

The retention of aluminium and iron in the biochar averaged around 105-110% and 115% respectively (Figure 4.24).

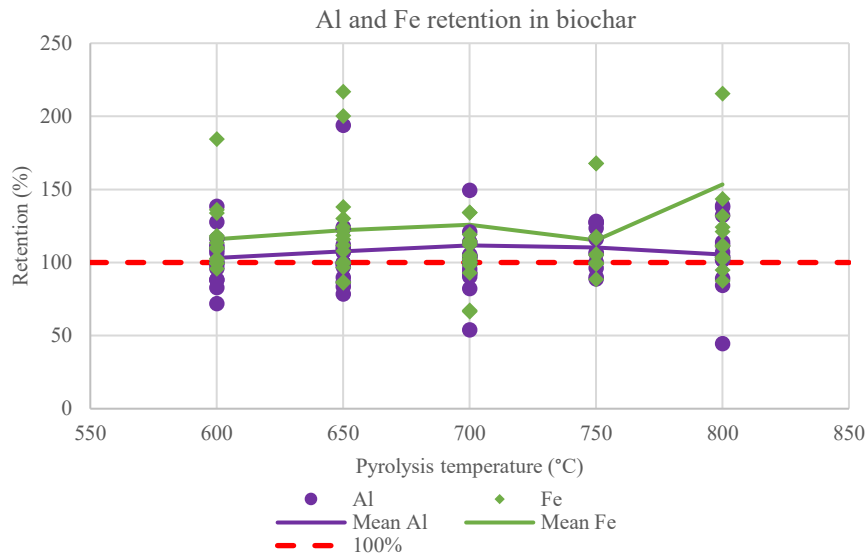


Figure 4.24. Retention of aluminium and iron in the biochar.

While it was expected that close to 100% of the metals would be retained in the biochar due to high boiling points (Hossain *et al*, 2011; Yuan *et al.*, 2015), the mean retention exceeding 100% could be indicative of erosion of the machinery (see 4.3.3 *Chromium and Nickel*). As the retention was close to 100% it is not certain whether the elevated levels were the result of random errors (possibly caused by a heterogeneity in the biochar or measurement errors) or whether it is indicative of erosion, but it is possible considering that the pilot plant was constructed using stainless steel and that the construction material of the pyrolysis unit was an iron-chromium-aluminium alloy (Kanthal). Two batches with iron retentions around 500 % cause the mean retention in Figure 4.23 to be especially high at 700 °C and 800 °C. This may be the result of a heterogeneity in the biochar but could also be attributed to a particularly impactful erosion event.

There was a high variability in the aluminium and iron concentrations of biochars from different WWTPs, ranging from about 10-100 and 20-200 g/kg DS respectively (Figure 4.25).

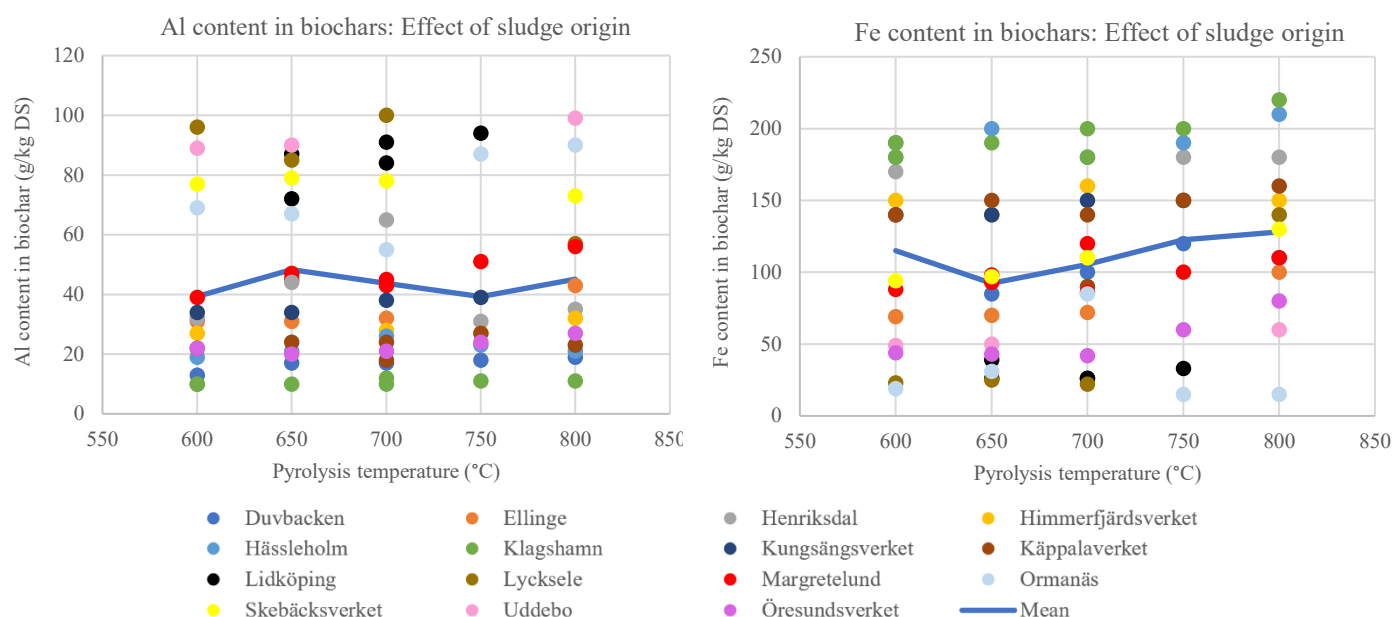


Figure 4.25. Aluminium and iron contents in biochars.

Sludges with high aluminium and iron contents produced biochars with correspondingly high metal contents and vice versa (Figures 4.25-4.26).

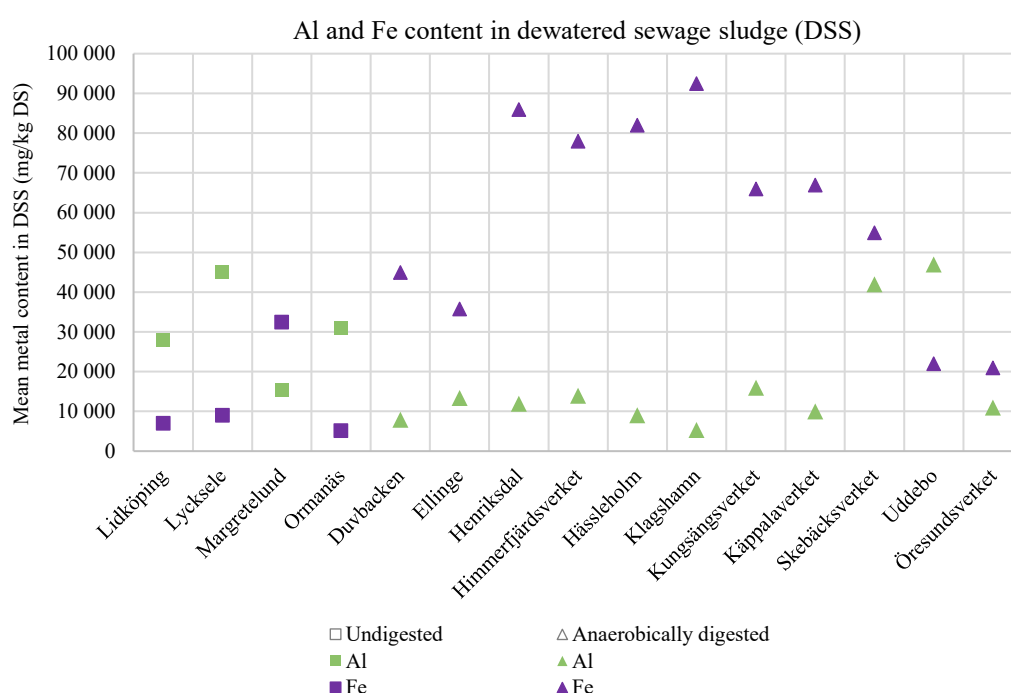


Figure 4.26. Aluminium and iron contents in dewatered sewage sludge (DSS). Statistical information in Table B3 in Appendix B.

The varying aluminium and iron concentrations in the sludges largely depended on the dosage of aluminium or iron containing coagulants in the wastewater treatment process (Table 3.1). This is especially visible for iron, where there are differences as large as 70 g/kg DS between anaerobically digested sludges that dose iron and digested sludges that do not (Figure 4.26), leading to large differences in biochar concentrations (Figure 4.25). This relationship is not always seen, as for example the sludge from Duvbacken had a

higher iron concentration than the sludge from Ellinge WWTP, despite the sludge from Duvbacken having been made with EBPR (no iron dosing) and the sludge from Ellinge WWTP having been treated with iron containing chemicals. This could possibly be explained by differences in iron concentrations in the influent wastewaters, possibly due to different geologies or pipe materials. Dosing likely also contributed to the variations in aluminium and iron concentrations between biochars from different sludges (Figure 4.25) but data on the dosing used for specific sludge batches are needed to analyse this further.

The aluminium and iron concentrations in the sludges and biochars were also affected by the carbon content, where biochars from anaerobically digested sludges generally had higher iron concentrations than biochars from undigested sludges, when comparing iron dosed sludges (see Klagshamn WWTP and Margretelund WWTP in Figure 4.26). This effect was less prevalent for aluminium, possibly due to smaller dosages of aluminium compared to iron.

4.4 Concluding Discussion

Comparing sludges from fifteen WWTPs using varying treatment methods, this thesis increases the knowledge about what factors affect sludge biochar contents. The pilot plant also contributes with operational experience of pyrolysis of sewage sludge on a pilot scale, identifying challenges regarding varying sludge characteristics and operating parameters.

4.4.1 The Effect of Pyrolysis Temperature on Biochar Contents

Regarding heavy metals, the pyrolysis temperature mainly affected the cadmium contents of the biochar. Increased cadmium volatilisation could be seen up to 750 °C, over which point the pyrolysis temperature essentially had no further effect. A pyrolysis temperature of 650 °C was sufficient to meet the legislative requirements as well as the long-term goal of Revaq regarding cadmium, and 600 °C was enough to reach levels below the detection limit of mercury. The remaining metals, as well as phosphorus and potassium, were essentially unaffected by the pyrolysis temperature in the range investigated in this study, with an average retention of about 100% (or higher, in the case of aluminium, chromium, iron and nickel).

Higher pyrolysis temperatures caused greater losses of nitrogen and slightly increased losses of carbon, although the effect was not as prevalent as with nitrogen. The same was seen for the biochar yield, which was mostly affected by the carbon content. With these parameters in mind, an optimal pyrolysis temperature (when only considering mass flows) could be between 650-750 °C, depending on whether nutrient and carbon retention or maximal cadmium removal is prioritised. These temperatures are sufficient for decreasing cadmium and mercury concentrations below legislative and Revaq limits and a lower pyrolysis temperature can give a higher nitrogen and carbon content and thus produce a fertilizer of higher value and maximise carbon sequestration while a higher temperature maximises cadmium removal. Other sludge biochar properties should, however, also be considered when determining an optimal pyrolysis temperature, such as phosphorus availability and the removal of organic micropollutants.

4.4.2 The Effect of Treatment Methods and Sludge Characteristics on Biochar Contents

Anaerobic digestion significantly influenced both sludge and biochar contents, mainly by contributing to a lower carbon content and thereby a higher ash content, likely due to carbon losses during the digestion process. This generally resulted in higher concentrations of

heavy metals, but also of phosphorus, in both the digested sludges and their resulting biochars. As a consequence, biochars from digested sludges were more likely to exceed limit concentrations of heavy metals. It was estimated that a lesser amount of the total carbon in a raw sludge could be sequestered as biochar if the sludge was anaerobically digested before pyrolysis, due to carbon conversion into biogas during digestion. A higher fraction of pyrolysis gas could furthermore be produced by undigested sludges, suggesting that a self-sustaining pyrolysis process may be more achievable when using an undigested sludge if the pyrolysis gas is recirculated back into the system as fuel.

The variability of aluminium and iron concentrations in the biochars was largely determined by dosing of aluminium or iron containing coagulants in the wastewater treatment process and by anaerobic digestion of the sludge, which led to increased biochar concentrations.

Sludge origin significantly influenced sludge biochar contents due to factors that could not be explained by treatment methods alone but could rather be the result of local conditions such as industrial load, geology and pipe materials. The potential influence of these factors contributed to a high variability in heavy metal contents in biochars originating from different WWTPs, which led to them being more or less likely to meet legislative and Revaq requirements regarding heavy metal content. Sludge origin should therefore be considered when determining appropriate sludge treatment methods and upstream work to reduce the heavy metal load in the influent wastewater can reduce the heavy metal content in the sludge biochar.

4.4.3 Legislative and Revaq Requirements

Table 4.2 provides an overview of how concentrations of heavy metals in the sludge biochars were affected by a pyrolysis temperature of 700 °C, which could potentially be optimal when only considering mass flows.

Table 4.2. Overview of the performance of sludges and biochars in meeting legislative and Revaq requirements regarding heavy metal contents. Green = Meets both legislative and Revaq requirements, Yellow = Fails to meet either legislative or Revaq requirements, Red = Fails to meet both legislative and Revaq requirements. UD = Undigested, AD = Anaerobically digested.

	Sludge pretreatment	WWTP	Pyrolysis temperature	Cd	Cr	Cu	Hg	Ni	Pb	Zn
Sludges	UD	Lidköping		Yellow	Green	Green	Green	Green	Green	Green
	UD	Lycksele		Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow
	UD	Margretelund		Green	Green	Green	Green	Green	Green	Green
	UD	Ormanäs		Yellow	Green	Red	Yellow	Green	Green	Green
	AD	Duvbacken		Yellow	Green	Red	Green	Green	Green	Green
	AD	Ellinge		Yellow	Green	Green	Green	Green	Green	Green
	AD	Henriksdal		Green	Green	Green	Green	Green	Green	Green
	AD	Himmerfjärdsverket		Green	Green	Green	Green	Green	Green	Green
	AD	Hässleholm		Green	Green	Green	Yellow	Green	Green	Green
	AD	Klagshamn		Yellow	Green	Yellow	Green	Green	Green	Green
	AD	Kungsängsverket		Green	Green	Green	Yellow	Green	Green	Green
	AD	Käppalaverket		Green	Green	Green	Yellow	Green	Green	Green
	AD	Skebäcksverket		Green	Green	Green	Yellow	Green	Green	Green
	AD	Uddebo		Yellow	Green	Green	Green	Green	Green	Green
	AD	Öresundsverket		Green	Green	Green	Yellow	Green	Green	Green
Sludge biochars	UD	Lidköping	700°C	Green	Green	Green	Green	Green	Green	Green
	UD	Lycksele	700°C	Green	Yellow	Green	Green	Yellow	Green	Green
	UD	Margretelund	700°C	Green	Green	Green	Green	Green	Green	Yellow
	UD	Ormanäs	650°C	Green	Green	Red	Green	Green	Green	Green
	AD	Duvbacken	700°C	Green	Green	Red	Green	Green	Green	Yellow
	AD	Ellinge	700°C	Green	Green	Green	Green	Green	Green	Green
	AD	Henriksdal	700°C	Green	Red	Green	Green	Red	Green	Yellow
	AD	Himmerfjärdsverket	700°C	Green	Green	Yellow	Green	Yellow	Green	Yellow
	AD	Hässleholm	700°C	Green	Green	Yellow	Green	Green	Green	Yellow
	AD	Klagshamn	700°C	Green	Green	Red	Green	Yellow	Green	Yellow
	AD	Kungsängsverket	700°C	Green	Green	Yellow	Green	Green	Green	Yellow
	AD	Käppalaverket	700°C	Green	Green	Red	Green	Yellow	Green	Yellow
	AD	Skebäcksverket	700°C	Green	Yellow	Green	Green	Red	Green	Green
	AD	Uddebo	650°C	Green	Green	Green	Green	Green	Green	Yellow
	AD	Öresundsverket	700°C	Green	Green	Green	Green	Green	Green	Yellow

As cadmium and mercury were removed during pyrolysis, the biochars performed better than the sludges in meeting legal requirements regarding these metals, as well as in meeting the requirements of Revaq. The concentrations of other heavy metals were generally higher in the biochars compared to the sludges due to a high retention and losses of organic matter during pyrolysis. This led to fewer of the biochars meeting the legislative requirements regarding most heavy metal concentrations compared to the sludges. This was particularly true for zinc and copper, possibly as a consequence of relatively high sludge concentrations already approaching limit values before pyrolysis. The metal/P ratios of these stable metals were generally unchanged during the pyrolysis since both phosphorus and metals were

retained in the biochar, but increased for chromium and nickel, potentially due to erosion of the machinery.

In conclusion, while the conversion of sewage sludge into sludge biochar can effectively reduce cadmium and mercury contents below legislative limits, increased concentrations of other heavy metals can prevent sludge biochar from being permitted to be used as a fertilizer under current legislation. The likelihood of a biochar meeting legislative requirements increases if the sludge is undigested and by upstream work to reduce the heavy metal load on the influent wastewater. Efforts to reduce potential erosion of chromium and nickel in the pyrolysis plants can also mitigate increased biochar concentrations of these metals. Considering that the total amount of heavy metals in the sludge decreases during pyrolysis, and that the bioavailability of the metals likely decreases (Kistler & Widmer, 1987; He *et al.*, 2010; Hossain *et al.*, 2011), the spreading of sludge biochar may reduce the exposure risk of heavy metals in the soil compared to the spreading of sewage sludge, despite increased concentrations of certain metals. When also considering the removal of organic micropollutants (Zielińska & Oleszczuk, 2015a) and the potential of the biochar to act as an adsorbent for heavy metals already existing in the soil (Ji *et al.*, 2022), as well as the potential for carbon sequestration and reduced emissions of greenhouse gases due to eliminated needs for sludge storage, the spreading of sludge biochar may on the whole pose a lower environmental risk than the spreading of sewage sludge.

4.4.4 The Water Scrubber

Although the water scrubber was mainly intended for particle removal it may be of interest to evaluate its potential in removing cadmium and mercury from the flue gas since these metals are volatilised during pyrolysis. With higher pyrolysis temperatures the water scrubber removed increasing amounts of cadmium from the flue gas but no more than approximately 30%. Similarly, at most 30% of the mercury content in the flue gas was removed by the scrubber, indicating that it had a low potential of removing cadmium and mercury from the flue gas. While legislative limit values regarding the heavy metal content of the flue gas can be guiding in whether further gas treatment is necessary or not, it may be considered in full scale pyrolysis plants to limit atmospheric pollution. Otherwise, the transfer of these metals from the sludge to the atmosphere may potentially increase the exposure risk of these metals compared with sludge spreading.

4.4.5 Uncertainties

As only four of the fifteen sludges studied were undigested, the analysis of more undigested sludges would give more reliable conclusions regarding the effect of anaerobic digestion on biochar contents and give more insight to factors affecting variation within undigested sludges. Similarly, more replicates of biochars produced at certain pyrolysis temperatures would reduce uncertainties regarding the influence of heterogeneity in the biochars and provide more reliable data.

5 Conclusions

This thesis examined what factors affect sludge biochar contents of nutrients and heavy metals by studying the pyrolysis products of sewage sludges from fifteen Swedish WWTPs. The following main conclusions could be drawn:

- Pyrolysis can effectively remove cadmium and mercury from the sludge biochar while concentrations of other heavy metals increase during pyrolysis due to high retention and loss of mainly organic matter, leading to a lower likelihood of the biochars meeting legislative requirements. Considering the removal of cadmium, mercury and organic micropollutants as well as an increased immobilization of the remaining heavy metals, pyrolysis of sewage sludge can, despite increased concentrations of certain heavy metals, on the whole be an environmentally advantageous sludge treatment method.
- Pyrolysis temperature mainly affected the biochar concentrations of cadmium and nitrogen and a pyrolysis temperature between 650-750 °C could be optimal depending on if nitrogen retention or cadmium removal is prioritised.
- Anaerobic digestion and the use of aluminium or iron containing chemicals in the wastewater treatment process are important factors affecting biochar contents of nutrients and heavy metals. Anaerobically digested sludge produces higher biochar yields but a lower carbon content, leading to higher heavy metal and nutrient concentrations compared to undigested sludge.
- The water scrubber had a low potential of removing cadmium and mercury from the flue gas and further gas treatment may be considered to avoid atmospheric pollution.
- Elevated levels of chromium and nickel in the biochar may be caused by erosion of the machinery, possibly due to friction between screws and outer walls or feedstock. Solutions to this need to be investigated.

6 Future Research

This study focused on the fate of heavy metals and nutrients during pyrolysis of sewage sludge, but more knowledge about what affects biochar properties is needed. The following research areas are of especial interest.

- A more advanced analysis of how sludge characteristics and operating conditions affect agronomic properties of biochar, regarding for example phosphorus availability, heavy metal bioavailability and porosity.
- Field trials assessing the effectiveness of different biochars as fertilizers and soil amendments, where the impact of sludge characteristics and operating conditions are investigated.
- A more in-depth analysis of the effect of different wastewater treatment methods on sludge and biochar properties, regarding different nitrogen removal methods, biological phosphorus removal and different types of sludge (e.g. primary, secondary).
- The potential of co-pyrolysis of sewage sludge with for example agricultural waste to increase phosphorus availability and reduce heavy metal bioavailability.
- An evaluation of the erosion risk in pyrolysis reactors to avoid unnecessarily high chromium, iron and nickel contents in the biochar. Attention should be given to the impact of different reactor types, e.g. continuous screw reactors and rotary drum reactors, optimized process conditions and possibly different reactor materials.

References

- Adamiec, J. (2009). High temperature corrosion of power boiler components clad with nickel alloys. *Materials Characterization*, 60(10), 1093–1099. <https://doi.org/10.1016/j.matchar.2009.03.017>
- Adhikari, S., Gasco, G., Mendez, A., Surapaneni, A., Jegatheesan, V., Shah, K., & Paz-Ferreiro, J. (2019). Influence of pyrolysis parameters on phosphorus fractions of biosolids derived biochar. *Sci Total Environ*, 695, Article 133846. <https://doi.org/10.1016/j.scitotenv.2019.133846>
- Agrafioti, E., Bouras, G., Kalderis, D., & Diamadopoulos, E. (2013). Biochar production by sewage sludge pyrolysis. *Journal of Analytical and Applied Pyrolysis*, 101, 72–78. <https://doi.org/10.1016/j.jaap.2013.02.010>
- Andersson, L. (2012). Avsättningsalternativ för avloppsslam från Gästrike Vatten. Möjligheter att återföra fosforföreningar i avloppsslam till produktiv mark. Uppsala Universitet. Examensarbete, W12004. URN: urn:nbn:se:uu:diva-172300. <https://www.diva-portal.org/smash/get/diva2:513933/FULLTEXT01.pdf>
- Andersson, P. G. (2015). Slamspridning på åkermark—fältförsök med kommunalt avloppsslam från Malmö och Lund under åren 1981–2014 (Sludge spreading on agricultural land—field tests with municipal sewage sludge from Malmö and Lund in the years 1981–2014). Hushållningssällskapens rapportserie 17. <http://hushallningssallskapet.se/wpcontent/uploads/2015/05/slamrapport-2015.pdf>
- Bagreev, A., Bandosz, T. J., & Locke, D. C. (2001). Pore structure and surface chemistry of adsorbents obtained by pyrolysis of sewage sludge-derived fertilizer. *Carbon*, 39(13), 1971–1979. [https://doi.org/10.1016/S0008-6223\(01\)00026-4](https://doi.org/10.1016/S0008-6223(01)00026-4)
- Bjarne, K. & Kull, P. O. (2016). Miljökonsekvensbeskrivning. MKB enligt 6 kap miljöbalken inför tillståndsprövning enligt 9 kap miljöbalken. Duvbackens reningsverk. Aquasvea. <https://www.gastrikevatten.se/download/18.39f98b4f1722babea8d252e6/1590577851603/MKB%20Duvbacken.pdf>
- Buss, W., Graham, M. C., Shepherd, J. G. & Mašek, O. (2016). Suitability of marginal biomass-derived biochars for soil amendment. *Science of The Total Environment*, 547, 314–322. <https://doi.org/10.1016/j.scitotenv.2015.11.148>
- Börjesson, G., Kirchmann, H., Kätterer, T. (2014). Four Swedish long-term field experiments with sewage sludge reveal a limited effect on soil microbes and on metal uptake by crops. *J Soils Sediments*, 14, 164–177. <https://doi.org/10.1007/s11368-013-0800-5>
- Börjesson, G., Kätterer, T. (2018). Soil fertility effects of repeated application of sewage sludge in two 30-year-old field experiments. *Nutr Cycl Agroecosyst*, 112, 369–385. <https://doi.org/10.1007/s10705-018-9952-4>

- Cao, Y. & Pawłowski, A. (2012). Sewage sludge-to-energy approaches based on anaerobic digestion and pyrolysis: Brief overview and energy efficiency assessment. *Renewable and Sustainable Energy Reviews*, 16(3), 1657-1665. <https://doi.org/10.1016/j.rser.2011.12.014>.
- Council Directive 86/278/EEC of 12 June 1986 on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture (OJ L 181, 4.7.1986, pp. 6–12). <http://data.europa.eu/eli/dir/1986/278/oj>
- Dai, Z., Zhang, X., Tang, C., Muhammad, N., Wu, J., Brookes, P. C. & Xu, J. (2017). Potential role of biochars in decreasing soil acidification - A critical review. *Science of The Total Environment*, 581–582, 601-611. <https://doi.org/10.1016/j.scitotenv.2016.12.169>
- De Figueiredo, C. C., Moura Chagas, J. K., Da Silva, J. & Paz-Ferreiro, J. (2019). Short-term effects of a sewage sludge biochar amendment on total and available heavy metal content of a tropical soil. *Geoderma*, 344, 31-39. <https://doi.org/10.1016/j.geoderma.2019.01.052>
- De la Rosa, J. M., Sánchez-Martín, Á. M., Campos, P. & Miller, A. Z. (2019). Effect of pyrolysis conditions on the total contents of polycyclic aromatic hydrocarbons in biochars produced from organic residues: Assessment of their hazard potential. *Science of The Total Environment*, 667, 578-585. <https://doi.org/10.1016/j.scitotenv.2019.02.421>.
- Di Bianca, M., Borch, C., Lotti, G., Rizzo, A., & Salimbeni, A. (2026). Phosphorus recovery and adsorbents production from municipal sewage sludge by the integration of slow pyrolysis and chemical leaching. *Cleaner Engineering and Technology*, 30, Article 101140. <https://doi.org/10.1016/j.clet.2025.101140>
- EBC. (2025). 'European Biochar Certificate - Guidelines for a Sustainable Production of Biochar.' Carbon Standards International (CSI), Frick, Switzerland. Version 10.5E from 14th August 2025. <http://carbon-standards.com/ebc>
- EGLE, L., MARSCHINSKI, R., JONES, A., YUNTA MEZQUITA, F., SCHILLACI, C. HUYGENS, D. (2023). Feasibility study in support of future policy developments of the Sewage Sludge Directive (86/278/EEC). Publications Office of the European Union, Luxembourg. <https://data.europa.eu/doi/10.2760/305263>, JRC134591.
- Figueiredo, C., Lopes, H., Coser, T., Vale, A., Busato, J., Aguiar, N., Novotny, E., & Canellas, L. (2017). Influence of pyrolysis temperature on chemical and physical properties of biochar from sewage sludge. *Archives of Agronomy and Soil Science*, 64(6), 881–889. <https://doi.org/10.1080/03650340.2017.1407870>
- Gascó, G., Blanco, C.G., Guerrero, F. & Méndez Lázaro, A.M. (2005). The influence of organic matter on sewage sludge pyrolysis. *J. Anal. Appl. Pyrolysis*, 74, 413–420. <https://doi.org/10.1016/j.jaap.2004.08.007>
- Gästrike Vatten. (2026). Miljörapport 2024 Duvbackens reningsverk Gävle Vatten AB. Ärende nr: 309393. <https://www.gastrikevatten.se/download/18.2b20a76219927b3521ccc052/1758694530487/Milj%C3%B6rapport%20Duvbacken%202024.pdf>

- Han, H., Hu, S., Syed-Hassan, S. S. A., Xiao, Y., Wang, Y., Xu, J., Jiang, L., Su, S., & Xiang, J. (2017). Effects of reaction conditions on the emission behaviors of arsenic, cadmium and lead during sewage sludge pyrolysis. *Bioresource Technology*, 236, 138–145. <https://doi.org/10.1016/j.biortech.2017.03.112>
- He, Y.D., Zhai, Y.B., Li, C.T., Yang, F., Chen, L., Fan, X.P., Peng, W.F. and Fu, Z.M. (2010). The fate of Cu, Zn, Pb and Cd during the pyrolysis of sewage sludge at different temperatures. *Environmental Technology*, 31(5), 567–574. https://www.tandfonline.com/doi/epdf/10.1080/09593330903514466?src=getft&utm_source=sciencedirect_contenthosting&getft_integrator=sciencedirect_contenthosting
- Hoffman, T.C., Zitomer, D.H. & McNamara, P.J. (2016). Pyrolysis of wastewater biosolids significantly reduces estrogenicity. *Journal of Hazardous Materials*, 317, 579–584. <https://doi.org/10.1016/j.jhazmat.2016.05.088>.
- Hossain, M., Strezov, V. & Nelson, P. F. (2009). Thermal characterisation of the products of wastewater sludge pyrolysis. *Journal of Analytical and Applied Pyrolysis*, 85(1–2), 442–446. <https://doi.org/10.1016/j.jaap.2008.09.010>.
- Hossain, M. K., Strezov, V., Chan, K. Y., Ziolkowski, A., & Nelson, P. F. (2011). Influence of pyrolysis temperature on production and nutrient properties of wastewater sludge biochar. *J Environ Manage*, 92(1), 223–228. <https://doi.org/10.1016/j.jenvman.2010.09.008>
- Hossain, M. Z., Bahar, M. M., Sarkar, B., Donne, S. W., Ok, Y. S., Palansooriya, K. N., Kirkham, M. B., Chowdhury, S. & Bolan, N. (2020). Biochar and its importance on nutrient dynamics in soil and plant. *Biochar*, 2, 379–420. <https://doi.org/10.1007/s42773-020-00065-z>
- Hörsing, M., Eriksson, E., Gissén, C., La Cour Jansen, J., Ledin, A. (2014). Organic pollutants in beet and haulm cultivated on farmland fertilized with sludge from municipal wastewater treatment plants. Svenskt Vatten Utveckling. Report Nr 2014-12. URI: <https://res.slu.se/id/publ/60446>
- Ji, M., Wang, X., Usman, M., Liu, F., Dan, Y., Zhou, L., Campanaro, S., Luo, G. & Sang, W. (2022). Effects of different feedstocks-based biochar on soil remediation: A review. *Environmental Pollution*, 294, Article 118655. <https://doi.org/10.1016/j.envpol.2021.118655>
- Jin, J., Li, Y., Zhang, J., Wu, S., Cao, Y., Liang, P., Zhang, J., Wong, M., Wang, M., Shan, S., & Christie, P. (2016). Influence of pyrolysis temperature on properties and environmental safety of heavy metals in biochars derived from municipal sewage sludge. *Journal of Hazardous Materials*, 320, 417–426. <https://doi.org/10.1016/j.jhazmat.2016.08.050>
- Juhanson, J., Malmström, E., Pell, M., Hallin, S. (2021). Microbial soil tests for quality assessment of sewage sludge. Swedish University of Agricultural Sciences, Department of Forest Mycology and Plant Pathology. https://vattenbokhandeln.svensktvatten.se/wp-content/uploads/2021/04/C_Mikrobiologiska_marktester.pdf

Karaca, C., Sözen S., Orhon, D. & Okutan, H. (2018). High temperature pyrolysis of sewage sludge as a sustainable process for energy recovery. *Waste Management*, 78, 217-226. <https://doi.org/10.1016/j.wasman.2018.05.034>.

Khawer, M. U. B., Naqvi, S. R., Ali, I., Arshad, M., Juchelková, D., Anjum, M. W. & Naqvi, M. (2022). Anaerobic digestion of sewage sludge for biogas & biohydrogen production: State-of-the-art trends and prospects. *Fuel*, 329, Article 125416. <https://doi.org/10.1016/j.fuel.2022.125416>.

Kistler, R., C. and Widmer, F. (1987). Behavior of Chromium, Nickel, Copper, Zinc, Cadmium, Mercury, and Lead during the Pyrolysis of Sewage Sludge. *Environ. Sci. Technol.*, 21, 704-708. https://pubs.acs.org/doi/pdf/10.1021/es00161a012?ref=article_openPDF

Kleemann, R., Chenoweth, J., Clift, R., Morse, S., Pearce, P., & Saroj, D. (2017). Comparison of phosphorus recovery from incinerated sewage sludge ash (ISSA) and pyrolysed sewage sludge char (PSSC). *Waste Management*, 60, 201–210. <https://doi.org/10.1016/j.wasman.2016.10.055>

Kumar, A., Joseph, S., Tsechansky, L., Privat, K., Schreiter, I. J., Schüth, & Graber E. R. (2018). Biochar aging in contaminated soil promotes Zn immobilization due to changes in biochar surface structural and chemical properties. *Science of The Total Environment*, 626, 953-961. <https://doi.org/10.1016/j.scitotenv.2018.01.157>

Kärrman, A., Fredriksson, F., Särnholm, E. (2024). Slamspridning på åkermark – PFAS i slam, jord, gröda och mask. Svenskt Vatten Utveckling. Rapport Nr 2024-5. <https://vattenbokhandeln.svensktvatten.se/wp-content/uploads/2024/06/svu-rapport-2024-05.pdf>

Lazic, A. (2025). Val av första fullskaliga pyrolysanläggning vid ett avloppsvattenreningsverk i Sverige. *Tidskriften Vatten*, 4, 216-224. <https://www.tidskriftenvatten.se/tsv-artikel/val-av-forsta-fullskaliga-pyrolysanlaggning-vidett-avloppsvattenreningsverk-i-sverige/>

Lee, G., Jeong, W., Tsang, Y., & Baek, K. (2026). Stability assessment of heavy metals in sewage sludge pyrolysis biochar based on the chemical-enhanced primary treatment (CEPT) process. *Process Safety and Environmental Protection*, 206, Article 108338. <https://doi.org/10.1016/j.psep.2025.108338>

Lehmann, J., Pereira da Silva, J., Steiner, C., Nehls, T., Zech, W. & Glaser, B. (2003). Nutrient availability and leaching in an archaeological Anthrosol and a Ferralsol of the Central Amazon basin: fertilizer, manure and charcoal amendments. *Plant and Soil*, 249, 343–357. <https://doi.org/10.1023/A:1022833116184>

Li, B., Ding, S., Fan, H., & Ren, Y. (2021). Experimental Investigation into the Effect of Pyrolysis on Chemical Forms of Heavy Metals in Sewage Sludge Biochar (SSB), with Brief Ecological Risk Assessment. *Materials* (Basel), 14(2). <https://doi.org/10.3390/ma14020447>

Magnér, J., Rosenqvist, L., Rahmberg, M., Graae, L., Eliaeson, K., Örtlund L., Fång, J. & Brorström-Lundén, E. (2016). Fate of pharmaceutical residues - in sewage treatment and

on farmland fertilized with sludge. IVL Swedish Environmental Research Institute, No. B 2264. ISBN 978-91-88319-04-3. URN: urn:nbn:se:ivl:diva-2919

Majumder, R., Livesley, S. J., Gregory, D. & Arndt, S. K. (2014). Biosolid stockpiles are a significant point source for greenhouse gas emissions. *Journal of Environmental Management*, 143, 34-43. <https://doi.org/10.1016/j.jenvman.2014.04.016>.

Méndez, A., Gómez, A., Paz-Ferreiro, J. & Gascó, G. (2012). Effects of sewage sludge biochar on plant metal availability after application to a Mediterranean soil. *Chemosphere*, 89(11), 1354-1359. <https://doi.org/10.1016/j.chemosphere.2012.05.092>

Méndez, A., Terradillos, M., & Gascó, G. (2013). Physicochemical and agronomic properties of biochar from sewage sludge pyrolysed at different temperatures. *Journal of Analytical and Applied Pyrolysis*, 102, 124–130. <https://doi.org/10.1016/j.jaap.2013.03.006>

Mittskåne Vatten. (2023). ORMANÄS AVLOPPSRENINGSVVERK HÖÖRS KOMMUN. MILJÖRAPPORT 2022. Version 1. <https://www.mittskanevatten.se/wp-content/uploads/2024/03/miljorapport-ormanas-2022.pdf>

Mittskåne Vatten. (2024). ORMANÄS AVLOPPSRENINGSVVERK HÖÖRS KOMMUN. MILJÖRAPPORT 2023. Version 1. <https://www.mittskanevatten.se/wp-content/uploads/2025/10/mr-ormanas-2023-ver-1.pdf>

Mittskåne Vatten. (2025). ORMANÄS AVLOPPSRENINGSVVERK HÖÖRS KOMMUN. MILJÖRAPPORT 2024. Version 1. <https://www.mittskanevatten.se/wp-content/uploads/2025/10/miljorapport-2024.pdf>

Moško, J., Pohořelý, M., Cajthaml, T., Jeremiáš, M., Robles-Aguilar, A.A., Skoblia, S., Beňo, Z., Innemanová, P., Linhartova, L., Michalíková, K. & Meers, E. (2021). Effect of pyrolysis temperature on removal of organic pollutants present in anaerobically stabilized sewage sludge. *Chemosphere*, 265, Article 129082. <https://doi.org/10.1016/j.chemosphere.2020.129082>.

Naturvårdsverket. (2025). *Avloppsslam på jordbruksmark, användning, innehåll och spridningsplatser*. Last reviewed 1 September 2025. <https://www.naturvardsverket.se/data-och-statistik/avlopp/avloppsslam-pa-jordbruksmark-anvandning-innehall-och-spridningsplatser/> [2026-04-09]

NSVA. (2023). Miljörapport Öresundsverket 2023. Helsingborgs kommun. <https://nsva.se/wp-content/uploads/2024/06/miljorapport-2023-oresundsverket.pdf> [2026-04-14]

NSVA. (2024). Miljörapport Öresundsverket 2024. Helsingborgs kommun. <https://nsva.se/wp-content/uploads/2025/08/miljorapport-2024-oresundsverket.pdf> [2026-04-14]

NSVA. (2025). Miljörapport Öresundsverket 2025. Helsingborgs kommun. <https://nsva.se/wp-content/uploads/2026/04/miljorapport-2025-oresundsverket.pdf> [2026-04-14]

- Paz-Ferreiro, J., Nieto, A., Méndez, A., Askeland M. P. J. & Gascó, G. (2018). Biochar from Biosolids Pyrolysis: A Review. *International Journal of Environmental Research and Public Health*, 15(5), 956. <https://doi.org/10.3390/ijerph15050956>
- Pelagalli, V., Langone, M., Matassa, S., Race, M., Tuffi, R., Papirio, S., Lens, P., Lazzazzara, M., Frugis, A., Petta, L., & Esposito, G. (2024). Pyrolysis of municipal sewage sludge: challenges, opportunities and new valorization routes for biochar, bio-oil, and pyrolysis gas. *Environmental Science-Water Research & Technology*, 10(10), 2282–2312. <https://doi.org/10.1039/d4ew00278d>
- Radwaniecka, E., Chwialkowski, W., Mulkiewicz, E., & Kumirska, J. (2025). Impact of biochar from sewage sludge on vegetable growth and metal uptake: an example of radish and butterhead lettuce. *Nafta-Gaz*, 2, 117-127. <https://doi.org/10.18668/ng.2025.02.04>
- Revaq. (2025). *Årsrapport 2024*. Rapport R2025-03. <https://www.svensktvatten.se/globalassets/dokument/avlopp-och-miljo/revaq/revaq-arsrapport-2024.pdf> [2026-04-05]
- Revaq. (2026). *Regler för certifieringssystemet*. Utgåva 11.1. <https://www.svensktvatten.se/globalassets/dokument/avlopp-och-miljo/revaq/regler-2026-utgava-11.1-vit.pdf> [2026-04-05]
- Roslagsvatten. (2025). Först i Sverige med pyrolysis. Last updated 2025-11-14. <https://projektmarginetelund.roslagsvatten.se/content/forst-i-sverige-med-pyrolysis>. [2026-05-11]
- Salimbeni, A., Di Bianca, M., Lombardi, G., Rizzo, A. M., & Chiaramonti, D. (2023). Opportunities of Integrating Slow Pyrolysis and Chemical Leaching for Extraction of Critical Raw Materials from Sewage Sludge. *Water*, 15(6). <https://doi.org/10.3390/w15061060>
- Sanei, H., Rudra, A., Möller Moltesen Przyswitt, Z., Kousted, S., Benkhettab Sindlev, M., Zheng, X., Bom Nielsen, S. & Ingermann Petersen, H. (2024). Assessing biochar's permanence: An inertinite benchmark. *International Journal of Coal Geology*, 281, Article 104409. <https://doi.org/10.1016/j.coal.2023.104409>.
- SFS 1998:944. Förordning om förbud m.m. i vissa fall i samband med hantering, införsel och utförsel av kemiska produkter.
- Shong, W. J., Liu, C. K., Wu, S. H., Liu, H. C., Yang, P. (2014). Oxidation behavior of nickel coating on ferritic stainless steel interconnect for SOFC application. *International Journal of Hydrogen Energy*, 39(34), 19737-19746. <https://doi.org/10.1016/j.ijhydene.2014.09.138>
- SNFS 1994:2. Kungörelse med föreskrifter om skydd för miljön, särskilt marken, när avloppsslam används i jordbruket.
- Sousa, A. A. T. C., & Figueiredo, C. C. (2016). Sewage sludge biochar: effects on soil fertility and growth of radish. *Biological Agriculture & Horticulture*, 32(2), 127–138. <https://doi.org/10.1080/01448765.2015.1093545>

- Statistiska Centralbyrån. (2025). *Utsläpp till vatten och slamproduktion 2022: Kommunala avloppsreningsverk, massa- och pappersindustri samt viss övrig industri*. Sveriges Officiella Statistik, Statistiska meddelanden. MISAM BR2501. Urn:nbn:se:scb-2025-misambr2501_pdf
- Sweden Water Research. (2026). *Testbed Ellinge*.
<https://swedenwaterresearch.se/en/projekt/testbed-ellinge/> [2026-04-02]
- Thoma, E. D., Wright, R. S., George, I., Krause, M., Presezzi, D., Villa, V., Preston, W., Deshmukh, P., Kauppi, P. & Zemek, P. G. (2022). Pyrolysis processing of PFAS-impacted biosolids, a pilot study. *Journal of the Air & Waste Management Association*, 72(4), 309–318. <https://doi.org/10.1080/10962247.2021.2009935>
- Uppsala Vatten. (2022). MILJÖRAPPORT 2021 Kungsängsverket. Version 2.
https://www.uppsalavatten.se/download/18.50f686c4182e8f93c11239f/1663932328850/miljorapport-kungsangsverket-2021_v2.pdf
- Uppsala Vatten. (2023). MILJÖRAPPORT 2022 Kungsängsverket. Version 1.
<https://www.uppsalavatten.se/download/18.69711d1218b14705ff91361/1697012412386/Milj%C3%B6rapport%20Kungs%C3%A4ngsverket%202022.pdf>
- Uppsala Vatten. (2024). MILJÖRAPPORT 2023 Kungsängsverket. Version 1.
<https://www.uppsalavatten.se/download/18.5ee3b46d19271686b47dd39/1729522836080/Milj%C3%B6rapport%20Kungs%C3%A4ngsverket%202023.pdf>
- Vakin. (2023). Umeå Energi planerar Sveriges första fullskaliga anläggning för cirkulär återvinning av avloppsslam vid Dåva Industripark. Last updated 2023-03-31.
<https://www.vakin.se/omvakin/nyheter/nyheter/umeaenergiplanerarssverigesforstafullskaligaanlaggningforcirkularatervinningavavloppsslamviddavaindustripark.5.266ce28c1872be9d88f10857.html> [2026-05-11]
- Vali, N., Combres, A., Hosseinian, A., & Pettersson, A. (2023). The Effect of the Elemental Composition of Municipal Sewage Sludge on the Phosphorus Recycling during Pyrolysis, with a Focus on the Char Chemistry-Modeling and Experiments. *Separations*, 10(1), Article 31. <https://doi.org/10.3390/separations10010031>
- Vassilev, S. V., Vassileva, C. G., & Petrova, N. L. (2022). Thermal behaviour of biomass ashes in air and inert atmosphere with respect to their decarbonation. *Fuel*, 314.
<https://doi.org/10.1016/j.fuel.2021.122766>
- Wang, J. and Tomita, A. (2003). A Chemistry on the Volatility of Some Trace Elements during Coal Combustion and Pyrolysis. *Energy & Fuels*, 17, 954-960.
https://pubs.acs.org/doi/pdf/10.1021/ef020251o?ref=article_openPDF
- Wei, L., Wen, L., Yang, T., & Zhang, N. (2015). Nitrogen Transformation during Sewage Sludge Pyrolysis. *Energy & Fuels*, 29(8), 5088–5094.
<https://doi.org/10.1021/acs.energyfuels.5b00792>
- Willén, A., Jönsson, 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>

- Willén, A., Junestedt, C., Rodhe, L., Pell, M. & Jönsson, H. (2017). Sewage sludge as fertiliser – environmental assessment of storage and land application options. *Water Sci Technol*, 75(5): 1034–1050. <https://doi.org/10.2166/wst.2016.584>
- 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. <https://doi.org/10.1016/j.scitotenv.2016.06.081>.
- Yuan, H., Lu, T., Zhao, D. *et al.* (2013). Influence of temperature on product distribution and biochar properties by municipal sludge pyrolysis. *J Mater Cycles Waste Manag*, 15, 357–361. <https://doi.org/10.1007/s10163-013-0126-9>
- Yuan, H., Lu, T., Huang, H., Zhao, D., Kobayashi, N., & Chen, Y. (2015). Influence of pyrolysis temperature on physical and chemical properties of biochar made from sewage sludge. *Journal of Analytical and Applied Pyrolysis*, 112, 284–289. <https://doi.org/10.1016/j.jaap.2015.01.010>
- Yuan, H., Lu, T., Wang, Y., Chen, Y., & Lei, T. (2016). Sewage sludge biochar: Nutrient composition and its effect on the leaching of soil nutrients. *Geoderma*, 267, 17–23. <https://doi.org/10.1016/j.geoderma.2015.12.020>
- Zaragüeta, A., Enrique, A., Virto, I., Antón, R., Urmeneta, H. & Orcaray, L. (2021). Effect of the Long-Term Application of Sewage Sludge to A Calcareous Soil on Its Total and Bioavailable Content in Trace Elements, and Their Transfer to the Crop. *Minerals*, 11, 356. <https://doi.org/10.3390/min11040356>
- Zhou, C., Heal, K., Tigabu, M., Xia, L., Hu, H., Yin, D., & Ma, X. (2020). Biochar addition to forest plantation soil enhances phosphorus availability and soil bacterial community diversity. *Forest Ecology and Management*, 455, Article 117635. <https://doi.org/10.1016/j.foreco.2019.117635>
- Zhu, Y., Zhao, Q., Li, D., Li, J., & Guo, W. (2022). Performance comparison of phosphorus recovery from different sludges in sewage treatment plants through pyrolysis. *Journal of Cleaner Production*, 372. <https://doi.org/10.1016/j.jclepro.2022.133728>
- Zielińska, A., & Oleszczuk, P. (2015a). The conversion of sewage sludge into biochar reduces polycyclic aromatic hydrocarbon content and ecotoxicity but increases trace metal content. *Biomass and Bioenergy*, 75, 235–244. <https://doi.org/10.1016/j.biombioe.2015.02.019>
- Zielinska, A., Oleszczuk, P., Charnas, B., Skubiszewska-Zieba, J., & Pasieczna-Patkowska, S. (2015b). Effect of sewage sludge properties on the biochar characteristic. *Journal of Analytical and Applied Pyrolysis*, 112, 201–213. <https://doi.org/10.1016/j.jaap.2015.01.025>
- Österås, A. H., Allmyr, M. & Sternbeck, J. (2015). SCREENING OF ORGANIC POLLUTANTS IN SEWAGE SLUDGE AMENDED ARABLE SOILS. Naturvårdsverket. URN: urn:nbn:se:naturvardsverket:diva-6233

Appendix A

Analysis Methods

Table A1. Analysed parameters and measurement methods of the dewatered sewage sludge.

Parameter	Unit	Analysis method	Laboratory	Accreditation
Sample preparation crushing, grinding		SS-EN 16179:2012 mod.	Eurofins Biofuel & Energy Testing Sweden AB, SWEDEN	ISO/IEC 17025:2017 SWEDAC 1820
Moisture content	%	SS-EN 15934:2012 mod.		
Loss of ignition	% of DS	SS-EN 12879:2000	Eurofins Environment Testing Sweden AB, SWEDEN	ISO/IEC 17025:2017 SWEDAC 1125
pH		SS-EN ISO 10390:2022		
Ash content	% of DS	SS-EN 15403:2011	Eurofins Biofuel & Energy Testing Sweden AB, SWEDEN	ISO/IEC 17025:2017 SWEDAC 1820
C		SS-EN ISO 21663:2020	Eurofins Biofuel & Energy Testing Sweden AB, SWEDEN	ISO/IEC 17025:2017 SWEDAC 1820
H				
N (Kjeldahl+dewardas)	mg/kg	SS 28101:1992 mod.	Eurofins Food & Feed Testing Sweden (Lidköping), SWEDEN	ISO/IEC 17025:2017 SWEDAC 1977
O		ASTM D3176-2015; SS-EN 15400:2011; SS-EN ISO 18125:2017; SS-EN ISO 21654:2021	Eurofins Biofuel & Energy Testing Sweden AB, SWEDEN	ISO/IEC 17025:2017 SWEDAC 1820
Pb	mg/kg DS	SFS-EN ISO 16967:2015 mod.; SFS-EN ISO 16968:2015 mod.; SFS-EN ISO 17294-2:2023	Eurofins Ahma - Oulu, FINLAND	SFS-EN ISO/IEC 17025:2017 FINAS T131
Fe				
Cd				
Cu				
Cr				
Hg				
Ni				
Zn				
P		SFS-EN ISO 11885:2009; SFS-EN ISO 16967:2015 mod.; SFS-EN ISO 16968:2015 mod.		
K				
Ni				
Hg		SS-EN ISO 54321:2021 mod./SS-EN 16175-2:2016 mod.	Eurofins Environment Testing Sweden AB, SWEDEN	IS+C63:G940/IEC 17025:2017 SWEDAC 1125
Pb		SS 028150:1993/SS-EN ISO 17294-2:2023		
P		SS 028150:1993/SS-EN ISO 11885:2009.		
Fe				
Cd				
K				
Cu				
Cr				
Na				
Ni				
Zn				

Table A2. Analysed parameters and measurement methods of the sludge biochar.

Parameter	Unit	Analysis method	Laboratory	Accreditation
Sample preparation crushing, grinding		SS-EN 15002:2015-07	Eurofins Biofuel & Energy Testing Sweden AB, SWEDEN	ISO/IEC 17025:2017 SWEDAC 1820
Moisture content	%	SIS-CEN/TS 15414-2:2014		
Loss of ignition	% of DS	SS-EN 12879:2000	Eurofins Environment Testing Sweden AB, SWEDEN	ISO/IEC 17025:2017 SWEDAC 1125
pH		SS-EN ISO 10390:2022		
Ash content	% of DS	SS-EN 15403:2011	Eurofins Biofuel & Energy Testing Sweden AB, SWEDEN	ISO/IEC 17025:2017 SWEDAC 1820
C		SS-EN ISO 21663:2020		
H				
N				
O				
TOC/TIC	% of DS	SS-EN 15936:2022 metodappl. A / fd SS-EN 13137:2001 metodappl. A		
Pb	mg/kg DS	SFS-EN ISO 16967:2015 mod.; SFS-EN ISO 16968:2015 mod.; SFS-EN ISO 17294-2:2023	Eurofins Ahma - Oulu, FINLAND	SFS-EN ISO/IEC 17025:2017 FINAS T131
Fe				
Cd				
Cu				
Cr				
Hg				
Ni				
Zn				
P		SFS-EN ISO 11885:2009; SFS-EN ISO 16967:2015 mod.; SFS-EN ISO 16968:2015 mod.		
K				

Table A3. Analysed parameters and measurement methods of the dryer condensate, scrubber water and drinking water.

Parameter	Unit	Analysis method	Laboratory	Accreditation
SS	mg/l	SS-EN 872:2005	Eurofins Water Testing Sweden, SWEDEN	ISO/IEC 17025:2017 SWEDAC 10300
VSS		fd SS 028112:1983		
pH	SS-EN ISO 10523:2012			
T at pH measurement		°C		
Alkalinitet	mmol/l	SS EN ISO 9963-2:1996 mod		
TOC	mg/l	SS EN 1484:1997 mod.		
COD-Cr		ISO 15705:2002		
P		SS-EN ISO 15681-2:2018		
N		ISO 29441:2010		
Fe		SS-EN ISO 15587-2:2002/SS-EN ISO 17294-2:2023	Eurofins Environment Testing Sweden AB, SWEDEN	ISO/IEC 17025:2017 SWEDAC 1125
Pb				
Cd				
Cu				
Cr				
Zn				
Ni				
Hg				

Appendix B

Sludge and Biochar Contents

Table B1. Sludge and biochar characteristics and carbon contents.

Sludge pretreatment	Sewage sludge	Pyrolysis temperature °C	Dry solids (DS) %	Ash % of DS	pH	Biochar yield %	C	
							Retention %	Content % of DS
Undigested	Lidköping		21	21	7.3			46
		650	89±5	58±1	7.8±0.4	36±1	28±1	36±1
		700	90±0.1	59±0.1	7.7±0.1	35±0.1	28±0.5	36±0.6
		750	89	60	7.8	35	27	36
Undigested	Lycksele		22	27	7.4			42
		600	92	58	8.4	47	35	31
		650	89	62	8.5	44	34	32
		700	91	66	8.3	42	31	31
		800	85	78	7.9	35	24	28
Undigested	Margretelund		21±2	19±0	7.0±0.8			48±1
		600	87	56	7.6	34	26	38
		650	90±1	54±0	8±0.4	35±0	29±1	41±0
		700	88±1	61±4	8.2±0.4	31±2	24±4	37±5
		750	83	61	7.8	31	26	40
		800	88	62	7.8	31	25	41
Undigested	Ormanäs		16	21	6.3			48
		600	82	56	7.8	37	25	33
		650	82	57	7.7	36	28	36
		700	83	69	7.5	30	12	19
		750	77	61	7.9	34	25	35
		800	78	58	7.5	36	29	39
Anaerobically digested	Duvbacken		19	31	8.3			38
		600	96	71	7.4	44	32	28
		650	93	68	8.3	46	37	30
		700	94	67	8.2	46	37	31
		750	92	70	8.0	44	35	30
		800	91	73	7.8	42	33	30

Sludge pretreatment	Sewage sludge	Pyrolysis temperature °C	Dry solids (DS) %	Ash % of DS	pH	Biochar yield %	C	
							Retention	Content
							%	% of DS
Anaerobically digested	Ellinge		16±1	28±2	8.4±0.1			39±1
		600	87	61	8.5	43	36	35
		650	86	62	8.5	42	35	34
		700	81	65	7.5	42	36	34
		800	87	66	7.8	42	38	36
Anaerobically digested	Henriksdal		27	37	8.4			35
		600	93	72	6.8	52	42	28
		650	95	70	8.0	53	43	28
		700	90	72	7.8	52	43	29
		750	96	78	8.9	48	37	27
		800	86	82	7.8	45	34	26
Anaerobically digested	Himmerfjärdsverket		23	35	8.1			36
		600	94	70	7.8	50	40	29
		700	90	73	8.0	48	38	29
		750	91	75	8.3	46	36	28
		800	90	76	8.3	45	35	28
Anaerobically digested	Hässelholm		26	39	6.8			34
		600	96	72	7.7	54	44	28
		650	93	72	8.9	53	44	28
		700	90	74	8.0	52	39	26
		750	90	77	8.6	50	38	26
		800	93	80	8.3	48	37	27
Anaerobically digested	Klagshamn		22±1	38±1	8.1±0.8			34±0
		600	95±2	73±0	7.3±0.2	53±1	41±3	27±1
		650	95	75	7.9	52	41	27
		700	95±1	76±0	8±0.3	50±1	38±2	26±0
		750	93	79	7.9	49	38	27
		800	90	84	7.6	46	34	25
Anaerobically digested	Kungsängsverket		24	33	8.3			37
		600	93	71	8.0	47	39	30
		650	94	71	7.5	47	38	30
		700	92	76	7.4	44	35	30
		750	91	76	7.4	44	31	26

Sludge pretreatment	Sewage sludge	Pyrolysis temperature °C	Dry solids (DS) %	Ash % of DS	pH	Biochar yield %	C	
							Retention %	Content % of DS
Anaerobically digested	Käppalaverket		24	37	8.5			36
		600	93	70	7.7	53	42	28
		650	86	71	7.7	52	41	28
		700	90±1	74±0	8.1±0.2	50±0	39±0	28±0
		750	90	78	7.9	47	36	27
		800	94	79	7.9	47	35	27
Anaerobically digested	Skebäcksverket		27	42	7.6			31
		600	96	73	8.4	58	46	24
		650	97	75	8.5	57	44	24
		700	94	76	8.3	56	44	24
		800	90	83	7.7	51	40	24
Anaerobically digested	Uddebo		17	36	8.3			35
		600	93	70	7.5	52	43	29
		650	93	70	7.5	52	42	28
		800	86	75	7.5	48	39	28
Anaerobically digested	Öresundsverket		23	43	8.1			29
		600	92	67	8.2	64	64	29
		650	93	70	8.3	61	56	27
		700	91	72	8.1	60	53	26
		750	89	73	8.0	59	56	28
		800	92	75	7.7	57	54	28

Table B2. Nutrient content of sludge and biochar, including retention. DS = Dry solids.

Sludge pretreatment	WWTP	Pyrolysis temperature °C	K		N		P	
			Retention	Content	Retention	Content	Retention	Content
			%	% of DS	%	% of DS	%	% of DS
Undigested	Lidköping	DSS		0.27		1.1		1.9
		650	114±20	0.87±0.2	24±3	2.5±0.3	104±1	5.5±0.1
		700	105±7	0.81±0.05	18±1.9	1.9±0.2	103±3	5.6±0.1
		750	100	0.79	14	1.5	107	5.9
Undigested	Lycksele	DSS		0.41		0.8		1.2
		600	89	0.78	35	2.4	121	3.1
		650	94	0.87	28	2.0	118	3.2
		700	93	0.92	19	1.5	114	3.3
		800	70	0.81	11	1.0	156	5.3
Undigested	Margretelund	DSS		0.19±0.03		1.1±0		2.1±0.1
		600	178	0.89	29	3.6	105	6.2
		650	111±8	0.61±0.05	26±1	3.1±0.1	97±9	5.7±0.3
		700	102±7	0.63±0.1	14±2	1.9±0.1	93±0.5	6.2±0.7
		750	130	0.70	14	1.8	102	6.5
		800	113	0.63	11	1.5	107	7.0
Undigested	Ormanäs	DSS		0.31		1.0		2.4
		600	107	0.89	18	2.9	110	7.1
		650	103	0.88	21	3.5	108	7.1
		700	86	0.88	7	1.4	99	7.8
		750	103	0.93	13	2.2	99	6.9
		800	112	0.97	13	2.2	103	6.9
Anaerobically digested	Duvbacken	DSS		0.34		1.2		4.1
		600	53	0.41	23	2.8	76	7.1
		650	95	0.70	22	2.6	93	8.3
		700	97	0.72	21	2.3	95	8.5
		750	92	0.71	15	1.8	93	8.6
		800	96	0.77	11	1.4	96	9.3
Anaerobically digested	Ellinge	DSS		0.85±0.05		1.2±0.1		3±0.3
		600	98	1.9	24	3.9	99	6.5
		650	95	1.9	20	3.3	96	6.5
		700	109	2.0	17	2.6	102	6.7
		800	583*	1.4	10	1.5	100	7.2

(*Not included in analysis due to an unrepresentatively significant deviation from batches of the same WWTP, likely due to the heterogeneous character of the sludge.)

Sludge pretreatment	WWTP	Pyrolysis temperature °C	K		N		P	
			Retention	Content	Retention	Content	Retention	Content
			%	% of DS	%	% of DS	%	% of DS
Anaerobically digested	Henriksdal	DSS		0.35		1.6		3.3
		600	101	0.68	27	2.5	90	5.7
		650	109	0.72	26	2.4	85	5.3
		700	128	0.86	14	1.3	77	4.9
		750	101	0.74	13	1.3	91	6.3
		800	100	0.77	10	1.0	88	6.4
Anaerobically digested	Himmerfjärdsverket	DSS		0.33		1.0		3.8
		600	89	0.59	26	2.6	88	6.7
		700	88	0.61	15	1.5	88	7.0
		750	89	0.64	12	1.4	87	7.2
		800	96	0.70	10	1.1	91	7.6
Anaerobically digested	Hässleholm	DSS		0.25		1.3		4.0
		600	98	0.46	22	2.0	88	6.6
		650	105	0.49	22	2.0	92	6.9
		700	107	0.51	16	1.5	90	6.9
		750	95	0.47	13	1.3	87	6.9
		800	103	0.53	10	1.0	98	8.1
Anaerobically digested	Klagshamn	DSS		0.19±0		1.2±0.1		3.9±0.1
		600	93±3	0.34±0.01	30±1	2.6±0.2	94±5	6.9±0.1
		650	107	0.39	25	2.1	101	7.4
		700	105±15	0.4±0.05	18±2	1.6±0.1	95±8	7.2±0.4
		750	109	0.42	13	1.2	105	8.1
		800	104	0.43	9	0.8	99	8.2
Anaerobically digested	Kungsängsverket	DSS		0.21		1.4		3.8
		600	108	0.48	23	2.5	97	7.8
		650	107	0.48	23	2.5	88	7.1
		700	105	0.50	11	1.3	97	8.4
		750	129	0.62	10	1.2	96	8.4
Anaerobically digested	Käppalaverket	DSS		0.29		1.4		3.8
		600	99	0.54	31	2.9	104	7.5
		650	100	0.56	24	2.3	106	7.8
		700	84±31	0.49±0.18	16±0.01	1.6±0.01	88±29	6.6±2.2
		750	103	0.63	11	1.1	103	8.3
		800	108	0.67	9	0.9	104	8.5

Sludge pretreatment	WWTP	Pyrolysis temperature °C	K		N		P	
			Retention	Content	Retention	Content	Retention	Content
			%	% of DS	%	% of DS	%	% of DS
Anaerobically digested	Skebäcksverket	DSS		0.61		1.1		2.3
		600	91	0.95	30	2.0	99	3.9
		650	89	0.95	25	1.7	97	3.9
		700	92	1.0	19	1.3	117	4.8
		800	80	0.95	10	0.7	120	5.4
Anaerobically digested	Uddebo	DSS		0.38		0.9		3.1
		600	98	0.71	22	1.9	101	6.0
		650	99	0.72	22	1.9	101	6.0
		800	104	0.82	10	0.9	106	6.8
Anaerobically digested	Öresundsverket	DSS		0.61		1.5		6.0
		600	115	1.1	32	3.0	88	8.3
		650	99	1.0	25	2.5	97	9.6
		700	98	1.0	20	2.1	109	11
		750	106	1.1	14	1.5	98	10
		800	102	1.1	11	1.2	92	9.8

Table B3. Heavy metal content of sludge and biochar, including retention and metal/P ratios. DS = Dry solids.

Sludge pretreatment	WWTP	Pyrolysis temperature	Al		Cd		Cr			Cu			Fe		
			Retention	Content	Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio	Retention	Content
			%	g/kg DS	%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P	%	g/kg DS
Undigested	Lidköping	DSS		28		0.55	29		21	1 100		160	8 400		6.8
		650	101±20	79±10	47±59*	0.74±0.9*	13±17*	141±9	83±4	1 500±80	97±7	430±20	7 800±480	169±40	32±9
		700	110±6	87±5	5.4±0.2	0.085±0.003	1.5±0.1	130±10	77±7.8	1 400±100	104±1	470±7	8 500±90	134±0.2	26±0
		750	116	94	4.4	0.070	1.2	150	91	1 500	108	500	8 500	168	33
Undigested	Lycksele	DSS		45		0.49	41		49	4 100		170	14 000		9.1
		600	100	96	64	0.67	22	63	66	2 100	105	380	12 000	118	23
		650	84	85	9.0	0.10	3.1	61	68	2 100	112	430	13 000	122	25
		700	92	100	3.4	0.040	1.2	54	64	1 900	73	300	9 100	101	22
		800	45	57	4.0	0.055	1.0	62	86	1 600	133	640	12 000	542	140
Undigested	Margretelund	DSS		16±1		0.29±0.03	14±2		12±1	560±20		200±20	9 500±700		33±4
		600	88	39	62	0.57	9.2	151	49	790	89	470	7 600	100	88
		650	103±8	46±1	8.6±2.1	0.071±0.01	1.3±0.3	125±30	41±8	720±110	104±8	580±20	10 000±900	102±8	95±4
		700	88±8	44±1	12±5	0.11±0.04	1.7±0.5	174±20	64±1	1 000±110	87±7	550±60	8 900±100	97±5	100±20
		750	107	51	6.7	0.066	1.0	166	58	890	107	610	9 400	105	100
		800	114	56	5.1	0.052	0.7	122	44	630	110	650	9 300	112	110
Undigested	Ormanäs	DSS		31		0.82	34		11	460		630	26 000		5.2
		600	83	69	59	1.3	18.0	163	48	680	95	1 600	23 000	136	19
		650	79	67	18	0.40	5.6	165	50	700	92	1 600	23 000	217	31
		700	54	55	63*	1.7*	22*	387	140	1 800	63	1 300	17 000	497	85
		750	96	87	5.4	0.13	1.9	122	39	570	109	2 000	29 000	99	15
		800	104	90	3.4	0.080	1.1	98	30	430	108	1 900	28 000	104	15

(*Not included in analysis due to an unrepresentatively significant deviation from batches of the same WWTP, likely due to the heterogeneous character of the sludge.)

Sludge pretreatment	WWTP	Pyrolysis temperature	Al		Cd			Cr			Cu			Fe	
			Retention	Content	Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio	Retention	Content
			°C	%	g/kg DS	%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P	%
Anaerobically digested	Duvbacken	DSS		8		1.1	27		33	800		930	23 000		45
		600	72	13	72	1.8	25	68	51	720	56	1 200	17 000	185	190
		650	99	17	37	0.89	11	111	80	960	94	1 900	23 000	87	85
		700	99	17	14	0.34	4.0	96	69	810	94	1 900	22 000	102	100
		750	100	18	5.2	0.13	1.5	92	69	800	95	2 000	23 000	118	120
		800	102	19	5.8	0.15	1.6	88	69	740	96	2 100	23 000	103	110
Anaerobically digested	Ellinge	DSS		13±2		0.84±0.1	28±3		26±2	840±30		110±10	3 500±300		34±9
		600	110	31	48	0.95	15	109	64	980	97	250	3 800	113	69
		650	107	31	15	0.30	4.6	103	62	950	94	250	3 800	112	70
		700	105	32	3.4	0.059	0.88	113	61	910	113	250	3 700	113	72
		800	138	43	1.8	0.040	0.56	87	50	690	206	470	6 500	144	100
Anaerobically digested	Henriksdal	DSS		12		0.59	18		25	760		360	11 000		86
		600	139	32	42	0.48	8.4	187	90	1 600	94	650	11 000	103	170
		650	194	44	29	0.32	6.0	186	88	1 700	88	600	11 000	86	140
		700	282	65	4.3	0.049	1.0	270	130	2 700	87	600	12 000	67	110
		750	124	31	3.2	0.040	0.6	140	73	1 200	97	730	12 000	100	180
		800	132	35	3.1	0.040	0.6	163	90	1 400	87	690	11 000	95	180
Anaerobically digested	Himmerfjärdsverket	DSS		14		0.61	16		26	680		370	9 700		78
		600	96	27	26	0.32	4.8	107	56	840	88	650	9 700	96	150
		700	96	28	3.1	0.040	0.6	132	72	1 000	89	690	9 900	98	160
		750	89	27	5.2	0.069	1.0	112	63	870	82	660	9 200	89	150
		800	104	32	6.9	0.092	1.2	119	68	890	91	740	9 700	87	150

Sludge pretreatment	WWTP	Pyrolysis temperature	Al		Cd			Cr			Cu			Fe	
			Retention	Content	Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio	Retention	Content
		°C	%	g/kg DS	%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P	%	g/kg DS
Anaerobically digested	Hässleholm	DSS		9.1		0.85	21		20	500		350	8 800		82
		600	112	19	63	1.0	15	136	51	770	118	770	12 000	117	180
		650	123	21	14	0.23	3.3	176	66	960	128	840	12 000	130	200
		700	150	26	4.4	0.071	1.0	157	60	870	139	930	13 000	115	180
		750	127	23	4.4	0.074	1.1	153	61	880	125	870	13 000	117	190
		800	112	21	2.3	0.040	0.5	155	64	790	119	860	11 000	124	210
Anaerobically digested	Klagshamn	DSS		5±1		1.1±0.4	29±10		32±6	830±160		550±30	14 000±1 000		93±9
		600	99±15	10±0	54±8	1.1±0.3	16±4	275±150	170±120	2 500±1 700	105±4	1 100±0	16 000±330	106±5	180±7
		650	87	10	14	0.22	3.0	71	49	660	91	1 000	14 000	100	190
		700	103±1	11±1	8.6±5	0.21±0.2	3±2.8	97±36	60±13	830±220	101±4	1 100±0	15 000±740	103±1	190±14
		750	90	11	2.4	0.040	0.49	70	51	630	95	1 100	14 000	99	200
		800	84	11	9.1	0.16	2.0	73	57	700	89	1 100	13 000	102	220
Anaerobically digested	Kungsängsverket	DSS		16		0.42	11		27	710		300	7 900		66
		600	100	34	32	0.29	3.7	103	59	760	97	620	7 900	100	140
		650	100	34	23	0.21	3.0	101	58	820	94	600	8 500	100	140
		700	105	38	4.6	0.044	0.50	121	74	880	91	620	7 400	100	150
		750	106	39	4.1	0.040	0.48	95	59	700	94	650	7 700	99	150

Sludge pretreatment	WWTP	Pyrolysis temperature	Al		Cd			Cr			Cu			Fe	
			Retention	Content	Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio	Retention	Content
			°C	%	g/kg DS	%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P	%
Anaerobically digested	Käppalaverket	DSS		10		0.71	19		24	630		420	11 000		67
		600	116	22	65	0.87	12	106	48	640	113	900	12 000	111	140
		650	124	24	20	0.28	3.6	108	50	640	-1	970	12 000	116	150
		700	106±22	21±4	3.5±0.4	0.05±0.01	0.76±0.17	100±2	47±1	750±240	111±11	920±92	15 000±6 300	86±27	110±40
		750	128	27	2.7	0.040	0.48	106	54	650	96	850	10 000	106	150
		800	107	23	3.0	0.046	0.54	126	65	760	109	980	12 000	111	160
Anaerobically digested	Skebäckverket	DSS		42		0.46	20		24	1 000		200	8 700		55
		600	107	77	63	0.50	13	172	71	1 800	111	380	9 700	100	94
		650	107	79	16	0.13	3.3	164	69	1 800	111	390	10 000	100	97
		700	104	78	7.7	0.063	1.3	208	89	1 900	123	440	9 200	112	110
		800	89	73	8.0	0.072	1.3	162	76	1 400	139	540	10 000	121	130
Anaerobically digested	Uddebo	DSS		47		0.68	22		27	870		260	8 400		22
		600	99	89	19	0.25	4.2	104	54	900	98	490	8 200	116	49
		650	100	90	62	0.81	13	128	66	1 100	104	520	8 700	119	50
		800	102	99	10	0.14	2.1	113	63	930	106	570	8 400	132	60
Anaerobically digested	Öresundsverket	DSS		11		0.50	8.0		43	720		320	5 300		21
		600	128	22	150	1.2	14	123	83	1 000	136	680	8 200	134	44
		650	110	20	110	0.90	9.4	109	77	800	121	640	6 700	124	43
		700	114	21	68	0.57	5.2	91	66	600	97	520	4 700	119	42
		750	128	24	23	0.20	2.0	102	75	750	110	600	6 000	168	60
		800	139	27	12	0.11	1.1	122	93	950	122	690	7 000	216	80

Table B4. Heavy metal content of sludge and biochar, including retention and metal/P ratios (continued). DS = Dry solids, LOD = Limit of detection.

Sludge pretreatment	WWTP	Pyrolysis temperature	Hg			Ni			Pb			Zn		
			Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio
		°C	%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P
Undigested	Lidköping	DSS		0.12	6.3		10	530		8.1	426		240	13 000
		650	<LOD	<LOD	<LOD	135±7	38±3	680±40	121±12	27±2	500±40	103±1	690±7	13 000±30
		700	<LOD	<LOD	<LOD	130±0.2	37±0	660±20	115±20	26±4	470±50	102±2	700±10	13 000±60
		750	<LOD	<LOD	<LOD	149	43	730	111	26	440	98	680	12 000
Undigested	Lycksele	DSS		0.39	33		18	1 500		14	1 200		380	32 000
		600	<LOD	<LOD	<LOD	99	38	1 200	104	31	1 000	96	780	25 000
		650	<LOD	<LOD	<LOD	101	41	1 300	104	33	1 000	93	800	25 000
		700	<LOD	<LOD	<LOD	88	38	1 200	98	33	1 000	78	710	22 000
		800	<LOD	<LOD	<LOD	129	66	1 200	101	40	750	93	1 000	19 000
Undigested	Margretelund	DSS		0.13±0.02	6.1±1		6.7±0.2	320±1		3.8±0	190±20		230±14	11 000±300
		600	<LOD	<LOD	<LOD	151	29	470	119	14	230	99	640	10 000
		650	<LOD	<LOD	<LOD	128±8	24±1	430±9	105±1	11±1	202±2	104±10	680±20	12 000±220
		700	<LOD	<LOD	<LOD	185±7	40±3	650±30	114±0.01	14±0	227±30	96±10	720±180	12 000±1500
		750	<LOD	<LOD	<LOD	194	40	620	102	13	200	99	690	11 000
		800	<LOD	<LOD	<LOD	136	29	410	92	12	170	96	690	9 900
Undigested	Ormanäs	DSS		0.25	10		10	400		12	500		500	21 000
		600	<LOD	<LOD	<LOD	142	37	520	103	33	460	97	1 300	18 000
		650	<LOD	<LOD	<LOD	143	38	540	100	33	460	95	1 300	18 000
		700	<LOD	<LOD	<LOD	502	160	2 100	99	39	500	120	1 900	24 000
		750	<LOD	<LOD	<LOD	124	35	510	94	33	480	96	1 400	20 000
		800	<LOD	<LOD	<LOD	126	34	490	84	28	410	93	1 300	19 000

Sludge pretreatment	WWTP	Pyrolysis temperature °C	Hg			Ni			Pb			Zn		
			Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio
			%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P
Anaerobically digested	Duvbacken	DSS		0.32	7.8		21	510		27	660		750	18 000
		600	<LOD	<LOD	<LOD	83	40	560	50	31	440	64	1 100	15 000
		650	<LOD	<LOD	<LOD	114	52	630	97	57	690	92	1 500	18 000
		700	<LOD	<LOD	<LOD	103	47	550	95	56	660	92	1 500	18 000
		750	<LOD	<LOD	<LOD	105	50	580	93	57	660	94	1 600	19 000
		800	<LOD	<LOD	<LOD	113	56	600	88	56	600	96	1 700	18 000
Anaerobically digested	Ellinge	DSS		0.090±0.02	3.0±1		13±1	430±20		8.8±1	290±48		300±18	10 000±700
		600	<LOD	<LOD	<LOD	125	38	580	100	21	320	100	700	11 000
		650	<LOD	<LOD	<LOD	118	37	570	97	21	320	100	720	11 000
		700	<LOD	<LOD	<LOD	131	37	550	107	23	340	100	720	11 000
		800	<LOD	<LOD	<LOD	135	39	540	62	15	210	89	600	8 300
Anaerobically digested	Henriksdal	DSS		0.33	10		24	730		15	450		520	16 000
		600	<LOD	<LOD	<LOD	149	69	1 200	100	29	510	98	980	17 000
		650	<LOD	<LOD	<LOD	139	63	1 200	109	31	580	99	970	18 000
		700	<LOD	<LOD	<LOD	180	83	1 700	114	33	670	100	1 000	20 000
		750	<LOD	<LOD	<LOD	124	62	980	99	31	490	100	1 100	17 000
		800	<LOD	<LOD	<LOD	130	69	1 100	103	34	530	96	1 100	17 000
Anaerobically digested	Himmerfjärdsverket	DSS		0.25	6.6		22	580		17	450		500	13 000
		600	<LOD	<LOD	<LOD	104	46	690	85	29	430	88	880	13 000
		700	<LOD	<LOD	<LOD	124	57	810	90	32	460	87	910	13 000
		750	<LOD	<LOD	<LOD	98	47	650	89	33	460	86	930	13 000
		800	<LOD	<LOD	<LOD	118	57	750	88	33	430	84	920	12 000

Sludge pretreatment	WWTP	Pyrolysis temperature °C	Hg			Ni			Pb			Zn		
			Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio
			%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P
Anaerobically digested	Hässleholm	DSS		1.0	13		16	400		15	380		500	13 000
		600	<LOD	<LOD	<LOD	127	38	580	100	28	420	120	1 100	17 000
		650	<LOD	<LOD	<LOD	160	48	700	107	30	430	140	1 300	19 000
		700	<LOD	<LOD	<LOD	154	47	680	105	30	430	130	1 200	17 000
		750	<LOD	<LOD	<LOD	142	45	650	104	31	450	130	1 300	19 000
		800	<LOD	<LOD	<LOD	145	48	590	100	31	380	120	1 200	15 000
Anaerobically digested	Klagshamn	DSS		0.34±0.17	8.9±5		23±3	600±80		13±2	330±61		540±42	14 000±1 400
		600	<LOD	<LOD	<LOD	259±140	120±80	1 700±1 100	109±10	25±1	370±3	100±10	1 000±70	15 000±1 300
		650	<LOD	<LOD	<LOD	79	38	510	93	25	340	100	1 100	15 000
		700	<LOD	<LOD	<LOD	105±30	47±10	650±170	102±20	25±0	350±20	100±7	1 100±0	15 000±740
		750	<LOD	<LOD	<LOD	77	39	480	88	25	310	95	1 100	14 000
		800	<LOD	<LOD	<LOD	92	50	610	89	27	330	89	1 100	13 000
Anaerobically digested	Kungsängsverket	DSS		0.64	17		14	370		15	390		360	9 500
		600	<LOD	<LOD	<LOD	101	30	380	75	24	310	100	780	10 000
		650	<LOD	<LOD	<LOD	104	31	440	63	20	280	100	770	11 000
		700	<LOD	<LOD	<LOD	145	46	550	59	20	240	120	950	11 000
		750	<LOD	<LOD	<LOD	106	34	400	61	21	250	94	780	9 300

Sludge pretreatment	WWTP	Pyrolysis temperature °C	Hg			Ni			Pb			Zn		
			Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio	Retention	Content	Metal/P ratio
			%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P	%	mg/kg DS	mg metal/ kg P
Anaerobically digested	Käppalaverket	DSS		0.44	12		21	550		14	370		540	14 000
		600	<LOD	<LOD	<LOD	113	45	600	106	28	370	110	1 100	15 000
		650	<LOD	<LOD	<LOD	118	48	620	104	28	360	120	1 200	15 000
		700	<LOD	<LOD	<LOD	111±10	46±6	760±340	95±7	26±2	430±170	100±10	1 100±163	18 000±8 300
		750	<LOD	<LOD	<LOD	97	43	520	81	24	290	83	950	11 000
		800	<LOD	<LOD	<LOD	115	52	610	126	38	450	100	1 200	14 000
Anaerobically digested	Skebäcksverket	DSS		0.28	12		15	650		18	780		330	14 000
		600	<LOD	<LOD	<LOD	159	41	1 100	100	31	790	110	640	16 000
		650	<LOD	<LOD	<LOD	156	41	1 100	104	33	850	110	660	17 000
		700	<LOD	<LOD	<LOD	205	55	1 100	103	33	690	130	780	16 000
		800	<LOD	<LOD	<LOD	157	46	850	97	34	630	130	860	16 000
Anaerobically digested	Uddebo	DSS		0.31	10		14	450		11	350		490	16 000
		600	<LOD	<LOD	<LOD	119	32	530	109	23	380	98	920	15 000
		650	<LOD	<LOD	<LOD	134	36	600	114	24	400	100	950	16 000
		800	<LOD	<LOD	<LOD	131	38	560	92	21	310	89	900	13 000
Anaerobically digested	Öresundsverket	DSS		0.94	16		22	370		16	270		470	7 800
		600	<LOD	<LOD	<LOD	142	49	590	132	33	400	140	1 000	12 000
		650	<LOD	<LOD	<LOD	130	47	490	118	31	320	130	970	10 000
		700	<LOD	<LOD	<LOD	103	38	350	112	30	270	110	870	7 900
		750	<LOD	<LOD	<LOD	115	43	430	121	33	330	120	950	9 500
		800	<LOD	<LOD	<LOD	141	55	560	110	31	320	110	940	9 600

Table B5. Statistical information regarding Figure 4.9. Means and standard deviations are calculated using data from Table B2.

Pyrolysis temperature	600 °C	650 °C	700 °C	750 °C	800 °C
Mean retention					
K	101±25	103±9.5	100±14	105±13	99±13
N	27±4.7	24±2.3	16±3.4	13±1.4	10±1.2
P	97±11	99±8.4	97±12	97±7.0	105±17
Mean nutrient contents					
K	0.74±0.4	0.74±0.4	0.76±0.4	0.70±0.2	0.81±0.3
N	2.7±0.6	2.5±0.5	1.7±0.4	1.5±0.3	1.2±0.4
P	6.4±1.4	6.3±1.6	6.7±1.8	7.6±1.2	7.4±1.4

Appendix C

Mass Flows

Table C1. Mass flows of carbon. C_{total} includes mass flows from anaerobic digestion. UD = Undigested, AD = Anaerobically digested, BC = Sludge biochar, DC = Dryer condensate, SW = Scrubber water, SG = Scrubbed flue gas, BG = Biogas.

Sludge pretreatment	WWTP	Pyrolysis temperature °C	C					C_{total}			
			BC	DC	SW	SG	BG	BC	DC	SW	SG
			% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS
UD	Margretelund	600	26	3.4	0.10	70	0	26	3.4	0.10	70
UD	Ormanäs	600	25	0.071	13	62	0	25	0.071	13	62
AD	Duvbacken	600	32	0.24	0.092	67	41	19	0.14	0.054	40
AD	Henriksdal	600	42	1.9	0.14	56	48	22	1.0	0.073	29
AD	Himmerfjärdsverket	600	40	0.33	0.14	59	69	12	0.10	0.045	18
AD	Hässleholm	600	44	0.25	0.11	56	50	22	0.13	0.055	28
AD	Kungsängsverket	600	39	0.99	0.16	60	50	19	0.50	0.080	30
AD	Käppalaverket	600	42	0.33	0.14	58	57	18	0.14	0.060	25
AD	Skebäcksverket	600	46	0.61	0.14	53	57	20	0.26	0.063	23
AD	Uddebo	600	43	0.29	0.17	57	44	24	0.16	0.094	32
AD	Öresundsverket	600	64	0.32	0.25	35	-	-	-	-	-
UD	Lycksele	650	34	3.1	0.19	63	0	34	3.11	0.19	63
UD	Margretelund	650	29±0.5	4.3±0.4	0.099±0.01	66±0.2	0	29±0.5	4.3±0.4	0.10±0.01	66±0.2
UD	Ormanäs	650	28	6.4	0.16	66	0	28	6.4	0.16	66
AD	Duvbacken	650	37	0.29	0.14	63	41	22	0.17	0.083	37
AD	Henriksdal	650	43	1.2	0.12	56	48	22	0.62	0.065	29
AD	Hässleholm	650	44	0.26	0.12	56	50	22	0.13	0.06	28
AD	Kungsängsverket	650	38	0.64	0.1	62	50	19	0.32	0.05	31
AD	Käppalaverket	650	41	0.42	0.13	58	57	18	0.18	0.057	25
AD	Skebäcksverket	650	44	0.54	0.14	55	57	19	0.24	0.059	24
AD	Uddebo	650	42	0.37	0.21	58	44	23	0.21	0.12	32
AD	Öresundsverket	650	56	0.56	0.2	43	-	-	-	-	-
UD	Lycksele	700	31	2.2	0.12	67	0	31	2.2	0.12	67
UD	Margretelund	700	24±4.1	3.6±0.8	0.088±0.01	72±4.9	0	24±4.1	3.6±0.8	0.088±0.01	72±4.9
UD	Ormanäs	700	12	4.6	0.12	83	0	12	4.6	0.12	83
AD	Duvbacken	700	37	0.35	0.13	62	41	22	0.21	0.079	36
AD	Henriksdal	700	43	2.0	0.18	55	48	23	1.1	0.10	29
AD	Himmerfjärdsverket	700	38	0.43	0.15	61	69	12	0.13	0.048	19
AD	Hässleholm	700	39	0.35	0.13	60	50	20	0.18	0.068	30
AD	Kungsängsverket	700	35	0.55	0.1	64	50	18	0.28	0.051	32
AD	Käppalaverket	700	39	0.44	0.13	60	57	17	0.19	0.055	26
AD	Skebäcksverket	700	44	0.59	0.16	55	57	19	0.26	0.068	24
AD	Öresundsverket	700	53	0.38	0.2	46	-	-	-	-	-

Sludge pretreatment	WWTP	Pyrolysis temperature °C	C					C _{total}			
			BC	DC	SW	SG	BG	BC	DC	SW	SG
			% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS
UD	Margretelund	750	26	5.3	0.12	69	0	26	5.3	0.12	69
UD	Ormanäs	750	25	7.1	0.16	67	0	25	7.1	0.16	67
AD	Duvbacken	750	35	0.23	0.11	64	41	21	0.14	0.067	38
AD	Henriksdal	750	37	0.91	0.09	62	48	19	0.48	0.049	33
AD	Himmerfjärdsverket	750	36	0.32	0.13	64	69	11	0.1	0.040	20
AD	Hässleholm	750	38	0.34	0.14	62	50	19	0.17	0.068	31
AD	Kungsängsverket	750	31	0.77	0.12	68	50	16	0.39	0.062	34
AD	Käppalaverket	750	36	0.47	0.14	64	57	15	0.2	0.059	27
AD	Öresundsverket	750	56	0.45	0.28	43	-	-	-	-	-
UD	Lycksele	800	24	2.1	0.093	74	0	24	2.1	0.093	74
UD	Margretelund	800	25	5.1	0.11	69	0	25	5.1	0.11	69
UD	Ormanäs	800	29	4.9	0.14	66	0	29	4.9	0.14	66
AD	Duvbacken	800	33	0.29	0.12	66	41	20	0.17	0.070	39
AD	Henriksdal	800	34	1.4	0.19	64	48	18	0.74	0.10	34
AD	Himmerfjärdsverket	800	35	0.34	0.13	65	69	11	0.11	0.041	20
AD	Hässleholm	800	37	0.55	0.13	62	50	19	0.28	0.063	31
AD	Käppalaverket	800	35	0.64	0.13	64	57	15	0.27	0.054	27
AD	Skebäcksverket	800	40	0.50	0.15	59	57	17	0.21	0.064	26
AD	Uddebo	800	39	0.43	0.19	60	44	22	0.24	0.10	34
AD	Öresundsverket	800	54	0.33	0.19	46	-	-	-	-	-

Table C2. Mass flows of nutrients. UD = Undigested, AD = Anaerobically digested, BC = Sludge biochar, DC = Dryer condensate, SW = Scrubber water, SG = Scrubbed flue gas.

Sludge pretreatment	WWTP	Pyrolysis temperature °C	N				P			
			BC	DC	SW	SG	BC	DC	SW	SG
			% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS
UD	Margretelund	600	29	13	0.18	58	100	0.084	0.19	-5.2*
UD	Ormanäs	600	18	0.12	18	64	95	0.044	0.030	-8.9
AD	Duvbacken	600	23	14	0.086	62	78	0.025	0.031	25
AD	Henriksdal	600	27	20	0.11	53	93	0.45	0.049	10
AD	Himmerfjärdsverket	600	26	14	0.12	60	90	0.0041	0.011	12
AD	Hässleholm	600	22	4.7	0.1	73	88	0.19	0.093	11
AD	Kungsängsverket	600	23	27	0.13	50	99	0.0054	0.052	3.5
AD	Käppalaverket	600	31	14	0.13	55	97	0.0073	0.039	-4.2
AD	Skebäcksverket	600	30	9	0.17	61	103	0.12	0.13	1.1
AD	Uddebo	600	22	17	0.14	61	98	0.0066	0.074	-1.1
AD	Öresundverket	600	32	15	0.14	53	104	0.038	0.075	13.5
UD	Lycksele	650	28	18	0.35	53	101	0.016	0.47	-16
UD	Margretelund	650	26±1	23±7	0.19±0.02	51±8	94±5	0.051±0.06	0.37±0.1	3.1±8
UD	Ormanäs	650	21	11	0.17	69	92	0.017	0.20	-6.7
AD	Duvbacken	650	22	21	0.12	56	95	0.0085	0.058	7.1
AD	Henriksdal	650	26	16	0.10	59	88	0.17	0.085	15
AD	Hässleholm	650	22	5.6	0.11	72	92	0.086	0.094	7.7
AD	Kungsängsverket	650	23	19	0.083	58	90	0.0031	0.058	13
AD	Käppalaverket	650	24	14	0.18	61	99	0.0089	0.045	-6.0
AD	Skebäcksverket	650	25	10	0.15	65	101	0.12	0.12	3.4
AD	Uddebo	650	22	23	0.19	55	98	0.0054	0.082	-1.1
AD	Öresundverket	650	25	13	0.12	62	114	0.32	0.064	3.1
UD	Lycksele	700	19	12	0.20	69	98	0.0099	0.24	-13
UD	Margretelund	700	14±1.7	17±3.8	0.14±0.04	68±5.5	91±3.6	0.044±0.05	0.27±0.002	6.7±0.2
UD	Ormanäs	700	7.2	8.6	0.12	84	85	0.014	0.053	0.87
AD	Duvbacken	700	21	19	0.13	60	98	0.0071	0.066	4.7
AD	Henriksdal	700	14	20	0.18	66	80	0.17	0.22	23
AD	Himmerfjärdsverket	700	15	15	0.13	70	91	0.0036	0.04	12
AD	Hässleholm	700	16	5.8	0.13	78	90	0.055	0.11	9.5
AD	Kungsängsverket	700	11	14	0.090	75	100	0.0034	0.059	2.7
AD	Käppalaverket	700	16	15	0.12	69	101	0.0060	0.054	-8.2
AD	Skebäcksverket	700	19	13	0.18	68	122	0.14	0.15	-18
AD	Öresundverket	700	20	16	0.12	63	129	0.048	0.033	-11

(*Negative flows are a consequence of the calculated contents of the SG, and indicate that the total outflows were larger than the inflows.)

Sludge pretreatment	WWTP	Pyrolysis temperature °C	N				P			
			BC	DC	SW	SG	BC	DC	SW	SG
			% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS
UD	Margretelund	750	14	22	0.22	64	97	0.062	0.27	-2.5
UD	Ormanäs	750	13	12	0.17	75	85	0.016	0.21	0.87
AD	Duvbacken	750	15	14	0.13	71	95	0.0069	0.058	7.6
AD	Henriksdal	750	13	11	0.07	76	94	0.33	0.037	8.6
AD	Himmerfjärdsverket	750	12	14	0.13	73	90	0.0061	0.034	13
AD	Hässleholm	750	13	5.8	0.12	81	87	0.19	0.089	13
AD	Kungsängsverket	750	10	20	0.10	69	99	0.0049	0.062	3.7
AD	Käppalaverket	750	11	15	0.12	74	96	0.0075	0.058	-3.1
AD	Öresundsverket	750	14	23	0.19	63	115	0.045	0.028	2.4
UD	Lycksele	800	11	16	0.15	73	133	0.098	0.11	-48
UD	Margretelund	800	11	24	0.19	65	102	0.058	0.36	-6.9
UD	Ormanäs	800	13	8.6	0.13	78	89	0.015	0.02	-2.9
AD	Duvbacken	800	11	18	0.11	70	98	0.0052	0.06	4.0
AD	Henriksdal	800	10	22	0.16	68	91	0.20	0.07	12.1
AD	Himmerfjärdsverket	800	10	13	0.12	77	93	0.0039	0.03	9.3
AD	Hässleholm	800	10	5.2	0.12	85	98	0.49	0.08	1.2
AD	Käppalaverket	800	9	16	0.10	75	97	0.013	0.06	-3.9
AD	Skebäcksverket	800	10	10	0.18	80	130	0.17	0.21	-21.8
AD	Uddebo	800	10	17	0.17	73	103	0.0068	0.11	-6.0
AD	Öresundsverket	800	11	15	0.11	74	110	0.027	0.03	8.9

Table C3. Mass flows of heavy metals. UD = Undigested, AD = Anaerobically digested, BC = Sludge biochar, DC = Dryer condensate, SW = Scrubber water, SG = Scrubbed flue gas.

Sludge pretreatment	WWTP	Pyrolysis temperature °C	Al				Fe				Hg			
			BC	DC	SW	SG	BC	DC	SW	SG	BC	DC	SW	SG
			% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS
UD	Margretelund	600	88	0.052	0.13	12	100	0.052	0.072	-8.3	19	7.0	14	60
UD	Ormanäs	600	83	0.018	0.066	17	136	0.11	0.24	-14	12	5.5	11	72
AD	Duvbacken	600	72	0.023	0.063	28	185	0.012	0.047	-111	11	2.4	9.0	78
AD	Henriksdal	600	139	0.42	0.072	-39*	103	0.53	0.03	-7.0	13	2.7	14	71
AD	Himmerfjärdsverket	600	96	0.012	0.026	3.9	96	0.0086	0.012	7.6	16	4.0	19	61
AD	Hässleholm	600	112	0.11	0.088	-12	117	0.18	0.042	-35	8.4	1.5	18	72
AD	Kungsängsverket	600	100	0.020	0.071	-0.07	100	0.013	0.02	24	5.9	1.8	17	76
AD	Käppalaverket	600	116	0.025	0.052	-17	111	0.033	0.024	-4.3	10	2.3	18	70
AD	Skebäcksverket	600	107	0.14	0.098	-7.0	100	0.18	0.084	6.0	17	3.2	18	62
AD	Uddebo	600	99	0.0075	0.045	1.1	116	0.030	0.054	-26	13	3.8	14	69
AD	Öresundsverket	600	128	0.014	0.044	-28	134	0.030	0.069	-15	5.4	1.3	15	78
UD	Lycksele	650	84	0.03	0.19	16	122	0.083	0.47	39	9.1	4.2	23	63
UD	Margretelund	650	103±8	0.061±0.07	0.37±0.2	-3.5±8	102±8	0.047±0.05	0.20±0.2	1.4±13	22±4	24±24	15±1	38±29
UD	Ormanäs	650	79	0.020	0.099	21	217	0.11	0.59	-113	12	6.1	22	60
AD	Duvbacken	650	99	0.038	0.14	0.98	87	0.023	0.092	30	11	3.8	30	55
AD	Henriksdal	650	194	0.25	0.13	-94	86	0.18	0.026	11	13	2.4	14	71
AD	Hässleholm	650	123	0.083	0.066	-23	130	0.14	0.028	-35	8.4	1.6	25	65
AD	Kungsängsverket	650	100	0.011	0.064	0.15	100	0.0072	0.023	-24	5.9	1.2	8.9	84
AD	Käppalaverket	650	124	0.025	0.071	-24	116	0.025	0.035	-11	9.4	2.2	25	64
AD	Skebäcksverket	650	107	0.13	0.063	-7.3	100	0.19	0.035	10	16	3.1	25	55
AD	Uddebo	650	100	0.0096	0.046	-0.027	119	0.052	0.057	-10	13	5.0	19	62
AD	Öresundsverket	650	110	0.041	0.040	-10	124	0.14	0.062	-61	5.2	1.2	13	80

(*Negative flows are a consequence of the calculated contents of the SG, and indicate that the total outflows were larger than the inflows.)

Sludge pretreatment	WWTP	Pyrolysis temperature °C	Al				Fe				Hg			
			BC	DC	SW	SG	BC	DC	SW	SG	BC	DC	SW	SG
			% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS
UD	Lycksele	700	92	0.019	0.16	7.4	101	0.031	0.26	10	8.5	2.7	9.3	79
UD	Margretelund	700	88±8	0.086±0	0.21±0	12±8	97±5	0.075±0	0.079±0	-20±14	20±2	6.2±1	12±2	61±1
UD	Ormanäs	700	54	0.017	0.054	46	497	0.10	0.31	-416	10	4.7	11	74
AD	Duvbacken	700	99	0.032	0.089	0.9	102	0.013	0.054	-7.3	11	3.4	36	49
AD	Henriksdal	700	282	0.48	0.51	-183	67	0.21	0.11	77	13	3.4	23	61
AD	Himmerfjärdsverket	700	96	0.013	0.068	4.2	98	0.01	0.040	2.2	15	4.4	33	48
AD	Hässleholm	700	150	0.10	0.13	-50	115	0.11	0.040	-10	8.2	1.8	16	74
AD	Kungsängsverket	700	105	0.012	0.053	-4.7	100	0.0083	0.018	-14	5.5	1.2	11	82
AD	Käppalaverket	700	121	0.022	0.11	-21	105	0.018	0.074	-7.4	9.2	2.1	20	69
AD	Skebäcksverket	700	104	0.14	0.11	-4.2	112	0.35	0.11	-0.02	16	3.6	28	52
AD	Öresundsverket	700	114	0.016	0.028	-14	119	0.04	0.042	-29	5.1	1.2	11	83
UD	Margretelund	750	107	0.13	0.30	-7.5	105	0.12	0.23	4.2	18	8.5	19	55
UD	Ormanäs	750	96	0.025	0.12	3.5	99	0.29	0.25	10	11	6.5	21	62
AD	Duvbacken	750	100	0.027	0.084	-0.6	118	0.014	0.048	-42	11	2.8	22	64
AD	Henriksdal	750	124	0.23	0.10	-24	100	0.48	0.051	35	12	2.0	8.7	78
AD	Himmerfjärdsverket	750	89	0.019	0.047	11	89	0.018	0.025	4.6	15	3.6	22	59
AD	Hässleholm	750	127	0.15	0.10	-27	117	0.27	0.050	-13	7.9	1.8	22	69
AD	Kungsängsverket	750	106	0.014	0.060	-6.3	99	0.014	0.021	-39	5.4	1.5	15	78
AD	Käppalaverket	750	128	0.027	0.11	-28	106	0.021	0.068	-6.0	8.6	2.3	19	70
AD	Öresundsverket	750	128	0.13	0.034	-28	168	0.68	0.029	-15	5.0	1.6	16	77

Sludge pretreatment	WWTP	Pyrolysis temperature °C	Al				Fe				Hg			
			BC	DC	SW	SG	BC	DC	SW	SG	BC	DC	SW	SG
			% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS
UD	Lycksele	800	45	0.047	0.039	55	542	0.57	0.35	-455	7.2	2.1	13	77
UD	Margretelund	800	114	0.081	0.28	-14	112	0.07	0.13	-30	17	7.2	14	61
UD	Ormanäs	800	104	0.013	0.022	-4.4	104	0.075	0.094	-13	11	4.9	10	74
AD	Duvbacken	800	102	0.028	0.093	-1.9	103	0.011	0.049	-5.5	11	3.1	38	48
AD	Henriksdal	800	132	0.17	0.15	-33	95	0.25	0.043	-4.1	11	4.0	19	66
AD	Himmerfjärdsverket	800	104	0.016	0.053	-4.0	87	0.0088	0.021	5.4	15	3.8	12	69
AD	Hässleholm	800	112	0.37	0.094	-12	124	0.84	0.079	-23	7.6	1.7	20	71
AD	Käppalaverket	800	107	0.035	0.066	-7.2	111	0.048	0.028	-12	8.5	2.2	22	67
AD	Skebäcksverket	800	89	0.11	0.091	11	121	0.21	0.11	-23	15	3.4	33	49
AD	Uddebo	800	102	0.0087	0.050	-1.9	132	0.027	0.049	-28	12	4.4	27	56
AD	Öresundsverket	800	139	0.015	0.034	-39	216	0.039	0.066	-125	4.8	1.1	10	84

Table C4. Mass flows of heavy metals (continued). UD = Undigested, AD = Anaerobically digested, BC = Sludge biochar, DC = Dryer condensate, SW = Scrubber water, SG = Scrubbed flue gas.

Sludge pretreatment	WWTP	Pyrolysis temperature °C	Cd				Cr				Cu			
			BC	DC	SW	SG	BC	DC	SW	SG	BC	DC	SW	SG
			% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS
UD	Margretelund	600	62	3.2	13	21	151	0.44	3.0	-55*	89	0.043	0.13	11
UD	Ormanäs	600	59	0.17	11	30	163	0.62	2.4	-66	95	0.022	0.10	5.2
AD	Duvbacken	600	72	0.69	2.2	26	68	0.11	0.64	32	56	0.0063	0.018	44
AD	Henriksdal	600	42	1.5	5.5	51	187	1.1	1.4	-90	94	0.23	0.040	5.9
AD	Himmerfjärdsverket	600	26	0.16	4.3	69	107	0.19	0.93	-8.4	88	0.014	0.028	12
AD	Hässleholm	600	63	0.15	7.5	29	136	0.19	1.0	-38	118	0.11	0.13	-18
AD	Kungsängsverket	600	32	0.28	11	56	103	0.22	1.1	-4.1	97	0.022	0.065	2.7
AD	Käppalaverket	600	65	0.14	5.9	29	106	0.21	0.99	-7	113	0.022	0.038	-13
AD	Skebäckverket	600	63	0.19	13	23	172	0.93	1.7	-75	111	0.15	0.12	-11
AD	Uddebo	600	19	0.17	3.8	77	104	0.22	0.96	-5.6	98	0.023	0.11	1.5
AD	Öresundsverket	600	153	0.24	5.3	-59	123	0.14	0.90	-24	136	0.019	0.083	-36
UD	Lycksele	650	9.0	3.3	28	60	61	0.17	1.3	37	112	0.069	0.385	-12
UD	Margretelund	650	8.6±2	9.9±9	25±11	57±23	125±34	0.42±0	3.4±1	-28±33	104±8	0.047±0.04	0.34±0.2	-4.8±8
UD	Ormanäs	650	18	0.19	21	61	165	0.7	11	-77	92	0.024	0.175	7.4
AD	Duvbacken	650	37	1.1	8.8	53	111	0.18	1.2	-13	94	0.012	0.063	6.1
AD	Henriksdal	650	29	1.4	10	60	186	0.55	0.9	-88	88	0.10	0.042	12
AD	Hässleholm	650	14	0.1	19	66	176	0.23	1.9	-78	128	0.082	0.098	-28
AD	Kungsängsverket	650	23	0.18	16	60	101	0.14	0.74	-1.7	94	0.013	0.056	6.0
AD	Käppalaverket	650	20	0.14	16	63	108	0.21	1.1	-9.2	120	0.019	0.047	-20
AD	Skebäckverket	650	16	0.19	19	65	164	0.81	1.2	-66	111	0.145	0.106	-11
AD	Uddebo	650	62	0.23	11	27	128	1.7	1.0	-30	104	0.052	0.12	-4.6
AD	Öresundsverket	650	109	0.22	10	-19	109	0.66	0.9	-10	121	0.048	0.095	-21

(*Negative flows are a consequence of the calculated contents of the SG, and indicate that the total outflows were larger than the inflows.)

Sludge pretreatment	WWTP	Pyrolysis temperature °C	Cd				Cr				Cu			
			BC	DC	SW	SG	BC	DC	SW	SG	BC	DC	SW	SG
			% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS
UD	Lycksele	700	3.4	2.2	17	78	54	0.11	0.74	45	73	0.032	0.18	26
UD	Margretelund	700	12±5	2.7±1	21±2	65±2	174±22	0.35±0	1.8±0	-76±22	87±7	0.063±0.06	0.20±0.01	13±7
UD	Ormanäs	700	63	0.14	17	19	387	0.53	1.8	-290	63	0.019	0.074	37
AD	Duvbacken	700	14	1.0	17	68	96	0.17	1.2	2.5	94	0.0077	0.042	6.0
AD	Henriksdal	700	4.3	1.9	29	65	270	0.73	1.8	-173	87	0.14	0.14	13
AD	Himmerfjärdsverket	700	3.1	0.18	65	32	132	0.21	0.81	-33	89	0.015	0.066	11
AD	Hässleholm	700	4.4	0.11	28	67	157	0.37	1.5	-59	140	0.087	0.16	-39
AD	Kungsängsverket	700	4.6	0.19	21	74	121	0.14	0.7	-22	91	0.014	0.57	8.4
AD	Käppalaverket	700	3.8	0.13	21	75	101	0.19	0.99	-2.0	100	0.017	0.061	-3.3
AD	Skebäcksverket	700	7.7	0.37	26	66	208	1.3	1.5	-110	120	0.20	0.15	-24
AD	Öresundsverket	700	68	0.22	14	18	91	0.13	0.72	7.7	97	0.023	0.067	3.1
UD	Margretelund	750	6.7	3.8	31	59	166	0.54	3.0	-70	107	0.099	0.26	-7.1
UD	Ormanäs	750	5.4	0.2	37	58	122	0.74	2.7	-25	109	0.036	0.28	-9.3
AD	Duvbacken	750	5.2	0.8	21	73	92	0.14	1.1	6.6	95	0.0093	0.042	5.1
AD	Henriksdal	750	3.2	1.1	14	81	140	0.99	0.94	-42	97	0.22	0.047	2.8
AD	Himmerfjärdsverket	750	5.2	0.15	38	56	112	0.17	1.2	-13	82	0.017	0.058	18
AD	Hässleholm	750	4.4	0.19	27	68	153	0.71	1.1	-55	125	0.16	0.13	-25
AD	Kungsängsverket	750	4.1	0.23	34	61	95	0.18	0.78	3.8	94	0.02	0.076	5.5
AD	Käppalaverket	750	2.7	0.14	20	77	106	0.21	0.99	-7.6	96	0.018	0.066	4.2
AD	Öresundsverket	750	23	0.31	31	45	102	0.18	0.86	-3.5	110	0.092	0.09	-10

Sludge pretreatment	WWTP	Pyrolysis temperature °C	Cd				Cr				Cu			
			BC	DC	SW	SG	BC	DC	SW	SG	BC	DC	SW	SG
			% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS
UD	Lycksele	800	4.0	1.6	15	80	62	0.16	0.43	38	133	0.11	0.12	-33
UD	Margretelund	800	5.1	3.2	24	68	122	0.46	3.1	-26	110	0.062	0.36	-11
UD	Ormanäs	800	3.4	0.15	12	85	98	0.56	2.4	-1.0	108	0.018	0.043	-8.4
AD	Duvbacken	800	5.8	0.90	27	66	88	0.15	1.3	10	96	0.010	0.053	4.4
AD	Henriksdal	800	3.1	2.3	27	68	163	0.51	1.3	-65	87	0.13	0.081	13
AD	Himmerfjärdsverket	800	6.9	0.16	19	74	119	0.18	0.81	-20	91	0.013	0.038	9.0
AD	Hässleholm	800	2.3	0.45	32	65	155	2.1	1.7	-59	119	0.49	0.11	-20
AD	Käppalaverket	800	3.0	0.13	22	74	126	0.20	0.79	-27	109	0.025	0.045	-8.7
AD	Skebäcksverket	800	8.0	0.25	35	56	162	0.48	1.6	-65	139	0.15	0.18	-39
AD	Uddebo	800	10	0.20	30	60	113	0.25	1.2	-14	106	0.026	0.15	-6.2
AD	Öresundsverket	800	12	0.21	41	47	122	0.12	2.7	-25	122	0.021	0.072	-22

Table C5. Mass flows of heavy metals (continued). UD = Undigested, AD = Anaerobically digested, BC = Sludge biochar, DC = Dryer condensate, SW = Scrubber water, SG = Scrubbed flue gas.

Sludge pretreatment	WWTP	Pyrolysis temperature °C	Ni				Pb				Zn			
			BC	DC	SW	SG	BC	DC	SW	SG	BC	DC	SW	SG
			% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS
UD	Margretelund	600	151	0.80	4.2	-56*	119	1.2	2.4	-22	99	0.089	0.2	0.98
UD	Ormanäs	600	142	0.70	14	-57	103	0.23	0.46	-3.2	97	0.055	0.21	2.8
AD	Duvbacken	600	83	0.18	0.58	16	50	0.14	0.28	49	64	0.020	0.040	36
AD	Henriksdal	600	149	1.1	1.2	-52	100	0.42	0.60	-1.5	98	0.37	0.070	1.6
AD	Himmerfjärdsverket	600	104	0.23	0.55	-4.9	85	0.12	0.24	15	88	0.040	0.080	12
AD	Hässleholm	600	127	0.44	0.93	-28	100	0.15	0.21	-0.23	118	0.19	0.18	-18
AD	Kungsängsverket	600	101	1.9	0.91	-3.7	75	0.16	0.32	24	102	0.072	0.13	-2.1
AD	Käppalaverket	600	113	0.24	0.47	-14	106	0.14	0.28	-6.3	108	0.037	0.074	-7.9
AD	Skebäcksverket	600	159	1.4	1.3	-62	100	0.20	0.20	-0.66	113	0.21	0.15	-13
AD	Uddebo	600	119	0.48	1.2	-21	109	0.21	0.43	-10	98	0.048	0.096	1.8
AD	Öresundswerket	600	142	0.35	0.96	-44	132	0.15	0.3	-32	136	0.051	0.10	-36
UD	Lycksele	650	101	0.61	5.8	-7.3	104	0.58	1.2	-6.1	93	0.086	0.40	6.3
UD	Margretelund	650	128±8	1.1±0.2	3.6±2	-33±7	105±1	1.3±0.01	2.5±0.01	-8.7±0.8	104±10	0.10±0.04	0.61±0.5	-4.3±10
UD	Ormanäs	650	143	0.79	2.5	-46	100	0.26	0.61	-0.9	95	0.061	0.48	4.9
AD	Duvbacken	650	114	0.38	7.85	-22	97	0.22	0.45	2.4	92	0.036	0.71	7.4
AD	Henriksdal	650	139	0.57	0.66	-40	109	0.27	0.54	-10	99	0.17	0.074	1.1
AD	Hässleholm	650	160	0.37	0.72	-61	107	0.11	0.29	-7.3	139	0.134	0.16	-39
AD	Kungsängsverket	650	104	0.27	0.62	-4.9	63	0.1	0.21	37	100	0.043	0.12	-0.6
AD	Käppalaverket	650	118	0.24	0.54	-19	104	0.14	0.3	-4.1	115	0.049	0.077	-15
AD	Skebäcksverket	650	156	1.2	1.29	-58	104	0.21	0.22	-4.8	114	0.211	0.11	-14
AD	Uddebo	650	134	0.62	1.69	-37	114	0.28	0.62	-15	101	0.064	0.13	-1.4
AD	Öresundswerket	650	130	0.99	0.85	-31	118	0.14	0.27	-18	125	0.063	0.09	-25

(*Negative flows are a consequence of the calculated contents of the SG, and indicate that the total outflows were larger than the inflows.)

Sludge pretreatment	WWTP	Pyrolysis temperature	Ni				Pb				Zn			
			BC	DC	SW	SG	BC	DC	SW	SG	BC	DC	SW	SG
		°C	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS
UD	Lycksele	700	88	0.34	1.2	11	98	0.38	0.77	0.85	78	0.056	0.29	22
UD	Margretelund	700	185±7	0.83±0.2	2.0±0.3	-88±7	114±0.01	1.0±0.27	2.1±1	-17±1	96±11	0.094±0.06	0.52±0.05	3.4±11
UD	Ormanäs	700	502	0.60	1.2	-400	99	0.19	0.43	0.47	116	0.047	0.24	-16
AD	Duvbacken	700	103	0.31	0.94	-4.2	95	0.20	0.41	4.0	92	0.029	0.073	7.9
AD	Henriksdal	700	180	0.80	1.0	-82	114	0.38	0.76	-16	100	0.26	0.37	-0.65
AD	Himmerfjärdsverket	700	124	0.25	0.7	-25	90	0.13	0.54	9.3	87	0.044	0.15	13
AD	Hässleholm	700	154	0.47	1.2	-55	105	0.12	0.61	-5.4	126	0.11	0.55	-26
AD	Kungsängsverket	700	145	0.28	0.79	-46	59	0.10	0.25	41	116	0.043	0.11	-16
AD	Käppalaverket	700	101	0.22	0.64	-1.7	90	0.13	0.60	9.3	91	0.034	0.15	9.3
AD	Skebäcksverket	700	205	1.8	1.6	-110	103	0.28	0.37	-3.3	132	0.31	0.17	-33
AD	Öresundsverket	700	103	0.42	9.6	-13	112	0.14	0.28	-12	110	0.047	0.095	-10
UD	Margretelund	750	194	1.1	3.0	-98	102	1.5	3.1	-7.0	99	0.12	1.2	-0.053
UD	Ormanäs	750	124	1.4	4.0	-29	94	0.27	2.7	2.6	96	0.084	1.2	2.6
AD	Duvbacken	750	105	0.21	0.77	-6.0	93	0.17	0.65	6.1	94	0.024	0.13	5.8
AD	Henriksdal	750	124	0.95	0.76	-25	99	0.35	0.5	0.33	101	0.30	0.12	-1.6
AD	Himmerfjärdsverket	750	98	0.24	0.65	0.74	89	0.11	0.73	9.8	86	0.036	0.15	14
AD	Hässleholm	750	142	0.84	0.98	-43	104	0.2	12 000	-12 000	131	0.21	0.46	-32
AD	Kungsängsverket	750	106	0.34	0.86	-7.0	61	0.13	0.62	38	94	0.053	0.27	5.3
AD	Käppalaverket	750	97	0.25	0.61	2.3	81	0.14	1.2	18	83	0.037	0.36	16
AD	Öresundsverket	750	115	0.68	0.98	-16	121	0.2	0.64	-22	119	0.21	0.13	-19

Sludge pretreatment	WWTP	Pyrolysis temperature °C	Ni				Pb				Zn			
			BC	DC	SW	SG	BC	DC	SW	SG	BC	DC	SW	SG
			% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS	% of DSS
UD	Lycksele	800	129	0.54	0.77	-31	101	0.29	1.1	-2.1	93	0.11	0.38	6.7
UD	Margretelund	800	136	0.96	3.1	-40	92	1.3	4.0	3.2	96	0.092	2.5	1.7
UD	Ormanäs	800	126	0.79	1.8	-29	84	0.2	1.3	15	93	0.049	0.16	6.4
AD	Duvbacken	800	113	0.24	0.95	-14	88	0.18	1.5	11	96	0.027	0.29	3.7
AD	Henriksdal	800	130	0.78	0.67	-32	103	0.44	1.7	-5.0	96	0.19	0.39	3.4
AD	Himmerfjärdsverket	800	118	0.23	0.46	-19	88	0.11	0.83	11	84	0.038	0.13	16
AD	Hässleholm	800	145	2.3	1.4	-49	100	0.51	1.8	-2.6	116	0.63	0.45	-17
AD	Käppalaverket	800	115	0.37	0.45	-16	126	0.14	1.8	-28	103	0.035	0.27	-3.8
AD	Skebäcksverket	800	157	0.89	1.5	-60	97	0.2	1.6	1.3	134	0.22	0.39	-34
AD	Uddebo	800	131	0.6	1.6	-33	92	0.25	5.7	1.7	89	0.055	0.50	11
AD	Öresundsverket	800	141	0.24	3.3	-45	110	0.13	1.0	-11	113	0.044	0.089	-13