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Analysis of biosolids and biochar from sewage sludge with its application on soil in Norwegian context

Master's thesis in Industrial Ecology
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Preface

This thesis concluded my Master of Science in Hydroscience and Engineering at the Norwegian University of Science and Technology (NTNU) and Technische Universität Dresden (TUD). It was written in collaboration with the Industrial Ecology department at NTNU. This thesis delves into the analysis of biosolids and biochars, focusing on their impact on soil quality and the environment, with a particular emphasis on the Norwegian context.

I would like to express my gratitude to my supervisor Prof. Francesco Cherubini, and my co-supervisors, Dr. Marjorie Morales and Prof. Jude Ndzifon Kimengsi, for all their guidance and valuable feedback. Working on the thesis has been an intensive learning process, and the result would not have been possible without your continuous help and motivation. I also want to thank my family and friends for supporting me during my studies.

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1. Abbreviations

AD	Anaerobic Digestion
ARGs	Antibiotic Resistance Genes
BET Method	Brunauer Emmett and Teller Method
BOD	Biochemical Oxygen Demand
CAS	Chemical Abstracts Service
CEC	Cation Exchange Capacity
CFU	Colony Forming Units
CHP	Combined Heat and Power
DO	Dissolved Oxygen
DTPA	Diethylenetriaminepentaacetic Acid
EC	Exchangable Cations
EEA	European Economic Area
EOPs	Emerging Organic Pollutants
EPA	Environmental Protection Agency
ESR	Effort Sharing Regulation
ETS	Emission Trading System
EU	European Union
HM(s)	Heavy metal(s)
HOC	Hazardous Organic Chemical
HTL	Hydrothermal Liquefaction
ILCD	International Reference Life Cycle Data System
LCA	Life Cycle Assessment
LCIA	Life Cycle Impact Assessment

LOQ	Limit Of Quantification
LPI	Leaching Potential Index
LULUCF	Land Use, Land-Use Change, and Forestry regulation
MPN	Most Probably Numbers
OPFRS	Organophosphorus Flame Retardants
PAH	Polycyclic Aromatic Hydrocarbons
POP(s)	Persistant Organic Pollutant(s)
PPB	Parts Per Billion
PCB	Polychlorinated Biphenyls
PFAA	Perfluoroalkyl or Perfluoroalkyl(poly)ether Acids
PFAS	Per- and Polyfluoroalkyl Substances
PFNA	Perfluorononanoic Acid
PFOA	Perfluorooctanoic Acid
PFOS	Perfluorooctanesulfonic Acid
PLFA	Phospholipid-derived Fatty Acids
PTE	Potentially Toxic Element
REACH	Registration, Evaluation, Authorisation and Restriction of Chemicals
SS	Sewage sludge
TH	Thermal Hydrolysis
VOC	Volatile Organic Compounds
WWTP	Waste Water Treatment Plant

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2. Abstract

Sewage sludge management presents a multifaceted challenge requiring comprehensive strategies to mitigate environmental and health risks while maximizing resource recovery. There is insufficient research done on biosolids and biochar, especially with respect to their feedstock-specific properties and long-term effects. This study investigates the efficacy of biosolids and biochar on soil application. Through a combination of experimental analysis and theoretical modeling, the study assesses the removal efficiency and fate of contaminants across different treatment scenarios. High accumulation potential in soil is observed for Cr, Cu, Cd, and Pb, with Cu and Zn having higher leaching potential. Amongst the HMs analyzed, Cu, Zn, Cd,

Pb and Cr are found to have higher toxicity potential in soil application. Pb, As, Zn, and Cd are found to be more bioavailable than other HMs in soil. In the context of soil application, biochar is deemed a much more viable option than biosolids due to the low mobility, leaching, and toxic effects of the pollutants present. Furthermore, biochar produced at low temperatures without AD leads to less leaching, toxicity, and bioavailability.
Recommendations for optimizing biochar application to soil, including thermal hydrolysis, co-pyrolysis, hydrothermal liquefaction (HTL), and biochar amendment are proposed to enhance treatment efficiency and minimize risks. Overall, this study underscores the need for comprehensive monitoring and regulation of treated sludge to ensure environmental protection and human health, particularly in the context of emerging pollutants and evolving treatment technologies.

2.1. Keywords

Biosolids, Biochar, sewage sludge, pyrolysis, PFAS, OPFRs, Heavy metals, bioavailability, toxicity, mobility, leaching, Norway

3. Introduction

Exponential growth of the population, coupled with climate change and acceleration in demand for natural resources has resulted in a search for different ways of procuring resources. The wastewater sector is no exception. With the recent increase in interest and demand for circular economy and sustainability, there has been an evident shift from the "removal and treat" approach to "recovery and reuse" (Arulrajah et al., 2011; Bagheri et al., 2023; Mulchandani & Westerhoff 2016; Shaddel et al., 2019). Sewage sludge, a byproduct of wastewater treatment, poses a significant environmental challenge worldwide in terms of its safe and sustainable disposal. It contains a variety of organic and inorganic matter along with pathogens and other microbial pollutants in dissolved and suspended states. These make it a potential source of secondary environmental pollution[1] (Raheem et al. 2018). As populations grow, so does the volume of sewage sludge[2], necessitating comprehensive and sustainable management strategies (Bagheri et al., 2023; Kim, Choi, & Lee 2024; Marchuk et al., 2023; Mohajerani et al. 2017; Patel et al., 2020). The traditional disposal methods of landfilling and incineration have proven to be highly detrimental to the environment. Landfilling leads to soil, air, and water contamination, which later enters the food chain and harms living beings. Incineration, despite being low-cost, high energy recovery, and potential destruction of contaminants and pathogens, can emit harmful substances like acid gases, dioxins, particulate matter, and NOx (Kodešová et

[1] Secondary pollution happens when a primary pollutant reacts with another primary pollutant, sunlight, and water to create a different pollutant.
[2] According to reports, 45 million tons of dry sewage sludge was produced globally in 2017 (Bagheri et al., 2023). The global sewage production is expected to reach approximately 200 million tons per year by 2025 (Kim, Choi, & Lee 2024; Mohajerani et al. 2017).

al. 2023; Kumar et al., 2022; Liang et al. 2021; Manikandan et al., 2023; Patel et al., 2020; Paz-Ferreiro et al. 2018; Rigby et al., 2021; Zhang et al., 2022). These have led to the search and practice of different ways of disposal.

There are several several types of SS (sewage sludge) treatment processes, each with its own objectives and methods (DOMBOR 2023). After thickening or dewatering[3], SS is usually subjected to biological, chemical, or thermal treatment. The biological treatment includes aerobic digestion (microbial degradation in the presence of oxygen) and anaerobic digestion (microbial decomposition in the absence of oxygen). Whereas chemical treatment involves adding chemicals such as lime, polymers, or coagulants to the sludge to enhance dewatering, remove contaminants, or stabilize metals. Thermal treatment involves subjecting sludge to high temperatures to destroy pathogens, reduce volume, and produce energy or beneficial byproducts (DOMBOR 2023; Menahem and Álvaro 2012). All these methods produce dewatered sludge or biosolids[4] and biochar[5] as end products and a wide variety of by-products like biogas (a combination of CH_4, CO_2, H_2, H_2O, N, and other gases), water, and Hydrogen Sulfide (H_2S) (Castro et al., 2023). Each of them has different constituents and properties, depending on the way and conditions in which they are produced. These products are often rich in a plethora of minerals and nutrients, which can be used for a variety of purposes such as fertilizers, composts, substrates for urban green infrastructures, concrete additives, etc (Paz-Ferreiro et al. 2018). However, they also contain a lot of harmful substances, which if not pre-treated properly, could lead to toxicity in the environment where they are released, further harming the biosphere of the region. The two major such products are biosolids and biochars.

Biosolids are treated sewage sludge and are a major by-product (usually 15% - 90% solid) of the wastewater treatment process. It contains macronutrients, such as N, P, K, and S, and micronutrients, such as Cu, Zn, Ca, Mg, Fe, B[6], Mo, and Mn. It may also contain traces of synthetic organic compounds and metals, including As[7], Cd, Cr, Pb, Hg, Ni, and Se (AWA 2017; Patel et al., 2020; Paz-Ferreiro et al. 2018; US EPA 2019). These contaminants limit the extent to which biosolids can be used. Biosolids are graded according to chemical composition and the level of pathogens remaining after production. Biosolids are divided into “Class A” and “Class B”[8]. The different classes have specified treatment requirements for pollutants, pathogens, and

[3] They are considered the first step towards treatment, usually done through mechanical methods. involves removing water from the sludge to increase its solid content and reduce the volume of sludge for subsequent treatment steps. Dewatering includes centrifugation, belt presses, and vacuum filtration for further removal of water from the sludge to reduce its weight and volume before disposal (DOMBOR 2023).

[4] “Biosolids” or dewatered sludge, are treated sewage sludges, which are mainly a mixture of water and organic materials (AWA 2017).

[5] “Biochars”, also known as biocarbons, is the lightweight black residues comprising of carbon and ashes, and obtained by the pyrolysis of biomass (SINTEF).

[6] In regards to Boron (B), even though it is not heavy metal, it is also considered a PTE (Potentially Toxic Element), but there is still little known about the potential for toxicity to the soil-plant system (Vera et al., 2019), and no limit values are found in the Norwegian standard or in the EU (Amorim Júnior et al. 2021).

[7] For simplicity, metalloid As is grouped with metals for this paper.

[8] For Class A biosolids, MPN (Most Probably Numbers) for pathogens is less than 1000/gram of total dry solids; while in Class B biosolids, MPN and Colony Forming Units (CFU) both shall be less than 20,00,000/gram of total dry solids. It has been suggested that ClassA biosolids can be used without pathogen-reducing treatment for different

vector attraction reduction, as well as general requirements and management practices (AWA 2017; US EPA 2019; Wanare et al., 2022).

The term biochar refers to a carbon-rich material obtained from heating biomass (biosolids) in the absence of oxygen (also called pyrolysis[9]) (Johnson et al., 2012; Patel et al., 2020; Paz-Ferreiro et al. 2018). It is a black, highly porous, fine-grained, lightweight substance with a large surface area. It comprises mainly of carbon (≥65%), with a small amount of N, H, O, K, Ca, and other elements. organic content in biosolids is mainly composed of protein (24–42%), carbohydrate (7–18%) and lipid (1–14%). The constituents of biochar vary with the temperature of production. Usually, they have a higher content of heavy metals (HMs) as compared to biosolids (Patel et al., 2020). The functional group[10] present on the biochar surface influences their interaction with nutrients and contaminants present in soil (Janu et al., 2021; Lu et al., 2015; Méndez et al., 2012).

These two have substantive polarized effects when released in the environment, depending on the way they are produced and pre-treated. They can however be analyzed by observing their properties such as concentration of constituents, mobility in environment, bioavailability, toxicity, leaching, life cycle assessment, and other characteristics. There have been several studies conducted, theoretically as well as experimental, to understand the behavior of biosolids and biochar. One promising avenue is the utilization of biosolids and biochars derived from sewage sludge in agricultural soil, offering a potential solution to both waste disposal issues and soil fertility enhancement, both in terms of their physical properties like pH, porosity, water retention capacity, soil aggregation, and microbial activities, as well as enhancing its mineral and nutrient content (Bagheri et al., 2023; Jaya Nepal et al. 2023; Méndez et al., 2012; Patel et al., 2020). However, most of these studies are done for short-term application in soils. The extensivity of experimental results also falls short leading to a culmination of generic ideas regarding their characteristics and behavior. Additionally, studies on specific contaminants and their fate with the objective of contributing to the regulatory measures are heavily lacking. The unique behaviors of different contaminants even within the same group do not make it easier either. This paper thus also tries to reflect on these research gaps as well as contribute to insights regarding policies.

The data is obtained from a combination of 18 WWTP facilities in Norway (Blytt & Stang 2019). For this paper, four different cases are undertaken. The first case (C_1) is of a common

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applications such as filling applications below roads, and buildings and for the construction of embankments. Class A biosolids can be considered similar to organic soil, and also be used as partial fertilizer and/or soil conditioner for application in agriculture, home lawns/gardens, cemeteries, medians in highways and airports; and to prevent cracking, deterioration, and moisture percolation through landfill cap. Class B biosolids can be used at animal grazing sites, crop harvesting, and public access if their pathogen concentration is below permissible limits. Proper treatment and disposal/reutilization of sewage sludge/biosolids play an important role in the improvement of the quality of water and the surrounding environment (US EPA 2019).

[9] Pyrolysis is a thermal endothermic process usually occurring at 400-700℃. Pyrolysis of biosolids usually produces three products: Biochar, pyrolysis gas (syngas), and pyrolysis oil (bio-crude or bio-oil) (Patel et al., 2020).

[10] The main functional groups of biochar are aromatic and heterocyclic carbons (Li et al., 2013; Paz-Ferreiro et al. 2018; Wallace, Su, & Sun 2017).

treatment that involves lime stabilization[11] followed by dewatering to produce dewatered sludge (biosolids) which are then disposed of into the land. In the second case (C_2), anaerobic digestion is used instead of lime stabilization followed by dewatering. The biogas produced in this case is further upgraded to produce methane. Both C_1 and C_2 are the current options for sewage sludge treatment in Norway and are also used as benchmarks to observe the environmental performances of other treatments. In the third case (C_3), after anaerobic digestion (AD) and dewatering, the sludge is dried and undergoes the process of pyrolysis to produce biochar. In the 4th case (C_4), the sludge is directly exposed to dewatering, drying, and then pyrolysis without undergoing the process of stabilization. In both C_3 and C_4, the pyrolysis is evaluated at both low (500-600)℃ and high (700-800)℃. In all these cases, the end product is applied to the soil and thus it is essential to understand their interaction with the soil as well as the biosphere around it. Furthermore, their short and long-term effects are also needed to be examined.

Understanding the effects of biosolids and biochars on soil is not confined to academic curiosity; it has direct implications for real-world scenarios. The sustainable utilization of sewage sludge by converting it into beneficial agricultural inputs can potentially alleviate the burden on traditional waste disposal systems. Moreover, if proven safe, this practice could enhance soil fertility and contribute to sustainable agricultural practices. Conversely, a lack of understanding or oversight could lead to unintended environmental consequences, including soil contamination and potential impacts on human health. Moreover, the general consideration of biochar production is proven to be energy-intensive, high capital investments, with an additional need for mechanical dewatering and pre-drying of biosolids (Kumar et al., 2023). Thus, this research serves a practical purpose by informing policy decisions and promoting sustainable practices in the management of sewage sludge (AWA 2017; Méndez et al., 2012; SINTEF).

Table 1: Different ways of sewage sludge disposal in Norway (Source: Statistics Norway)

	Disposal of sewage sludge (tonnes dry weight)
	Year: 2022
Total disposal	**132 818**
Agriculture	**69 071**
Park and green spaces	**18 456**
Fertilizers	**22 817**
Landfill covers	**3 406**
Deposits	**5 680**
Incineration	**3 994**
Other uses	**9 395**
Mass loss as Biogass	**34 831**

[11] The procedure of sludge stabilization is done to break down the organic components of sludge to decrease its mass as well as order and make it less hazardous from a public health perspective. Stabilization can be done by various processes like anaerobic and aerobic digestion, composting, lime stabilization, and heat treatment. These processes can be applied individually or in combination (Farzadkia et al., 2014).

Norway, a country known for its pristine natural environment and stringent environmental regulations, places a premium on maintaining the integrity of its natural ecosystems. Norway’s comprehensive climate action plan was approved on the 8th of January, 2021 to meet climate targets under the Paris Agreement and promote green growth. It has agreed with the EU to take part in EU climate legislation for the period 2021-2030. The climate cooperation covers the EU Emission Trading System (EU ETS), the Effort Sharing Regulation for non-ETS emissions (ESR), and the land-use, land-use change and forestry regulation (LULUCF). Norway aims to be carbon neutral by 2030 (IEA 2022). With stringent environmental regulations in place, the management of sewage sludge has also been a critical concern. According to 2023 Statistics Norway data, there are about 2754 wastewater facilities in Norway treating about 88% of Norway’s population. Approximately 83 percent of sewage sludge produced from these plants is used in agriculture, parks, and other green spaces or delivered to soil producers, according to the 2023/50 report of Statistics Norway. The introduction of biosolids and biochars into agricultural practices requires a nuanced understanding of their impact on Norwegian soils, considering the unique environmental and regulatory landscape of the country. The importance of this research lies in its potential to contribute to the development of guidelines and policies that align with Norway's commitment to environmental stewardship while addressing the practical challenges associated with sewage sludge management (MCE 1981; Statistics Norway 2023).

This research delves into the analysis of biosolids and biochars, focusing on their impact on soil quality and the environment, with a particular emphasis on the Norwegian context. By examining the potential benefits and risks associated with the application of SS-derived materials in Norwegian soils, the study seeks to inform sustainable SS management practices aligned with the country's commitment to environmental conservation. It further aims to contribute to the knowledge base necessary for informed decision-making in waste management practices. However, it should be noted that this research does not extensively cover the speciation of contaminants, nor does it go into depth about the various sewage sludge removal processes mentioned within the paper’s premise. While the study focuses on the effects of pyrolysis on soil health, it does not extensively explore the potential impacts of biochar application on soil microbiota or ecosystem dynamics. Additionally, it falls short of a comprehensive exploration into the degradation or removal mechanisms of the contaminants discussed here. This limitation underscores the necessity for future investigations focusing on the mechanisms of these pollutants to contribute to informed management strategies.

3.1. Objective

The objective of this study is to comprehensively analyze the impact of biosolids and biochar from sewage sludge on soils. The paper first analyzes the data for the four different types of sludge treatment and their byproducts, especially for biosolids and biochar production. This takes into account the temperature of pyrolysis for the production of biochar. Consequently, the focus is shifted in the direction of constituent analysis of biosolids and biochar. Specifically, the research focuses on two major classes of contaminants: Heavy Metals (HMs) and Organic Compounds, including Organophosphorus Flame Retardants (OPFRs) and Per- and Polyfluoroalkyl Substances (PFAs). Through rigorous examination, the study aims to provide insights into the leaching behavior, toxicity, bioavailability, and mobility, general behavior of biosolids & biochar, Life cycle assessment (LCA) with respect to climate change and human toxicity, and adherence to normative limits set by Norwegian authorities, followed by potential effects on crops.

While analyzing the data, the key comparative parallels are drawn between the pretreatment processes of lime stabilization and anaerobic digestion, the effects of pyrolysis on the contaminants' fate, and the temperature aspect of pyrolysis. Several parameters like removal efficiency (RE %), biochar yield, and heavy metals' retention rate are calculated to understand the characteristics of the processes. Through a systematic overview of these factors, the study seeks to provide a holistic understanding of the risks and benefits associated with the application of sewage sludge-derived materials (biosolids and biochar) in Norwegian agriculture. It also delves into the Norwegian limits and regulations set for different contaminants and their specified disposal ways.

Finally, it provides certain recommendations on the favorable method of the SS treatment process as well as its byproduct(s) application to the environment followed by the identification of further research scope.

3.2. Research Question

The central research question guiding this investigation is: ***How do biosolids and biochars derived from sewage sludge concerning the leaching behavior, toxicity, bioavailability, adherence to normative limits, mobility, Life cycle impact assessment (LCIA), and potential effects on crops, with a focus on Heavy Metals (HMs) and Organic Compounds (OPFRs, PFAs), impact Norwegian soils?***

Through a systematic examination of these factors, this study aims to provide a comprehensive understanding of the potential risks and benefits associated with the application of sewage sludge-derived materials on soil in the Norwegian context. In exploring these dimensions, the research seeks to contribute valuable insights to the scientific community, policymakers, and environmental practitioners, promoting evidence-based decision-making in the sustainable management of sewage sludge.

4. Literature Review

The literature review aims to provide a comprehensive overview of studies conducted between 2011 and 2024 in order to reflect the latest advancements, findings, and trends in the field. It also helps in maintaining accuracy and relevancy, while reflecting on emerging trends. The key keywords utilized in the search process include biosolids, biochars, sewage sludge, PFAS, emerging pollutants, LCA, OPFRs, heavy metals, agriculture, recovery, and soil application. These keywords were used individually as well as in combination. During the evaluation process, papers out of scope or duplicates were meticulously identified and removed to ensure the inclusion of only relevant and novel findings. A thorough search was conducted using prominent academic databases including SCOPUS, ScienceDirect, Google Scholar, PubMed, ResearchGate, and Web of Science. The selected databases were chosen to ensure comprehensive coverage of scholarly literature, incorporating a range of sources such as review journals, articles, experiments, theses, conference papers, and books. The main themes that were explored include biosolids and biochars in agriculture, PFAs and OPFRs in soil, heavy metals uptake and mitigation, and soil recovery and sustainable practices. In addition, the government regulations on discharge and limits of the substances were also considered.

The abstracts were evaluated in accordance with the theme of this paper. In case of changes made during the inductive process of categorizing the material, abstracts were reread and adapted to the latest standard of categories. Some additional papers were retrieved from a reference search of the review papers, which might have been missed in the initial search due to having broad titles and keywords. There was a substantive amount of research done on the physical and geotechnical properties and/or application of biosolids and biochars for engineering purposes; however, due to the irrelevancy of the scope of the paper, these were discarded with a few exceptions for basic understanding purposes. One limitation of the findings is that only English peer-reviewed scientific articles were examined.

Table 2: Details on material collected from different databases as a basis for this study.

Website	Year	No. of articles	Relevant articles*
SCOPUS	2011-2024	132	16
ScienceDirect	2011-2024	123	15
Google Scholar	2011-2024	91	14
PubMed	2011-2024	456	18
Research Gate	2011-2024	562	12
Web of Science	2011-2024	49	9

**papers out of scope or duplicates were identified and removed during the evaluation process*

Several additional journals, websites, books, reports, and other credible sources were also taken into account for additional information. A detailed table (see Appendix A) is prepared mentioning the key findings of major journals considered for this paper.

Based on the literature review, some interesting things were observed. There has been a sizable amount of research in the past decade which made it easier to access the relatively new information. The scope of this paper was a general overview of biosolids and biochar and later a comparative study, furthered by the analysis of data presented with that of other results found in different studies. However, a large number of such studies were site and feedstock-specific, which made it difficult to have a comparative study. Amongst the three contaminants analyzed for the paper - OPFRs, PFASs, and HMs, there were very few studies done on OPFRs and their behavior in soil. Most of the literature was focused on HMs and their toxicity, bioavailability & mobility, as well as analysis of biochar and biosolids. Few studies have quantified the effect of adding biochar from sewage sludge to soil and their stability, and even fewer on the leaching of HMs and other phenomena in biochar. There is a theoretical possibility of the reaction of HMs with organic contaminants like PFASs and OPFRs within the biosolid itself, however, there was not a single paper in this regard. In the case of biochar, most of the research on biosolids returning to the agricultural fields focused on the impact of short-term biosolids application on soil's ARGs (Qin et al. 2022; Tiwari et al., 2023).

Understanding the findings of the journals, AD coupled with pyrolysis emerges as a viable route for waste treatment, offering benefits such as nutrient recovery and minimal accumulation of potentially toxic elements (PTEs). However, AD alone proves inefficient for the removal of organophosphate flame retardants (OPFRs), highlighting a need for integrated approaches. Moreover, it also leads to significant ecotoxicity. Pyrolysis, particularly at optimal temperatures, demonstrates promising results in adsorbing per- and poly-fluoroalkyl substances (PFASs) to maximum allowable concentrations in wastewater, offering a sustainable solution for their removal. Furthermore, biochar derived from pyrolysis exhibits excellent potential in immobilizing heavy metals (HMs) and reducing their bioavailability in contaminated soil. The efficiency of biochar in HM immobilization can be enhanced through modifications, such as Fe/Mn modification, which significantly improves its effectiveness in removing arsenic (As) from soil and water. Moreover, pyrolysis temperature plays a crucial role in altering the characteristics and behavior of biochar. Higher temperatures lead to a reduction in biochar yield but enhance its stability, surface area, and pH, thereby influencing its sorption capacity and ability to immobilize pollutants. While slow pyrolysis is favored for biochar production, careful monitoring is necessary to prevent adverse effects such as an increase in metal content.

The SS treatment process and the byproducts of transportation have a major effect on climate change (Pradel et al., 2014; Tarpani et al., 2020; Yoshida et al., 2018).

A few interesting results were also found. For instance, according to some research (Duwiejuah et al., 2020; Lu et al., 2015; Wang et al., 2022), contrary to popular opinion, it was found that the Pyrolysis temperature of biochar had little to no significant effect on bioavailable Cd, Pb, and As contaminated soil but rather on feedstock type of biochar. Similarly, there were a few other conflicting results and theories as well. Subsequently, for soil application of biosolids and biochar, most of them were either theoretical information or crop-specific.

Overall, integrated waste treatment approaches involving AD, pyrolysis, and temperature optimization hold promise for pollutant degradation, removal, and stabilization/immobilization. At last, there is a general realization of the lack of extensive study in the matter which was highlighted by several of the papers.

5. Methodology

The different SS management cases that were incorporated to obtain the values for biochar and biosolids are reflected upon. A short overview of the toxins which would be the key topic of discussion throughout along with their concentration is provided. The characteristics of these pollutants that are key for this analysis are overviewed followed by a general overview of the characteristics of biosolid and biochar. The temperature of pyrolysis, which too is the basis of the analysis is discussed from the theoretical and previous studies' perspectives. Finally, a careful consideration of the current regulations regarding these contaminants prevalent in the EU and Norway is elucidated. This ensures the basis for their soil application and its effects on the environment.

5.1. Data collection

The data has been a culmination of a mixed sample collected from 18 treatment plants across Norway over a duration of 5 months from October 2017 to February 2018. The samples were contained in Rilsan bags, stored in a freezer box at a temperature of – 20°C, and sent via overnight mail where they were collected in the laboratory at the end of the sampling period. These samples were analyzed at the Laboratory of Environmental Chemistry and Biochemistry (LECHB) at the Faculty of Fisheries and Protection of Waters (FFPW), University of South Bohemia, České Budějovice (USB CB), and at the water laboratory Povodí Labe, státní podnik, in the Czech Republic (Blytt & Stang 2019).

Collected samples' specifications	Total in Norway	WW treatment processes in project	Total % of Norway
No. of plants	**18**	**6**	**-**
Total Capacity (PE)	**3 004 150**	**1 177 400**	**41 %**
Total sludge (tonnes TS/year)	**62 829**	**25 789**	**39.2 %**

5.2. Sewage sludge management cases

A detailed overview in Figure 1 of the types of treatment taken into consideration provides information on the different results obtained in terms of end products.

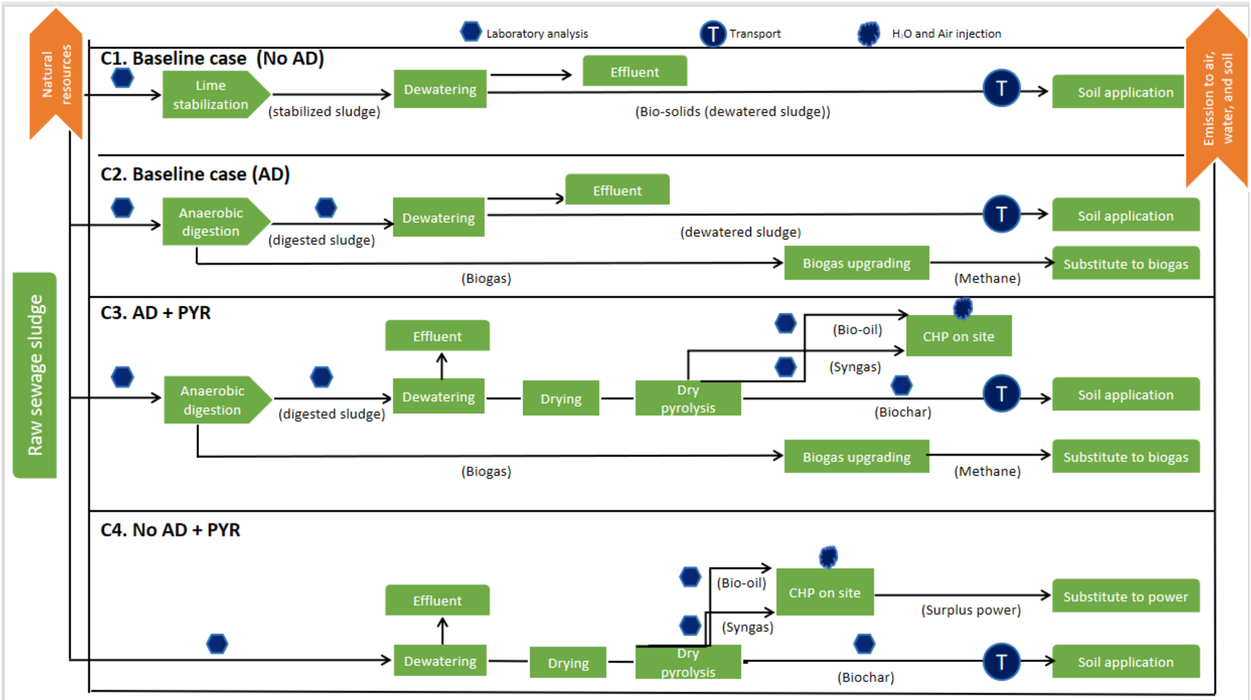


Figure 1: Sewage sludge management cases (AD: Anaerobic digestion; PYR: Pyrolysis; CHP: Combined heat and power)

Here, Case C_1 represents a baseline scenario that involves lime stabilization followed by a dewatering process. The two end products are liquid stream (effluent) which is released to the water bodies, and biosolids, which are spread on land. Case C_2 is similar to C_1, except that it involves AD instead of lime stabilization, which is followed by dewatering. The AD of sludge leads to the production of biogas, which after upgrading transforms into methane, which can be used as a substitute for natural gas, while a byproduct CO_2 is released into the atmosphere. In case C_3, the dewatered sludge after AD is further fed into the dryer to decrease the water content. This makes it suitable for pyrolysis. This dried SS enters the yield reactor, from where three main products are obtained: Biochar, Bio-oil, and pyrolytic gas (syngas). The CHP on site makes use of bio-oil and syngas to further produce energy (heat and power). The energy is partially used to meet the internal energy demand of the treatment plants, and the rest is exported to the energy market (as power, heat is only produced for internal use). The final three emissions to the atmosphere are ash-free combustion gas, ash, and residual stream from the CHP site, while biochar is released in soil. In the case of C_4, the process includes dewatering, drying, pyrolysis, and CHP. It includes the same reactor and operational parameters of C_3, except skipping on the AD part. The main product is biochar, while the output streams emitted to the environment are residual streams, combustion gas, ash, and liquid emissions. The pyrolysis cases were evaluated at both low (500-600°C, averaged results indicated as 550 °C) and high (700-800 °C, averaged results indicated as 750 °C) temperatures.

5.3. Toxins to be examined

The biosolids and biochars produced in the above-mentioned processes contain numerous toxins. A few decades ago, the primary focus of SS was on PTEs like Cd, Hg, Pb, Cu, Zn, & Cr, polychlorinated biphenyls (PCB) and polycyclic aromatic hydrocarbons (PAHs) (Eggen et al., 2019). With time, a wide range of potentially hazardous substances has been identified. For the purpose of this research, however, the focus here is limited to 12 HMs and Organic compounds (PFASs and OPFRs). Let us look categorically at each of these toxin groups from the perspective of soil application.

The term **heavy metal** refers to metals and metalloids that possess relatively high density and biological toxicity, even at the PPB (parts per billion) level. They enter the soil agroecosystem through natural processes (from parent material) as well as anthropogenic activities. These pose a great threat to human as well as plant and animal health by potential accumulation risks through the food chain and adsorption from soil (Li et al., 2019; Rigby et al., 2021). Their chemical speciation determines the metal mobility, toxicity, carcinogenicity, leaching capacity, and bioavailability of biochar for plant uptake (Harcer et al., 2016; Wang, Victor, et al., 2022).

Per- and Polyfluorinated Alkyl Substances (PFAS) is a broad term for synthetic organic (aliphatic) compounds with at least one carbon-fluorine (C-F) bond with functional groups containing oxygen (O), hydrogen (H), nitrogen (N), and sulfur (S) (Pozzebon et al., 2023; Shahsavari et al., 2021). They are also referred to as “forever chemicals” due to their very high chemical and thermal stability (most notably perfluorooctanesulfonic acid (PFOS) and perfluorooctanoic acid (PFOA)), unique physicochemical properties of PFAS compounds (i.e., both hydrophobic and oleophobic behaviors), bioaccumulation potential, and resistance to environmental degradation[12] (Bamdad et al., 2022; Bolan, Sarkar, Yan, et al. 2021; Pozzebon et al., 2023; Scheringer 2023; Shahsavari et al., 2021; Thoma et al., 2022). PFAS have been traditionally used since 1950s in fabrics and industrial materials with nonstick and oil/water repellent properties, such as carpets, food packaging, Teflon coatings, and fire-fighting foams & sprays (Bamdad et al., 2022; Blytt & Stang 2019; Morales et al., 2023; Pozzebon et al., 2023; Rigby et al., 2021; Thoma et al., 2022). There are currently about 8000 different types of known PFAS in the market (Bamdad et al., 2022; Blytt & Stang 2019; Thoma et al., 2022). Several studies indicate that they can cause severe health impacts even at ultra-low concentrations due to their ability to bioaccumulate in animals and humans in the lungs, kidneys, liver, brain, and bone tissue, with some findings suggesting immune system dysfunction, cancer, and thyroid hormone disruption. They are even linked to phyto, aquatic, and terrestrial ecotoxicity and biomagnification in different tropic levels (Evich et al., 2022; Pozzebon et al., 2023). Human exposure to PFAS happens through a variety of pathways namely contaminated drinking water,

[12] Even though PFAS have electron-donating groups at the end of their backbone (sulfate and carboxyl, respectively), they are very persistent to biodegradation due to the highly fluorinated and strongly electron-attracting backbone (Eggen et al., 2019).

food[13], inhalation of air and contact with other contaminated media (Pozzebon et al., 2023; Shahsavari et al., 2021). Being usually resistive to degradation in soil and water, they have the ability to undergo disintegration (Shahsavari et al., 2021). Short-chain PFAS have a lower tendency to be absorbed or leached into the soil and bioaccumulate but tend to be more mobile in the environment than the longer-chain (C_8) compounds due to their higher solubility and lower density (Bamdad et al., 2022; Pozzebon et al., 2023; Thoma et al., 2022). Irrespective, both still persist in the environment. There are also some studies showing that the environmental degradation of PFASs creates even more highly persistent end products, usually perfluoroalkyl or perfluoroalkyl(poly)ether acids (PFAAs) (Holmquist et al., 2020). PFAS with sulfonate groups sorb more than carboxylates. Similarly, iron and aluminum oxides also appear to be key parameters for the adsorption of PFAS (Shahsavari et al., 2021). Recent monitoring and research have revealed that when different PFAS occur together, they have a combined detrimental effect (Blytt & Stang 2019). Their existence is even detected in the arctics where PFOS (Perfluorooctanesulfonic Acid) and PFOA (Perfluorooctanoic Acid) are found in the blood of women in Northern Norway and Siberia (Blytt & Stang 2019). However, PFAS's degradation rate and pathways are highly uncertain (Holmquist et al., 2020). As largely considered under "chemicals of intermediate toxicity", PFAS's characteristics of persistency need to be observed with outstanding concern (Harder et al., 2016; Scheringer 2023). The PFASs group analyzed here are further categorized into several subgroups based on their chemical composition: FTS, PFCA, PFSA, and PReFOS along with 5 compounds in the uncategorized list.

Organophosphate flame retardants (OPFRs) refer to a group of chemicals used as flame retardants, plasticizers, anti-foaming agents, and additives in lubricants, hydraulic oils, coatings, floor polishes and adhesives (Bika et al., 2022; Blytt & Stang 2019; Morales et al., 2023; Pantelaki & Voutsa 2019; Yang et al., 2019). These groups consist mainly of inorganic and organic compounds based on halogens, phosphorus, nitrogen, and metallic hydroxides (Cristale et al., 2016). OPFRs have a strong covalent bond with their host compounds but get easily drawn out through deposition, dissolution, leaching, abrasion, infiltration, and volatilization (Bika et al., 2022). They are of rising concern due to their bioaccumulation and long-lasting persistence in the environment and food products (Bika et al., 2022; Morales et al., 2023; Pantelaki & Voutsa 2019). They are not easily biodegradable and hydrolyzed (Cristale et al., 2016). OPFRs have been detected in the atmosphere, dust, sediment, soil, surface water, and even in the Arctic (Yang et al., 2019). A prominent way for them to enter the environmental matrices is through informal e-waste handling facilities which include heating, leaching of acids, and burning materials containing OPFRs (Bika et al., 2022). They have harmful effects on humans and animals. They are carcinogenic, and mutagenic, and lead to neurodevelopmental, endocrine disruption, and fertility challenges (Bika et al., 2022; Kumar et al., 2022; Yang et al., 2019; Zhang et al., 2021;

[13] PFAS transmission to agriculture happens through the application of recycled water from wastewater treatment plants, landfill leachates and biosolids applied to agricultural land. In the agricultural land, they are transported through the root system of plants to other parts. PFAS with higher chain lengths are usually restricted to the roots, whereas shorter chains compounds can extend further (Shahsavari et al., 2021).

Zhang et al., 2023). OPFRs cause phytotoxicity in plants and affect their physiological conditions (Zhang et al., 2021).

Collectively, OPFRs and PFASs are broadly classified under the umbrella term of "emerging pollutants". It is further hypothesized that, when present in mixtures, these emerging pollutants tend to show cumulative interaction, namely the so-called "cocktail effect", making it further difficult for risk analysis (Pozzebon et al., 2023; NEA 2023).

A descriptive list of these toxins is given in Appendix B.

5.4. Concentration of different byproducts

Dewatered sludge typically contains a mixture of organic and inorganic substances, including heavy metals, pathogens, and nutrients, while biochar has more HM content concentrated into the mass or adsorbed to the surface. Concentrations of contaminants vary depending on the wastewater treatment process and the industrial sources contributing to the sludge. The chemical composition of biosolids produced even in the same plant varies with time and season (Paz-Ferreiro et al. 2018).

5.5. Properties to be examined

The properties of biosolids and biochar vary prominently since they depend on factors such as the composition of wastewater, the method of the treatment process, retention time, temperature of pyrolysis, and the age of biosolids and biochar. This variation is evident even within the same treatment plant due to variations in wastewater composition (AWA 2017; Arulrajah et al. 2011; Paz-Ferreiro et al. 2018). The principal properties of biosolids and biochar that are to be examined here are the bioavailability of nutrients, the toxicity of components examined, and the mobility of toxins along with their potential effects on crops. These properties of dewatered sludge are crucial for assessing its environmental impact and potential beneficial uses.

The **bioavailability** of sewage sludge derivatives refers to the extent to which contaminants in biosolids and biochar can be taken up by living organisms. It is influenced by the chemical form of the contaminants, their solubility, and their interactions with soil or other matrices. In the case of organic contaminants, the residence time is also a factor that affects its bioavailability in soil. With the increase in soil pollutant contact time, pollutant bioavailability and extractability decreases as they slowly diffuse into the soil matrix via isomorphic dissolution reactions, thus becoming increasingly inaccessible for biodegradation and bioaccumulation (Pozzebon et al., 2023). The bioavailability of HMs is determined by DTPA[14] extractable fraction (Lu et al., 2015). Biosolids have higher bioavailability as compared to biochar. The **toxicity** of biosolids and biochar is influenced by the presence of heavy metals, organic pollutants, and pathogens.

[14] The DTPA (diethylenetriaminepentaacetic acid) micronutrient extraction method is a non-equilibrium extraction for estimating the potential soil availability of heavy metals.

Contaminants such as Pb, Cd, and persistent organic pollutants (POPs)[15] can pose environmental and human health risks (Rigby et al., 2021). The **mobility** of contaminants in dewatered sludge depends on their physicochemical properties and the environmental conditions. Adsorption to soil particles and other solid matrices can limit the mobility of contaminants (Evich et al., 2022; Kumar et al., 2022). In order to understand the LCIA of biosolids and biochar, the two impacts focused here are **Human toxicity** and **Climate change** (Harder et al., 2016; Pradel et al., 2014; Tarpani et al., 2020; Yoshida et al., 2018).

5.6. Behavior of biosolids and biochar

Apart from their physical structures and chemical compositions, biosolids and biochar also have distinct behaviors in several aspects. Some of those behaviors that affect their soil application are adsorption capacity, moisture content, presence of volatile matter and functional group(s), and fixed carbon.

Adsorption capacity is a crucial aspect of biosolids and biochar, both of which possess remarkable abilities to adsorb contaminants from various environmental matrices. Biosolids typically contain organic matter, microorganisms, and inorganic compounds, contributing to their adsorptive properties. They can effectively sequester HMs, organic pollutants, and nutrients through surface complexation, ion exchange, and precipitation mechanisms. Conversely, biochar, with its high surface area, porous structure (microporosity), and presence of functional groups, demonstrates superior adsorption capabilities, particularly for organic contaminants and nutrients. Relatively high pyrolysis temperatures generally produce biochars that are effective in the sorption of organic contaminants due to an increase in their surface area, microporosity, and hydrophobicity; whereas the biochars obtained at low temperatures are more suitable for removing inorganic/polar organic contaminants by oxygen-containing functional groups, electrostatic attraction, and precipitation. Its carbonaceous nature enhances interactions with contaminants, leading to their immobilization and reduction in environmental availability (Ahmad et al., 2014; Lehmann & Joseph 2015; Li et al., 2017; Paz-Ferreiro et al., 2017; Wang et al., 2022; Zhang et al., 2022).

Biosolids, derived from sewage sludge, typically contain a significant amount of moisture due to their origin from wastewater treatment processes. The moisture content in biosolids can affect their handling, transportation, and storage, as well as their potential for biological degradation and odor generation. On the other hand, biochar, produced through pyrolysis or carbonization of biomass, typically has a lower moisture content compared to biosolids (Ahmad et al., 2014; Lehmann & Joseph 2015; Li et al., 2013; US EPA 2019).

The presence of volatile matter in both biosolids and biochar refers to the organic compounds that can be released at relatively low temperatures. In biosolids, volatile matter can contribute to

[15] Persistent organic pollutants (POPs) are chemicals of global concern due to their potential for long-range transport, persistence in the environment, ability to bio-magnify and bio-accumulate in ecosystems, as well as their significant negative effects on human health and the environment (ECHA 2022).

odor issues, and may undergo microbial degradation over time. In contrast, biochar's volatile matter content is reduced during the pyrolysis process, resulting in a more stable carbonaceous material with improved properties for soil amendment and carbon sequestration (Ahmad et al., 2014; Lehmann & Joseph 2015; Li et al., 2013; Li et al., 2017).

Functional Groups in biosolids mainly comprise carboxylic acids, phenols, amines, and sulfhydryl groups. They contribute to the complex nature of biosolids and their ability to retain nutrients and contaminants in soil environments. Carboxylic acids, for example, can facilitate cation exchange and enhance the retention of positively charged ions like Ca, Mg, and HMs. Phenolic groups can influence the sorption of organic pollutants and HMs through various mechanisms, including hydrogen bonding and π-π interactions. Biochar contains a range of functional groups derived from the precursor feedstock and pyrolysis conditions. Common functional groups in biochar include carboxyl (-COOH), hydroxyl (-OH), and carbonyl (C=O) groups, among others. These functional groups contribute to biochar's sorption capacity, surface chemistry, and reactivity. For instance, carboxyl groups can act as sites for metal complexation and cation exchange reactions, while hydroxyl groups can participate in hydrogen bonding and water retention. These functional groups mediate crucial processes such as nutrient cycling, contaminant sorption, and microbial activity, highlighting their importance in environmental remediation and soil fertility management (Ahmad et al., 2014; Lehmann & Joseph 2015; Li et al., 2013; Li et al., 2017; US EPA 2019).

Fixed carbon refers to the stable carbonaceous material remaining in biosolids and biochar after volatile matter and other organic compounds have been driven off through processes such as pyrolysis or combustion. In biosolids, it represent the organic matter that remains after the removal of volatile components during the treatment process in wastewater treatment plants. This residual carbonaceous material contributes to the nutrient content and organic matter content of biosolids, influencing their suitability for soil amendment and agricultural use. Meanwhile, in biochar, fixed carbon refers to the carbon-rich structure that remains after the biomass feedstock undergoes pyrolysis or carbonization at high temperatures in the absence of oxygen. The fixed carbon content of biochar is an important indicator of its stability and potential for long-term carbon sequestration when applied to soils. Higher fixed carbon content in biochar indicates greater resistance to decomposition and mineralization, leading to enhanced soil carbon storage and improvement of soil health and fertility over time. Moreover, it helps in climate change mitigation by making a stable carbon sink as compared to release in the atmosphere in free/elemental form (Ahmad et al., 2014; Lehmann & Joseph 2015; Li et al., 2017; Paz-Ferreiro et al., 2017).

5.6.1. Temperature aspect of pyrolysis

Pyrolysis temperature greatly impacts the chemical properties of biochar (Zhang et al., 2022). Increasing pyrolysis temperature (within the range of 300 - 700 ℃) significantly decreases the yield[16] of biochar (Janu et al., 2021; Kundu et al., 2021). Molar ratios H/C, O/C, and N/C are reduced when biochar is produced at higher temperatures, indicating an aromatization of its structure and volatilization of matters. Biochar produced at a low temperature (300–400 °C) was acidic while the solid product shifted to an alkaline pH at a high temperature (700 °C). A study showed that the biochar pH value increased from 5.87 to 10.50 and the specific area grew from 5.26 m2/g to a maximum of 15.23 m2/g with pyrolysis temperature rose (Li et al., 2017; Zhang et al., 2022). A different study concluded that EC (exchangeable cations) increased slowly with temperatures of up to 500 °C but halved at higher temperatures. The concentration of K, P, and micronutrients increases significantly with temperature, while N-concentration tends to decrease. Several VOCs (Volatile organic compounds) content also decreases significantly with temperature rise. BET surface area[17], pH, porosity, and total concentration of Cu, Ni, Zn, Cd, and Pb as well as EC and CEC (cation exchange capacity) in biosolids biochar increase with temperature (Paz-Ferreiro et al. 2018; Bilias et al., 2021; Zhang et al., 2022). The abundance of functional groups (mainly those composed of lignocellulosic materials) in biochar decreases with increasing temperature due to a higher degree of carbonization (Li et al., 2017). A high temperature of pyrolysis can immobilize more HMs in a biochar mix, converting them into stable forms. (Wang, Victor, et al., 2022; Zhang et al., 2022). The study showed that the leaching rate was reduced by 5.5% on average with high temperatures (Zhang et al., 2022). However, a study conducted showed that Pb, Ni, Cd, and Cr, their respective leaching rates had no significant difference between 300 and 700 °C, while for Zn, there was no significant difference during the range of 400 and 700 °C (Lu et al., 2015). Pyrolysis temperature also affects biochar's affinity towards functional groups. A study showed that a pyrolysis temperature of 600 °C leads to a partial and a 750 °C to a nearly complete loss of biochar surface functional groups (Janu et al., 2021).

Increasing the temperature of pyrolysis also maximizes the yield of the gaseous fraction and decreases the solid fraction, while the liquid fraction remains constant. The heating rate is also an important parameter, but only for temperatures below 650 °C. PFAs are partially or completely destructed[18] (Kumar et al., 2023; Patel et al., 2020; Paz-Ferreiro et al. 2018). OPFRs are seen to achieve REs (Removal efficiency) ~100% at pyrolysis temperatures ≥ 500 °C (Sørmo et al. 2023).

Research was conducted to observe the characteristics of biochar with increasing pyrolysis temperature. The result compiled in Table 3 shows that the percentage of biochar yield decreased

[16] Biochar yield refers to the amount of biochar produced relative to the starting biomass feedstock used in the pyrolysis or carbonization process.It is typically expressed as a percentage and calculated by dividing the mass of biochar obtained by the mass of the initial biomass feedstock, multiplied by 100 (Kundu et al., 2021; Zhang et al., 2022).

[17] The specific surface area of a powder is estimated from the amount of nitrogen adsorbed in relationship with its pressure, at the boiling temperature of liquid nitrogen under normal atmospheric pressure. The observations are interpreted following the model of Brunauer, Emmett, and Teller (BET Method) (Naderi 2015).

[18] According to experiments done in Canada, most of the PFAS compounds are observed below detection limits in biochar and other byproducts of biosolid pyrolysis at 500-700 ℃. The elemental carbon mass fraction decreases with increasing pyrolysis temperature for the biochar of all biosolids (Bamdad et al., 2022).

with the increase in temperature, while pH value, surface area, pore volume, and C, H, O, and N content decreased (Source: Cheng et al., 2023).

Table 3: Characteristics of biochar at different temperatures (Source: Cheng et al., 2023)

Pyrolysis temp (°C)	**Heating rate (°C/min)**	**Yield (%)**	**pH**	**C (%)**	**H (%)**	**O (%)**	**N (%)**	**Surface area (m^2/g)**	**Pore volume (cm^3/g)**
300	7	70.1	6.8	30.72	3.11	11.16	4.11	4.5	0.01
400	7	57.4	6.6	26.62	1.93	10.67	4.07	14.1	0.02
500	7	53.8	7.9	20.19	1.08	9.81	2.84	26.2	0.04
600	7	51.2	8.3	24.76	0.83	8.41	2.78	35.8	0.04
700	7	50.3	8.1	22.04	0.57	7.09	1.73	54.8	0.05

However, water-soluble concentrations of mineral components (K, Ca, Mg, and P) behave differently. A study conducted showed that their concentration increased when heated at 200 ℃ but decreased beyond that temperature. The reason hypothesized was the possible increase in crystallization as evidenced by the formation of whitlockite [(Ca, Mg)3(PO4)2] or incorporation into the silicon structure at a pyrolysis temperature of 500 C, which is less soluble (Li et al., 2017).

5.7. LCIA

LCA is applied in order to quantify the environmental burdens and benefits of treating and utilizing SS as its land application is a major contributor to global warming and eutrophication[19] (Yoshida et al. 2018). In the scope of this paper, the term human toxicity is broadly classified into carcinogenic and non-carcinogenic based on the type of pollutant. For instance, the impacts caused by Zn and Cu would come under non-carcinogenic toxicity, while Hg and Pb impacts would be categorized under carcinogenic toxicity. Meanwhile, gaseous emissions associated with the land application of SS (e.g. CH_4, N_2O, NH_3, CO_2, SF_6) are the main contributors to climate change (Harder et al., 2016; Pradel et al., 2014; Tarpani et al., 2020; Yoshida et al., 2018). The SS treatment process significantly affects climate change. This included their energy requirements, particulate matter formation, and GHG emissions (Pradel et al. 2014). Notingly, the energy substitution via biogas utilization in the SS treatment process contributes significantly to savings in climate change (Yoshida et al., 2018).

5.8. Current regulations

[19] It is the process of rapid nutrient accumulation in water bodies resulting in an exponential growth of microorganisms that may deplete the dissolved oxygen (DO) of water (Chislock et al. 2013).

The implementation of the Urban Waste Water Treatment Directive 91/271/EEC[20] in European Member States have increased the amounts of biosolids requiring disposal. The Sewage Sludge Directive 86/278/EEC[21] promotes the use of biosolids while preventing detrimental effects on the lives of plants, animals, and humans. For instance, it prevents the direct application of untreated sludge on agricultural land. Thus, the sludge has to be biologically, chemically, or thermally treated, or any other appropriate processes to reduce its hazardous effects. This was first implemented in attempts to reduce discharges of P and N into the coastal area at the Swedish border to Lindesnes. This originated from the North Sea Protocol, signed in 1987 (Paz-Ferreiro et al. 2018; Shaddel et al., 2019; Statistics Norway 2023). The quality of wastewater sludge used as a fertilizer[22] is regulated by Regulation no. 951 of July 4th, 2003 on fertilizer products of organic origin. The regulation although not establishing any limits for organic pollutants (only HMs), does mention the phrase *"organic pollutants, pesticides, antibiotics/chemotherapeutics or other manmade organic substances in quantities that can damage public health or the environment when used"* (Blytt & Stang 2019). ISO 14040 is used as a standard for LCA. Furthermore, there are recommendations provided in the International Reference Life Cycle Data System (ILCD) handbook (Yoshida et al. 2018).

The regulations on PFASs and OPFRs are primarily governed by the Registration, Evaluation, Authorisation, and Restriction of Chemicals (REACH)[23] regulation and the Stockholm Convention[24] on Persistent Organic Pollutants (POPs). Due to environmental concerns, the production and use of long-chain (≥ 8 carbons) PFAS in North America, Europe, and Australia were voluntarily phased out in the early 2000s and replaced with shorter-chain PFAS (Pozzebon et al., 2023). At the international level, persistent organic pollutants (POPs) are regulated in the Stockholm Convention[25]. Since there is very limited information on the interaction of PFAS with the environment due to the difficulty associated with its detection and their sources of emission, the only ones that are strictly regulated in this group are PFOS[26] (since 2009) and PFOA (since 2019) (Antunes et al., 2021; Blytt & Stang 2019; Rigby et al., 2021; Shahsavari et al., 2021). The current limit value is set at 0.025 mg/kg for PFOA including its salts, and at 1 mg/kg for the

[20] As part of the European Economic Area (EEA-agreement), Norway has implemented EU's directives on Urban Waste Water Directive (91/271/EEC and 98/15/EEC) into Norwegian law (Statistics Norway 2023).

[21] Biosolid land application in the EU is controlled by the Sludge Directive of 1984. However, pollutants such as organic compounds and pathogens were not included in it.Most pathogen standard limits allowable in biosolids are less than or equal to 1000 CFU per gram dry matter (CFU/gDM); this has been adopted in many other states in Europe today (Lekan et al., 2023).

[22] The draft regulation has introduced the term "sludge-based fertiliser product" (Blytt & Stang 2019).

[23] REACH Regulation (EC) No 1907/2006 aims to ensure a high level of protection of human health and the environment from the risks that can be posed by chemicals (ECHA 2022).

[24] The Stockholm Convention is a global treaty aimed at eliminating or restricting the production and use of persistent organic pollutants, including certain PFAS compounds (United Nations 2009).

[25] Pertaining to the hazardous properties of pentaBDE and octaBDE of BFR (Brominated Flame Retardants) group, they were added in the POPs group and have since seen the elimination of production and use in the EU and North America (Martín-Pozo et al., 2019).

[26] PFOS have been added to the priority list of the Norwegian authorities in 2002. PFOA was then added in 2007 (Blytt & Stang 2019). EU regulations 2020/784 decided to add PFOA to the list of chemicals that are required to be prohibited for production, use, import, and export (European Parliament 2020).

individual PFOA-related compounds or a combination of those compounds (European Parliament 2020). The EU Regulation 2019/2021 states that concentrations of PFOS equal to or below 10 mg/kg (0.001 % by weight)[27] where it is present in substances or in mixtures. The four PFASs, PFOA, Perfluorononanoic acid (PFNA), PFHxS, and PFOS are being considered by the EC for inclusion in the regulatory framework on food contaminants (Rigby et al., 2021).In Norway, the regulations regarding HMs, PFAS, and OPFRs are primarily based on the EU regulations, as Norway is a member of the European Economic Area (EEA) and thus adopts many EU laws and regulations. Additionally, there are Norwegian national regulations and guidelines on marketing fertilizing products which is applicable to sewage sludge as it comes under the term "organic fertilizers". According to EU 2019/1009, there is a limit value for As, Cd, Cr/Cr(VI), Cu, Pb, Hg, Ni, and Zn (Eggen et al., 2022; Whipps & Tornes 2018). In Norwegian regulations, organic fertilizers/soil improvers are divided based on the content of the PTEs - Cd, Pb, Zn, Cu, Ni, Cr, and Hg (FOR-2003-07-04-951). Maximum level (ML) for As has also been suggested, but it is under review and currently not approved. The quality class determines the restrictions regarding application to soil. PFHxA, PFHxS, PFBS, C_9-C_{14} PFCAs, and PFSAs have also been added to the national priority list which is meant to ensure the maximum possible discharge (NEA 2023; Milijøstatus 2019). The different quality classes and corresponding maximum application to agricultural soils allowed are given in Table 4 while, the maximum limit (ML) for the different PTEs in the different quality classes is given in Table 5.

Table 4: Quality classes for organic fertilizers and corresponding maximum allowed application to agricultural soils (Source: Eggen et al., 2022; Whipps & Tornes 2018)

Quality class	Applications	Max. amount
0	Agriculture -private gardens and green amenity areas	No restrictions (adopted to agronomical properties)
I		40 tonnes DW ha^{-1} 10 $years^{-1}$
II		20 tonnes DW ha^{-1} 10 $years^{-1}$
III	Green amenity areas	5 cm thickness blended with local soil

Quality class	0	I	II	III
mg kg^{-1} DW				
As*	5	8	16	32
Cd	0.4	0.8	2	5
Cr	50	60	100	150

Table 5: Present MLs for different PTEs in different quality classes for different fertilizers and soil mixtures (Source: Eggen et al., 2022; Norwegian Agricultural Agency 2018; Whipps & Tornes 2018)

[27] This is applied to *"concentrations of PFOS in semi-finished products or articles, or parts thereof, if the concentration of PFOS is lower than 0,1 % by weight calculated with reference to the mass of structurally or micro-structurally distinct parts that contain PFOS or, for textiles or other coated materials, if the amount of PFOS is lower than 1 μg/m2 of the coated material"* (EU Regulation 2019/2021).

Cu	50	150	650	1000
Pb	40	60	80	200
Hg	0.2	0.6	3	5
Ni	20	30	50	80
Zn	150	400	800	1500

*Suggested Value

Table 6: Maximum HMs concentration allowed in soils treated with sewage sludge (mg/Kg DW) (Source: Eggan et al., 2022; Paz-Ferreiro et al. 2018; Whipps & Tornes 2018)

Metals	Cr	Ni	Cu	Zn	Cd	Pb	Hg
EU	100-150	30-75	50-140	150-300	1-3	50-300	1-1.5
Norway	100	30	50	150	1.0	50	1

Table 7: Limit values for heavy metal concentrations in biosolids for use in agriculture (mg kg−1 of dry matter) (Source: Healy et al., 2016; Paz-Ferreiro et al. 2018)

	Cr	Ni	Cu	Zn	Cd	Pb	Hg
EU	-	300-400	1000-1750	2500-4000	20-40	750-1200	16-25

It is worth noting that the missing HMs mentioned in the list of toxins for this paper scope have not been evaluated and thus not included in the report. To this day, there is no international standards for Sb or Sn in biosolids for reuse in agriculture (Healy et al., 2016).
According to the Norwegian Geological Institute, the recommended normative values for PFOS and PFOA are above the current limit of quantification (LOQ) used by commercial chemical laboratories operating in the Norwegian market (0.1 µg/kg d.w.) (NGI 2020).
This information provides insight into the regulatory framework, permissible pollutant concentrations, and ongoing discussions around emerging contaminants like PFASs and OPFRs. Understanding these regulations is crucial for assessing the environmental impact and safe use of biosolid-derived products in soil management practices.

6. Results and Discussion

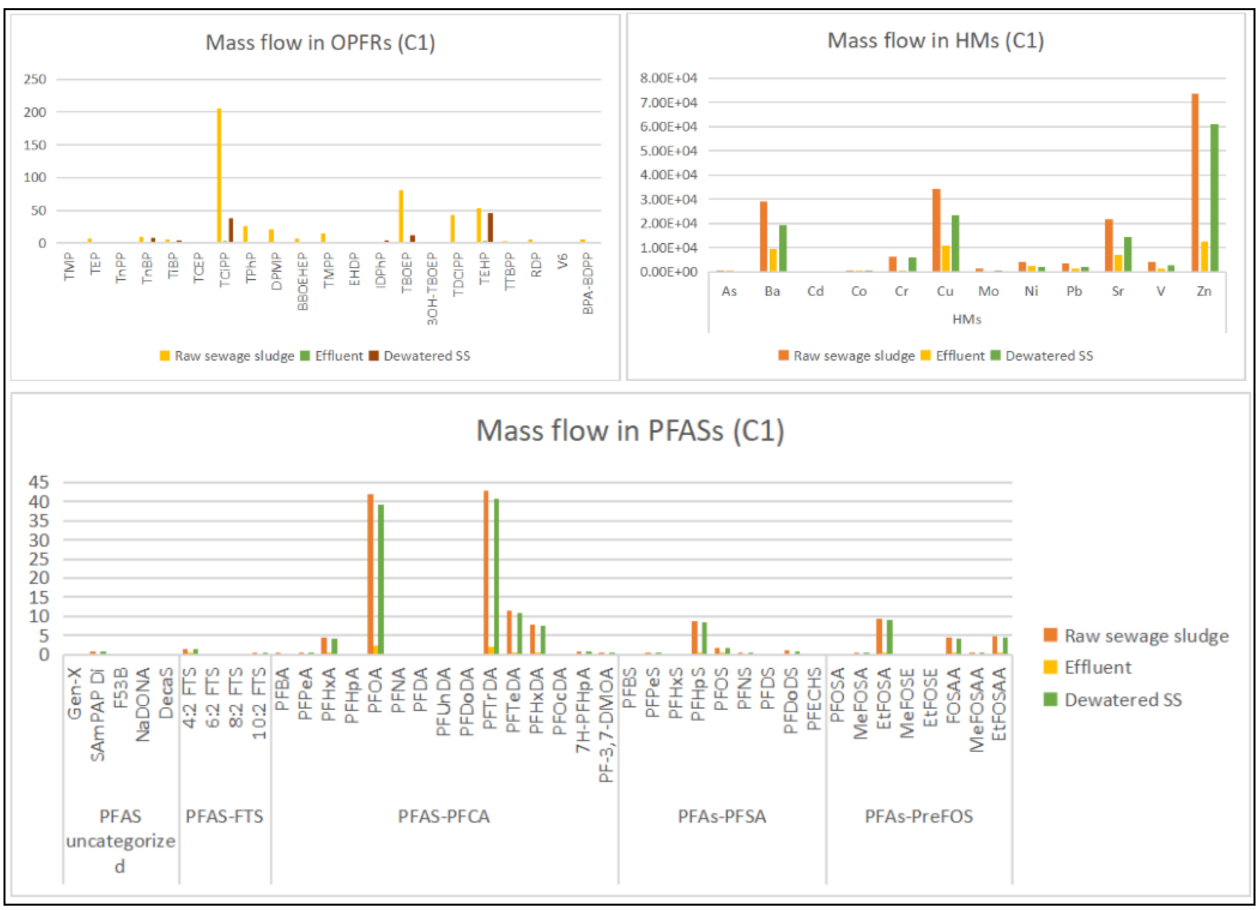
In this section, a descriptive data analysis is done. Based on the analysis, an observation of the results is sketched comparing the different SS treatment processes. The analysis is further related to the Norwegian context. It is followed by an overview of the effects of biosolids and biochar in soil. The relatively more viable option, biochar is described based on its benefits in soil application. Finally, some recommendations are postulated based on the understanding of previous studies done, which is followed by concluding the findings and hypotheses.

6.1. Data analysis

The data procured is presented with its analysis for different aspects including lime stabilization, AD, and pyrolysis at different temperatures. Several factors such as Removal efficiency of organic pollutants, biochar yield, and retention rate of HMs are calculated and their results are discussed to analyze the various processes of SS treatment and their impacts on the fate of pollutants. The three main foci of comparison are: (i) amongst the two pretreatment processes (lime stabilization and AD), which one is more efficient, (ii) what role does pyrolysis play in the characteristics of biochar production, and (iii) what is the role of temperature increase in pyrolysis process on the fate of pollutants removal.

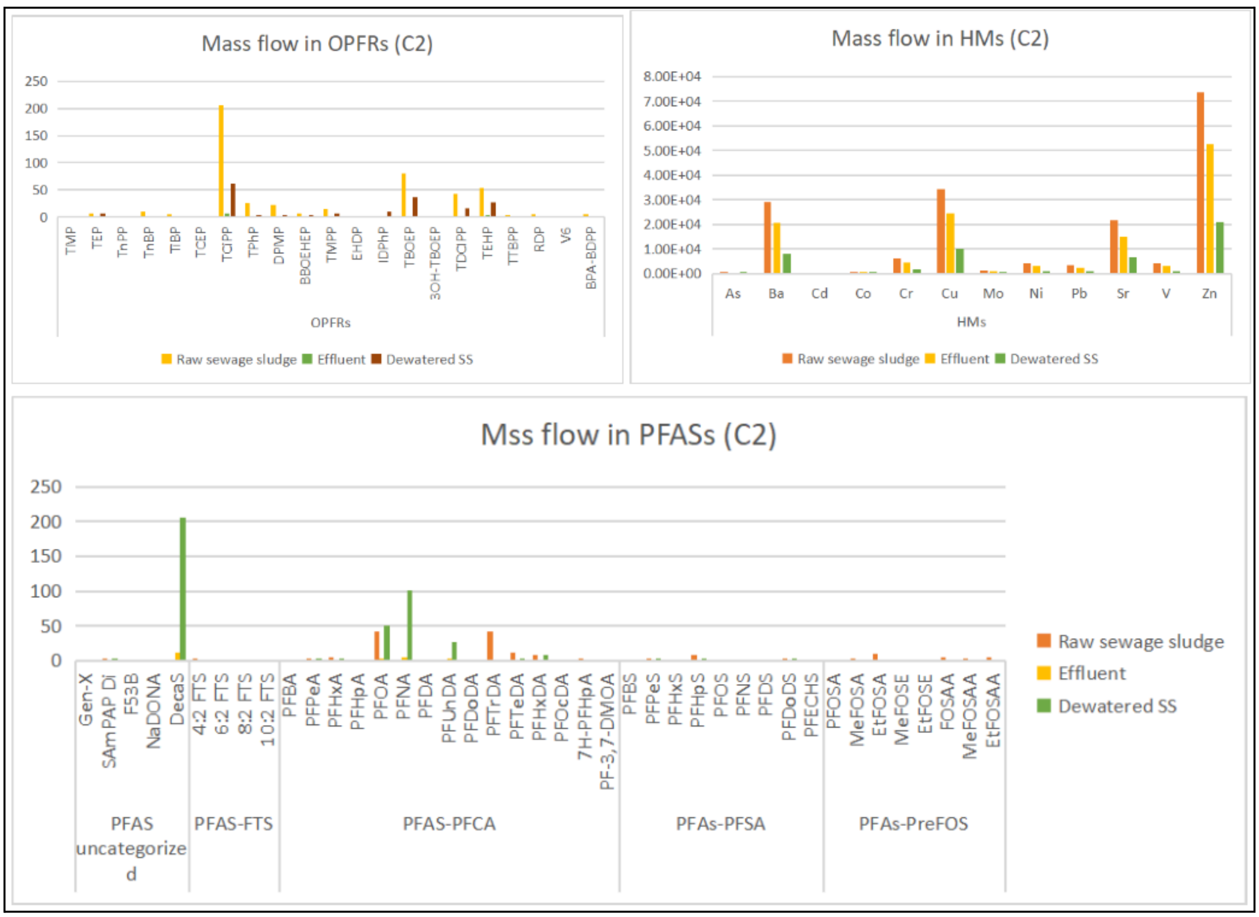
Based on the data available (see Appendices C and D for the data) for the 4 cases of SS treatment processes, graphs are prepared showing the mass flows of OPFRs, PFASs, and HMs. These graphs provide an overview of the different processes and their effects on the concentration of these substances. For each of the graphs, the end products are first compared with the raw SS to understand their degree of degradation, and then amongst the different byproducts to understand their mass flow.

Figure 2: Mass flow of contaminants in C_1



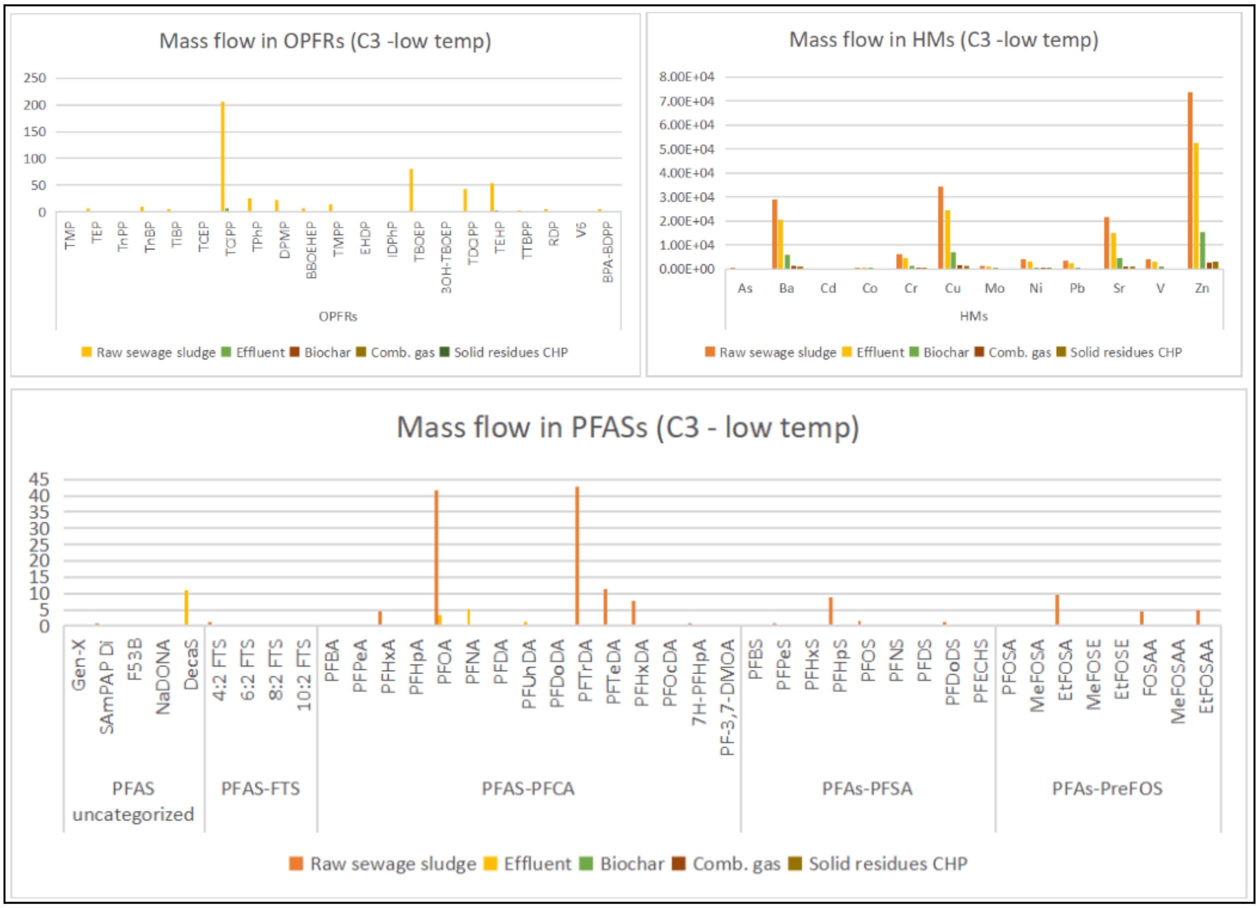
In C_1, lime stabilization, and dewatering are the two processes that the sewage sludge goes through. The mass of the raw SS was measured at 250 kg-dw/d, while the mass of the two byproducts - effluent and biosolid was found to be 12.5 and 237.5 kg-dw/d respectively (see Appendix C). This indicates that there was no release of chemicals in gaseous or any other form due to the reaction with lime. For the OPFR group, most of the ones that were present in the sewage sludge were reduced in the total mass of end products. On the other hand, PFASs show very little to no reduction in mass implying that lime stabilization has no reactionary effect on PFAS. In the case of HMs, there is no degradation of mass as well. Also, it seems that the process of dewatering removes a significant amount of OPFRs and PFAS from the effluent signaling that lime causes precipitation and accumulation of these compounds in the biosolid where these get concentrated. In the case of HMs, it's the same case. The exception is Ni, which sees more concentration in effluent than in dewatered sludge. This signifies that lime stabilization also helps in the accumulation of HMs in biosolid.

Figure 3: Mass flow of contaminants in C_2



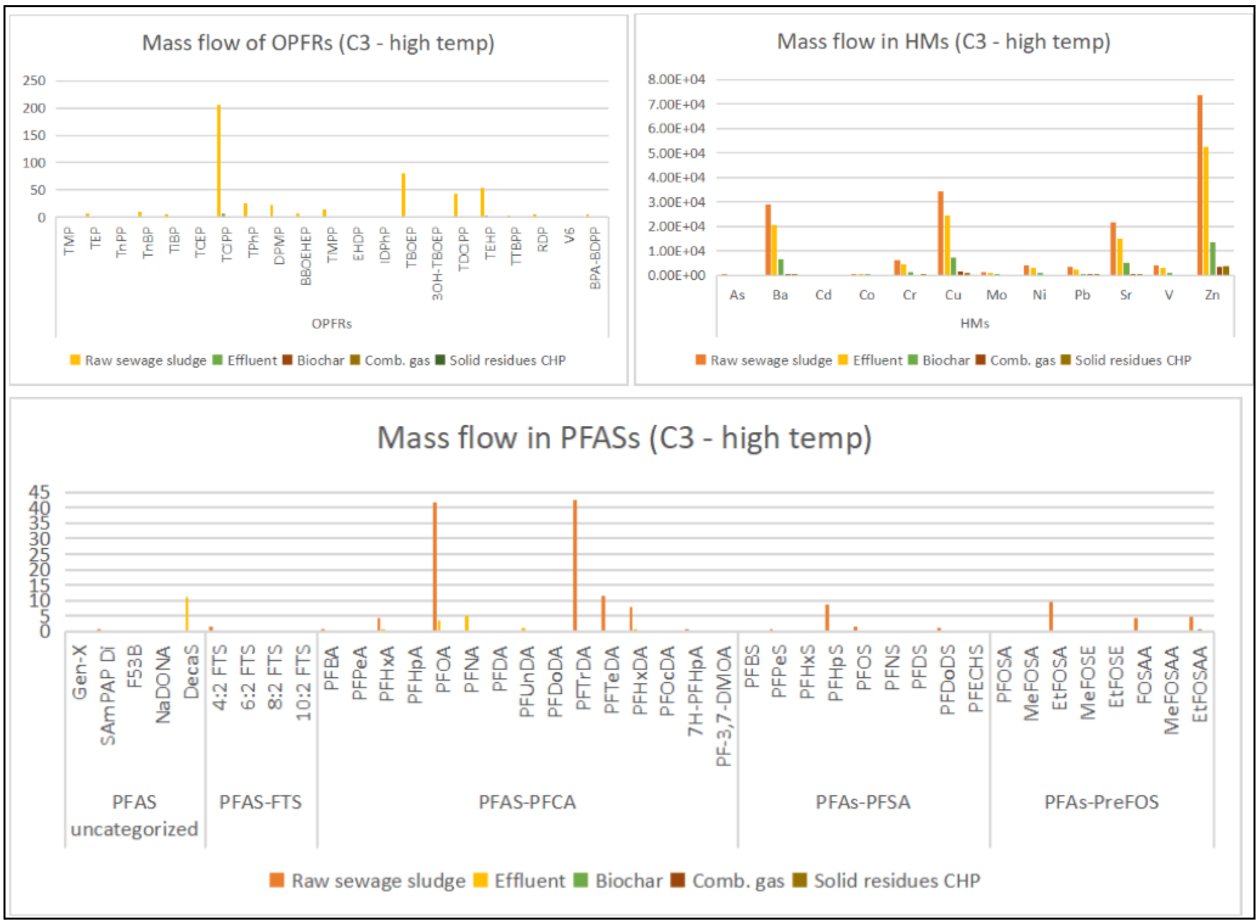
In case C_2, which sees AD followed by dewatering, the sum total mass of effluent and biosolid is 136.91 kg-dw/d (see Appendix C), which is less than that of raw sewage sludge. This indicates the chemical reactions occurring in AD which lead to a reduction in total mass. For the OPFR group, most of them get significantly reduced. PFASs, being a much more resilient compound group, see little to no reduction in mass except for a few exceptions where they get reduced or completely dissipated. Interestingly, in the case of DecaS, PFOA, PFNA, and PFUnDA, it rather emerges in gigantic amounts in effluent and biosolid. This could be due to the fact that AD changes the chemical composition of the sludge leading to the creation of some PFAS compounds. For HMs, the effluent has a much higher mass than biosolids. This shows that AD while leading to the sedimentation of PFASs and OPFRs in biosolid, is not as efficient in immobilizing HMs within the solid mass.

Figure 4: Mass flow of pollutants in C_3 at low pyrolysis temperature



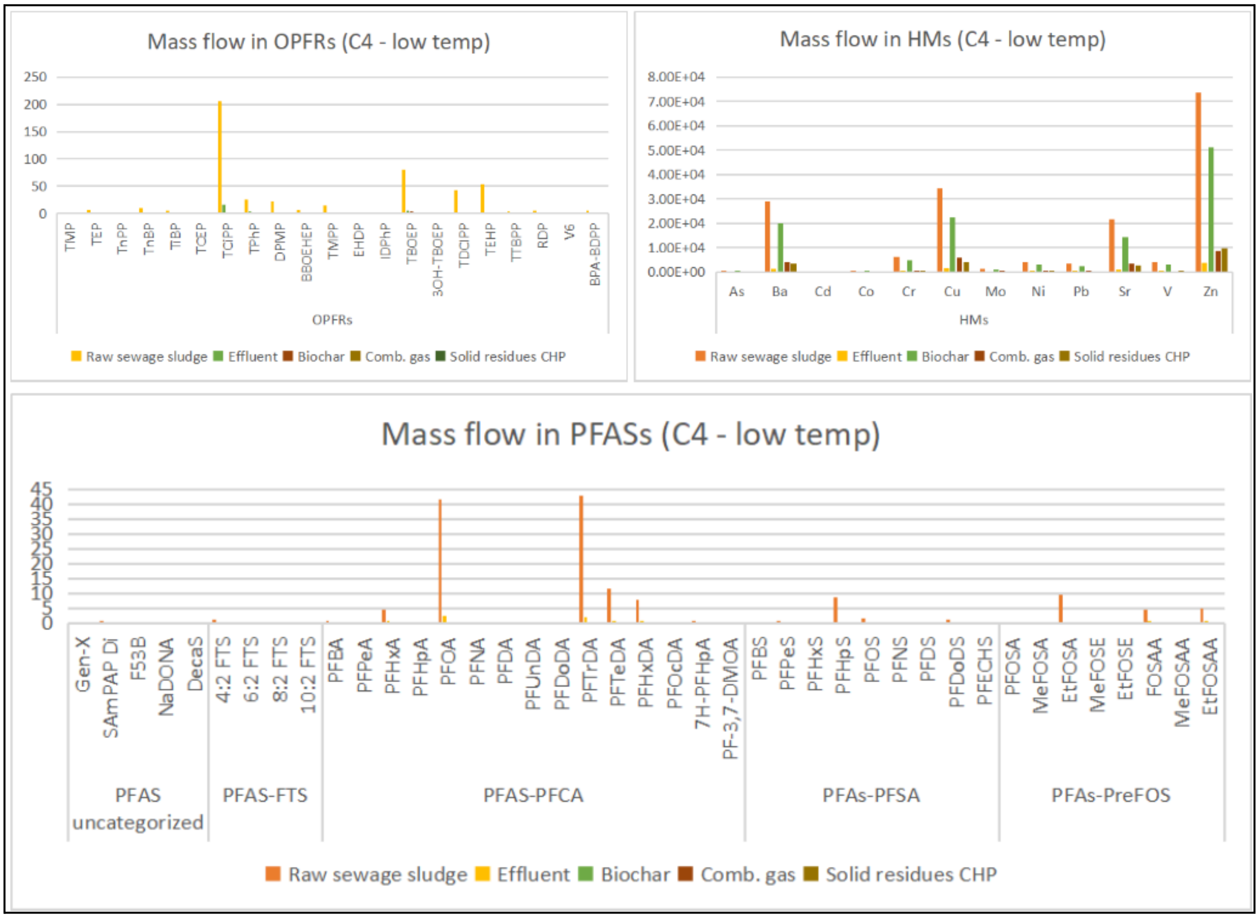
For case C_3, at low pyrolysis temperature, where the sludge undergoes AD followed by dewatering and pyrolysis, the sum of the mass of effluent, biochar, and CHP is much less (87.35 kg-dw/d, see Appendix C) than that of raw SS, which clearly shows the degradation/destruction of biomass. However, the mass of combustion gas is way higher. Upon exploration of the process, it is revealed that there is an injection of a large amount of H_2O and Air in syngas in the boiler to create the combustion gas (see Figure 1). This causes an increase in the mass flow of combustion gas. This is the case for C_3 at high pyrolytic temperatures and for C_4 at low and high pyrolytic temperatures as well. It is also notable that the total mass of biochar produced is lesser than the mass of biosolids in the case of C_1 and C_2. Thus it proves that pyrolysis reduces the mass of SS. It also plays an important role in reducing the HOC content in biochar. This is evident from the total value of OPFRs as compared to that in raw SS. This backs the idea that AD alone is not sufficient for the removal/elimination of OPFRs (Biel-Maeso et al., 2019; Castro et al., 2023). PFASs follow the same trend except for a few exceptions. DecaS, PFOA, and PFNA see a rather prominent emergence in effluent. This again could be inferred from the contribution of AD and its facilitation of chemical reactions leading to the formation of some PFASs. Notingly, in HMs, the mass flow seems to be distributed amongst the different byproducts, mainly in effluent. Comparing this trend of similarity to C_2, it could be postulated that AD leads to the leaching of HMs in effluent.

Figure 5: Mass flow of pollutants in C_3 at high pyrolysis temperature



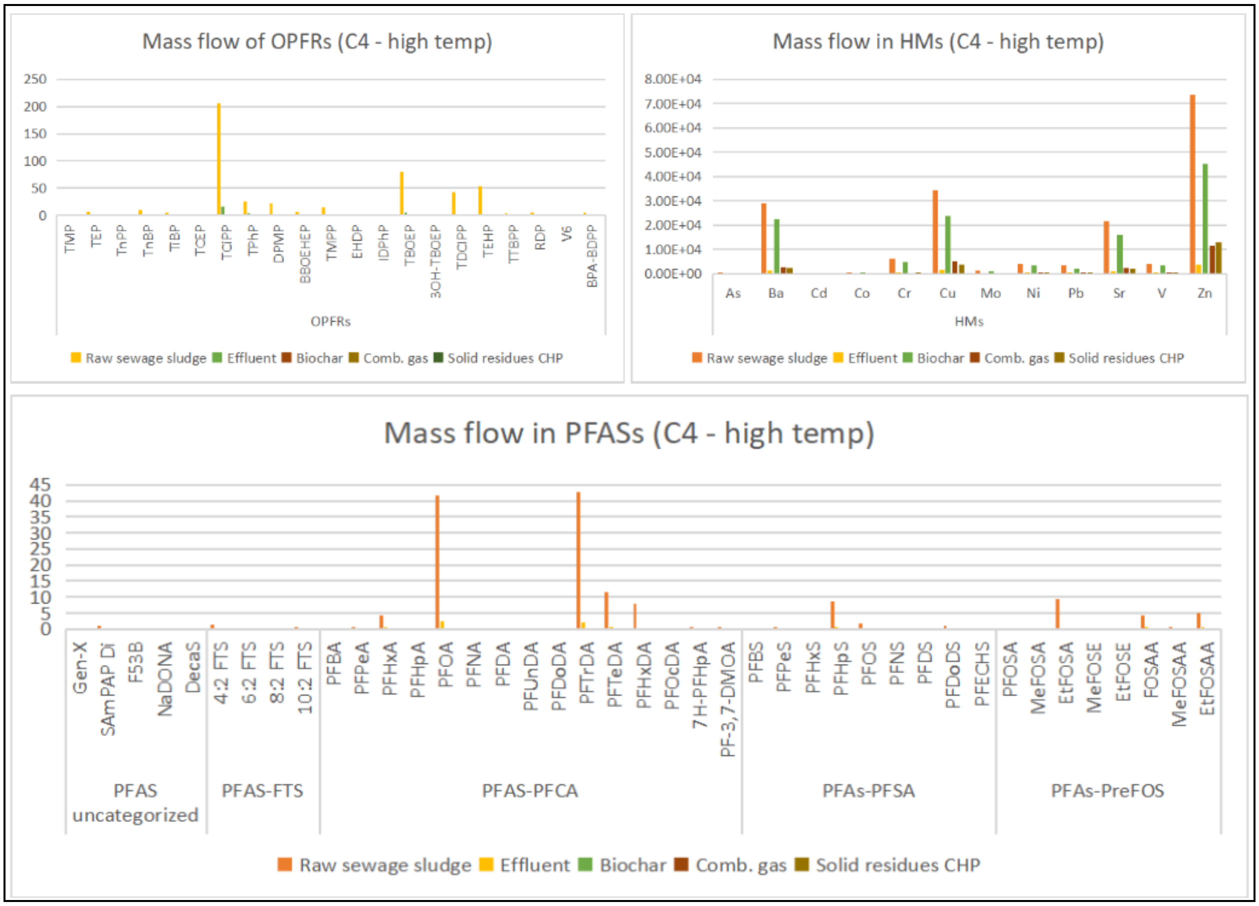
A similar trend in the reduction of total mass can be seen in the case of C_3 at high pyrolysis temperature. For OPFRs and PFASs, it follows the same trend as that in low pyrolysis temperatures. There is one exception of EtFOSAA from the PFAS group, where biochar contains some additional amount as compared to the low-temperature pyrolysis case (see Appendix C). An explanation of this could be the synthesis of new compounds from the existing HOCs through the high pyrolysis temperature. HMs do not show any difference in trend within themselves during the process or while comparing the results to that of the same process conducted at low temperatures. This result has been seen in other experiments as well (Lu et al., 2015). This signifies that there is no notable change in the HMs mobilization with high pyrolytic temperature.

Figure 6: Mass flow of contaminants in C_4 for low pyrolysis temperature



In case C_4, where the SS directly undergoes pyrolysis at low temperature without AD, the mass of biochar is less than that of raw SS but more than that of C_3 at low temperature. Lack of AD in this process could be the reason for it. This signifies that AD has a major impact on the reduction of the overall biomass of SS. Similar to that of C_3, the effluent sees a sizable reduction in their mass of OPFRs and PFAS, while biochar observes no residue except for a small amount in the case of TCiPP and TBOEP. HMs however see most of their mass being immobilized in biochar fraction. Combustion gas and CHP contain only HMs and no OPFRs and PFASs.

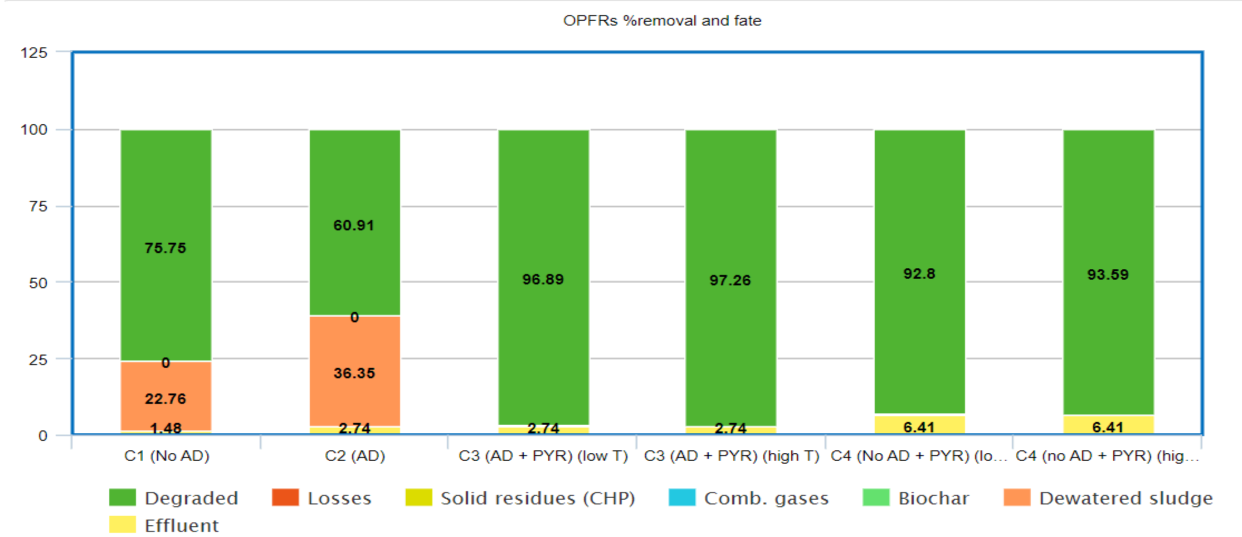
Figure 7: Mass flow of contaminants in C_4 at high pyrolysis temperature



In C_4 with high pyrolysis temperature, there is barely any residue of HOCs in biochar. However, continuing the trend at low pyrolysis temperatures, HMs have a significant amount of mass in biochar fraction. There is also not much of a difference visible in the figure as compared to that for C_4 with low pyrolysis temperature.

For further analysis of these pollutants and their removal fraction in different end products, a cumulative value of fraction in each step and end product is calculated. They are herein depicted in a stocked 2D-column graph for OPFRs, HMs, and PFASs respectively. Figures 8, 9, and 10 depict the %removal and fate of different contaminants.

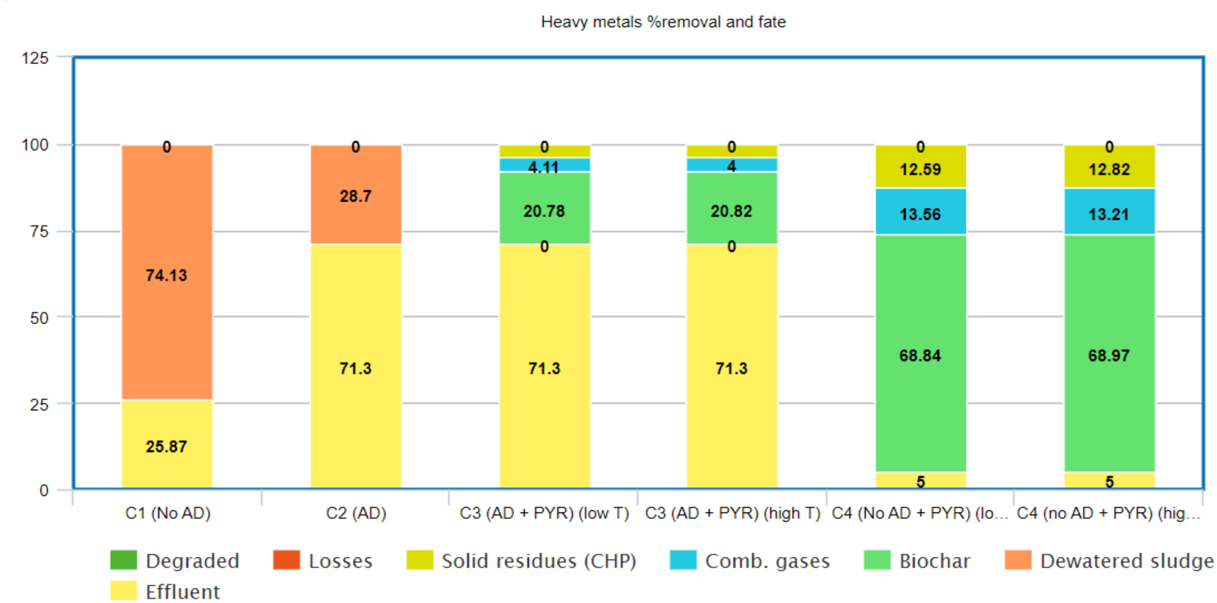
Figure 8: OPFRs % removal and fate



	C_1 (No AD)	C_2 (AD)	C_3 (AD+PYR) (low Temp)	C_3 (AD+PYR) (high Temp)	C_4 (No AD+PYR) (low Temp)	C_4 (No AD+PYR) (high Temp)
Degraded	75.75	60.91	96.89	97.26	92.80	93.59
Losses	0	0	0	0	0	0
Solid residues (CHP)	0	0	0	0	0	0
Comb. gases	0	0	0	0	0	0
Biochar	0	0	0.37	0	0.79	0
Dewatered sludge	22.76	36.35	0	0	0	0
Effluent	1.48	2.74	2.74	2.74	6.41	6.41

A substantive amount of OPFR removal happened in form of degradation. This includes the process of lime stabilization in the case of C_1, AD in the case of C_2, AD + Pyrolysis for C_3, and pyrolysis in C_4. Comparing C_1 and C_2, it can be deduced that AD leads to lesser degradation of OPFRs than lime stabilization. Between C_3 and C_4, a similar trend is seen which deduces the effects of AD. Looking at C_3 and C_4 for different temperatures, the increase in pyrolysis temperature has led to an increase in degradation of OPFRs. Biochar in particular gets completely free from OPFRs with temperature rise. Also, AD + pyrolysis combination at high temperatures allows the maximum degradation of OPFRs. It is worth noting that lime treatment contributes the least to the effluent for OPFRs.

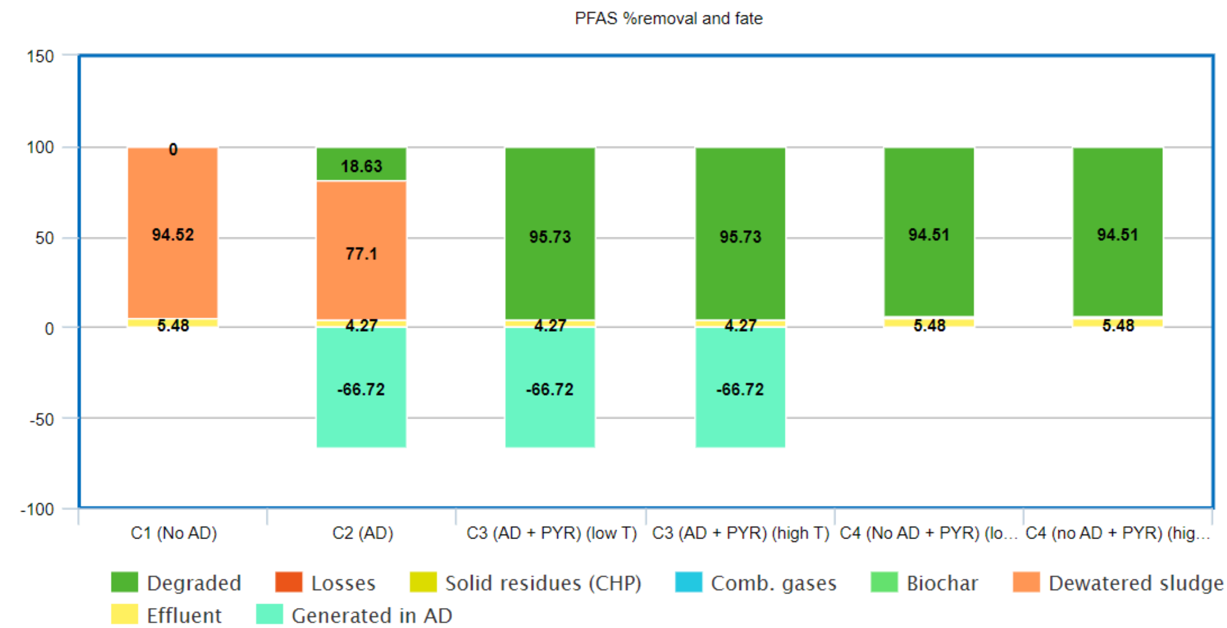
Figure 9: Heavy metals % removal and fate



	C_1 (No AD)	C_2 (AD)	C_3 (AD+PYR) (low Temp)	C_3 (AD+PYR) (high Temp)	C_4 (No AD+PYR) (low Temp)	C_4 (No AD+PYR) (high Temp)
Degraded	0	0	0	0	0	0
Losses	0	0	0	0	0	0
Solid residues (CHP)	0	0	3.81	3.88	12.59	12.82
Comb. gases	0	0	4.11	4.00	13.56	13.21
Biochar	0	0	20.78	20.82	68.84	68.97
Dewatered sludge	74.13	28.70	0	0	0	0
Effluent	25.87	71.30	71.30	71.30	5.00	5.00

In case of C_2, it gets more concentrated in effluent than C_1, which signifies that AD leads to HMs being released into the liquid mass while lime treatment coagulates a major part of them into the solid mass. The same is the trend for C_3, where AD leads to them being released in the liquid effluent. In C_4, the effluent sees a landslide plummet of HMs content. Pyrolysis without AD leads to a substantive fraction of HMs being contained in the biochar fraction. Now from several studies, it is known that biochar facilitates the immobilization of HMs. Pyrolysis temperature also shows a minute effect on the HM distribution amongst the end products. Thus, it can be observed that the majority of HM content in the original sewage sludge still remains in the biochar in the case of C_4, where there is no AD. This trend was also seen in the analysis by Lu et al. (Lu et al., 2015).

Figure 10: PFAS % removal and fate



	C_1 (No AD)	C_2 (AD)	C_3 (AD+PYR) (low Temp)	C_3 (AD+PYR) (high Temp)	C_4 (No AD+PYR) (low Temp)	C_4 (No AD+PYR) (high Temp)
Generated in AD	-	-66.72	-66.72	-66.72	-	-
Degraded	0	18.63	95.73	95.73	94.51	94.51
Losses	0	0	0	0	0	0
Solid residues (CHP)	0	0	0	0	0	0
Comb. gases	0	0	0	0	0	0
Biochar	0	0	0	0.08	0.01	0.01
Dewatered sludge	94.57	77.10	0	0	0	0
Effluent	5.48	4.27	4.27	4.27	5.48	5.48

The observation is done for PFASs show that in case of C_2 and C_3 where AD takes place, PFAS gets rather generated. This deduces that AD leads to the formation of new PFAS compounds from the synthesis of existing ones. Comparing C_1 and C_2, PFAS gets degraded in C_2 but in a rather small proportion. Pyrolysis temperature seems to have no effect whatsoever on the degradation of PFASs. However, in the case of C_3, a very small amount of PFAS seems to appear in biochar mass indicating the possibility of synthesis of some PFAS compounds with the rise in pyrolytic temperature.

6.1.1. Removal efficiency

In order to understand AD, pyrolysis, pyrolysis temperature, and its effect, some studies propose the calculation of Removal Efficiency RE (%) (Castro et al., 2023). It refers to the effectiveness of the AD or pyrolysis process in reducing or removing specific pollutants or organic matter from the influent feedstock (Grady et al., 2011; Li et al., 2018). RE can be quantified by comparing the concentration of the target pollutants or organic matter in the influent (feedstock) to the concentration in the effluent (digestate) (biosolids or biochar in these cases). It is typically expressed as a percentage and calculated using the following formula:

RE (%) = [((C1 - C2)* Sludge yield)/C1] * 100

where,

C1 and C2 are concentrations of SS before and after treatment respectively,

While sludge yield is the ratio of the dry weight mass after treatment divided by the dry weight mass before treatment.

The sludge yields for different cases presented here are:

C_1 = 0.95; C_2 = 0.52024

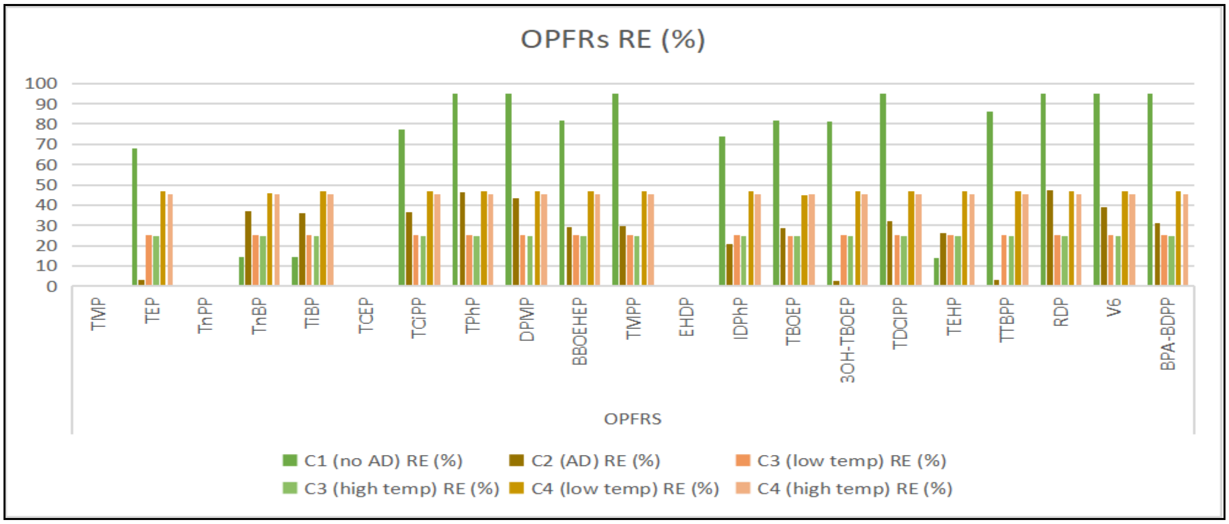
C_3 (low temp) = 0.25416; C_3 (high temp) = 0.24768

C_4 (low temp) = 0.47056; C_4 (high temp) = 0.45264

Calculating the values of RE (%) of OPFRs and PFASs for different cases, a detailed table (see Appendix E and F) is created. For the sake of ease in analysis, the values are considered up to 2 decimal points. The values of RE in positive form represent its efficiency in the removal of that compound. If the value of RE comes to be 0, it shows that there was no removal of that pollutant during the treatment process. Meanwhile, a negative value of RE means that the particular pollutant is rather created due to chemical and/or thermal reactions. It is crucial to understand that RE is a parameter calculated with respect to the amount of sludge yield in the process. Thus it denotes that in a certain amount of sludge yield, how much removal of the contaminant has taken place.

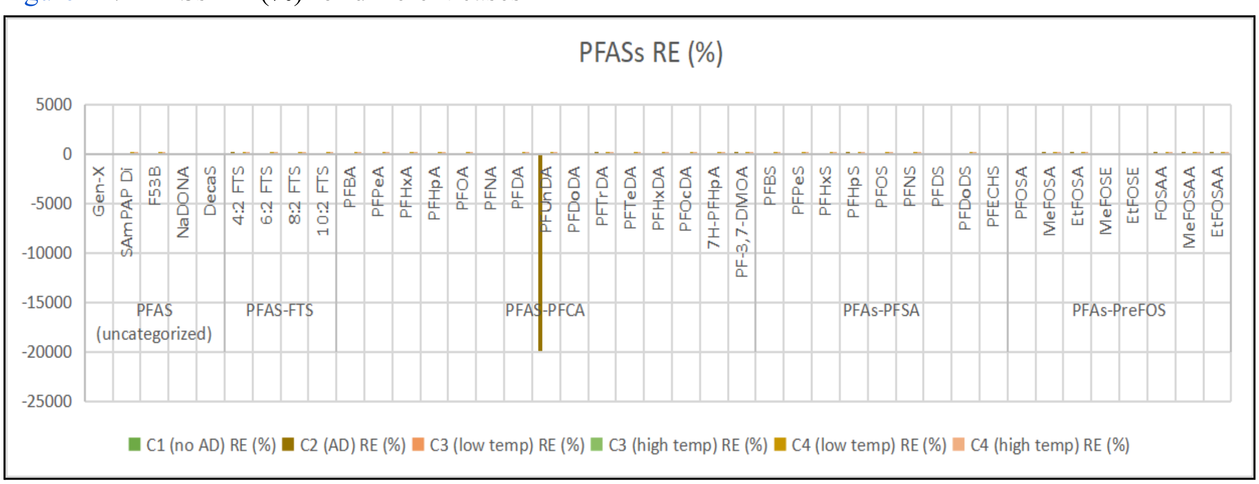
Based on the data obtained, a column graph is prepared for both OPFRs and PFASs.

Figure 11: OPFRs RE (%) for different cases



Based on the graph, it is observed that both lime stabilization, as well as AD, contribute to the removal of OPFRs. But between the two, lime stabilization shows a higher value of RE (%) than AD, with the exception of TnBP and TiBP. In fact, in the case of TEP, 3OH-TBOEP, and TTBBP, the difference is gigantic. The value ranges from 2.48% in 3OH-TBOEP to 47.24% in RDP for C_2 while for C_1, the value ranges from 13.77% in TEHP to as high as 95% for several of the compounds (see Appendices E and F). This cements the idea that lime stabilization is a much more efficient process for OPFR removal from SS. Comparing C_3 and C_4, the RE values are much higher for both temperatures of C_4 respectively. While comparing the values within C_3 and C_4 for different pyrolysis temperatures, there is a slight decrease in the RE value for higher pyrolysis temperature in C_3, which becomes more visible in case of C_4, with the exception of TBOEP for C_4. This shows that a rise in pyrolysis temperature does not necessarily lead to the removal of OPFRs from biochar. However, contrary to a result published by Castro et al., none of the RE values were found to be negative which indicates that no additional formation of OPFRs took place during the treatment (Castro et al., 2023).

Figure 12: PFASs RE (%) for different cases



The value of PFUnDA from the PFCA group shows such a high value of negative RE (%) (-19880.6%) in the case of C_2, that the entire graph gets skewed. However looking at the other cases for PFUnDA, the value is 0 (see Appendix F), denoting no removal efficiency. Thus to have a better look at the values and trends, the entire PFAS group is divided into two parts, and the value of PFUnDA is left out of the graphs.

Figure 13: PFASs (uncategorized, FTS, and PFCA group) RE (%) for different cases

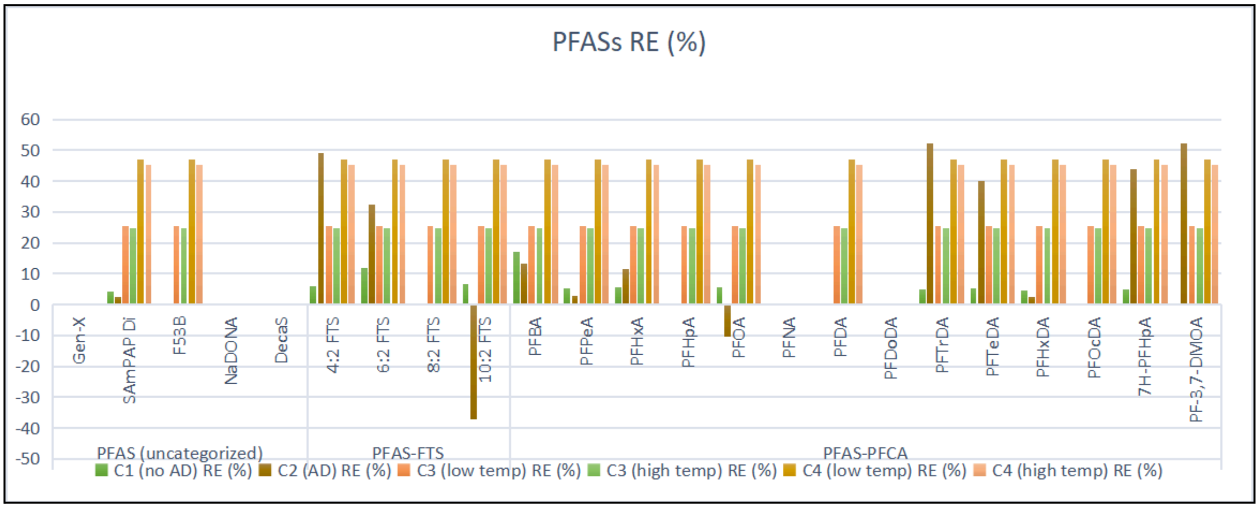
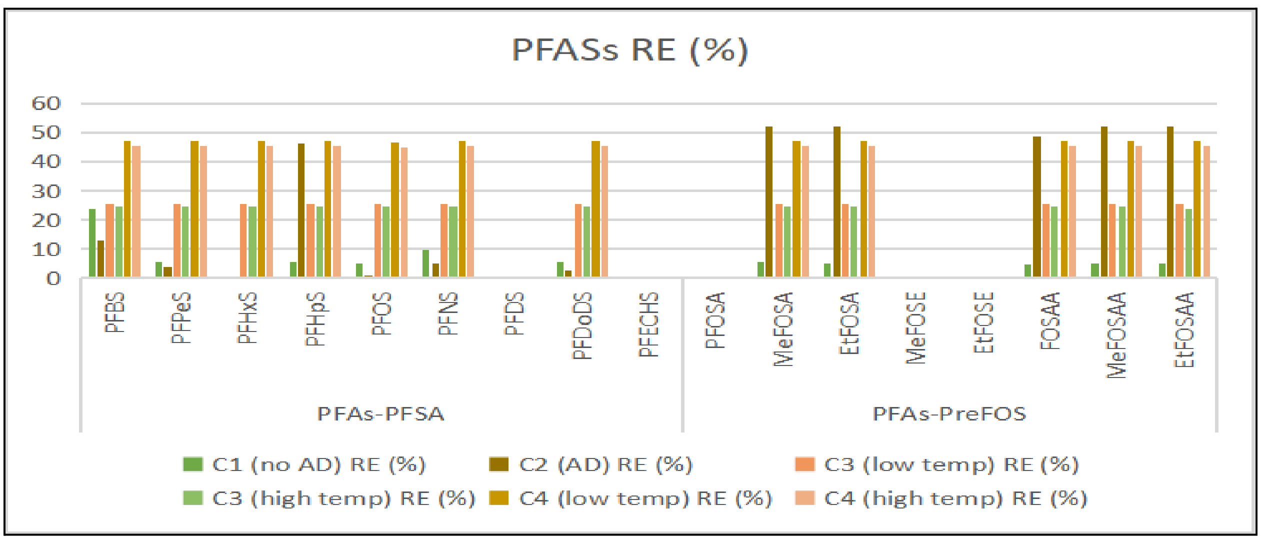


Figure 14: PFASs (PFSA and PreFOS group) RE (%)



The graphs (Figures 13 and 14) depict the trend of OPFRs. RE is very low in C_1 but has a fluctuating trend for C_2. 10:2 FTS, PFOA, and PFUnDA (see Figure 12) have a rather negative value denoting that they are formed during the AD process. It could be said that a majority of PFASs get removed with AD while synthesizing to form other PFAS compounds. It is also worth noting that some compounds have 0 RE during AD or lime stabilization, but get removed during pyrolysis processes. Now looking at the values of C_3 and C_4, RE has significantly increased values for C_4. In the case of temperature comparison amongst the two pyrolysis processes C_3 and C_4, for high pyrolytic temperature, the value of RE sees a slight dip. This is consistent for all PFASs. This phenomenon shows that the rise of pyrolysis temperature does not increase the RE of PFASs.

6.1.2. Biochar yield

Another parameter to analyze the reduction of organic matter from biosolids through pyrolysis is the calculation of Biochar yield. It indicates the amount of biochar produced relative to the starting biomass feedstock used in the pyrolysis or carbonization process (Kundu et al., 2022; Zhang et al., 2021). It is typically expressed as a percentage and calculated by dividing the mass of biochar obtained by the mass of the initial biomass feedstock, multiplied by 100.
For calculating the Biochar yield (%) at both low and high temperature for case C_3 and C_4, the formula is:

$$\text{Biochar yield (\%)} = \frac{\text{Biochar amount}}{\text{Biomass amount}} \times 100\ (\%)$$

Using the data obtained from Appendices C and D, the values of biochar yield for C_3 and C_4 are:

Case	**C_3 (low temp)**	**C_3 (high temp)**	**C_4 (low temp)**	**C_4 (high temp)**
Yield (%)	25.416	24.768	47.056	45.264

It shows a decrease in yield (%) with temperature in both the cases. It is worth understanding that as a substantial amount of organic content is reduced in AD, thus the value of yield for C_3 is lower than that of C_4. The value range of slightly different results was obtained in a study by Kundu et al. where the biochar yield obtained was in the range of 36–45% at 500–600 °C (Kundu et al., 2021). This shows that AD followed by pyrolysis at high temperatures leads to a decrease in biochar yield.

6.1.3. HMs retention rate

Now looking at the retention rate of HMs which is defined as the ratio of HMs' quantities in biochar to that of sewage sludge (Lu et al., 2015), the values obtained are presented as percentage (%) in Table 8.
The values of HM retention increase slightly with an increase in temperature with the exception of Cd, Pb, and Zn. The values obtained in C_3 are quite low as compared to C_4 for low and high temperatures respectively. The values for Cd are much lesser compared to other HMs. However, looking at the data of effluent (liquid), it could be hypothesized that some HM content is released into the liquid fraction through AD which later gets removed through dewatering. This shows that AD hinders the ability of biochar in the retention of HMs.

Based on these observations, several implications can be deduced regarding the different sewage sludge management processes and their effects on hazardous organic chemicals HOCs and HMs.

Table 8: Comparison of retention rate (%) for HMs in biochar at low and high-temperature

Heavy metal	C_3 (low temp)	C_3 (high temp)	C_4 (low temp)	C_4 (high temp)
As	19.39	20.90	66.80	72.13
Ba	20.41	22.97	68.97	77.59
Cd*	4.66	0.79	14.05	2.39
Co	20.08	21.97	73.11	79.92
Cr	22.22	23.32	75.43	78.87
Cu	19.91	21.19	65.12	69.48
Mo	21.43	24.45	70.50	80.67
Ni	19.30	22.0	70.77	80.51
Pb*	20.84	16.62	72.54	58.09
Sr	21.44	24.03	66.67	74.54
V	21.67	23.60	78.99	85.99
Zn*	21.06	18.61	69.57	61.41

The data suggests that different treatment processes have varying effectiveness in removing HOCs from sewage sludge. Lime stabilization leads to significant degradation of OPFRs but very little for PFASs. It further causes precipitation and accumulation of OPFRs, PFASs, and HMs in the biosolids mass. In fact, it allows the least amount of OPFRs in the effluent compared to any other process. The RE (%) of OPFRs is quite high, while for PFASs it's very low. AD facilitates a significant reduction of SS biomass. It causes degradation of some OPFRs and PFASs, while also resulting in the synthesis of substantive amounts of some PFASs. It also leads to sedimentation of HOCs in biomass but increases the leaching of HMs in the effluent.

Pyrolysis plays an important role in reducing the HOCs (PFAS and OPFR) content. A combination of AD + pyrolysis almost destroys the OPFR content of biosolids. However, AD leads to the synthesis of some PFAS compounds along with the leaching of HMs in the effluent. The biochar yield and HM retention rate are low. The increase in temperature for this combination of treatments only sees a slight increase in OPFR degradation. But this does not improve its RE. Moreover, there is almost no difference in HM immobilization. PFASs observe no degradation change with even a slight decrease in RE. The biochar yield decreases in this case while the HM retention rate slightly increases.

In the case of direct pyrolysis without subjecting SS to AD, there is a sizable amount of HOCs in effluent pertaining to the lack of a pretreatment process. However, most of the OPFRs and PFASs get destroyed and a major concentration of HMs is observed in the biochar fraction. Absences of AD also result in no synthesis of PFASs. The RE, biochar yield, and HM retention rate see a higher value. With an increase in pyrolytic temperature, there is no notable change in the fate of OPFRs and PFASs. The biochar yield decreases while the HM retention rate sees a small increase.

6.1.4. Observation

Based on the analysis of the data, the preferred SS treatment process among those mentioned appears to be direct pyrolysis without prior anaerobic digestion (AD) at low temperatures. This is in contradiction with the theory that AD coupled with pyrolysis can be a viable option (Raheem et al., 2018). This approach yields several advantages over other methods.

Firstly, it results in a substantial reduction of HOCs, particularly OPFRs and PFASs, without the need for additional pretreatment steps. Secondly, the process generates a significant amount of biochar, which serves as an effective means of carbon sequestration and aids in the stabilization of HMs, thus minimizing their environmental impact. In contrast, AD as a pretreatment method leads to the leaching of HMs into the effluent, posing potential risks to ecosystems. Meanwhile, low-temperature pyrolysis preserves the integrity of organic matter in the SS, leading to the production of high-quality biochar rich in carbon and beneficial for soil amendment and carbon sequestration. Moreover, it minimizes energy consumption and greenhouse gas emissions compared to high-temperature pyrolysis processes, aligning with sustainability objectives. It is also seen that the use of low pyrolytic temperatures is preferred as it does not significantly improve efficiency compared to higher temperatures. Therefore, direct pyrolysis at low temperatures emerges as a favorable SS treatment option for its effectiveness in HOC reduction, biochar production, and environmental considerations.

Interestingly, it is observed that the use of lime stabilization as a pretreatment reduces the mass of OPFRs (very high RE), and causes accumulation of pollutants in biosolid, which will facilitate a lesser load on pyrolysis for pollutant destruction/immobilization. Additionally, while comparing the data of C_1 to C_4, it can be seen that it significantly reduces the amount of pollutants leaching into the effluent fraction. This will ensure a better quality of effluent and be much safer to be released to water bodies/environment in general. The one drawback of this process seems to be the release of some HMs in the effluent which could be undesirable. But with proper monitoring of the effluent to ensure its HM content within the regulation limit could overcome this shortcoming. Additionally, the alkaline conditions created by lime stabilization can enhance the stabilization of heavy metals (HMs) present in the SS, reducing their mobility and potential for environmental contamination.

A theory could be postulated that lime stabilization can serve as an effective pretreatment benefit for SS, preparing it for subsequent pyrolysis at low temperatures to achieve maximum benefits. This pretreatment step serves to enhance the efficacy of pyrolysis by facilitating the removal of moisture, organic contaminants, and pathogens from the SS matrix, thereby improving the quality of the biochar produced during pyrolysis.

Further research and experimentation are warranted to validate and optimize this theoretical framework in practical applications.

6.2. Analysis in Norwegian context

These data are from the samples collected from 18 different WWTPs with a total capacity of 41% (Blytt & Stang 2019). This implies that they represent a significant proportion of SS composition in Norway. Analyzing these data underscores the complexity of treatment outcomes. The distribution of pollutants in byproducts varies with the process as well as with different compounds. The temperature of pyrolysis, which a lot of study claims to contribute to a significant reduction in HOCs and immobilization of HMs, also needs to be analyzed critically. From the data, it is clear that these reductions are not significant, and are contaminant & processing conditions specific. For treatment optimization, a combination treatment process may offer synergistic benefits in pollutant removal. Thus understanding the fate and distribution of each individual byproduct is essential to assess the environmental impact and potential reuse of treated sludge, no matter how tedious it sounds.

Table 9: Municipal Wastewater Treatment in Norway (Source: Statistics Norway)

	2017	2021	2022	Change in percentage	
				2017 - 2022	2021 - 2022
Percentage of population connected to municipal wastewater facilities (per cent)	84.9	86.0	85.6	0.8	-0.5
Percentage of households with water meter installed (per cent)	33.6	36.6	37.5	11.6	2.5
Length of separate stormwater pipelines (m)	18 318 610	20 016 073	20 548 245	12.2	2.7
Total length of sewage pipelines (m)	37 399 466	39 237 331	39 951 473	6.8	1.8
Percentage of total wastewater pipeline system renewed, 3-year-average (per cent	0.62	0.67	0.66	6.5	-1.5
Share of inhabitants connected complying with treatment permits (per cent)	57.7	69.0	69.6	20.6	0.9
Share of inhabitants connected not complying with treatment permits (per cent)	33.7	25.3	25.6	-24.0	1.2
Share of inhabitants connected where compliance with treatment permits is unknown (per cent)	8.6	5.8	4.8	-44.2	-17.2
Discharge phosporus, wastewater plants 50 pe or larger (tonnes)	974.1	1 023.5	1 033.8	6.1	1.0
Discharge BOD5, wastewater plants 50 pe or larger (tonnes)	37 819	34 620	34 532	-8.7	-0.3
Total amount of sewage sludge disposed (tonnes dry weight) (tonnes)	121 328	133 832	132 818	9.5	-0.8

From the information presented in Table 9, between the 5 years period of 2017 - 2022 in Norway, the percentage of inhabitants complying with the treatment permits has increased. But at the same time, the discharge of P and BOD_5 (Biochemical Oxygen Demand) has also increased indicating that the concentration of pollutants in SS is also increasing. Several factors such as urbanization, industrialization globalization, and general increased use of chemicals (especially HOCs) in anthropogenic processes (like agriculture) could be cited to this trend.

Table 10: HMs in sewage sludge (mg per kg dry weight) (Source: Statistics Norway)

HM	Zinc (Zn)	Copper (Cu)	Chromium (Cr)	Nickel (Ni)	Lead (Pb)	Cadmium (Cd)	Mercury (Hg)
Average content of HMs (2022 Data)	354.8	163.2	16.7	13.7	12.6	0.5	0.3

Table 10 represents the average concentration of HMs found in Norwegian SS. Comparing it to the Norwegian limits, all HMs are within the required limits except Cu and Zn which are way higher than their mandated maximum permissible limits. The other 5 HMs analyzed in this paper have no frame of reference to compare with. Moreover, it is also quite difficult to understand the contamination levels of OPFRs and PFASs as apart from a very few compounds, there is no limit set up by the regulating authorities.

Now based on Table 1, it is evident that a larger portion of the produced SS is applied in the soil in one form or the other, these contaminants which are attached to the molecular structure and/or surface of biosolid/biochar also get released into the soil. Thus it is quite important to understand their concentration and behavior in the environment as well as their interactions with other biotic and abiotic beings.

It could be postulated that the current regulations and basic essence of authorities in Norway indicate the overall mentality and inclination towards environmental conservation and sustainability in general. However, there is a lot that still needs to be accomplished in this sector. Due to the potential for chemical transformations and the complex distribution of contaminants in various by-products, comprehensive monitoring and regulation of treated sludge is essential to ensure environmental protection and human health. This may include monitoring not only the concentrations of targeted contaminants but also assessing the potential risks associated with newly formed compounds or transformation products. It is also worth noting that even a country like Norway is still far from providing clear regulations for most HOCs and several HMs. This makes it even more difficult for public and private entities to ensure the proper disposal of sewage sludge.

6.3. Effects of biosolid and biochar on soil

The actual fate of organic pollutants in soil is governed by many different factors including soil characteristics, compound properties, and environmental factors such as temperature, precipitation, and the ability of soil microbes to degrade the compound (Pozzebon et al., 2023). For a long time, the use of biosolids especially, has been restricted due to concerns about pollutants in it, pathogens, and odors (Arulrajah et al. 2011; Charlton et al. 2016; Marchuk et al., 2023; Pradel et al., 2014; US EPA 2019). However, in comparison with other sludge disposal methods, land application is deemed to be the most economical as well as an eco-friendly way to dispose of the sludge and alleviate the shortage of resources and energy in the process (Amorim Júnior et al. 2021; Bagheri et al. 2023; Charlton et al. 2016; Chen et al. 2016; Martín-Pozo et al., 2019; Qin et al. 2022; US EPA 2019). The biosolids have proven to have a significant nutrient and organic material content that can benefit the soil's physical conditions[28] (e.g. porosity, surface area, bulk density, water retention capacity, etc.), chemical fertility (e.g. cation exchange capacity, soil pH), and, consequently, its agronomical potential[29] in soil (Amorim Júnior et al. 2021; Bamdad et al., 2022; Bilias et al., 2021; Kumar Raja Vanapalli et al. 2021; Manikandan et al., 2023; Paz-Ferreiro et al. 2018). The mineralization (decomposition) of organic matter in biosolids releases several macronutrients (N, P, K, Ca, Mg, S) and micronutrients (Cu, Zn, Fe, B, Mo, and Mn) which can be used as a fertilizer substitute to improve and maintain productive soils and stimulate plant growth (Cheng et al., 2023). It enhances the water retention capacity, and erosion & surface runoff resistance (Paz-Ferreiro et al. 2018; Qin et al. 2022). In contrast to chemical fertilizers where the nutrients are readily plant available, nutrients in biosolids are typically slow-release, which could prove to be effective in the prevention of rapid nutrient washing and facilitating sustainable mineralization (Marchuk et al., 2023).

However, it is worth mentioning that not all qualities of biosolids can be used for soil application[30] (Arulrajah et al. 2011; AWA 2017). For instance, the interaction of emerging pollutants in biosolids after land application can vary based on the physicochemical properties of the organic compound, the treatment process used to generate the biosolids, and soil properties (e.g., pH and organic carbon), as well as climate. It can get degraded, and interchanged into compounds of similar or even greater toxicity, than the parent compound. Thus land application of biosolids could in theory result in sensitive environments being exposed to PFAS and OPFRs at levels much higher than previously anticipated. Land-applied biosolids enriched with longer-chain PFAS can be adsorbed to microplastics or dust and become airborne. It can also act as a sink for emerging pollutants (Martín-Pozo et al., 2019; Patel et al., 2020; Pozzebon et al., 2023). PFAS' long-term exposure leads to adsorption in the soil phase thus slowing the rate of microbial transformation (Evich et al., 2022; Harder et al., 2016). Biosolids can also release fine particles or colloids when subjected to natural drying and freeze-thaw cycles, which can carry PFAS to subsurface and groundwater (Pozzebon et al., 2023). They are also observed to be accumulating OPFRs (Cristale et al., 2016). Furthermore, there is also a concern about a low N:P ratio which means that prolonged application of biosolids can result in progressive build-up in soil P levels, increasing the risk of P transport to water courses by erosion and runoff (Marchuk et al., 2023; Rigby et al., 2021). The HMs, ill-biodegradable organic compounds, and pathogenic organisms accumulated in biosolids also have the risk of entering the environment and further into the food chain via leaching and surface runoff (Cheng et al., 2023; Harder et al., 2016;

[28] A field study in 2019 showed that biosolids are beneficial to soil-subjugated to intense weathering actions such as sandy soils (Amorim Júnior et al. 2021). It also reduces dependency on fossil fuels used for the production of chemical fertilizers (Qin et al. 2022).

[29] Agronomical potential refers to ability of germination, development of crops, native vegetation etc.

[30] Biosolids application may pose the risk of soil pollution due to the presence of pollutants (e.g., antibiotic resistance genes (ARGs), Potentially Toxic Elements (PTEs), heavy metals, etc.), which causes severe restrictions on field applications. Especially, the presence of ARGs in biosolids is considered a high-potential threat to the soil. Once the ARGs are transferred to bacteria, they would show a high probability of presence in natural ecosystems soil physicochemical properties and the presence of heavy metals affect the spread and profile of ARGs (Amorim Júnior et al. 2021; Chen et al. 2016; Qin et al. 2022; Tiwari et al., 2023).

Kumar et al., 2022; Méndez et al., 2012; Rigby et al., 2021). Moreover, their direct application in soil could cause significant ecotoxicity (Tarpani et al. 2020). Thus, Biosoilds should be applied to land at an appropriate agronomic rate depending on the vegetation type, geographic location, and soil characteristics (US EPA 2019).

In the case of biochar, numerous studies have found that it increases the soil's P[31], N, Ca, and Mg contents facilitating higher crop yield. The soil cation exchange capacity (CEC)[32] and pH value were also increased by biochar application. A major concern about the application of biochar in soil has been the presence of high levels of HMs which even increase compared to biosolids. However, the bioavailability of most of these HMs is tested to be very low. In fact, it is observed to lower the leaching of several HMs, while also reducing the polycyclic aromatic hydrocarbon (PAH) concentrations and the availability of potentially toxic elements (PTEs) by immobilizing PTEs in contaminated soils (Bilias et al., 2021; Bogusz et al., 2017; Cheng et al., 2020; Lu et al., 2015; Méndez et al., 2012; Patel et al., 2020; Xu et al., 2018). These HMs transform from easy-to-leach forms to hard-to-dissolve forms through pyrolysis. It is also seen that the compounds and functional groups of biochar suppress the release of HMs through the formation of organometallic complexes (Lu et al., 2015; Méndez et al., 2012; Zhang et al., 2022).This efficiency of biochar to immobilize HMs can even be increased by reducing the particle size, which can increase the surface area and expose the inner functional groups. These can be negatively charged when ionized resulting in a greater CEC of the soil (Fahmi et al., 2018; Wang, Victor, et al., 2022). By alleviating the metal toxicity and increasing total PLFAs[33] (phospholipid fatty acids) concentration, it enhances microbial activity and mitigates biotoxicity to microorganisms (Xu et al., 2018). It also increases the soil organic matter content, and enhances the water and nutrient holding capacity and aeration (Lu et al., 2015; Méndez et al., 2012). Biochar enhances the nutrient availability of acidic soil[34] by increasing the soil pH, consumption of OH^- (hydroxyl) ions released from the dissociating phenolic functional groups, and enhancing nutrient retention (Hachib Mohammad Tusar et al. 2023).

However, the immediate nutrient supply of biochar is limited. Some ecotoxicological studies have revealed that there is a tendency of biochar to immobilize soil N content[35] limiting their accessibility to plants and sorption of contaminants which could be released later posing a secondary pollution and ecological risk. Especially with a longer period of time, biochar could be oxidized and acidic functional groups from biosolids will be released into the soil solution, leading to a rather drop in the pH value, which can increase the phytotoxicity and the heavy

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[31] Application of biosolids biochar to soil can increase P availability by up to 38 times (Patel et al., 2020).

[32] Cation exchange capacity (CEC) is a measure of the total negative charges within the soil that adsorb plant nutrient cations such as calcium (Ca2+), magnesium (Mg2+), and potassium (K+). A soil's CEC affects fertilization and liming practices. Soils with high CEC retain more nutrients than low-CEC soils (NSW Govt. 2021).

[33] Phospholipid-derived fatty acids (PLFAs) are widely used in microbial ecology as chemotaxonomic markers of bacteria and other organisms. It is used to quantify total viable biomass in water or soil samples and provide a general profile of the microbial community (Quideau et al., 2016).

[34] A research conducted combining biochar and lime saw the increased nutrient availability of P by 137% (Hachib Mohammad Tusar et al. 2023).

[35] Experiments revealed that content was significantly reduced at pyrolysis temperatures above 500 °C (Zhang et al., 2022).

metal availability (Cheng et al., 2020; Dike et al., 2021; Manikandan et al., 2023; Paz-Ferreiro et al. 2018; Zhang et al., 2022). There's also a risk of unconscious discharge of toxic elements that are used in biochar production or from the biochar-based material production process (Duwiejuah et al., 2020). According to some research, HMs adsorbed by biochars through the cation exchange mechanism are readily bioavailable for plants (Fahmi et al., 2018). There are also some cases of greenhouse gas (GHG) such as CO_2, N_2O, and CH_4 emissions under specific conditions. Some research has indicated that biochar inhibits the efficacy of soil pesticides and their biodegradation effects, which could result in an efficiency decrease of insecticides and pesticides. Furthermore, while improving the biological activities of bacteria such as *Geobacter, Anaeromyxobacter,* and *Clostridium*, it implicates the possibility of a negative impact on the survival, growth, and diversity of other soil living communities like acidophilic earthworms and fungi (Cheng et al., 2020). Sometimes the particle size of biochar also leads to toxicity to microorganisms (Manikandan et al., 2023). Moreover, as new HOCs are being discovered every day, their interaction/destruction with pyrolysis is still a matter of research (Pozzebon et al., 2023). It is also observed that the soil respiration increment of biochar application is less as compared to SS or biosolids (Méndez et al., 2012). These all put biochar in a "double-edge" position.

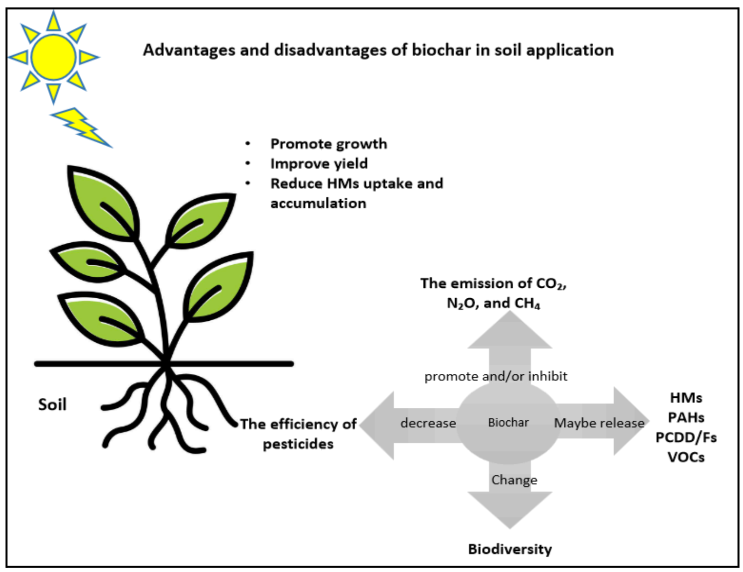

Figure 15: Advantages and disadvantages of biochar in soil application

A detailed analysis of mobility, toxicity, and bioavailability in terms of soil application of biosolids and biochar is provided below.

6.3.1. Mobility/leaching

In the case of HMs, determining the mobility and leaching potential is a complex affair that depends on factors like pH, redox conditions, mineralogy, and organic matter content of the soil. A general order trend of solubility is Ba < Sr < Ni < Co < Cu < Cr < Pb < Zn < As < Cd < V < Mo (Trevors & Alloway 2013; Wang et al., 2022; Zhang et al., 2022). Usually, HMs with higher solubility and lower affinity for soil particles are more likely to leach. A general ranking for leaching potential for HMs is V < Sr < Ba < Mo < Zn < Co < Ni < Cu < Pb < Cr < As < Cd (Trevors & Alloway 2013; Wang et al., 2022; Zhang et al., 2022). According to several studies, biosolids have more leaching potential than biochar due to high organic matter content, high water content, of Chelating Agents like organic acids that can bind to heavy metals and increase their solubility (Wang et al., 2022; Wieczorek, Baran, & Bubak 2023). A measure of bioaccumulation was proposed by Müller in

1969 which is still widely used is **Geoaccumulation Index (Igeo)**. It is a measure used in environmental geochemistry to assess the degree of contamination or accumulation of heavy metals or other pollutants in sediments, soils, or other environmental matrices compared to natural background levels (Abdullah, Sah, & Haris 2020; Nowrouzi & Pourkhabbaz 2014).
The formula for calculating Igeo is:

$$Igeo = \log_2 \left(\frac{C_n}{1.5 \times B_n} \right)$$ where

C_n is the concentration of the heavy metal in the sample.
B_n is the background or reference value of the heavy metal. This could be the concentration found in uncontaminated or pristine environments or established regulatory guidelines.
Igeo provides a numerical value that indicates the degree of contamination or accumulation of the heavy metal in the sample compared to the background/reference level. The index values are interpreted as follows:
Igeo $<$ 0 - uncontaminated to moderately contaminated
0 $\leq$ Igeo $\leq$ 1 - moderately contaminated
1 $<$ Igeo $\leq$ 2 - moderately to heavily contaminated
2 $<$ Igeo $\leq$ 3 - heavily contaminated
3 $<$ Igeo $\leq$ 4 - heavily to extremely contaminated
4 $\geq$ Igeo - extremely contaminated
Calculating the values of Igeo based on the concentration of HMs from Appendices C and D and the normative limit from Table 6, the values obtained are:

Table 11: Igeo values for HMs

	C1	C2	C3 (low temp)	C3 (high temp)	C4 (low temp)	C4 (high temp)
Cd	5.28	4.79	2.02	-0.53	3.62	1.06
Cr	18.63	16.86	16.53	16.60	18.29	18.36
Cu	19.58	18.34	17.80	17.89	19.51	19.60
Ni	15.32	14.45	14.02	14.21	15.90	16.08
Pb	16.06	14.94	14.55	14.23	16.35	16.03
Zn	22.54	21.01	20.56	20.39	22.29	22.11

Based on the table, some interesting results are observed. Amongst all the SS management processes, biosolids produced in C_1 have the highest bioaccumulation Index. Amongst the four different types of biochar produced, C_4 at low temperatures has the highest Igeo values. However, it has been established by several studies that biochar showcases lower mobility than biosolids. Also, from the aspect of concentration and further removal of pollutants, it is important to achieve a higher Igeo value.

The table further indicates that most of the degree of accumulation of these 6 HMs is quite high. The trend seen here is Cd <Ni <Pb <Cr <Cu <Zn. Comparing this with the leaching potential trend indicates that for the SS treatments done, the HMs that are of major concern are Cr, Cu, Cd, and Pb.

Another index to determine the leaching value is the **Leaching Potential Index (LPI)**. It is calculated using the following formula:
[LPI = Cs/Cr] where,
Cs = Concentration of heavy metal in soil (mg/kg or ppm)
Cr = Concentration of heavy metal in soil that causes harmful effects (mg/kg or ppm)
The concentration Cr can be derived from regulatory guidelines, threshold values, or soil quality standards for the specific heavy metal. A lower LPI value indicates a lower potential for leaching, while a higher LPI value indicates a higher potential for leaching (Guzmán-Martínez et al. 2020; Li, Yohey et al., 2018).
Using the data from Appendices C and D, and the normative limits from Table 6, the values obtained are:

Table 12: LPI values for HMs

	C1	C2	C3 (low temp)	C3 (high temp)	C4 (low temp)	C4 (high temp)
Cd	58.10	41.40	6.10	1.04	18.40	3.13
Cr	60.70	17.90	14.20	14.90	48.20	50.40
Cu	470.00	199.40	137.00	145.80	448.00	478.00
Ni	68.33	37.33	27.73	31.60	101.67	115.67
Pb	41.00	18.84	14.42	11.50	50.20	40.20
Zn	406.00	140.67	103.33	91.33	341.33	301.33

Here it is quite clear that biochar has significantly lower leaching potential than biosolids. But unlike the previous inclination, here C_3 at low temperatures has on average a lower leaching potential. However, recalling the retention rate of HMs (see Table 8), the values are quite low for C_3 at low temperatures as compared to C_4 at the same temperature. This means that even though the biochar in the case of C_3 will lead to low leaching, it would retain a significantly smaller amount of HMs in the mass to begin with. Adding, it is also evident from the mass flow that a sizable amount of HMs get released in effluent (see Figure 9). Thus, keeping these in mind, it is still more efficient for C_4 at low pyrolysis temperature to be preferred. Lastly, based on the values, Cu and Zn have the highest leaching potential.
In the case of OPFRs, their solubility and adsorption affinity depend on their chemical structure. Generally, OPFRs have high mobility in soil due to their relatively high solubility and low

affinity for soil particles. However, in the case of biochar, most of the OPFRs get destroyed during the pyrolysis process (US EPA). The remaining OPFRs attach to the biochar mass and get immobilized. Thus biochar is a much viable option in terms of OPFRs.

PFASs are known for their persistence and tendency to accumulate in soil and water. Their mobility depends on factors such as molecular size, chemical structure, and soil properties. PFASs with smaller molecular sizes and lower adsorption affinities may have higher mobility in soil. Shorter-chain PFASs have a lower tendency to be absorbed or leached into the soil but are more mobile than long-chained compounds. In biochar, these compounds tend to be adsorbed on the biochar surface and fraction, thus reducing their mobility and leaching in soil (US EPA).

Mobility of pollutants in soil further depends on soil properties such as texture, pH, organic matter content, and mineral composition can significantly influence the mobility of contaminants. For example, sandy soils with low organic matter content may have higher leaching potential compared to clay soils with higher organic matter content. Environmental conditions such as rainfall, temperature, and microbial activity can also affect the transport and fate of contaminants in soil. Heavy rainfall events, for instance, can promote the leaching of contaminants from the soil into groundwater (Cheng et al., 2020; Li et al. 2019).

6.3.2. Toxicity

The toxicity of pollutants varies with concentration, bioavailability, chemical forms, and interactions with soil components (Trevors & Alloway 2013; Zerizghi et al. 2020). For HMs, the general trend[36] of toxicity for the analyzed HMs is: Ba＜Sr＜ Ni＜ Co＜ Cu＜ Zn＜ Cr＜ Pb＜ As＜ Cd＜ V＜ Mo. A similar result has been observed in other studies done as well (González Henao & Ghneim-Herrera 2021; Zerizghi et al. 2020). Comparing toxicity trend with the accumulation potential from Table 11 and leaching potential from Table 12, Cu and Zn from non-carcinogenic groups are the most toxic HMs found in the samples, while from carcinogenic group- Cd, Pb, and Cr seem to be the available HMs with most toxicity. However, from the accumulation and leaching potential, it is evident that biochar without AD at low pyrolysis temperature is an efficient solution to diminish their toxicity to human health and the environment.

For OPFRs, there is limited information available on the carcinogenic potential. However, they have severe ecotoxic and non-carcinogenic toxicity. The persistence and accumulation potential of PFAS compounds results in biomagnification through the food chain (US EPA). However as seen from the mass flow values, since most of them get degraded during biochar formation, the actual effect due to soil application is projected to be very low (well within the permissible limits).

It's important to note that the calculation of heavy metal toxicity in soil often involves interdisciplinary considerations, including environmental chemistry, toxicology, and risk assessment principles. Additionally, regulatory guidelines and site-specific factors may influence

[36] This trend is a culmination of ecotoxicity, human carcinogenic toxicity, and human non-carcinogenic toxicity.

the selection of appropriate methods and interpretation of results. Consulting relevant regulatory agencies, scientific literature, or environmental professionals with expertise in soil quality and contamination assessment can provide further guidance on assessing heavy metal toxicity in soil (Li et al. 2019).

6.3.3. Bioavailability

The Bioavailability of HMs depends on factors such as their chemical form, soil properties, and mobility & solubility in soil. In biosolids and biochar, HMs form complexes with organic matter often reducing their availability for uptake by plants or leaching into groundwater (Ahmad et al. 2014). Usually, Pb, As, Zn, and Cd have a higher degree of bioavailability as compared to other HMs (Wang et al., 2022; Wieczorek, Baran, & Bubak 2023). Although the concentration of HMs in biochar is much higher than that of biosolids, they are immobilized by the biochar mass and surface, thus reducing their bioavailability. Also, the biochar formed at low pyrolysis temperature without AD has the highest biochar yield (see results of section 6.1.2). Thus, the process leads to the least bioavailable HMs among different SS treatments.
Limited research is available on the bioavailability of OPFRs in biosolids and biochar for soil application. However, studies suggest that OPFRs can undergo degradation and transformation in soil, influencing their bioavailability and environmental fate (Cheng et al. 2020; Zhang et al. 2021).
PFASs in biosolids and biochar can exhibit varying degrees of bioavailability depending on their chemical properties, soil characteristics, and environmental conditions. Some PFASs have been shown to leach from biosolids or biochar into soil or groundwater, while others may remain bound to soil particles.
It is quite challenging to determine a general trend of bioavailability of pollutants in soil from biosolids and biochar. It is because these are soil and plant-specific.

6.4. Benefits of biochar

Biochar production overall is seen as a highly efficient, economically viable, and environmentally sustainable option (Fahmi et al. 2018; Kumar et al. 2023; Méndez et al. 2012; Zhang et al. 2022). It has an extremely long storage life and can hold carbon for many centuries. Therefore, if the carbon is put in the earth instead of burning it, it can efficiently remove CO_2 from the atmosphere. Recent estimation indicates that the application of biochar to the soil can promote a C sequestration rate of 0.54 Mg C ha−1 year−1. According to a hypothesis by SINTEF, one of Norway's prominent research organizations, using just one cubic meter of biochar in the earth would equal a reduction of 1,000 kg of CO_2 emissions. If 4,000 Norwegian farms and gardens created biochar and mixed it in the soil, it could halve emissions from the agricultural sector. Several researches have also established the efficacy of biochar to sorb heavy metals, Emerging contaminants (e.g., PFAS), and agricultural chemicals from contaminated

soils, thus effectively reducing the potential for metal and chemical contamination of surface and ground waters. Some experiments have been done to use biochar to remove HMs from stormwater. Biochar can also be used in the remediation of hydrocarbon-contaminated soil (Antunes et al., 2021; Cheng et al. 2023; Dike et al., 2021; Jaya Nepal et al., 2023; Johnson 2011; Johnson, Maynard, & Nico 2012; Méndez et al., 2012; Patel et al., 2020; SINTEF; Thoma et al., 2022; Wallace, Su, & Sun 2017). It significantly reduces the waste volume and thus the transport cost for disposal (Méndez et al., 2012). Biochar produced from the pyrolysis of biosolids has the potential to partially/completely destroy contaminants such as pharmaceuticals, antibiotics, pesticides, microplastics, and most PFAS. It diminishes the amount of acidic gases and dioxins formed and helps in soil amendment by improving its physical, chemical, and biological properties. It can complement chemical fertilizers and organic sources like compost and manure. Long-term biochar application promotes nutrient & water retention and soil productivity, helping reduce chemical fertilizer needs over time. Because of its special adsorption and chemical characteristics, which can help to capture and immobilize as well as reduce[37] the bioavailability of pollutants like HMs, organic pollutants, and dangerous emerging contaminants like microplastics, biochar has a great deal of potential for remediating soil and water. This will greatly improve the quality of the soil and water. By activating biochar with nutrients, metals, and other materials, it is possible to maximize its properties and produce biochar that is appropriate for a variety of uses, including the remediation of hazardous waste sites and landfills. Biochar can also be used as an alternative product to activated carbon in wastewater treatment (Bilias et al., 2021; Cheng et al., 2023; Duwiejuah et al., 2020; Jaya Nepal et al., 2023; Johnson et al., 2012; Méndez et al., 2012; Mulchandani & Westerhoff 2016; Patel et al., 2020; Paz-Ferreiro et al. 2018; Wang, Victor, et al., 2022; Wu et al., 2022; Zhang et al., 2022).
Because of its ability to gradually mineralize into simple organic matter and other nutrients required for soil, biochar enhances soil enzyme and microbial activities. Additionally, it enhances the saturated hydraulic conductivity of soil to facilitate nutrient absorption and plant growth. It also dilutes the HM content of plant tissue to reduce its phytotoxicity (Castro et al., 2023; Cheng et al., 2020).

6.5. Recommendations

A detailed analysis of biochar highlights both its positive as well as negative/questionable aspects. However, as it is implied through the discussion that comparatively, it is a more viable option for SS disposal, the question arises for remediating the ill properties of biochar. Recommendations on biochar application to soil are guided by various factors aimed at optimizing its benefits while minimizing potential risks. A few processes that could be incorporated are thermal hydrolysis, co-pyrolysis, hydrothermal liquefaction, and biochar amendment.

[37] The lower contents of bio-available and leachable HMs in biochar as compared to SS indicates that pyrolysis process could repress the release of heavy metal in DTPA extractant (Lu et al., 2015).

6.5.1. Thermal hydrolysis

Thermal hydrolysis (TH) is a pretreatment process that effectively disintegrates and solubilizes suspended solids at high temperatures and pressure (typically at 150–200°C and 5–10 bar) for about 30 minutes. It is successfully coupled with AD. Its main purpose is to improve the bioavailability of the organic matter in the sludge thereby increasing the biogas production and reducing the final sludge production. The combined high heat and pressure may also contribute to the hydrolysis of the selected HOCs. It offers several advantages including simple operation, short treatment time, no chemical usage, high solubilization efficiency, and improved sludge settleability and dewaterability, despite its high energy demand. According to a study, the combination of CAMBI (a commercially available thermal hydrolysis process (Abu-Orf and Goss 2012) and AD increased the removal percentage of OPFRs up to 95 % (Castro et al., 2023). The energy consumption can be further reduced by recovering and reusing the steam generated during decompression. Many full-scale TH-AD plants have been installed and operated worldwide, especially in Europe and China, since the first one installed in Hamar, Norway was started up in 1995 (Eggen et al., 2019; Kim, Choi, & Lee 2024; Zhang et al., 2023).

6.5.2. Copyrolysis

Copyrolysis is a process in which two or more feedstocks are co-pyrolyzed together to produce biochar. Copyrolysis of biosolids should be considered in the future to improve the characteristics of biochar (like pH) and reduce HMs' concentration in the resultant biochar (Jaya Nepal et al., 2023; Lekan et al., 2023; Patel et al., 2020; Quan & Gao., 2016; Wang, Victor, et al., 2022). During co-pyrolysis, the organic portion of the biomass aggregates to form larger specific surface areas with increased pore structure, and enhance syngas properties, while the inorganic minerals react with HMs to form stable forms of crystalline compounds. Research conducted in China concludes that a SS/FWD (Food waste digestate) of 2:2 is optimal for HMs immobilization in biochar (Wang, Victor, et al., 2022). Copyrolysis can optimize resource utilization and increase the overall efficiency of biochar production (Quan & Gao., 2016).

6.5.3. Hydrothermal Liquefaction (HTL)

It is a thermal process adapted from the algae biofuel industry that can directly convert liquid biomass to energy in the form of bio-oil, thereby avoiding energy and costs associated with sludge dewatering. In HTL, liquid biomass reacts at a high temperature (250–350 °C) and pressure (10–15 MPa). The four products of HTL are bio-crude oil, bio-char, an aqueous component containing water-soluble compounds, and CO2 gas. The technology occupies a minimal land footprint, reduces biomass by nearly half the concentration of the metals and nutrients in it, and operates 100 times faster than AD. It can be adapted for sewage sludges, which have a wet biomass medium similar to algae (Mulchandani & Westerhoff 2016).

6.5.4. Biochar amendment

Biochar amendment or modification of biochar refers to the process of altering its properties or surface characteristics to enhance its performance for specific applications. This can involve physical, chemical, or biological treatments aimed at tailoring biochar's structure, surface chemistry, porosity, or functionality (Liu et al., 2022; Bao et al., 2022). Several methods used for biochar modification are chemical activation, functionalization[38], impregnation[39], doping[40], composite formation[41], and surface coating (Ahmad et al., 2014; Liu et al., 2022; Tiziana Crovella et al. 2024). Some benefits of biochar amendments could be the increase in the activity of specific functional groups, efficient soil nutrient management, improved crop growth and productivity, reduction in the use of fertilizers, reduced bioavailability of contaminants in soil, and overall reduction in soil GHG emissions. Subsequently, its ability of carbon sequestration would also benefit climate change mitigation. It also helps in retaining water, thus reducing the climate and economic cost of irrigation (Bao et al., 2022; Kumar Raja Vanapalli et al. 2021; Méndez et al., 2012; Paz-Ferreiro et al. 2018). Biochar can also be modified to further facilitate the retention of HMs (Liu et al., 2022; Paz-Ferreiro et al. 2018).

7. Conclusion

Sewage sludge management is a complex process that demands careful consideration of multiple factors to ensure both environmental protection and resource recovery. Through this investigation, the effectiveness of biosolids and biochar in reducing HOCs and HMs in sewage sludge is demonstrated, which is further analyzed for soil application in the Norwegian context.
The two types of biosolids produced - one with lime stabilization and the other with AD are compared with biochar produced with and without AD at both low and high pyrolysis temperatures. The findings underscore the pivotal role of biochar in reducing the HOC content and overall mass of SS. It leads to a substantive fraction of HMs being contained in the biochar fraction, which is believed to be immobilizing HMs. The higher temperature of pyrolysis allows the maximum (~100 %) degradation of OPFRs but also requires more energy for the process.

[38] Functionalization entails introducing functional groups onto biochar surfaces through chemical reactions. This can be achieved by treating biochar with acids, bases, or reactive organic compounds to modify its surface chemistry and enhance its affinity for specific contaminants or nutrients.
[39] Impregnation involves infusing biochar with additives, such as nanoparticles, metal oxides, or organic compounds, to impart additional functionalities or catalytic properties. This approach can enhance biochar's performance in catalyzing chemical reactions, facilitating pollutant degradation, or promoting nutrient transformation in soil.
[40] Doping refers to incorporating dopant materials, such as metals, metal oxides, or heteroatoms like nitrogen or sulfur, into the biochar matrix during its production or post-treatment to make them suitable for diverse environment and soil application.
[41] Composite biochars are synthesized by combining biochar with other materials, such as clay minerals, zeolites, or organic polymers, to create hybrid materials with synergistic properties.

Observing the behavior of pollutants present in biosolids and biochar during soil application, biochar produced without AD at low temperatures has the highest geoaccumulation Index, the highest retention rate of HMs, highest biochar yield and maximum mass flow of HMs in biochar fraction. Moreover, it is also efficient in diminishing the toxic effects of HMs on human health and the environment, while also leading to the least bioavailable HMs. OPFRs and PFASs mostly get degraded during the pyrolysis process, leading to less mobility, toxicity, and bioavailability. Therefore amongst the processes explored, pyrolysis without AD at low temperature (500 - 600 ℃) is optimum for the SS treatment option for its effectiveness in HOC reduction, biochar production, and environmental considerations.

Amongst HMs, Cr, Cu, Cd, and Pb have a higher accumulation rate in soil, while Cu and Zn have higher leaching potential. There is a higher toxicity observed for Cu and Zn for non-carcinogenic HMs, and Cd, Pb, and Cr in carcinogenic toxins. Pb, As, Zn, and Cd are found to be more bioavailable as compared to other HMs. In the case of OPFRs and PFASs, it is stipulated to be compound and treatment process specific. Pyrolysis significantly reduces the leaching potential of HMs.

In summary, these observations suggest that while each SS management process has its unique effects on HOCs and HMs, there is a need for integrated approaches that consider the interactions between different treatment methods to optimize the removal and transformation of these contaminants. Additionally, further research is required to elaborate the mechanisms underlying the observed trends and to develop more effective and sustainable sewage sludge management strategies. The advantages and disadvantages between the economic cost (production) and benefit value (application) of biochar need to be carefully measured. Looking ahead, the study recommends the optimization of biochar application to soil, leveraging techniques such as thermal hydrolysis, co-pyrolysis, and hydrothermal liquefaction (HTL) to enhance treatment efficiency and minimize environmental risks. Additionally, robust monitoring and regulatory frameworks are imperative to ensure the safe management of treated sludge, particularly in the context of emerging pollutants and evolving treatment technologies.

8. Scope for further research

The current state of research reveals several key areas where further investigation is warranted within the realm of biosolids and biochar utilization in agriculture. Firstly, it has been pointed out that there is a notable lack of awareness regarding the economic feasibility and practicality of employing biosolids and biochar in agricultural practices, necessitating efforts at regional, societal, and scientific levels (Bilias et al., 2021; Nicholas et al., 2022). Moreover, the absence of adequate regulations poses a significant challenge, particularly concerning emerging pollutants. Lack of data on the effects of chronic human exposure, significant variations in units, types of research, sampling & experimentation units contribute to uncertainties in results and trends, underlining the need for standardized diagnostic testing methods and comprehensive epidemiological and toxicological studies (Pozzebon et al., 2023; Tiziana Crovella et al., 2024).

The inconsistency in policy framework and guidelines for both biochar production and use can be extrapolated at different governing body levels (Patel et al., 2020).

Addressing the lack of studies is another crucial aspect of further research. As sludge is a complex matrix where contaminants are often found at trace levels, developing an efficient pre-treatment method to extract the target contaminants is extremely challenging (Martín-Pozo et al., 2019). Following this, there is a study shortfall of biosolids' effect on PFAS and OPFRs, along with their degradation removal from the environment interfaces, their behaviors in plant systems, and their human exposure risks (Castro et al., 2023; Jaya Nepal et al., 2023; Lekan et al., 2023; Patel et al., 2020; Pantelaki & Voutsa 2019; Yang et al., 2019; Zhang et al., 2021). Studies that evaluate the chemical properties of tropical soils subjected to long-term applications of biosolids are scarce, especially concerning Emerging Organic Micropollutants (EOPs) and heavy metals (Amorim Júnior et al. 2021; Jaya Nepal et al., 2023; Wang, Victor, et al., 2022). In the case of biochar, its a major research gap, since specific natural conditions like temperature, rainfall, wind, pH, etc. may significantly influence the sorption capacity and consequently the immobilization mechanisms. Although the effects of fresh biochar on soil characteristics have seen some studies, the influence of biochar aging effects on soil properties and immobilization mechanisms has been the subject of comparatively fewer scientific attempts (Jaya Nepal et al., 2023; Lekan et al., 2023; Patel et al., 2020). Thus more studies focused on biochar application in the field under natural conditions, are required to fully understand and elucidate the above mechanism (Bilias et al., 2021). Along with the field application, future studies should address factors related to metal removal efficiencies, such as application rate, dosing and recovery approaches, and regeneration and disposal of metal-sorbed biochars (Li et al., 2017). There is also a lack of information on the roles of different sorption mechanisms for different metals (Li et al., 2017). Moreover, there has only been a handful of pilot-scale demonstrations of biosolids pyrolysis reported in the literature, that too are batch or semi-continuous types. Additionally, biochar itself varies manifold based on the constituent of the biosolid in which it is made. Biochar amendment, which is seen as the next step towards carbon sequestration and improved microbial activities also needs attention. Thus to provide more precise data and regulatory suggestions, they need to be studied separately in detail (Jaya Nepal et al., 2023; Lekan et al., 2023; Patel et al., 2020; Xu et al., 2018).

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10. Appendix

A: Literature Review of key findings

Journal	Year	Location	Contaminant(s) discussed (HMs/OPFRs/PFAs)			Key objective						Methodology (if any)	Type of study		Key finding(s)
			HMs	OPFRs	PFAs	Appl. of Biosolid	Appl. of Biochar	Bioavailability	Toxicity	Mobility	LCA		Research/ Report	Review	
Júnior et al., 2021	2021	Brazil	❌	❌	❌	✅	❌	❌	❌	❌	❌	Turkey Method and Principal Component Analysis	✅	❌	Biosolids improve soil chemical parameters, and macronutrient levels with minimal accumulation of potentially toxic elements
Bagheri et al., 2023	2023	-	✅	❌	❌	✅	❌	❌	❌	❌	❌	-	❌	✅	Trend shift from “removal and treat” approach to “recovery and reuse”
Aboughaly et al., 2023	2023	-	❌	❌	✅	✅	✅	✅	✅	✅	❌	-	❌	✅	Optimal pyrolysis reaction conditions to adsorb PFAS to maximum allowable concentrations in wastewater up to EPA is 70 ng/L
Raheem et al. 2018	2018	-	✅	❌	❌	✅	✅	✅	✅	❌	❌	-	✅	✅	AD coupled with pyrolysis, co-combustion and co-incineration can be the viable routes
Paz-Ferreiro et al. 2018	2018	-	✅	❌	❌	✅	✅	✅	✅	✅	❌	-	✅	✅	Pyrolysis of biosolids have several benefits; concerns

Journal	Year	Location	Contaminant(s) discussed (HMs/OPFRs/PFAs)			Key objective							Methodology (if any)	Type of study		Key finding(s)
			HMs	OPFRs	PFAs	Appl. of Biosolid	Appl. of Biochar	Bioavail ability	Toxicity	Mobility	LCA		Research/ Report	Review	
															about the toxicity from Biochar is irrational
Wallace, Su, & Sun 2017	2017	North America	✅	❌	❌	❌	✅	✅	✅	✅	❌	Fourier-transform infrared spectra (FTIR)	✅	❌	Biochar can be used to remove HMs from stormwater
(Mulchandani & Westerhoff 2016)	2016	-	✅	❌	❌	✅	❌	✅	✅	❌	❌	Various thermal and liquid solvent purposes	✅	✅	HTL (Hydrothermal liquefaction) can be adapted for sewage sludges transformation to bio-oil; it occupies a minimal land footprint and operates 100 times faster than anaerobic digestion
Arulajah et al., 2011	2011	Australia	✅	❌	❌	✅	❌	❌	❌	❌	❌	Various geotechnical tests	✅	❌	HMs, biological and other prime contaminants were found within the permissible limits for biosolids
Charlton et al., 2016	2016	UK	✅	❌	❌	❌	❌	✅	✅	✅	❌	Long-term sludge experiments (LTSE)	✅	❌	significant decreases (7-12%) in Cmic (Soil microbial biomass carbon) have occurred in soils where the total concentrations of Zn and Cu fall below the statutory limits

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Chen et al., 2016	2016	-	❌	❌	❌	✅	❌	❌	❌	❌	❌	Quantitative PCR, Illumina sequencing, Statistical analysis	✅	❌	Long-term application of sewage sludge (and chicken manure) can increase the abundance and diversity of ARGs and bacteria.
Cheng et al., 2023	2023	-	✅	❌	❌	✅	✅	✅	✅	✅	❌	-	✅	❌	Slow pyrolysis is a better biochar production method; biochar can be used as activated carbon for HMs and Organo micropollutants
Kim, Choi & Lee 2024	2024	-	❌	❌	❌	✅	❌	✅	❌	❌	❌	Anaerobic reactor operations, analytical methods	✅	❌	The thermal hydrolysis (TH) pretreatment is proven effective in solubilizing DSS and improving its bioavailability
Li et al., 2019	2019	-	✅	❌	❌	❌	❌	❌	✅	❌	❌	-	✅	❌	Phytoremediation has proven to be an effective method for heavy metals removal from soil
Liang et al., 2021	2021	-	✅	❌	❌	✅	❌	❌	✅	❌	❌	-	✅	❌	Sewage sludge incineration needs to be carefully monitored; pyrolysis is much viable option
Patel et al., 2020	2020	-	✅	❌	✅	❌	✅	✅	❌	✅	❌	-	❌	✅	Biosolids to biochar via pyrolysis can be an

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																effective option for biosolids management.
Shaddel et al., 2019	2019	-	❌	❌	❌	✅	❌	✅	❌	❌	❌	-	✅	❌	Nutrient recovery from sewage sludge requires a sustainable approach by utilization of appropriate technical options.
Qin et al., 2022	2022	-	✅	❌	❌	✅	❌	❌	❌	❌	❌	-	✅	❌	Long-term biosolids application increased the relative abundance of ARGs in soil.
Pozzebon et al., 2023	2023	US	❌	❌	✅	✅	❌	✅	✅	✅	❌	-	❌	✅	Current regulations on pollutants do not have provisions for PFAs; significant lack of information on the effects of PFAs application and interaction with soil
Jaya Nepal et al., 2023	2023	-	✅	❌	❌	❌	✅	✅	✅	✅	❌	-	✅	❌	Long-term biochar application can promote nutrient retention and soil productivity, helping reduce chemical fertilizer needs over time; A knowledge gap remains in understanding the long-term persistence of biochar on agroecosystem

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Antunes et al., 2021	2021	-	❌	❌	✅	❌	✅	❌	❌	❌	❌	-	✅	❌	Biochar can be used as an adsorbent for removal of contaminants.
Shahsavari et al., 2021	2021	-	❌	❌	✅	✅	❌	❌	✅	❌	❌	-	❌	✅	A combination of phytoremediation and PFAS-degrading bacteria can be used for biodegradation of PFAS.
Morales et al., 2023	2023	-	✅	✅	✅	✅	✅	✅	✅	✅	❌	Experimentation and data analysis using different methods	✅	❌	Pyrolysis and incineration degrade from 94% to 99% of hazardous organic compounds (PFAS, OPFRs, and BPA)
Eggen et al., 2022	2022	Norway	✅	❌	❌	❌	❌	✅	✅	✅	❌	Modeling and assessment from data in a 100-year perspective	✅	❌	There is little information available on the speciation of PTEs As, Hg, Cr, as well as the significance of Hg volatilization as a removal process from soil. Climate change results in changes in temperature and precipitation that will affect the release, transport, and use of PTEs.

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Eggen et al., 2019	2019	Norway	❌	❌	✅	✅	✅	✅	✅	✅	❌	-	✅	❌	Pyrolysis produce C-rich (>50%) biochar; justifies excluding sewage sludge from positive input material list to ensure human health and environmental safety.
Bamdad et al., 2022	2022	Canada	❌	❌	✅	✅	✅	❌	✅	❌	❌	Pyrolysis reactor, Biosolids & Biochar sampling analysis; PFAS analysis	✅	❌	Treatment process at higher pyrolysis temperatures can remarkably reduce or eliminate the level of PFAS (by ~97–100 wt%) in the resulting biochar samples
Bilias et al., 2021	2021	-	✅	❌	❌	✅	✅	✅	✅	✅	❌	-	✅	✅	Biochar helps in significant reduction of mobility, bioavailability and leachability of potentially toxic elements (PTEs)
Bogusz et al., 2017	2017	-	✅	❌	❌	❌	✅	✅	✅	✅	❌	Batch sorption experiment	✅	❌	2.5% addition of biochar to sewage sludge increased the soil's sorption capacity toward the PTEs and the mobility of PTEs was reduced.
Xu et al. 2018	2018	-	✅	❌	❌	❌	✅	✅	✅	✅	❌	Soil spiking, biochar amendment and	✅	❌	Biochar addition reduced metal toxicity and enhanced microbial ability in immobilization of soil

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													incubation experiment			carbon under contaminated soils.
Wu et al. 2022	2022	-	✅	❌	❌	❌	✅	✅	✅	✅	❌		Fungal HiSeq sequencing and bioinformatics analysis; Statistical analyses	✅	❌	Biochar can reduce the bioavailability of heavy metals.
Wang et al., 2022	2022	China	✅	❌	❌	❌	✅	✅	✅	✅	❌		Soil bacterial analysis, statistical analysis	✅	❌	Pyrolysis temperature of biochar had no significant effect on bioavailable Cd in Cd, Pb and As contaminated soil, but the bioavailable Pb reduced and As increased with pyrolysis temperature raising.
Tiziana Crovella et al. 2024	2024	-	❌	❌	❌	❌	✅	❌	✅	❌	✅		-	❌	✅	The recovery of nutrients biosolids for soil amendment can generate a GWP (Global Warming Potential) gain up to - 37 kg CO2-eq.
Thompson et al., 2023	2023	United States	❌	❌	✅	✅	❌	❌	❌	✅	❌		LC-MS/MS Analysis	✅	❌	Anaerobic digestion, heat treatment and drying helps remove PFAS from the sewage sludge.

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Thoma et al., 2022	2022	-	❌	❌	✅	✅	✅	❌	❌	❌	❌	-	✅	❌	PFAS were significantly removed from biosolids while converting to biochar.
Sørmo et al. 2020	2020	Norway	❌	❌	✅	❌	✅	✅	❌	✅	❌	Potentiometry, DIN 38414-S14 method	✅	❌	Biochar can be used to reduce the leaching of PFAS from contaminated soil.
Rosa et al., 2022	2022	Spain	✅	❌	❌	❌	✅	✅	✅	✅	❌	Elemental, chromatographic, and spectrospic analyses	✅	❌	Biochar helps in stabilization of heavy metals in contaminated soil.
Nicholas et al., 2022	2022	Wales	❌	❌	❌	✅	✅	❌	❌	❌	❌	-	✅	❌	Males are almost twice as likely than females to have a positive perception of biosolids (OR 1.91, p value 0.004) and fecal sludge biochar (OR 2.02, p value 0.03); Older age people (65+) are more likely to have a positive view of fecal sludge biochar than the youngest age group (OR 4.88, p value 0.001).

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Marchuk et al., 2023	2023	Australia	✅	❌	✅	✅	✅	✅	✅	✅	❌	-	✅	✅	Heavy metals can be immobilized in biochar derived from biosolids, reducing their bioavailability and reducing the risk of soil-plant contamination.
Manikandan et al., 2023	2023	-	✅	❌	❌	✅	✅	✅	✅	✅	❌	-	✅	❌	Biochar has enormous potential for removing heavy metal ions and pesticides from soil.
Lekan et al., 2023	2023	UK	✅	❌	✅	✅	❌	❌	✅	❌	❌	-	❌	✅	Government regulations used in contaminant limits in biosolids need upgrading and extensive research
Kundu et al., 2021	2021	-	✅	❌	✅	✅	✅	❌	❌	❌	❌	-	✅	❌	Yield of Biochar: 36–45% at 500–600 °C; >90% removal of PFOS and PFOA from biosolids derived biochar in pyrolysis-combustion integration process; >80% adsorption of long-chain PFASs and 19–27% adsorption of short-chain PFASs in biochar

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Kumar et al., 2023	2023	-	❌	❌	✅	✅	✅	✅	✅	✅	❌	-	✅	❌	Pyrolysis and gasification are efficient solutions to mitigate PFAS and convert biosolids into biochar which can be applied in agriculture.
Kumar Raja Vanapalli et al. 2021	2021	-	❌	❌	❌	❌	✅	✅	❌	❌	❌	-	✅	❌	Biochar has excellent agronomic properties, helps in soil quality improvement, facilitates crop productivity and is economically profitable.
Kończak and Oleszczuk 2020	2020	Italy	✅	❌	❌	❌	✅	✅	✅	✅	❌	-	✅	✅	Pyrolysis in CO2 caused the increase the metal content in biochar; metal content varied by elements and pyrolysis temperature.
Jingzi Beiyuan et al. 2021	2021	-	❌	❌	❌	❌	✅	✅	✅	✅	❌	-	✅	❌	Fe/Mn-modified biochar is even more effective in removing As from contaminated soil and water.

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Fahmi et al., 2018	2018	Malaysia	✅	❌	❌	❌	✅	✅	❌	✅	❌	-	✅	❌	The efficiency of biochar to immobilize heavy metals can be increased by reducing the particle size, which can increase the surface area and the cation exchange capacity (CEC).
Duwiejuah et al., 2020	2020	-	✅	❌	❌	❌	✅	✅	✅	✅	❌	PRISMA (Preferred Reporting Items for Systematic Reviews and Meta-Analyses)	❌	✅	The characteristics of biochar are largely dependent on the feed stock biomass and pyrolysis conditions.
Dike et al., 2021	2021	-	❌	❌	❌	❌	✅	✅	✅	✅	❌	-	❌	✅	Biochar can act as a biostimulator in the bioremediation of hydrocarbon-contaminated soils.
Cheng et al., 2020	2020	-	✅	❌	❌	❌	✅	✅	✅	✅	❌	-	✅	❌	The HM uptake and accumulation of plants can be suppressed by biochar through pH value change and DOC (dissolved organic carbon) content.
Braine et al., 2024	2024	-	✅	❌	✅	✅	❌	✅	✅	✅	❌	-	❌	✅	Substantial variations are found between different country's guidelines;

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Bolan et al., 2024	2024	-	❌	❌	❌	✅	✅	✅	✅	✅	❌	-	✅	❌	Biochar promotes microbial activity and function, and indirectly by altering soil physical and chemical properties.
Bolan, Sarkar, Yan, et al. 2021	2021	-	❌	❌	✅	✅	✅	✅	✅	✅	❌	-	❌	✅	Mobilization techniques can be used for the complete removal of PFAS compounds through abiotic and biotic degradation.
Biel-Maeso et al., 2019	2019	Spain	❌	✅	❌	❌	❌	✅	✅	❌	❌	-	✅	❌	AD alone is proven inefficient for the removal/elimination of OPFRs.
Bika et al., 2022	2022	-	❌	✅	❌	❌	❌	✅	✅	❌	❌	-	✅	❌	OPFRs are proven to be detrimental to plants, animals, and humans alike; there's still no proper guidelines regarding OPFRs
Campo & Yolanda 2020	2020	-	❌	✅	✅	❌	❌	❌	✅	✅	❌	-	❌	✅	Chromatography coupled with mass spectrometry is an efficient method for analysis of OPFRs and PFAs.

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[Castro et al., 2023]()	2023	Norway	❌	✅	❌	❌	✅	❌	❌	✅	❌	Matrix-solid phase dispersion (MSPD), ultra-performance liquid chromatography (UPLC), tandem mass spectrometry (MS/MS)	✅	❌	Pyrolysis at 500 °C successfully removed >99 % OPFRs from digested sludge.
[Cristale et al., 2016]()	2016	Spain	❌	✅	❌	❌	❌	❌	✅	❌	❌	-	✅	❌	OPFRs present in wastewater influents have low degradability during the conventional activated sludge treatment.
[Evich et al., 2022]()	2022	-	❌	❌	✅	✅	❌	✅	✅	✅	❌	-	✅	❌	Long-term PFAS preferentially adsorb to soil phases slowing the rate of microbial transformation.
[Healy et al., 2016]()	2016	Ireland	❌	❌	❌	✅	❌	❌	✅	❌	❌	X-ray fluorescence (XRF) analyser	✅	❌	Land application of biosolids are determined by their nutrient content and not the HM content; Sb and Sn remain omitted from EU regulation.
[Holmquist et al., 2020]()	2020	-	❌	❌	✅	❌	❌	❌	✅	❌	✅	Life cycle impact assessment	✅	❌	PFAS's degradation rate and pathways are highly uncertain.
[Kumar et al., 2022]()	2022	Australia	✅	✅	✅	✅	❌	✅	✅	✅	❌	-	❌	✅	Physicochemical properties of ECs like hydrophobicity play a vital role in the

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															adsorption of ECs on the sludge.
Martín-Pozo et al., 2019	2019	-	❌	✅	❌	✅	❌	❌	✅	❌	❌	-	❌	✅	Liquid chromatography (LC) and gas chromatography (GC) coupled to mass spectrometry are generally employed as the analytical technique for OPFRs.
Lu et al., 2015	2015	China	✅	❌	❌	❌	✅	✅	✅	✅	❌	-	✅	❌	HM concentration is more in biochar than SS, it increases with pyrolysis temperature.
Méndez et al., 2012	2012	Spain	✅	❌	❌	❌	✅	✅	✅	✅	❌	-	✅	❌	Pyrolysis process decreased the plant-available of Cu, Ni, Zn and Pb, the mobile forms of Cu, Ni, Zn, Cd and Pb and also the risk of leaching of Cu, Ni, Zn and Cd.
Pantelaki & Voutsa 2019	2019	-	❌	✅	❌	❌	❌	✅	✅	✅	❌	-	❌	✅	There is a significant knowledge gap in the understanding and fate of OPFRs.
Rigby et al., 2021	2021	UK	❌	✅	✅	✅	❌	❌	✅	❌	❌	-	✅	❌	The most significant of PFAS groups are PFOS and PFOA; TCPP, DBDPE, TDCPP and TCEP are the most found compounds of OPFR group in SS.

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Wang, Victor, et al., 2022	2022	China	✅	❌	❌	❌	✅	✅	✅	✅	❌	-	✅	❌	Co-pyrolysis of SS with food waste digestate (FWD) (SS/FWD 2:2) reduces the bioavailability of HMs in biochar.
Yang et al., 2019	2019	-	❌	✅	❌	❌	❌	❌	✅	✅	❌	-	❌	✅	There is very little research on the degradation/removal of OPFRs.
Zhang et al., 2021	2021	-	❌	✅	❌	✅	✅	✅	✅	❌	❌	-	❌	✅	OPFRs cause phytotoxicity in plants and affect their physiological conditions.
Zhang et al., 2022	2022	China	✅	❌	❌	❌	✅	✅	✅	✅	❌	-	✅	✅	Rise in pyrolytic temperature led to decrease in biochar yield and environmental risk, increase in pH, specific surface area, stability of biochar and HMs immobilization.
Zhang et al., 2023	2023	China	❌	✅	❌	❌	❌	❌	✅	❌	❌	Cambi® thermal hydrolysis (TH) + advanced anaerobic digestion (AAD) + plate-frame pressure filtration	✅	✅	Cambi® thermal hydrolysis + advanced anaerobic digestion + plate-frame pressure filtration" could reduce the tri-OPFR content in sludge.
Hachib Mohammad Tusar et al. 2023	2023	-	❌	❌	❌	❌	✅	✅	✅	❌	❌	-	❌	✅	Biochar enhances the nutrient availability of acidic soil by increasing the soil pH, consumption of OH^- (hydroxyl) ions

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															released from the dissociating phenolic functional groups and enhancing nutrient retention.
Ahmad et al., 2014	2014	-	✅	❌	❌	❌	✅	✅	✅	✅	❌	XAFS spectroscopy	❌	✅	Sorption capacity depends on the surface area, microporosity, and hydrophobicity of biochar.
Li et al., 2017	2017	-	✅	❌	❌	✅	✅	❌	❌	❌	❌	-	❌	✅	Pyrolytic temperature has a profound impact on the characteristics and behavior of biochar.
Li et al., 2013	2013	China	❌	❌	❌	❌	✅	❌	❌	❌	❌	Solid-state ^{13}C NMR Spectroscopy, 2D Correlation Spectroscopy, Statistical analysis	✅	❌	The main functional groups of biochar are aromatic and heterocyclic carbons.
Janu et al., 2021	2021	-	✅	❌	❌	❌	✅	❌	❌	✅	❌	Diffusive Reflection Fourier Transformation Infrared Spectroscopy	✅	❌	Pyrolysis temperature of 600 °C lead to a partial and a 750 °C to a nearly complete loss of biochar surface functional groups.

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Bao et al., 2022	2022	-	✅	❌	❌	❌	✅	✅	✅	❌	❌	-	❌	✅	Biochar modification can increase the activity of specific functional groups, adjust pH, nutrients, moisture, and enzyme activity, and suppress greenhouse gases through surface structure and microorganisms.
Liu et al., 2022	2022	-	✅	❌	❌	❌	✅	✅	✅	✅	❌	-	✅	❌	Biochar modifications can increase the specific surface areas, active sites, pore volumes and functional groups of biochar, thereby enhancing the sorption and fixation, and catalytic reduction/degradation of heavy metals and organic contaminants in the environment.
Yoshida et al., 2018	2018	-	❌	❌	❌	✅	✅	✅	✅	✅	✅	Life cycle impact assessment	✅	✅	LCA due to C and N emission, pointed at human toxicity non-carcinogenic and ecotoxicity as being the impact categories of highest concern for sewage treatment technologies
Tarpani et al., 2020	2020	UK	✅	❌	❌	✅	✅	✅	✅	❌	✅	ISO 14040/14044 LCA guidelines application	✅	❌	AD has highest total freshwater ecotoxicity; Thermal processes are environmentally beneficial

Journal	Year	Location	Contaminant(s) discussed (HMs/OPFRs/PFAs)			Key objective						Methodology (if any)	Type of study		Key finding(s)
			HMs	OPFRs	PFAs	Appl. of Biosolid	Appl. of Biochar	Bioavail ability	Toxicity	Mobility	LCA		Research/ Report	Review	
															only at high resource recovery.
Harder et al., 2016	2016	Sweden	✅	❌	❌	✅	✅	✅	✅	❌	✅	LCIA model	✅	❌	There is an uncertainty associated with human toxicity in LCA.
Pradel et al. 2014	2014	France	✅	❌	❌	✅	❌	❌	✅	❌	✅	LCIA model	✅	❌	Atleast 60% of the impact on climate change is due to the treatment processes.
Lanko et al. 2020	2020	-	✅	❌	❌	✅	❌	❌	✅	❌	✅	-	✅	❌	Products as nutrients and energy recovered from the AD systems and incorporated into the sludge treatment create an amount of credits that make the whole WWTP more environmentally friendly.

B: List of hazardous organic compounds and heavy metals analyzed

Classification	IUPAC Name	Abbreviation	CAS number[42]
OPFRs (organophosphate flame retardants)			
OPFRs-Alkyl	Trimethyl phosphate	TMP	000512-56-1
	Triethyl phosphate	TEP	000078-40-0
	Tripropyl phosphate	TnPP	000513-08-6
	Tributyl phosphate	TnBP	000126-73-8
	Triisobutyl phosphate	TiBP	000126-71-6
	bis(2-butoxyethyl) 2-hydroxyethyl phosphate	BBOEHEP	1477494-86-2
	Trimethylolpropane phosphate	TMPP	001005-93-2
	Tris(2-butoxyethyl) phosphate	TBOEP	000078-51-3
	Bis(2-butoxyethyl) 3-hydroxyl-2-butoxyethyl phosphate	3OH-TBOEP	1477494-87-3
	TriS(2-ethylhexyl) phosphate	TEHP	000078-42-2
OPFRs -Chlorinated	Tris(2-chloroethyl) phosphate	TCEP	000115-96-8
	Tris(1-chloro-2-propyl) phosphate	TCiPP	013674-84-5
	Tris(1,3-dichloro-2-propyl) phosphate	TDCIPP	013674-87-8
	Commercial products of 2,2-bis(chloromethyl) trimethylene bis [bis (2 chloroethyl) phosphate]	V6	038051-10-4
OPFRs- Aryl	Triphenyl phosphate	TPhP	000115-86-6
	Diphenyl methylphosphonate	DPMP	007526-26-3
	2-Ethylhexyl diphenyl phosphate	EHDP	001241-94-7
	Isodecyl diphenyl phosphate	IDPhP	029761-21-5
	Tris(4-tert-butylphenyl) phosphate	TTBPP	000078-33-1
	Rersorcinol bis(diphenylphosphate)	RDP	057583-54-7
	Bisphenol A bis (diphenyl phosphate)	BPA-BDPP	005945-33-5

[42] A CAS Registry Number is a unique identification number, assigned by the Chemical Abstracts Service in the US to every chemical substance described in the open scientific literature, in order to index the substance in the CAS Registry.

Classification	IUPAC Name	Abbreviation	CAS number[42]
PFAs (Poly-and perfluoroalkylated substances)			
PFAs-Uncategorized	2,3,3,3-tetrafluoro-2-(1,1,2,2,3,3,3 heptafluoropropoxy)propanoate	Gen-X	062037-80-3
	bis[2-(N-ethylperfluorooctane-1-sulfonamido) ethyl] phosphate	SAmPAP Di	030381-98-7
	9-chlorohexadecafluoro-3-oxanonane-1-sulfonate	F53B	073606-19- 6
	dodecafluoro-3H-4,8-dioxanonanoate	NaDONA	958445-44-8
	Sodium 1- decanesulfonate	DecaS	013419-61-9
PFAs-FTS (Fluorotelomer sulfonates)	1H,2H-Perfluorohexan sulfonate (4:2)	4:2 FTS	757124-72-4
	1H,2H-Perfluorooctane sulfonate (6:2)	6:2 FTS	027619-97-2
	1H,2H-Perfluorodecan sulfonate (8:2)	8:2 FTS	039108-34-4
	1H,2H-Perfluorododecan sulfonate (10:2)	10:2 FTS	120226-60-0
PFAs-PFCA (Perfluoroalkyl carboxylates)	Perfluorobutanoic acid	PFBA	000375-22-4
	Perfluoropentanoic acid	PFPeA	002706-90-3
	Perfluorohexanoic acid	PFHxA	000307-24-4
	Perfluoroheptanoic acid	PFHpA	000375-85-9
	Perfluorooctanoic acid	PFOA	000335-67-1
	Perfluorononanoic acid	PFNA	000375-95-1
	Perfluorodecanoic acid	PFDA	000335-76-2
	Perfluoroundecanoic acid	PFUnDA	002058-94-8
	Perfluorododecanoic acid	PFDoDA	000307-55-1
	Perfluorotridecanoic acid	PFTrDA	072629-94-8
	Perfluorotetradecanoic acid	PFTeDA	000376-06-7
	Perfluoro-n-hexadecanoic acid	PFHxDA	0067905-19-5
	Perfluorooctadecanoic acid	PFOcDA	016517-11-6
	7H-Dodecafluoroheptanoic Acid	7H-PFHpA	001546-95-8
	Perfluoro-3,7-dimethyloctanoic acid	PF-3,7-DMOA	172155-07-6
PFAs-PFSA (Perfluoroalkane sulfonates)	Perfluorobutanoic acid sulfonate	PFBS	108427-52-7
	Perfluoropentane sulfonic acid	PFPeS	002706-91-4

Classification	IUPAC Name	Abbreviation	CAS number[42]
	Perfluorohexane sulfonic acid	PFHxS	000355-46-4
	Perfluoro-1-heptanesulfonate	PFHpS	146689-46-5
	Perfluorooctano sulfonic acid	PFOS	001763-23-1
	Perfluorononane sulfonic acid	PFNS	068259-12-1
	Perfluorodecane sulfonic acid	PFDS	000335-77-3
	Perfluorododecane sulfonic acid	PFDoDS	079780-39-5
	Perfluoroethylcyclohexane sulfonic acid	PFECHS	000335-24-0
PFAs-PreFOS (Perfluorooctane sulfonate precursors)	Perfluorooctane sulfonamide	PFOSA	000754-91-6
	N-methylPerfluoro-1-octanesulfonamide	MeFOSA	031506-32-8
	N-ethyl perflurooctane sulfonamide	EtFOSA	004151-50-2
	N-(2-hydroxyethyl)-N-methylperfluorooctane sulfonamide	MeFOSE	024448-09-7
	N-ethyl-N-(2-hydroxyethyl)-N-methylperfluorooctane sulfonamide	EtFOSE	001691-99-2
	Perfluoro-1-octanesulfonamidoacetic acid	FOSAA	002806-24-8
	2-(N-methylPerfluoro-1-octansulfonamido) acetic acid	MeFOSAA	002355-31-9
	N-ethylPerfluoro-1-octanesulfonamide acetic acid	EtFOSAA	001336-61-4
Heavy metals			
	Arsenic	As	17428-41-0
	Barium	Ba	22541-12-4
	Cadmium	Cd	22537-48-0
	Cobalt	Co	22541-53-3
	Chromium	Cr	18540-29-9
	Copper	Cu	15158-11-9
	Molybdenum	Mo	16065-87-5
	Nickel	Ni	14701-22-5
	Lead	Pb	14280-50-3
	Strontium	Sr	22537-39-9
	Vanadium	V	15121-26-3
	Zinc	Zn	23713-49-7

C: Mass flow for HOCs and HMs in output flows in C1, C2, and C3 (low and high temperature)

Mass flow for HOCs and HMs in output flows in C1, C2, and C3 (low and high temperature)															
				C1		C2		C3 (low temp)				C3 (High temp)			
Component		Unit	Raw sewage sludge	Effluent	Dewatered SS	Effluent	Dewatered SS	Effluent	Biochar	Comb. gas	Solid residues CHP	Effluent	Biochar	Comb. gas	Solid residues CHP
Biomass, biochar, solid residues (dry basis)→		kg-dw/d	250	12.5	237.5	6.85	130.06	6.85	63.54	830.86	16.96	6.85	61.92	856.95	17.49
OPFRs	TMP	mg/d	0	0	0	0	0	0	0	0	0	0	0	0	0
	TEP	mg/d	7.22	0.12	2.05	0.4	6.82	0.4	0	0	0	0.4	0	0	0
	TnPP	mg/d	0	0	0	0	0	0	0	0	0	0	0	0	0
	TnBP	mg/d	9.46	0.46	8.04	0.16	2.72	0.16	0.07	0	0	0.16	0	0	0
	TiBP	mg/d	5.37	0.29	4.55	0.11	1.67	0.11	0	0	0	0.11	0	0	0
	TCEP	mg/d	0	0	0	0	0	0	0	0	0	0	0	0	0
	TCiPP	mg/d	206	3.25	37.95	6.75	61.23	6.75	0	0	0	6.75	0	0	0
	TPhP	mg/d	26.08	0	0	0.15	2.87	0.15	0	0	0	0.15	0	0	0
	DPMP	mg/d	21.65	0	0	0.21	3.56	0.21	0	0	0	0.21	0	0	0
	BBOEHEP	mg/d	7.08	0.06	1	0.18	3.09	0.18	0	0	0	0.18	0	0	0
	TMPP	mg/d	14.26	0	0	0.36	6.13	0.36	0	0	0	0.36	0	0	0
	EHDP	mg/d	0	0	0	0	0	0	0	0	0	0	0	0	0
	IDPhP	mg/d	16.78	0.21	3.72	0.6	10.06	0.6	0	0	0	0.6	0	0	0
	TBOEP	mg/d	80.39	0.66	11.39	2.19	36.44	2.19	1.81	0	0	2.19	0	0	0
	3OH-T	mg/d	0.63	0.01	0.09	0.04	0.6	0.04	0	0	0	0.04	0	0	0

Mass flow for HOCs and HMs in output flows in C1, C2, and C3 (low and high temperature)															
				C1		C2		C3 (low temp)				C3 (High temp)			
Component		Unit	Raw sewage sludge	Effluent	Dewatered SS	Effluent	Dewatered SS	Effluent	Biochar	Comb. gas	Solid residues CHP	Effluent	Biochar	Comb. gas	Solid residues CHP
	BOEP														
	TDCIPP	mg/d	42.77	0	0	1.04	16.45	1.04	0	0	0	1.04	0	0	0
	TEHP	mg/d	53.87	2.42	46.06	1.33	26.61	1.33	0	0	0	1.33	0	0	0
	TTBPP	mg/d	2.93	0.02	0.28	0.16	2.77	0.16	0	0	0	0.16	0	0	0
	RDP	mg/d	5.33	0	0	0.03	0.49	0.03	0	0	0	0.03	0	0	0
	V6	mg/d	0.32	0	0	0	0.08	0	0	0	0	0	0	0	0
	BPA-BDPP	mg/d	5.67	0	0	0.14	2.28	0.14	0	0	0	0.14	0	0	0
PFAS (Uncategorized)	Gen-X	mg/d	0	0	0	0	0	0	0	0	0	0	0	0	0
	SAmPAP Di	mg/d	0.89	0.05	0.85	0.05	0.85	0.05	0	0	0	0.05	0	0	0
	F53B	mg/d	0.04	0	0.04	0	0.04	0	0	0	0	0	0	0	0
	NaDONA	mg/d	0	0	0	0	0	0	0	0	0	0	0	0	0
	DecaS	mg/d	0	0	0	11.01	205.52	11.01	0	0	0	11.01	0	0	0
PFAS-FTS	4:2 FTS	mg/d	1.4	0.09	1.31	0.01	0.08	0.01	0	0	0	0.01	0	0	0
	6:2 FTS	mg/d	0.08	0	0.07	0	0.03	0	0	0	0	0	0	0	0
	8:2 FTS	mg/d	0.02	0	0.02	0	0.02	0	0	0	0	0	0	0	0
	10:2 FTS	mg/d	0.14	0.01	0.13	0.01	0.24	0.01	0	0	0	0.01	0	0	0

Mass flow for HOCs and HMs in output flows in C1, C2, and C3 (low and high temperature)															
				C1		C2		C3 (low temp)				C3 (High temp)			
Component		Unit	Raw sewage sludge	Effluent	Dewatered SS	Effluent	Dewatered SS	Effluent	Biochar	Comb. gas	Solid residues CHP	Effluent	Biochar	Comb. gas	Solid residues CHP
PFAS-PFCA	PFBA	mg/d	0.39	0.07	0.32	0.1	0.29	0.1	0	0	0	0.1	0	0	0
	PFPeA	mg/d	0.54	0.03	0.51	0.03	0.51	0.03	0	0	0	0.03	0	0	0
	PFHxA	mg/d	4.48	0.26	4.22	0.23	3.49	0.23	0	0	0	0.23	0	0	0
	PFHpA	mg/d	0.06	0	0.06	0	0.06	0	0	0	0	0	0	0	0
	PFOA	mg/d	41.73	2.44	39.28	3.38	50.13	3.38	0	0	0	3.38	0	0	0
	PFNA	mg/d	0	0	0	5.23	100.97	5.23	0	0	0	5.23	0	0	0
	PFDA	mg/d	0.04	0	0.04	0	0.04	0	0	0	0	0	0	0	0
	PFUnDA	mg/d	0.07	0	0.07	1.37	26.82	1.37	0	0	0	1.37	0	0	0
	PFDoDA	mg/d	0	0	0	0	0	0	0	0	0	0	0	0	0
	PFTrDA	mg/d	42.66	2.14	40.52	0	0	0	0	0	0	0	0	0	0
	PFTeDA	mg/d	11.43	0.63	10.79	0.16	2.66	0.16	0	0	0	0.16	0	0	0
	PFHxDA	mg/d	7.86	0.39	7.47	0.37	7.49	0.37	0	0	0	0.37	0	0	0
	PFOcDA	mg/d	0.02	0	0.02	0	0.02	0	0	0	0	0	0	0	0
	7H-PFHpA	mg/d	0.76	0.04	0.72	0.01	0.12	0.01	0	0	0	0.01	0	0	0
	PF-3,7-DMOA	mg/d	0.14	0.01	0.14	0	0	0	0	0	0	0	0	0	0

Mass flow for HOCs and HMs in output flows in C1, C2, and C3 (low and high temperature)															
				C1		C2		C3 (low temp)				C3 (High temp)			
Component		Unit	Raw sewage sludge	Effluent	Dewatered SS	Effluent	Dewatered SS	Effluent	Biochar	Comb. gas	Solid residues CHP	Effluent	Biochar	Comb. gas	Solid residues CHP
PFAs-PFSA	PFBS	mg/d	0.04	0	0.03	0	0.03	0	0	0	0		0	0	0
	PFPeS	mg/d	0.69	0.04	0.65	0.05	0.64	0.05	0	0	0	0.05	0	0	0
	PFHxS	mg/d	0.03	0	0.03	0	0.03	0	0	0	0	0	0	0	0
	PFHpS	mg/d	8.82	0.52	8.3	0.07	0.98	0.07	0	0	0	0.07	0	0	0
	PFOS	mg/d	1.72	0.09	1.63	0.09	1.69	0.09	0.01	0	0	0.09	0	0	0
	PFNS	mg/d	0.1	0	0.09	0	0.09	0	0	0	0	0	0	0	0
	PFDS	mg/d	0	0	0	0	0	0	0	0	0	0	0	0	0
	PFDoDS	mg/d	1.05	0.05	0.99	0.05	1	0.05	0	0	0	0.05	0	0	0
	PFECHS	mg/d	0	0	0	0	0	0	0	0	0	0	0	0	0
PFAs-PreFOS	PFOSA	mg/d	0	0	0	0	0	0	0	0	0	0	0	0	0
	MeFOSA	mg/d	0.5	0.03	0.47	0	0	0	0	0	0	0	0	0	0
	EtFOSA	mg/d	9.41	0.5	8.92	0	0	0	0	0	0	0	0	0	0
	MeFOSE	mg/d	0	0	0	0	0	0	0	0	0	0	0	0	0
	EtFOSE	mg/d	0	0	0	0	0	0	0	0	0	0	0	0	0
	FOSAA	mg/d	4.44	0.22	4.21	0.01	0.29	0.01	0	0	0	0.01	0	0	0

Mass flow for HOCs and HMs in output flows in C1, C2, and C3 (low and high temperature)															
				C1		C2		C3 (low temp)				C3 (High temp)			
Component		Unit	Raw sewage sludge	Effluent	Dewatered SS	Effluent	Dewatered SS	Effluent	Biochar	Comb. gas	Solid residues CHP	Effluent	Biochar	Comb. gas	Solid residues CHP
	MeFOS AA	mg/d	0.55	0.03	0.52	0	0	0	0	0	0	0	0	0	0
	EtFOS AA	mg/d	4.88	0.27	4.61	0.01	0	0.01	0	0	0	0.01	0.22	0	0
HMs	As	mg/d	4.88E+02	1.38E+02	3.51E+02	3.54E+02	1.35E+02	3.54E+02	9.46E+01	1.60E+01	2.40E+01	3.54E+02	1.02E+02	1.29E+01	1.94E+01
	Ba	mg/d	2.90E+04	9.55E+03	1.94E+04	2.08E+04	8.15E+03	2.08E+04	5.92E+03	1.22E+03	1.01E+03	2.08E+04	6.66E+03	8.12E+02	6.72E+02
	Cd	mg/d	1.31E+02	7.30E+01	5.81E+01	8.97E+01	4.14E+01	8.97E+01	6.10E+00	3.53E+01	0.00E+00	8.97E+01	1.04E+00	4.03E+01	0.00E+00
	Co	mg/d	7.92E+02	2.61E+02	5.31E+02	5.85E+02	2.06E+02	5.85E+02	1.59E+02	2.59E+01	2.14E+01	5.85E+02	1.74E+02	1.78E+01	1.47E+01
	Cr	mg/d	6.39E+03	3.20E+02	6.07E+03	4.60E+03	1.79E+03	4.60E+03	1.42E+03	1.58E+02	2.10E+02	4.60E+03	1.49E+03	1.31E+02	1.74E+02
	Cu	mg/d	3.44E+04	1.09E+04	2.35E+04	2.44E+04	9.97E+03	2.44E+04	6.85E+03	1.81E+03	1.31E+03	2.44E+04	7.29E+03	1.55E+03	1.12E+03
	Mo	mg/d	1.19E+03	3.94E+02	8.00E+02	8.50E+02	3.44E+02	8.50E+02	2.55E+02	4.90E+01	4.06E+01	8.50E+02	2.91E+02	2.90E+01	2.40E+01
	Ni	mg/d	4.31E+03	2.26E+03	2.05E+03	3.19E+03	1.12E+03	3.19E+03	8.32E+02	1.43E+02	1.43E+02	3.19E+03	9.48E+02	8.48E+01	8.48E+01
	Pb	mg/d	3.46E+03	1.40E+03	2.05E+03	2.52E+03	9.42E+02	2.52E+03	7.21E+02	9.94E+01	1.22E+02	2.52E+03	5.75E+02	1.65E+02	2.02E+02
	Sr	mg/d	2.16E+04	7.11E+03	1.44E+04	1.50E+04	6.58E+03	1.50E+04	4.63E+03	1.07E+03	8.83E+02	1.50E+04	5.19E+03	7.63E+02	6.32E+02
	V	mg/d	4.14E+0	1.37E+0	2.78E+0	3.06E+0	1.08E+0	3.06E+0	8.97E+0	1.01E+0	8.33E+0	3.06E+0	9.77E+0	5.70E+0	4.72E+0

Mass flow for HOCs and HMs in output flows in C1, C2, and C3 (low and high temperature)														
				C1		C2		C3 (low temp)				C3 (High temp)			
Component		Unit	Raw sewage sludge	Effluent	Dewatered SS	Effluent	Dewatered SS	Effluent	Biochar	Comb. gas	Solid residues CHP	Effluent	Biochar	Comb. gas	Solid residues CHP
			3	3	3	3	3	3	2	2	1	3	2	1	1
	Zn	mg/d	7.36E+04	1.27E+04	6.09E+04	5.24E+04	2.11E+04	5.24E+04	1.55E+04	2.65E+03	2.99E+03	5.24E+04	1.37E+04	3.52E+03	3.97E+03

D: Mass flow for HOCs and HMs in output flows in C4 (low and high temperature)

Mass flow for HOCs and HMs in output flows in C4 (low and high temperature)											
				C4 (low temp)				C4 (High temp)			
Component		Unit	Raw sewage sludge	Effluent	Biochar	Comb. gas	Solid residues CHP	Effluent	Biochar	Comb. gas	Solid residues CHP
Biomass, biochar, solid residues (dry basis)→		kg-dw/d	250	12.5	117.64	1598.9	32.63	12.5	113.16	1603.32	32.72
OPFRs	TMP	mg/d	0	0	0	0	0	0	0	0	0
	TEP	mg/d	7.22	0.39	0	0	0	0.39	0	0	0
	TnPP	mg/d	0	0	0	0	0	0	0	0	0
	TnBP	mg/d	9.46	0.52	0.22	0	0	0.52	0	0	0
	TiBP	mg/d	5.37	0.32	0	0	0	0.32	0	0	0
	TCEP	mg/d	0	0	0	0	0	0	0	0	0
	TCiPP	mg/d	206	16.26	0	0	0	16.26	0	0	0
	TPhP	mg/d	26.08	1.32	0	0	0	1.32	0	0	0

Mass flow for HOCs and HMs in output flows in C4 (low and high temperature)											
				C4 (low temp)				C4 (High temp)			
Component		Unit	Raw sewage sludge	Effluent	Biochar	Comb. gas	Solid residues CHP	Effluent	Biochar	Comb. gas	Solid residues CHP
	DPMP	mg/d	21.65	1.18	0	0	0	1.18	0	0	0
	BBOEHEP	mg/d	7.08	0.39	0	0	0	0.39	0	0	0
	TMPP	mg/d	14.26	0.78	0	0	0	0.78	0	0	0
	EHDP	mg/d	0	0	0	0	0	0	0	0	0
	IDPhP	mg/d	16.78	0.92	0	0	0	0.92	0	0	0
	TBOEP	mg/d	80.39	4.42	3.77	0	0	4.42	0	0	0
	3OH-TBOEP	mg/d	0.63	0.03	0	0	0	0.03	0	0	0
	TDCIPP	mg/d	42.77	2.42	0	0	0	2.42	0	0	0
	TEHP	mg/d	53.87	2.69	0	0	0	2.69	0	0	0
	TTBPP	mg/d	2.93	0.16	0	0	0	0.16	0	0	0
	RDP	mg/d	5.33	0.29	0	0	0	0.29	0	0	0
	V6	mg/d	0.32	0.02	0	0	0	0.02	0	0	0
	BPA-BDPP	mg/d	5.67	0.31	0	0	0	0.31	0	0	0
PFAS (Uncategorized)	Gen-X	mg/d	0	0	0	0	0	0	0	0	0
	SAmPAP Di	mg/d	0.89	0.05	0	0	0	0.05	0	0	0
	F53B	mg/d	0.04	0	0	0	0	0	0	0	0
	NaDONA	mg/d	0	0	0	0	0	0	0	0	0
	DecaS	mg/d	0	0	0	0	0	0	0	0	0

Mass flow for HOCs and HMs in output flows in C4 (low and high temperature)											
				C4 (low temp)				C4 (High temp)			
Component		Unit	Raw sewage sludge	Effluent	Biochar	Comb. gas	Solid residues CHP	Effluent	Biochar	Comb. gas	Solid residues CHP
PFAS-FTS	4:2 FTS	mg/d	1.4	0.09	0	0	0	0.09	0	0	0
	6:2 FTS	mg/d	0.08	0	0	0	0	0	0	0	0
	8:2 FTS	mg/d	0.02	0	0	0	0	0	0	0	0
	10:2 FTS	mg/d	0.14	0.01	0	0	0	0.01	0	0	0
PFAS-PFCA	PFBA	mg/d	0.39	0.07	0	0	0	0.07	0	0	0
	PFPeA	mg/d	0.54	0.03	0	0	0	0.03	0	0	0
	PFHxA	mg/d	4.48	0.26	0	0	0	0.26	0	0	0
	PFHpA	mg/d	0.06	0	0	0	0	0	0	0	0
	PFOA	mg/d	41.73	2.44	0	0	0	2.44	0	0	0
	PFNA	mg/d	0	0	0	0	0	0	0	0	0
	PFDA	mg/d	0.04	0	0	0	0	0	0	0	0
	PFUnDA	mg/d	0.07	0	0	0	0	0	0	0	0
	PFDoDA	mg/d	0	0	0	0	0	0	0	0	0
	PFTrDA	mg/d	42.66	2.14	0	0	0	2.14	0	0	0
	PFTeDA	mg/d	11.43	0.63	0	0	0	0.63	0	0	0
	PFHxDA	mg/d	7.86	0.39	0	0	0	0.39	0	0	0
	PFOcDA	mg/d	0.02	0	0	0	0	0	0	0	0
	7H-PFHpA	mg/d	0.76	0.04	0	0	0	0.04	0	0	0

Mass flow for HOCs and HMs in output flows in C4 (low and high temperature)											
				C4 (low temp)				C4 (High temp)			
Component		Unit	Raw sewage sludge	Effluent	Biochar	Comb. gas	Solid residues CHP	Effluent	Biochar	Comb. gas	Solid residues CHP
	PF-3,7-DMOA	mg/d	0.14	0.01	0	0	0	0.01	0	0	0
PFAs-PFSA	PFBS	mg/d	0.04	0	0	0	0	0	0	0	0
	PFPeS	mg/d	0.69	0.04	0	0	0	0.04	0	0	0
	PFHxS	mg/d	0.03	0	0	0	0	0	0	0	0
	PFHpS	mg/d	8.82	0.52	0	0	0	0.52	0	0	0
	PFOS	mg/d	1.72	0.09	0.01	0	0	0.09	0.01	0	0
	PFNS	mg/d	0.1	0	0	0	0	0	0	0	0
	PFDS	mg/d	0	0	0	0	0	0	0	0	0
	PFDoDS	mg/d	1.05	0.05	0	0	0	0.05	0	0	0
	PFECHS	mg/d	0	0	0	0	0	0	0	0	0
PFAs-PreFOS	PFOSA	mg/d	0	0	0	0	0	0	0	0	0
	MeFOSA	mg/d	0.5	0.03	0	0	0	0.03	0	0	0
	EtFOSA	mg/d	9.41	0.5	0	0	0	0.5	0	0	0
	MeFOSE	mg/d	0	0	0	0	0	0	0	0	0
	EtFOSE	mg/d	0	0	0	0	0	0	0	0	0
	FOSAA	mg/d	4.44	0.22	0	0	0	0.22	0	0	0
	MeFOSAA	mg/d	0.55	0.03	0	0	0	0.03	0	0	0
	EtFOSAA	mg/d	4.88	0.27	0	0	0	0.27	0	0	0

Mass flow for HOCs and HMs in output flows in C4 (low and high temperature)											
				C4 (low temp)				C4 (High temp)			
Component		Unit	Raw sewage sludge	Effluent	Biochar	Comb. gas	Solid residues CHP	Effluent	Biochar	Comb. gas	Solid residues CHP
HMs	As	mg/d	4.88E+02	2.44E+01	3.26E+02	5.52E+01	8.28E+01	2.44E+01	3.52E+02	4.46E+01	6.69E+01
	Ba	mg/d	2.90E+04	1.45E+03	2.00E+04	4.12E+03	3.41E+03	1.45E+03	2.25E+04	2.74E+03	2.27E+03
	Cd	mg/d	1.31E+02	6.55E+00	1.84E+01	1.06E+02	0.00E+00	6.55E+00	3.13E+00	1.21E+02	0.00E+00
	Co	mg/d	7.92E+02	3.96E+01	5.79E+02	9.45E+01	7.82E+01	3.96E+01	6.33E+02	6.50E+01	5.38E+01
	Cr	mg/d	6.39E+03	3.20E+02	4.82E+03	5.37E+02	7.12E+02	3.20E+02	5.04E+03	4.45E+02	5.89E+02
	Cu	mg/d	3.44E+04	1.72E+03	2.24E+04	5.93E+03	4.30E+03	1.72E+03	2.39E+04	5.08E+03	3.68E+03
	Mo	mg/d	1.19E+03	5.97E+01	8.39E+02	1.62E+02	1.34E+02	5.97E+01	9.60E+02	9.54E+01	7.90E+01
	Ni	mg/d	4.31E+03	2.15E+02	3.05E+03	5.24E+02	5.24E+02	2.15E+02	3.47E+03	3.11E+02	3.11E+02
	Pb	mg/d	3.46E+03	1.73E+02	2.51E+03	3.47E+02	4.24E+02	1.73E+02	2.01E+03	5.76E+02	7.04E+02
	Sr	mg/d	2.16E+04	1.08E+03	1.44E+04	3.32E+03	2.75E+03	1.08E+03	1.61E+04	2.38E+03	1.97E+03
	V	mg/d	4.14E+03	2.07E+02	3.27E+03	3.66E+02	3.03E+02	2.07E+02	3.56E+03	2.08E+02	1.72E+02
	Zn	mg/d	7.36E+04	3.68E+03	5.12E+04	8.77E+03	9.89E+03	3.68E+03	4.52E+04	1.16E+04	1.31E+04

E: RE (%) of OPFRs for different cases

Analyte		Raw sewage	C_1 (no AD)		C_2 (AD)		C_3 (AD+PYR) (low temp)		C_3 (AD+PYR) (high temp)		C_4 (NoAD+PYR) (low temp)		C_4 (No AD+PYR) (high temp)	
			Biosolid	RE (%)	Biosolid	RE (%)	Biochar	RE (%)	Biochar	RE (%)	Biochar	RE (%)	Biochar	RE (%)
OPFRs	TMP	0	0	0	0	0	0	0	0	0	0	0	0	0
	TEP	7.22	2.05	68.03	6.82	2.88	0	25.42	0	24.77	0	47.06	0	45.26

Analyte		Raw sewage	C_1 (no AD)		C_2 (AD)		C_3 (AD+PYR) (low temp)		C_3 (AD+PYR) (high temp)		C_4 (NoAD+PYR) (low temp)		C_4 (No AD+PYR) (high temp)	
			Biosolid	RE (%)	Biosolid	RE (%)	Biochar	RE (%)	Biochar	RE (%)	Biochar	RE (%)	Biochar	RE (%)
	TnPP	0	0	0	0	0	0	0	0	0	0	0	0	0
	TnBP	9.46	8.04	14.26	2.72	37.07	0.07	25.23	0	24.77	0.22	45.96	0	45.26
	TiBP	5.37	4.55	14.51	1.67	35.85	0	25.42	0	24.77	0	47.06	0	45.26
	TCEP	0	0	0	0	0	0	0	0	0	0	0	0	0
	TCiPP	206	37.95	77.50	61.23	36.56	0	25.42	0	24.77	0	47.06	0	45.26
	TPhP	26.08	0	95	2.87	46.30	0	25.42	0	24.77	0	47.06	0	45.26
	DPMP	21.65	0	95	3.56	43.47	0	25.42	0	24.77	0	47.06	0	45.26
	BBOEHEP	7.08	1	81.58	3.09	29.32	0	25.42	0	24.77	0	47.06	0	45.26
	TMPP	14.26	0	95	6.13	29.66	0	25.42	0	24.77	0	47.06	0	45.26
	EHDP	0	0	0	0	0	0	25.42	0	0	0	0	0	0
	IDPhP	16.78	3.72	73.94	10.06	20.83	0	0	0	24.77	0	47.06	0	45.26
	TBOEP	80.39	11.39	81.54	36.44	28.44	1.81	25.42	0	24.77	3.77	44.85	0	45.26
	3OH-TBOEP	0.63	0.09	81.43	0.6	2.48	0	24.84	0	24.77	0	47.06	0	45.26
	TDCIPP	42.77	0	95	16.45	32.01	0	25.42	0	24.77	0	47.06	0	45.26
	TEHP	53.87	46.06	13.77	26.61	26.33	0	25.42	0	24.77	0	47.06	0	45.26
	TTBPP	2.93	0.28	85.92	2.77	2.84	0	25.42	0	24.77	0	47.06	0	45.26
	RDP	5.33	0	95	0.49	47.24	0	25.42	0	24.77	0	47.06	0	45.26
	V6	0.32	0	95	0.08	39.02	0	25.42	0	24.77	0	47.06	0	45.26
	BPA-BD	5.67	0	95	2.28	31.10	0	25.42	0	24.77	0	47.06	0	45.26

Analyte		Raw sewage	C_1 (no AD)		C_2 (AD)		C_3 (AD+PYR) (low temp)		C_3 (AD+PYR) (high temp)		C_4 (NoAD+PYR) (low temp)		C_4 (No AD+PYR) (high temp)	
			Biosolid	RE (%)	Biosolid	RE (%)	Biochar	RE (%)	Biochar	RE (%)	Biochar	RE (%)	Biochar	RE (%)
	PP													

F: RE (%) of PFASs for different cases

Analyte		Raw sewage	C_1 (no AD)		C_2 (AD)		C_3 (AD+PYR) (low temp)		C_3 (AD+PYR) (high temp)		C_4 (NoAD+PYR) (low temp)		C_4 (No AD+PYR) (high temp)	
			Biosolid	RE (%)	Biosolid	RE (%)	Biochar	RE (%)	Biochar	RE (%)	Biochar	RE (%)	Biochar	RE (%)
PFAS (Uncategorized)	Gen-X	0	0	0	0	0	0	0	0	0	0	0	0	0
	SAmPAP Di	0.89	0.85	4.27	0.85	2.34	0	25.42	0	24.77	0	47.06	0	45.26
	F53B	0.04	0.04	0	0.04	0	0	25.42	0	24.77	0	47.06	0	45.26
	NaDONA	0	0	0	0	0	0	0	0	0	0	0	0	0
	DecaS	0	0	0	205.52	0	0	0	0	0	0	0	0	0
PFAS-FTS	4:2 FTS	1.4	1.31	6.11	0.08	49.05	0	25.42	0	24.77	0	47.06	0	45.26
	6:2 FTS	0.08	0.07	11.88	0.03	32.52	0	25.42	0	24.77	0	47.06	0	45.26
	8:2 FTS	0.02	0.02	0	0.02	0	0	25.42	0	24.77	0	47.06	0	45.26
	10:2 FTS	0.14	0.13	6.79	0.24	-37.16	0	25.42	0	24.77	0	47.06	0	45.26
PFAS-PFCA	PFBA	0.39	0.32	17.05	0.29	13.34	0	25.42	0	24.77	0	47.06	0	45.26
	PFPeA	0.54	0.51	5.28	0.51	2.89	0	25.42	0	24.77	0	47.06	0	45.26
	PFHxA	4.48	4.22	5.51	3.49	11.50	0	25.42	0	24.77	0	47.06	0	45.26
	PFHpA	0.06	0.06	0	0.06	0	0	25.42	0	24.77	0	47.06	0	45.26

Analyte		Raw sewage	C_1 (no AD)		C_2 (AD)		C_3 (AD+PYR) (low temp)		C_3 (AD+PYR) (high temp)		C_4 (NoAD+PYR) (low temp)		C_4 (No AD+PYR) (high temp)	
			Biosolid	RE (%)	Biosolid	RE (%)	Biochar	RE (%)	Biochar	RE (%)	Biochar	RE (%)	Biochar	RE (%)
	PFOA	41.73	39.28	5.58	50.13	-10.47	0	25.42	0	24.77	0	47.06	0	45.26
	PFNA	0	0	0	100.97	0	0	0	0	0	0	0	0	0
	PFDA	0.04	0.04	0	0.04	0	0	25.42	0	24.77	0	47.06	0	45.26
	PFUnDA	0.07	0.07	0	26.82	-19880.6	0	25.42	0	24.77	0	47.06	0	45.26
	PFDoDA	0	0	0	0	0	0	0	0	0	0	0	0	0
	PFTrDA	42.66	40.52	4.77	0	52.02	0	25.42	0	24.77	0	47.06	0	45.26
	PFTeDA	11.43	10.79	5.32	2.66	39.92	0	25.42	0	24.77	0	47.06	0	45.26
	PFHxDA	7.86	7.47	4.71	7.49	2.45	0	25.42	0	24.77	0	47.06	0	45.26
	PFOcDA	0.02	0.02	0	0.02	0	0	25.42	0	24.77	0	47.06	0	45.26
	7H-PFHpA	0.76	0.72	5	0.12	43.81	0	25.42	0	24.77	0	47.06	0	45.26
	PF-3,7-DMOA	0.14	0.14	0	0	52.02	0	25.42	0	24.77	0	47.06	0	45.26
PFAs-PFSA	PFBS	0.04	0.03	23.75	0.03	13.01	0	25.42	0	24.77	0	47.06	0	45.26
	PFPeS	0.69	0.65	5.51	0.64	3.77	0	25.42	0	24.77	0	47.06	0	45.26
	PFHxS	0.03	0.03	0	0.03	0	0	25.42	0	24.77	0	47.06	0	45.26
	PFHpS	8.82	8.3	5.60	0.98	46.24	0	25.42	0	24.77	0	47.06	0	45.26
	PFOS	1.72	1.63	4.97	1.69	0.91	0.01	25.27	0	24.77	0.01	46.78	0.01	45.00
	PFNS	0.1	0.09	9.5	0.09	5.20	0	25.42	0	24.77	0	47.06	0	45.26
	PFDS	0	0	0	0	0	0	0	0	0	0	0	0	0
	PFDoDS	1.05	0.99	5.43	1	2.48	0	25.42	0	24.77	0	47.06	0	45.26

Analyte		Raw sewage	C_1 (no AD)		C_2 (AD)		C_3 (AD+PYR) (low temp)		C_3 (AD+PYR) (high temp)		C_4 (NoAD+PYR) (low temp)		C_4 (No AD+PYR) (high temp)	
			Biosolid	RE (%)	Biosolid	RE (%)	Biochar	RE (%)	Biochar	RE (%)	Biochar	RE (%)	Biochar	RE (%)
	PFECHS	0	0	0	0	0	0	0	0	0	0	0	0	0
PFAs-PreFOS	PFOSA	0	0	0	0	0	0	0	0	0	0	0	0	0
	MeFOSA	0.5	0.47	5.7	0	52.02	0	25.42	0	24.77	0	47.06	0	45.26
	EtFOSA	9.41	8.92	4.95	0	52.02	0	25.42	0	24.77	0	47.06	0	45.26
	MeFOSE	0	0	0	0	0	0	0	0	0	0	0	0	0
	EtFOSE	0	0	0	0	0	0	0	0	0	0	0	0	0
	FOSAA	4.44	4.21	4.92	0.29	48.63	0	25.42	0	24.77	0	47.06	0	45.26
	MeFOSAA	0.55	0.52	5.18	0	52.02	0	25.42	0	24.77	0	47.06	0	45.26
	EtFOSAA	4.88	4.61	5.26	0	52.02	0	25.42	0.22	23.65	0	47.06	0	45.26