



Sewage sludge and the circular economy

Version: 7

Date: 17/05/2021

EEA activity: 1.5.2

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Acknowledgements

Acknowledgements and thanks go to the following organisations who contributed to this report:

Estonian Environment Agency, Estonia
Ministry of the Environment of Estonia, Estonia
POPs Environmental Consulting, Germany
Italian National Institute of Health, Italy
Higher Institute for Environmental Protection and Research (ISPRA), Italy
Swedish Environmental Protection Agency, Sweden
Swedish Agency for Marine and Water Management, Sweden
European Commission, DG Environment
Aqua Publica Europea
EurEau
Joint Research Centre
European Commission, DG Environment

Furthermore, we would like to thank the representatives of Eurostat who assisted with queries on interpretation of relevant datasets.

This report

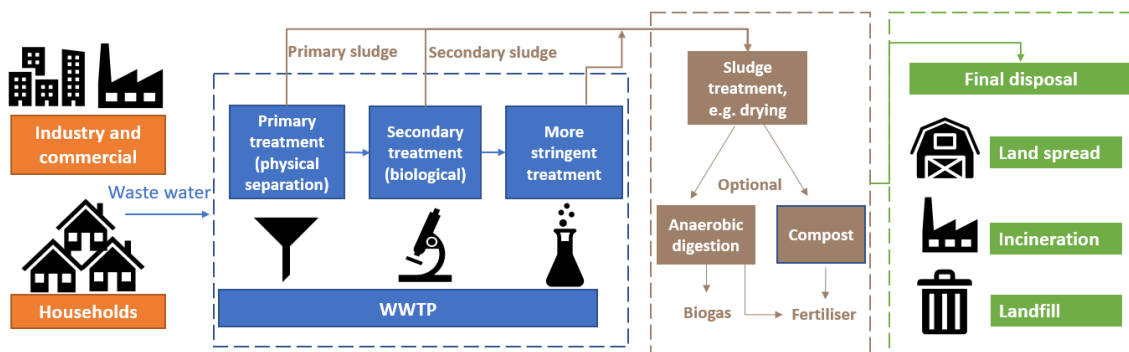
This is the final report for the project “Provision of services in the area of sewage sludge and the circular economy”, delivered under the contract No 3415/B2020.EEA.58102. The opinions expressed are those of the Contractor, reflecting verbal and written feedback from the European Environment Agency (EEA) and relevant stakeholders, and do not represent the Agency’s official position.



Executive summary

Sewage sludge is a material which results from the waste water treatment process and is a sink for contaminants. While recent European policy has focused on high standards in waste water treatment, less focus has been placed on the management and disposal of sewage sludge, and the recovery from it of valuable energy and key nutrients (such as Nitrogen (N) and Phosphorus (P)). These elements are of great interest as Europe works towards a more circular economy under the Green Deal¹ while also pursuing a zero-pollution ambition. Figure 0-1 shows a schematic of the sewage sludge life cycle, from its generation in waste water treatment plants, through treatment, to end-of-life management.

Figure 0-1 The sewage sludge life cycle



Source: own elaboration

The aim of this study is to provide an understanding of how sewage sludge is currently managed in Europe, and how it could be better managed in a circular economy. This has been explored by four country-specific case studies (Estonia, Germany, Italy and Sweden) and two case studies on reducing contamination of the sludge upstream (for Di(2-ethylhexyl)-phthalate (DEHP) and Benzo(a)pyrene (BaP)).

Key findings from the four country case studies

- The two predominant approaches to final sewage sludge disposal are agricultural use and incineration. All countries have set limits for metals in sludge used in agriculture and some have regulated additional pollutants. Limits set are generally much lower than those in the Sewage Sludge Directive (SSD).
- Public concern about health and environmental contamination is driving a shift away from agricultural uses, towards alternative sewage sludge management approaches that ensure nutrient recovery while controlling contaminants, such as P recovery from incineration ash.
- Currently, sewage sludge effectively acts as a sink for persistent contaminants removed from waste water, with the level of contamination being dependent on that in the influent to the treatment works. A balance needs to be found to manage contamination with the principles of a circular economy in recovering valuable nutrients.
- Some research exploring contaminants of emerging concern in sludge has uncovered potential risks to soil quality and human health. Analysis of sludge composition, the behaviour of those substances in sludge and a full assessment of sludge in an agricultural setting should be explored further. However, an enquiry conducted in Sweden concluded that currently, there is a lack of evidence for a total ban on sewage sludge use in agriculture.
- Upstream approaches preventing the contamination of the sludge, to enable its greater use as a resource, are not common.

¹ https://ec.europa.eu/info/strategy/priorities-2019-2024/european-green-deal_en



Current management of sewage sludge in Europe and potential value recovery

Analysis of the data available, and our calculations² show that for the EU-27:

- Around 10.4 million t of sewage sludge is produced annually, of which 94% is ‘disposed’.
- There is potential to recover up to an additional³ 3 500 GWh of energy from sewage sludge if the sludge currently landfilled and composted is instead anaerobically digested. This represents up to 14 % of the total waste water sector energy needs in the EU-27 in 2018.
- There is potential to recover up to 69 300 t of P and up to 96 300 t of N via landspreading. This represents 6.3 % of the amount of P fertiliser⁴ and 0.9 % of the amount of N fertiliser used in the EU in 2018. In a mono-incineration scenario, the P recovery potential rises to up to 105 500 t of P (10 % of P fertiliser use) at the expense of N recovery.

Study of specific pollutants and options for their reduction in sewage sludge

In a circular economy, prevention of contamination of sewage by persistent, toxic chemicals would allow recycling of sewage sludge on land without concerns that this might lead to diffuse pollution of soils, plants and water. This study looked into what kinds of effort this might entail, by considering two pollutants, one a “point source” in products, the other a diffuse pollutant.

DEHP: the following upstream options were identified:

- **Option 1** – Substituting DEHP in PVC with alternative chemicals
- **Option 2** – Substituting PVC with alternatives

While Option 1 would be feasible and is not associated with significant additional costs, there would be some risks associated with the increase of releases of DEHP substitutes. Option 2 is associated with positive environmental and human health impacts, but while it proved feasible and cost-effective for specific products considered (PVC flooring), full life cycle analysis would be needed for each product and potential replacement.

BaP: the following upstream options were identified:

- **Option 1** – Further reduce common sources of BaP emissions to air
- **Option 2** – Remove BaP from run-off water before it enters UWWTP using sustainable urban drainage systems

The analysis shows similar results for both options. Both would lead to significant environmental benefits, with Option 1 leading to benefits for air, soil and human exposure, and Option 2 to specific benefits for waste water. However, both options would also face challenges to feasibility. For both substances, a balanced combination of upstream and downstream options could be designed and implemented to maximise benefits and mitigate the risks.

Future considerations

Based on the experience of this study, the following suggestions are made for future work:

- **Improvements to the data on sludge management in Europe:** In particular, greater granularity in reporting of treatment and disposal routes.
- **Further elaboration of the energy and nutrient recovery potential estimates and assessment of the recovery of organic content:** Needs more detailed consultation with countries on existing practices and investment needs.
- **Application of the “upstream” methodology to other pollutants:** Including expansion to potential modifications to treatment at UWWTP and treatment of sludge, to ensure that optimal solutions can be identified.

² Subject to data limitations and assumptions detailed in relevant report sections.

³ Additional to that assumed already recovered.

⁴ The actual value to agriculture of any P recovery amount is dependent on the P-availability of the soil to which it is applied. This is discussed in more detail in Section 3.4.



List of abbreviations

Abbreviation	Name
AD	Anaerobic Digestion
ANH	Absolute Non-Hazardous
As	Arsenic
AT	Austria
ATBC	Acetyl Tributyl Citrate
BaP	Benzo(A)Pyrene
BAT	Best Available Techniques
BBP	Benzyl Butyl Phthalate
BE	Belgium
BG	Bulgaria
Cd	Cadmium
CEAP	Circular Economy Action Plan
CH	Switzerland
CO ₂	Carbon Dioxide
Cr	Chromium
Cu	Copper
CY	Cyprus
CZ	Czechia
DBP	Dibutyl Phthalate
DE	Germany
DEHP	Di(2-Ethylhexyl) Phthalate
DIBP	Diisobutyl Phthalate
DINCH	1,2-Cyclohexane Dicarboxylic Acid Diisononyl Ester
DK	Denmark
EC	European Commission
ECHA	European Chemicals Agency
EE	Estonia
EEA	European Environment Agency
EEA-32	European Economic Area Member Countries
EEE	Electrical and Electronic Equipment
EL	Greece
EQSD	Environmental Quality Standards Directive
ES	Spain
ESBO	Epoxidised Soybean Oil
EU	European Union
EU-27	European Union Member States
F	Fluorine
FCM	Food Contact Material
FI	Finland
FPR	Fertilising Products Regulation
FR	France
g	Gram



Abbreviation	Name
GHG	Greenhouse Gas
GWh	Giga-Watt Hour
HBCDD	Hexabromocyclododecanes
Hg	Mercury
HMW	High-Molecular-Weight
HR	Croatia
HU	Hungary
IE	Ireland
IED	Industrial Emissions Directive
IS	Iceland
ISPRA	Italian National Institute for Environmental Protection and Research
IT	Italy
JRC	Joint Research Centre
K	Potassium
kg	Kilogramme
K _{ow}	Octanol-Water Partitioning Coefficient
kWh	Kilo-Watt Hour
LI	Liechtenstein
LMW	Lower-Molecular-Weight
LRF	Federation of Swedish Farmers
LT	Lithuania
LU	Luxembourg
LV	Latvia
m	Metre
m ³	Cubic Meter
MAP	Magnesium Ammonium Phosphate
MEHP	Mono (2-Ethylhexyl) Phthalate
mg	Milligram
MT	Malta
MWth	Megawatt Thermal
N	Nitrogen
NH ₃	Ammonia
Ni	Nickel
NL	Netherlands
NMVOG	Non-methane volatile organic compounds
NO	Norway
NO _x	Nitrogen oxides
NP	Neptunium
NPE	Nonylphenol
NPP	Non-Phthalate Plasticisers
P	Phosphorus
p.e.	Population Equivalent ⁵
PAE	Phthalate Esters

⁵ 1 p.e. (population equivalent)" means the organic biodegradable load having a five-day biochemical oxygen demand (BOD5) of 60 g of oxygen per day



Abbreviation	Name
PAH	Polycyclic Aromatic Hydrocarbons
Pb	Lead
PCB	Polychlorinated biphenyl
PCDD/F	Polychlorinated dibenzo(p)dioxin and furan
PE	Polyethylene
PL	Poland
PM	Particulate matter
POP	Persistent organic pollutant
PP	Polypropylene
ProgRes	German Resource Efficiency Programme
PS	Perfluorinated Surfactants
PT	Portugal
PVC	Poly Vinyl Chloride
RAC	Committee for Risk Assessment
RAG	Red - Amber - Green
RO	Romania
Sb	Antimony
Se	Selenium
SE	Sweden
SEAC	Committee for Socio-Economic Analysis
SI	Slovenia
SK	Slovakia
SO ₂	Sulphur dioxide
SSD	Sewage Sludge Directive
SSWA	Swedish Water and Waste Water Association
SUDS	Sustainable urban drainage systems
SVHC	Substances of Very High Concern
t	Tonne
TOTM	Trioctyl Trimellitate
TR	Turkey
TWh	Tera-Watt Hour
UWWTP	Urban Waste Water Treatment Plants
Zn	Zinc



1. Introduction

Urban waste water treatment is essential in safeguarding both the health of people and the quality of the environment in Europe. In recent years, European policy has focused on treating the waste water to a high standard to protect the aquatic environment. Less focus has however been placed on the management, recovery and disposal of sewage sludge - the residual, semi-solid material produced as a result of waste water treatment. Figure 1-1 shows the schematic of the sewage sludge life cycle, from its generation in waste water treatment plants, through treatment to end-of-life management. Waste water treatment plants have three key stages – primary treatment, focusing on separating the solids from the rest of the waste water; secondary treatment, focusing on removing organic matter using biological processes and advanced treatment, focusing on the removal of nitrogen, phosphorus and others using biological and/or chemical processes.

1.1 Sewage sludge overview

As shown, there are two key types of sewage sludge arising from the waste water treatment process:

- Primary sludge – i.e. settleable solids separated in the primary treatment (physical separation such as screening) of waste water
- Secondary sludge – i.e. the sludge from the secondary, biological treatment (i.e. activated sludge treatment process, which is a secondary treatment process which removes nutrients through a biological process).

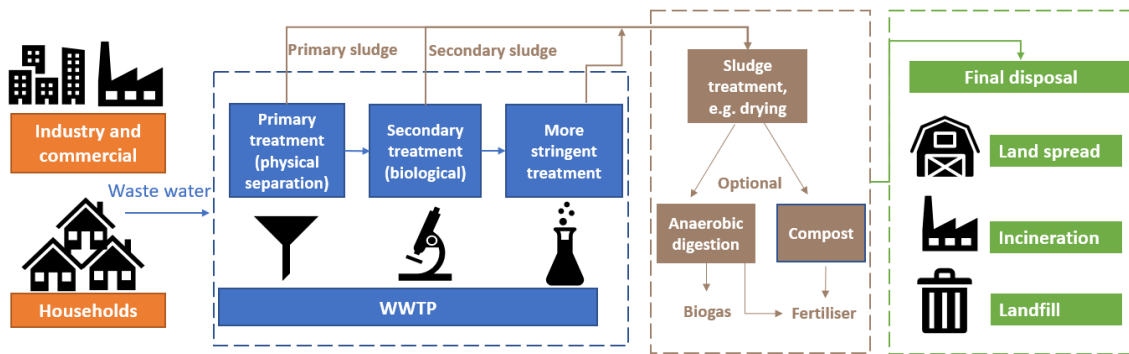
Sewage sludge is characterised by a high carbon and nutrient content, but also high water content, potential content of hazardous materials and unpleasant odour. Sludges from treatment of urban waste water are categorised as absolute non-hazardous (ANH) waste⁶ in the European List of Waste (waste code 19 08 05) (EC, 2018a).

Following its extraction from the waste water treatment process, sewage sludge requires treatment to enable more efficient and safer transport and ultimate recovery and/or disposal. Depending on the size of the plant, the primary and secondary sludges are either treated separately or mixed. Common treatment options involve thickening, stabilisation, dehydration and sometimes drying of sludge. Additional and well-established management techniques for sewage sludge involve lime treatment, anaerobic digestion (AD), composting with other organic waste, as well as in a few full-scale plants. Final recovery and disposal include spreading of treated sludge on land, incineration, and landfilling. There are already plants in Europe recovering nutrients from ash following sewage sludge incineration, particularly phosphorus and in rare cases nitrogen. Disposal and/or recovery, in particular in agriculture, generally occur in the regional surrounding area of the plant to minimise transport cost. Summaries of common sewage sludge treatment methods and nutrient and energy recovery techniques are presented in Annexes A and B to this report.

⁶ A waste classified as non-hazardous, without any further assessment of its hazardous properties. The only exception to this principle is described in Article 7 (2) of the Waste Framework Directive, whereby if the competent authority of the MS concerned considers that, based on adequate evidence, a given waste to which a ANH code is attributed, is in reality to be classified as hazardous, the waste in question will be classified as hazardous.



Figure 1-1 The sewage sludge life cycle



1.2 Relevance of sewage sludge in circular economy

Due to its high nutrient and organic matter content, and the energy content of dried sludge comparable to that of woody biomass, sewage sludge is a potential secondary resource which can contribute to Europe's transition to a circular economy. The driving principle of the circular economy is to retain materials in the economy at their maximum value for as long as possible (EC, 2020a). The principle is underpinned by the EU waste hierarchy⁷, which ranks waste management options according to their sustainability and gives top priority to preventing and recycling of waste, followed by recovery. In the context of sludge, circular economy concerns facilitating nutrient recycling and making use of secondary raw materials while ensuring environmental and health protection, followed by recovering energy (EC, 2017).

Applied as a fertiliser, sludge can supply carbon to increase the organic matter in the soil by carbon sequestration and part of the nitrogen (N), phosphorus (P) and other nutrients that most crops require. N, P and potassium (K) are the most widely used fertilisers in the EU and worldwide (EC, 2019c). On the EU level, N represents more than two-thirds of the total use of the three minerals, whereas P and K are applied in lower quantities and represent less than 20 % each of the overall use in volume (EC, 2019c). This pattern in fertilisers use is driven by plants' higher demand for N compared to P and K. Out of the three nutrients, P and K fertilisers are derived from natural mineral sources, while N fertilisers are manufactured from atmospheric N using fossil gas. Feedstocks for production of P fertilisers in Europe are significantly limited. This combined with the classification of P and Phosphate rock as critical raw materials⁸, make P recovery a strategic priority.

Sludge-derived biogas is a source of renewable energy for waste water treatment and associated processes. Excess energy can be exported off-site as electricity, heat or biogas for injection into the gas network or distributed for vehicle gas, depending on the local conditions.

The challenges to a greater use of sludge as a resource are technical constraints (e.g. its high moisture content which increases energy consumption and costs of preparation for recovery) and potential content of substances which may inhibit onward recycling, such as organic pollutants, metals, pathogens, and pollutants of emerging concern such as pharmaceuticals and microplastics. These substances, found in urban waste water, may be removed from the waste water during treatment and concentrated in the sewage sludge. Some of these, when applied to land via sewage sludge, would break down to non-harmful substances, but others (such as the metals) do not degrade easily in the environment and are what is classed as persistent toxic substances. The application to land of potentially contaminated sludge is particularly

⁷ Waste Framework Directive 2008/98/EC, <https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32008L0098>

⁸ https://ec.europa.eu/growth/sectors/raw-materials/specific-interest/critical_en



problematic if the pollutants present in sewage sludge accumulate over time to toxic concentrations.

Concerns over human health and environmental impacts resulting from the application of sewage sludge on land resulted in adoption of the EU Sewage Sludge Directive (SSD) in 1986. The SSD places a series of requirements on the application of sewage sludge to land, relating to concentration levels of metals in sludge and the receiving soil, quantity of sludge utilised and the scenarios in which its use is prohibited. Article 12 of the SSD allows MS to introduce stricter requirements if necessary. The SSD has not been revised since its introduction in 1986, however many EU Member States have set stricter limits in the intervening years. Recognising the SSD may no longer match current needs, it is currently subject to evaluation by the European Commission (EC). The evaluation is set in the context of the European Green Deal and the latest Circular Economy Action Plan (CEAP) (EC, 2020a) as well as the Commission's zero pollution ambition (with an action plan due to be published in 2021). These policy initiatives aim to accelerate the transition to circular economy through increasing the efficient use of resources, restoring biodiversity and safeguarding human health and the environment from pollution.

1.3 Aims and scope of this study

Given the above context, the aim of this study is to provide policy makers and the wider EEA audience with a better understanding of how sewage sludge is currently managed and how it can be treated, with respect to environmental protection, within a circular economy. This is achieved through:

- **Providing an overview of the current sewage sludge management practices:** This focuses on examining the current legislation, sewage sludge quality and its management in Europe, as well as potential for nutrient and energy recovery from sludge. This is supported by four country-specific case studies.
- **Identifying opportunities and challenges for applying circular economy thinking in sewage sludge management by reducing contamination of sewage sludge upstream:** These aspects are discussed in two case studies for specific substances showing how contamination can be avoided at source and discussing the costs and benefits of such options.

The focus of this study is sewage sludge from urban waste water treatment plants (UWWTP), and not sludge produced in waste water treatment plants located e.g. on industrial sites. Its outputs are likely to contribute to a future report by the EEA on the Assessment of Urban Waste Water.



2. Methodology

2.1 Overview

The study methodology was divided into two distinct phases, corresponding to the two aims as detailed in Section 1.3 above. The first phase was to describe the current situation of sewage sludge management in Europe, incorporating analysis of energy and nutrient recovery potential, and presenting different approaches to sewage sludge management using four country case studies. The second phase involved a review of potential opportunities to remove sludge contaminants ‘upstream’ i.e. by preventing them from entering the waste water stream and consequently avoiding contamination of sludge, thus enabling its use as a resource.

The bulk of the evidence used and discussed in this study is sourced from the literature list detailed in Section 7. Gaps in insights from available literature have been filled via targeted consultation with relevant stakeholders, both in specific countries and trade associations.

2.2 Calculation of nutrient and energy recovery potential

This section sets out the methodology used to calculate the potential for energy and nutrient recovery in EU-27 and EEA-32. Flow charts describing the calculation methodology are provided in Annex D.

Energy Recovery

To estimate the amount of unused energy recovery potential, we considered the production of biogas during anaerobic digestion (AD) of sewage sludge and the generation of energy from sewage sludge through incineration.

For the potential additional energy recovery using AD, we used data on the total amount of sewage sludge currently landfilled and composted from Eurostat as a starting point, assuming that all sewage sludge currently landfilled would first go through AD, that all sewage sludge composted can be pre-treated via AD and that a share of the sewage sludge currently composted is already pre-treated via AD.

For the potential additional energy recovery from incineration, we used data on the total amount of sewage sludge currently landfilled, used in agriculture and composted from Eurostat as a starting point, assuming that all sewage sludge currently landfilled and half of the sewage sludge currently used in agriculture and composted would be incinerated instead.

We also estimated the amount of sewage sludge with energy recovery potential if the Urban Waste Water Treatment Directive (UWWTD) would be fully implemented, using the production of sewage sludge per population equivalent (p.e.) connected to secondary and more stringent treatment, and assuming the same proportions in disposal routes as presently.

The energy consumption and production figures used in our calculations are estimated based on findings from a literature review.

This methodology is discussed in more detail in the later relevant section 3.3.



Nutrient recovery

The nutrient recovery potential has been estimated for P (P_2O_5) and N. Our calculations for total nutrient recovery potential are based on the assumption that nutrients would be recovered from the following amounts of sewage sludge:

- Total amount of sewage sludge currently landfilled,
- 50 % of sewage sludge currently disposed using “other” methods,
- 50 % of sewage sludge currently incinerated and
- 50 % of potential additional amount of sewage sludge that could be incinerated (i.e. that currently recorded as going to landfill)

Based on the same assumptions used for the estimations of the additional energy recovery potential, we also estimated the amount of sewage sludge with nutrient recovery potential if the UWWTD would be fully implemented.

Values of total P and N content of sludge generated in Europe were sourced from literature. We assumed that 100 % of the nutrient contained in the sewage sludge and sewage sludge ash would be recovered through landspreading and plant available. It is noteworthy that simultaneous recovery of P and N is possible, either through application of sludge to land⁹ or through chemical techniques such as precipitation (Lorick & Biljana Macura, 2020). A recovery rate of 90 % for P was used to estimate the potential nutrient recovery from sewage sludge ash using nutrient recovery technologies (The German Environment Agency, 2018). For N, no recovery from sewage sludge ash was calculated as N is removed from sewage sludge during the combustion process.

This methodology is discussed in more detail in the later relevant section 3.4.

2.3 Selection of case study countries to illustrate existing sewage sludge management practices and strategies

Four countries were selected for detailed case studies: Estonia, Germany, Italy and Sweden.

The objective of the case studies was to demonstrate differences in sludge management across Europe, what drives the management choices and what the costs and benefits of them are.

For that reason, the selection of the countries for the case studies was governed by the following criteria:

- **Current sludge management**
- **Difference in strategic approaches towards sludge management**
- **Level of centralisation of sludge management governance**
- **Location within Europe**

In conjunction with the EEA, a screening exercise of all EEA member countries was conducted against these criteria to select the most interesting combination of case studies. Table 2-1 shows the resulting countries selected.

Table 2-1 Selected case study countries

Country	Primary sludge management method	Centralisation	Location
Estonia	Compost	Centralised	Eastern
Germany	Incineration	Centralised	Central
Italy	Landfill & agricultural use	Decentralised	Southern
Sweden	Mixed	Centralised	Northern

⁹ Though N recovery could be limited depending on the sludge treatment applied



2.4 Selection of pollutants to illustrate potential for upstream reduction of sewage sludge contamination

The methodology for the selection of contaminant case studies consisted of the following steps:

1. Developing a list of chemicals/pollutants which represent a challenge to the circular management of sewage sludge, especially application on land.
2. Developing assessment criteria to shortlist the options.
3. Applying the selection criteria and shortlisting two key pollutants.

Chemicals/pollutants which represent a challenge to the circular management of sewage sludge

The list of chemicals considered in this task was based on the Water Framework Directive list of priority substances¹⁰ (45 chemicals/groups of chemicals posing significant risk to the aquatic environment and including pesticides, household and industrial chemicals) for which the Environmental Quality Standards Directive (EQSD – 2008/105/EC)¹¹ set the limits in water. The focus on these pollutants was because they have been identified as being of concern for human health and the environment, and waste water treatment. Among them, chemicals more likely to sorb on sludge during the treatment will find their way into sewage sludge. In addition, the list included substances that can pollute drinking waters and are in the “watch list” of chemicals of emerging concern under the EQSD as well as several other substances of increasing concern. The full list, consisting 54 representative micropollutants were considered in the recent modelling by the Joint Research Centre (JRC) to assess the impacts of the UWWTD (Pistocchi, et al., 2019). It is noteworthy that some substances are degraded to harmless or harmful metabolites and that the human exposure of these substances via use of sludge or its derivatives in agriculture is, in general, likely to be a very small part of the possible exposure to humans via other routes. In this study, chemicals were categorised in six classes depending on their behaviours in UWWTP:

1. Chemicals practically bypassing waste water treatment, virtually unaffected (i.e. chemicals that stay in treated water);
2. Slowly removed chemicals from the effluent in activated sludge bioreactor, with limited or no accumulation in sludge;
3. Moderately removed from the effluent in activated sludge bioreactor, with limited or no accumulation in sludge;
4. Removed from the effluent in activated sludge bioreactor, with limited or no accumulation in sludge;
5. Sorbed to sludge and slowly removed from the effluent in activated sludge bioreactor;
6. Sorbed to sludge (but not appreciably removed from the effluent in activated sludge bioreactor).

¹⁰ https://ec.europa.eu/environment/water/water-dangersub/pri_substances.htm

¹¹ <https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32008L0105>



Chemicals classed in categories 5 and 6 provided a good starting point for the selection of case study substances. This is because:

- These substances need to be removed from European waste waters to protect the aquatic environment
- Due to their properties, during waste water treatment these substances sorb on sludge and would therefore be expected to be found in resulting sewage sludge that subsequently needs to be managed¹²

This approach resulted in the following list of substances or groups:

- Metals: Antimony (Sb), Arsenic (As), Cadmium (Cd), Chromium (Cr), Copper (Cu), Lead (Pb), Mercury (Hg), Nickel (Ni), Zinc (Zn), Selenium (Se) (present naturally in the environment, also originate from industrial, agricultural, pharmaceutical, domestic waste waters and atmospheric sources, not degraded and tend to adsorb in sludge, persist in rivers)
- Benzo(a)pyrene (polycyclic aromatic hydrocarbon present in some oils and produced as a result of incomplete combustion, non-polar and persistent but settling in sludge; persistent organic pollutant (POP) which is part of UNECE POPs Protocol¹³)
- Anthracene (polycyclic aromatic hydrocarbon present in some oils, used in chemical production, non-polar and persistent in sludge)
- Fluoranthene (polycyclic aromatic hydrocarbon present in some oils, produced during low temperature combustion, Non-polar and persistent in sludge)
- Naphthalene (used in chemical production present in some oils, non-polar and persistent in sludge)
- Brominated diphenylethers (large family of substances, used in flame retardants, non-polar and persistent but settling in sludge)
- C10-13 Chloroalkanes (used in formulation or re-packaging, at industrial sites and in manufacturing, Non-polar and persistent)
- 4-Nonylphenol (used in manufacturing antioxidants, lubricating oil additives, emulsifiers, and solubilisers, partly removed and sorbed in UWWTP)
- Di(2-ethylhexyl)-phthalate (DEHP)¹⁴ (plasticiser in manufacturing of articles made of PVC, slowly degradable and sorbing to sludge)
- Pentachloro-benzene (persistent organic pollutant, by-product of the manufacture of carbon tetrachloride and benzene, released to the environment from waste burning, non-polar and persistent in sludge)
- Tributyltin compounds (Tributyltin-cation) (biocides, adsorbed to sludge and settling)
- Ciprofloxacin (antibiotic, adsorbed to sludge and settling)
- Sertraline (antidepressant, sorbing to sludge, persistent in rivers)
- Hexabromocyclododecanes (HBCDD) (flame retardant, non-polar and persistent in sludge)
- Triclosan (antibacterial and antifungal agent, mostly adsorbed to sludge but fast-dissipating in rivers)

The above list includes substances from all major pollutant groups identified at the proposal stage. Metals have been excluded from the scope owing to the fact that they have been widely

¹² It is noteworthy that some chemicals are degraded in further processing such as anaerobic digestion.

¹³ <https://www.unece.org/fileadmin/DAM/env/lrtap/full%20text/ece.eb.air.104.e.pdf>

¹⁴ The use of DEHP is restricted in several products under the Registration, Evaluation, Authorisation and Restriction of Chemicals regulation (REACH).



investigated and are already addressed by the SSD and national regulations in the EEA member countries.

Assessment criteria to select two case study substances

The assessment criteria to shortlist the options include:

- Presence of the substance in European sludge
- Extent to which presence of the substance in sludge creates issues for sludge management (particularly in the context of circular economy)
- Persistence, bioaccumulation and toxicity of the substance

It was agreed in discussion with the EEA that one point and one diffuse source pollutant should be captured in the case studies, and that contaminants of emerging concern should not be considered due a potential lack of available information.

Applying these considerations, the following pollutants were selected for case studies:

- DEHP, as a point source occurring in households, due to its presence in sewage sludge and its hazard classification (Repr 1 B - May damage fertility. May damage the unborn child) (Lamastra, et al., 2018) (ECHA, 2020d). DEHP is a chemical used to soften plastics and is therefore present in plastics and products used within households and industry.
- Benzo(a)pyrene, as a diffuse source, due to its general presence in sewage sludge documented in JRC (2012) and the expected good level of information availability. BaP has high toxicity and is present in sewage sludge (WCA, 2019).

2.5 General study limitations

Outside of the data gaps and limitations discussed in specific sections of this report, the following points should be noted as gaps and/or areas for further consideration:

- The Eurostat dataset on the generation and management of sewage sludge gives only limited insight into sewage sludge management, as 'composting' is a generally a pre-treatment technique and does not give information about the final destination. Also, the 'other' category is used for reporting large amounts of sewage sludge without any further information what this entails. This is discussed in Section 3.2.
- There is little information in the data and literature studied on the full detail of the typical sludge management processes in each country. For example, there was no information readily available on what volumes of sewage sludge were subject to AD as an intermediate step before the final disposal fates recorded. This pre-treatment will impact on the actual volumes of sludge finally disposed of (AD significantly reduces volume) and the amount of energy recovered.
- A full calculation of the potential for sewage sludge in Europe to contribute to increased soil quality and fertility via the addition of organic carbon and related humus content was not possible due to a variety of local factors determining such potential. These include, amongst others, frequency and duration of sludge application, cultivation practices, use of other soil improvers, types of crops grown and local climate.
- Similarly, our energy recovery calculations include both the electricity and heat generation potential from AD processes. While electricity is relatively easy to make use of via feeding into national grids in the absence of any direct demand, heat recovery is reliant on having an off-take such as district heating systems or industrial processes.



- It should be noted that for both sets of calculations we have derived figures for what could be technically possible based on data reported and stated assumptions. We have not discussed the financial or infrastructure-driven feasibility factors involved in delivering that potential in practice, which are important next-step considerations for any future strategic discussions or planning. A high proportion of the European population is served by a relatively small number of large UWWTP, with many smaller plants covering the remainder. To realise economies of scale in the potential energy and nutrient recovery levels discussed here, there may need to be development of new infrastructure or centralised collection and treatment processes. These may have additional environmental impacts related to e.g. construction and materials and increased transportation etc. Thus, additional analysis capturing the associated costs and benefits of realising these theoretical potential recovery levels would be required to understand their feasibility and economically and environmentally preferable way forward.



3. Sewage sludge – current situation

3.1 European policy framework

There are three main legislative acts covering sewage sludge in the EU: The Sewage Sludge Directive (SSD, 86/278/EEC); Urban Waste Water Treatment Directive (UWWTD, 91/271/EEC) and the Fertilising Products Regulation (FPR, 2019/1009). The specifics of these are presented below, with references to relevant policies included in brackets:

- The residual sewage sludge from UWWTP treating urban waste water can be used in agriculture only if it has been treated (e.g. biological, chemical or heat treatment, or long-term storage). However, the use of untreated sludge can be authorised if it is injected or worked into the soil (1).
- The accepted concentrations of six metals (Cd, Cu, Ni, Pb, Zn and Hg) in sewage sludge and in soil to which sewage sludge is applied are limited. The maximum annual quantities of these metals introduced into soil used for agriculture are also limited (1).
- Application of sewage sludge to land is prohibited in a number of scenarios, including on grassland that is for grazing and foraging crops when the crops are due to be harvested in three weeks or less, and on soil in which fruit and vegetable crops are growing (with the exception of fruit trees). Additionally, for ground intended for the cultivation of fruit and vegetable crops which are normally in direct contact with the soil and eaten raw, sludge use is not permitted for a period of ten months preceding the harvest of the crops and during the harvest itself (1).
- Sewage sludge must be used in a way that account is taken of the nutrient needs of the plants and that the quality of the soil and of the surface and ground water is not impaired. Also, where sewage sludge is used on soils of which the pH is below 6, the increased mobility and availability to the crop of metals must be considered and the limit values reduced if necessary (1).
- Industrial waste water entering collecting systems and urban waste water treatment plants must be subject to pre-treatment, if required, in order to ensure that (2):
 - The operation of the waste water treatment plant and the treatment of sludge are not impeded,
 - Discharges from the treatment plants do not adversely affect the environment,
 - Sludge can be disposed of safely and in environmentally acceptable manner.
- All agglomerations¹⁵ with a population equivalent (p.e.) of more than 2 000 must have collecting systems for urban waste water (2). A larger share of population connected means larger volumes of water treatment, and larger quantities of sewage sludge produced and in need of management.

European policies regulating sewage sludge

1. SSD (1986)
2. UWWTD (1991)
3. FPR (2019)

¹⁵ Area where the population and/or economic activities are sufficiently concentrated for urban waste water to be collected and conducted to an urban waste water treatment plant or to a final discharge point (EC, 2020b)



- For all discharges from agglomerations of more than 10 000 p.e., urban waste water entering collecting systems must be subject to secondary treatment or an equivalent treatment before discharge. It must also be subject to more stringent treatment before discharge into sensitive areas, since 1998. For discharges to fresh water and estuaries from agglomerations of between 2 000 and 10 000 p.e., urban waste water entering collecting systems must be subject to secondary treatment or an equivalent treatment before discharge. Urban waste water discharges from agglomerations of between 10 000 and 150 000 p.e. to coastal waters, and of between 2 000 and 10 000 p.e. to estuaries situated in less sensitive areas, may be subject to less stringent treatment providing that such discharges receive at least primary treatment and that comprehensive studies indicate that these discharges will not adversely affect the environment (2). These requirements for more advanced treatment lead to greater removal from waste water of substances which may then accumulate in sludge.
- Limit values are set for the concentrations of total P and N for discharges from UWWTP to sensitive areas which are subject to eutrophication (2). This means that in these areas, the concentration of P and N in sewage sludge might be higher.
- The load expressed in p.e. must be calculated on the basis of the maximum average weekly load entering the treatment plant during the year, excluding unusual situations (e.g. heavy rain) (2).
- Sewage sludge arising from UWWTP must be re-used whenever appropriate. Disposal routes must minimise the adverse effects on the environment and by the end of 1998, the disposal of sludge to surface waters by dumping from ships, by discharge from pipelines or by other means had to be phased out (2).
- Fertilisers made from sewage sludge cannot be placed on the EU market currently or traded from one state to another. However, the advancement of technologies to recover P from sewage sludge means that, as the manufacturing processes become scientifically established, fertilisers made from sewage sludge will be able to transition smoothly on the EU market in the near future (3).
- The European Commission has proposed adopting delegated acts to amend Annexes I to IV to the FPR for the purposes of adapting them to technical progress in the light of new scientific evidence. At the time of writing this report, three delegated acts were pending adoption. These acts would allow EU fertilising products to contain precipitated salts and derivatives and thermal oxidation materials obtained from processing of sewage sludge from municipal waste water treatment (JRC, 2019).

3.1.1 Sewage Sludge Directive (1986) (86/278/EEC)

The key instrument regulating sewage sludge use in agriculture in the EU is the SSD (1). The Directive was formulated on the basis of the precautionary principle: to protect human health and the environment against harmful effects of untreated, contaminated sludge. However, the Directive was introduced more than 30 years ago and its scope and limits were found to be outdated by the 2014 “Ex-post evaluation of certain waste stream directives” (EC, 2014). This is, for example, evidenced by the fact that many Member States have set stricter metal limits than those of the Directive, and, have introduced limits for organic pollutants not restricted in the Directive. A more in-depth evaluation of the Directive¹⁶ was launched in Q3 of 2020. The

¹⁶ https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=PI_COM%3AAres%282020%293116544



Evaluation will assess whether the Directive has had the initially intended effects, which provisions have achieved their objectives, and which have not.

There have been significant developments in diverting biodegradable wastes from landfill and encouraging the recycling of organic wastes through composting and AD in the last 30 years. Depending on the country, sewage sludge has been part of these developments yet public perception and the risks associated with hazardous micropollutants present in sludge remain a decisive factor when determining sludge recovery potential.

A key concern when using sewage sludge in agriculture is its contamination with pollutants removed from the waste water and sorbed to the sludge, e.g. polycyclic aromatic hydrocarbons (PAH) and perfluorinated surfactants, pharmaceuticals and microplastics. As the range of products used in modern society increases, so does the potential variety of contaminants which can make their way into sewage sludge, although some pollutants are broken down in the waste water treatment process and further during sludge treatment. Petersen (2020) points out that an increase in research exploring contaminants of emerging concern has uncovered potential impacts on all aspects of agricultural systems, from soil quality to human health. An in-depth review of the scientific knowledge about microplastics in the environment concluded that sources of microplastics in soils are not well known but that sewage sludge could be an important source (SAPEA, 2019). On the other hand, other recent studies conclude the following:

- “Overall, it is concluded that sewage sludge from contemporary Danish society does not constitute a higher risk to soil organisms or human health than cattle or pig slurry.” (Pedersen, et al., 2019)
- “Evidence for a total ban being necessary is lacking, however, research having failed to prove that crops grown with sludge have health impacts or have an adverse impact on ecosystems in agriculture. On the other hand, there is clear evidence that sludge fertiliser application supplies plant nutrients and humus that agriculture demands.” (Ministry of the Environment of Sweden, 2020)
- “Long-term application of Swedish sewage sludge on farmland does not cause clear changes in the soil bacterial resistome”
 - “Application of sewage sludge did not cause accumulation of antibiotics in soil.”
 - “There was no detected increase in phenotypic resistance after sludge application.”
 - “Long-term sludge-amendment did not cause enrichment of resistance genes in soil.” (Rutgersson, et al., 2020).

As a result, analysis of sludge composition, the behaviour of emerging chemicals in sludge and a full assessment of sludge in an agricultural setting should be explored further and defined within the SSD (EC, 2014). The European Commission’s evaluation of the UWWTD proposed that any revision to the Directive would need to explore the inclusion of pollutants of emerging concern in scope (EC, 2019d). As part of stakeholder engagement within this evaluation, trade associations and UWWTP operators expressed the need for these pollutants to be dealt with. With growing awareness and evidence base, pollutants of emerging concern may need to be dealt with through specific treatment or restrictions on sludge spreading on soils. This is to both reflect the precautionary principle and manage public perception and expectations.



3.1.2 Urban Waste Water Treatment Directive (1991) (91/271/EEC)

The Urban Waste Water Treatment Directive (UWWTD) (2) was adopted to protect the environment from the adverse effects of urban waste water discharges, as well as those from certain industrial sectors, and more specifically from discharges of organic pollution and nutrients. This Directive outlines necessary steps for the collection, treatment and recovery of treated waste water and sewage sludge (EC, 2020b).

The SSD and the UWWTD are strongly related since the SSD deals with the waste product from the waste water treatment processes regulated under the UWWTD. The recent evaluation of the UWWTD found that the link could be made more explicit, especially for areas such as measurement requirements, analytical methods, and reporting cycles. The evaluation found that, even though the UWWTD has led to a marginal increase in sludge recovery, requirements on doing this were not clear thereby leaving room for interpretation by Member States when implementing relevant provisions. Particularly unclear was the term ‘whenever appropriate’ in reference to sludge recovery.

As stated above, Annex I of the UWWTD includes provisions to protect the quality of sludge by requiring pre-treatment of waste water before it enters UWWTPs if required, in order to ensure treatment of resultant sludge is not impeded and that sludge can be disposed of safely.

Additionally, as part of the UWWTD, EU Member States are required to publish biennial reports on waste water treatment and can voluntarily report on sludge management. There is overall little information on sludge related practices from this reporting process. However, the European Commission (2019b) points out that both UWWTD and the Industrial Emissions Directive (IED) do not contain clear requirements for treating and recycling sludge. This highlights a gap where neither the UWWTD nor IED currently offers detailed and up to date rules for sludge monitoring, analysis and treatment and the SSD provides rules only for one type of use i.e. spreading on agricultural land. Current regulatory framework thus lacks clear rules on managing sludge for other uses and disposal routes.

3.1.3 Fertilising Products Regulation (2019) (EU 2019/1009)

The purpose of this regulation is to encourage the move away from the EU’s dependency on imports of mineral fertilisers, encourage a greater use of organic fertilisers and development of the circular economy for nutrients. This regulation intends to allow a number of organic fertilisers, soil improvers and growing media to be placed on the EU market and traded from one state to another. In doing this, producers who intend to place their products on the EU market will need CE certification¹⁷ for their products, the criteria of which currently stipulates that sewage sludge and industrial sludge are not allowed as input materials (EEA, 2020b).

Whilst sewage sludge is not currently a permitted input, the advancement of technologies in the field of phosphorus recycling means that, as the manufacturing processes become scientifically established, these products should be able to have a smooth transition to market. This will be further supported by upcoming amendments to the Fertilising Products Regulation which will allow fertiliser products to contain precipitated salts, thermal oxidation materials and derivatives obtained from sewage sludge as an input material (JRC, 2019). The power to define and introduce additional component materials eligible for use in the production of EU fertilising products, and their corresponding contaminant limit values, remains with the Commission. This power will only be exercised when this can be justified through technical progress.

¹⁷ A certification which confirms products have been producer/manufacture assessed and confirmed to meet all relevant EU health, safety and environmental protection requirements.



3.1.4 Sewage sludge and circular economy policies

Key legislation such as the SSD and the UWWTD focused on the control and management of waste water and sludge pre-dates the current EU climate neutrality and circular economy policy agendas. As a result, there is a lack of unified strategy to ensure that sludge governance also encourages circularity, recovery or recycling.

In 2014, the SSD was evaluated as part of an "Ex-post evaluation of certain waste stream directives", which collected a broad range of stakeholder inputs. The evaluation indicated that the SSD is not coherent with circular economy principles, and that this could be tackled by establishing more precise resource efficiency objectives e.g. on waste prevention, circular economy, product design or phosphorus recovery or establishing different categories of sludge quality (with "high" quality sludge being subject to fewer constraints and having access to a wider range of outlets) (EC, 2014).

A potential difficulty for legislation to promote sludge recovery is that it has been caught between potentially conflicting objectives. On the one hand, protecting the environment and human health requires specific precautions not to transfer the problem from one environment compartment to the other. On the other hand, if legislation on sludge were to contribute fully to the objectives of policy and legislation on resource efficiency, the use of sludge in agriculture would be more strongly encouraged to ensure the recycling and recovery of valuable and finite nutrients. Sludge could also contribute to renewable energy policy and targets as feedstock for energy generation, for example, through AD or incineration. When viewed in the context of the waste hierarchy, however, energy recovery is of lesser importance than nutrient recovery. There may therefore be a balance to be found in the use of low-quality sludge as energy feedstock after the higher value material has been subject to nutrient recovery, or combinations of both. In addition, there are differing approaches to human health, soil and groundwater protection in the EEA member countries; the latitude left by the legislation has resulted in some countries adopting policies to prevent the use of sludge in agriculture due to health and pollution concerns, whereas others consider agricultural use to be adequately managed within safety boundaries and view it as an important agricultural input (EC, 2014).

Recent policies which promote circular economy principles, such as the European Green Deal and the Circular Economy Action Plan, set a new direction towards resource efficiency and recovery and recovery of key nutrients wherever possible. In particular the second foresees the development of an Integrated Nutrient Management Plan, 'with a view to ensuring more sustainable application of nutrients and stimulating the markets for recovered nutrients' (EC, 2020a).

Phosphate rock and phosphorus are on the EU's critical raw materials list¹⁸, which means incentives for recovering phosphorus from waste water and sludge using both established and emerging methods, should be increased in the future. However, there are currently no established markets for recovered phosphorus.

¹⁸ https://ec.europa.eu/growth/sectors/raw-materials/specific-interest/critical_en



3.2 Management of sewage sludge in the EEA-32

Table 3-1 presents the quantities of sewage sludge produced and disposed in EU-27 and EEA-32 based on the data from Eurostat. According to the latest data available (see detailed explanation in the note below the table):

- In EU-27: a total of 10.4million t of dry sewage sludge were produced, of which 94 % were disposed.
- In EEA-32: 11.1 million t of dry sewage sludge were produced, of which 93 % were disposed.

The differences between the amount of sewage sludge produced and disposed can be explained by the fact that some countries store the sewage sludge produced before disposing of it and also by the fact that the anaerobic digestion process reduces the amount of sewage sludge.

The amount of sewage sludge produced is defined by Eurostat as “the quantity of decanted matter resulting from waste water treatment, including sludge treatment” (Eurostat, 2020a). Disposal methods include agricultural use, compost and other applications, landfill, incineration and other. It should be noted that there are some gaps and reporting delays in the Eurostat data as it currently does not provide the amount of sewage sludge disposed by Finland (FI) (total amount disposed, and amounts disposed by different methods), the amounts produced and disposed by Iceland (IS) and Liechtenstein (LI) (total amount disposed, and amounts disposed by method), and data from Denmark are from 2010.

Table 3-1 Quantities of dry sewage sludge produced and disposed in EU-27 and EEA-32 at the latest date available

	EU-27	EEA-32
Sludge produced in million t	10.44	11.09
Average Sludge produced in kg/capita	17	17
Sludge disposed in million t	9.78	10.36
Sludge disposed in % of sludge produced	94 %	93 %
% of population connected to urban waste water collecting systems	83%	84 %

Notes: For data on sludge production and disposal, we used 2018 data for AT, HR, CY, CZ, HU, LV, LT, MT, NL, NO, PL, RO, SK, SI, SE, TR. For the countries for which the 2018 data was missing, data from the latest available year has been used as follows: 2017 (BE, BG, FR, IE, LU, CH), 2016 (EE, DE, EL, IT, PT, ES), 2015 (FI), 2010 (DK).

For data on the population connected to urban waste water collection systems, we used 2017 data for BE, BG, HR, CZ, DK, EE, FR, HU, IE, LV, LT, LU, MT, NL, NO, PL, PT, RO, SK, SI, SE. For the countries for which the 2017 data was missing, data from the latest available year has been used as follows: 2016 (AT, DE, EL, TR), 2014 FI, ES), 2013 (CH), 2010 (IS), 2009 (IT).

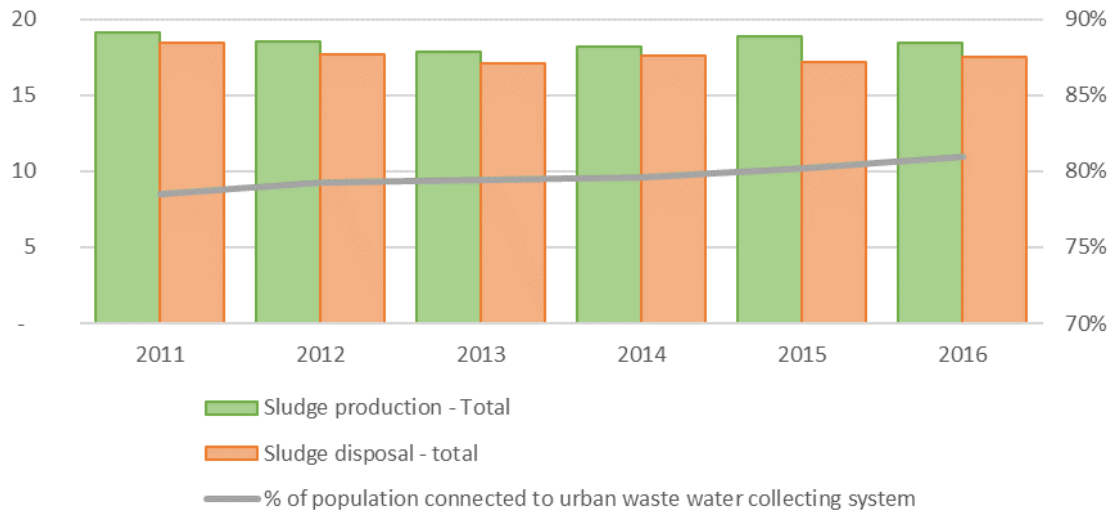
Source: (Eurostat, 2020a) and (Eurostat, 2020b)

Figure 3-1 shows the trend in sewage sludge production, disposal and percentage of population connected to urban waste water collecting system in the EU-27 (see note below figure) based on the data from Eurostat.

Between 2011 and 2016, the dry sewage sludge production per thousand inhabitants has decreased by 3.7 % and the dry sewage sludge disposal by 5.1 % while the percentage of population connected to urban waste water collecting systems increased by 3.2 percentage points. This decrease in generated amounts, in spite of increasing population and connection to collection systems, could come from e.g. an increase in AD as this reduces sludge amounts.



Figure 3-1 Trend in sewage sludge production and disposal in the EU-27 (in kg of dry sewage sludge/inhabitant), and population connected to urban waste water collecting systems (in % secondary axis)



Source: (Eurostat, 2020a)

Note: Information on trend not available for BE, CY, DK, FI, IT and PT.

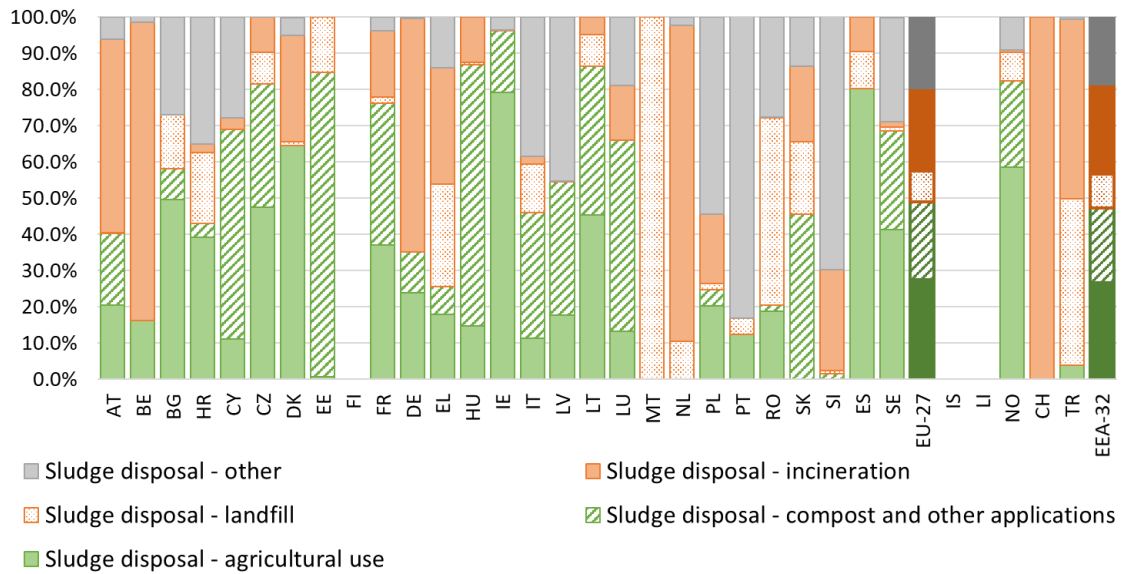
Figure 3-2 shows current sewage sludge management across EEA-32 member countries based on Eurostat. Eurostat gives the following definitions for each management route (Eurostat, 2020a):

- Agricultural use: all use of sewage sludge as fertiliser on arable land or pastures, irrespective of the method of application
- Compost and other uses: all use of sewage sludge after mixing it with other organic material and composting, e.g. in parks or for gardens
- Landfill: all sludge that is disposed of in tips, landfill areas or special depot sites and that serves no useful function
- Incineration: all sludge that is disposed of by direct incineration or by incineration after mixing with other waste

Overall, in the EU-27, 27 % of the sewage sludge disposed of is used in agriculture, 21 % in compost and other applications, 8 % is landfilled, 23 % is incinerated and 20 % is disposed of in another way. Similarly, in the EEA-32, 27 % of the sewage sludge disposed of is used in agriculture, 21 % in compost and other application, 9 % is landfilled, 25 % is incinerated and 19 % is used in another way.



Figure 3-2 Current sewage sludge management approaches in EEA-32



Notes: For data on sludge production and disposal, we used 2018 data for AT, HR, CY, CZ, HU, LV, LT, MT, NL, NO, PL, RO, SK, SI, SE, TR. For the countries for which the 2018 data was missing, data from the latest available year has been used as follows: 2017 (BE, BG, FR, IE, LU, CH), 2016 (EE, DE, EL, IT, PT, ES) and 2010 (DK).

Source: (Eurostat, 2020a)

Table 3-2 shows the dominant disposal method used by each country in EU-27 and EEA-32.

Table 3-2 Dominant sewage sludge disposal method/country

Disposal method	Countries
Agriculture	Bulgaria (BG), Croatia (HR), Czechia (CZ), Denmark (DK), Ireland (IE), Lithuania (LT), Norway (NO), Spain (ES) and Sweden (SE)
Compost and other application	Cyprus (CY), Estonia (EE), France (FR), Hungary (HU), Luxembourg (LU) and Slovakia (SK)
Landfill	Malta (MT) and Romania (RO)
Incineration	Austria (AT), Belgium (BE), Germany (DE), Greece (EL), Netherlands (NL), Switzerland (CH) and Turkey (TR)
Other	Italy (IT), Latvia (LV), Poland (PL), Portugal (PT) and Slovenia (SI)

The figure shows that for some countries, the share of sewage sludge management classified as ‘other’ is relatively significant. Liaison with Eurostat reveals that, while Member States have the option to specify what is included in their ‘other’ submissions, this is very rarely provided, with the only entry on record being from Czechia for the year 2013 stating:

“We provided “other purposes” until 2013. It contained “drying” as one of kind of sludge disposal. The Decree No. 428/2001 Coll., On Act No. 274/2001 Coll., On Water Supply and Sewerage Systems has been changed and drying has been excluded from sludge disposal since 2014. Data could not be provided for period 2014-2017 for that reason.”



Table 3-3 summarises findings of additional literature research on what category ‘other’ may mean in countries where this option applies to more than 10 % of total sludge disposed.

Table 3-3 ‘Other’ sludge disposal

Country	Potential fate of sludge marked as ‘Sludge disposal – other’ in Eurostat’s dataset
Bulgaria	Unknown (Uzunova, 2020)
Croatia	Temporarily stored within UWWTPs’ perimeter or unknown (Vouk, et al., 2017)
Cyprus	The source states “soil enrichment” but there is no further detail available on what that implies (Republic of Cyprus Statistical Service, 2007)
Greece	Re-use within the plant site and forestry (Karagiannidis, et al., 2011)
Italy	Blended, mixed and repackaged and used as a fuel and stored. The source is unclear on the purpose of “mixed and repackaged” sewage sludge (ISPRA, 2020)
Latvia	Forestry, land reclamation and greening (Palm & Jakobsson, n.d.)
Luxembourg	<i>Not found</i>
Poland	Used for the reclamation of land including land for agricultural purposes. Also includes sludge that is temporarily stored (Przydatek & Wota, 2020)
Portugal	In 2018, a Portuguese NGO, ZERO, found that the disposal of sewage sludge was not properly monitored in the country. Interrogation of national data revealed that 50 % of the sewage sludge produced in Portugal was classed as ‘no final destination identified’ and therefore potentially not disposed of according to legal procedures (ZERO, 2018)
Romania	According to the source, the sewage sludge is used for the reclamation of land (Sabau, 2015)
Slovakia	Temporary storage and storage on the landfill (Lew, 2020)
Slovenia	Export to Hungary (STA, 2019)
Sweden	Landscaping and landfill coverage to establish a plant layer (Statistics Sweden, 2018)



3.3 Unused energy recovery potential

Summary

- Estimations show that the EU-27 could potentially recover between 1 800 GWh and 3 200 GWh of energy (heat and electricity) from sewage sludge through pre-treatment by AD and 250 GWh (electricity) from sewage sludge incineration in addition to what is already recovered today. This respectively represents 7 % , 13 % and 1 % of the total waste water sector energy needs in the EU-27 in 2018. If there were a partial shift of 50 % from current use in agriculture and composting of sewage sludge to future incineration, the EU-27 total potential recovery from this route would rise to 1 100 GWh electricity.
- Estimations show that the EEA-32 could potentially recover between 1 900 GWh and 3 300 GWh of energy (heat and electricity) from sewage sludge through pre-treatment by AD and 350 GWh (electricity) from sewage sludge incineration. If a partial shift of 50 % from current landspreading of sewage sludge to future incineration would occur, the EEA-32 could potentially recover a total of 1 100 GWh of energy from sewage sludge incineration.
- If the UWWTD were to be fully implemented, estimations show that the recovery potential for the EU-27 could rise to between 2 000 GWh and 3 500 GWh of energy from sewage sludge pre-treated by AD and 300 GWh of electricity from sewage sludge incineration. This respectively represents 8 % to 14 % and 1 % of the total waste water sector energy needs in the EU-27 in 2018. If a partial shift of 50 % from current landspreading of sewage sludge to future incineration would occur, the EU-27 could potentially recover a total of 1 200 GWh of energy from sewage sludge incineration.
- Estimations are based on the total amount of sewage sludge currently landfilled and composted for the estimation of the energy recovery potential from AD pre-treatment and on the total amount of sewage sludge currently landfilled, used in agriculture and composted for the estimation of the energy recovery potential from incineration. The only sewage sludge not included in calculations for potential energy recover is that current recorded as disposed of via 'other' means. What this category refers to is undefined and can vary significantly across countries.
- The data presented must be taken with caution given numerous assumptions made in the estimations, including on:
 - The amount of sewage sludge currently subject to AD
 - The amount of sewage sludge currently landfilled, used in agriculture and composted that could be incinerated instead

Methodology to estimate the energy recovery potential of sewage sludge

The primary methods for recovering energy from sludge are:

- Generation of heat and electrical energy from the use of the biogas produced during AD pre-treatment of sewage sludge. It should be noted that AD also generates a stabilised by-product (sludge) with a reduction of about 40 % of the sludge quantity and a very significant reduction of odour, while retaining a usable nutrient content and a reduced amount of volatile organic compounds (ADEME, 2014).
- Recovery of energy from the thermal treatment of sewage sludge (e.g. incineration, co-incineration e.g. in cement industry or in waste incinerators, pyrolysis or gasification).



For simplicity in this analysis, combinations of various techniques (such as incineration with energy recovery of sludge that has been treated through anaerobic digestion, or energy generation from landfill gas obtained from landfills where sludge was disposed) have not been considered when estimating energy recovery potential. Flow charts describing the calculation methodology are provided in Annex D.

The starting point for the calculation of the energy recovery potential is data from Eurostat

- For the estimation of the energy recovery potential from incineration: the total amount of sewage sludge currently landfilled, and
- For the estimation of the energy recovery potential from AD pre-treatment: the total amount of sewage sludge currently landfilled and composted.

We did not include the disposal method “other” in our estimations as what it covers is unclear and differs significantly between countries.

To estimate the potential additional amount of sewage sludge that would be pre-treated via AD, we assumed that the sewage sludge that is currently landfilled would instead go through AD and then be landfilled. Also, while composting produces a useful fertiliser product, the energy recovery potential from sewage sludge would be maximised if the sludge was first subject to AD. Thus, for the purpose of the calculation it is assumed that sewage sludge composted can be pre-treated via AD.

In the absence of hard data on the amount of sewage sludge currently pre-treated by AD, assumptions have been made about the proportion of sewage sludge currently pre-treated in AD facilities. Calculations have been performed assuming hypothetical shares of a minimum of 10 % and a maximum of 50 % of sewage sludge currently landfilled and composted in Europe being subject to AD pre-treatment¹⁹. This means that we estimated the potential based on an additional minimum of 50 % and a maximum of 90 % of sewage sludge currently landfilled and composted being subject to AD pre-treatment²⁰. Table 3-4 shows the potential additional quantity of dry sewage sludge pre-treated by AD based on these assumptions.

Table 3-4 Estimation of the potential additional amount of sewage sludge that could be pre-treated via AD in the EU-27 and EEA-32 (in million t of dry sewage sludge)

Area	Landfill	Compost	% currently pre-treated by AD (assumed)	% additional AD pre-treatment potential (estimated)	Total mass of sewage sludge with additional energy recovery potential
EU-27	0.83	2.10	50 %	50 %	1.46
EU-27	0.83	2.10	10 %	90 %	2.63
EEA-32	0.97	2.12	50 %	50 %	1.55
EEA-32	0.97	2.12	10 %	90 %	2.78

Source: (Eurostat, 2020a) *Sewage sludge production and disposal [env_ww_spd]*

As shown in Table 3-5, to estimate a lower value of the total mass of sewage sludge with additional energy recovery potential from incineration, we assumed that all sewage sludge currently landfilled would be incinerated instead.

¹⁹ These shares are theoretical to demonstrate a range and we do not believe any country has achieved this to date.

²⁰ Reaching 90 % of sewage sludge pre-treated by AD would require a significant shift in current practices that is not yet ready for implementation and therefore theoretical.



To estimate an upper value of the total mass of sewage sludge with additional energy recovery potential, we assumed that the following would be incinerated instead:

- All sewage sludge currently landfilled
- 50 % of sewage sludge currently used in agriculture
- 50 % of sewage sludge currently composted

This upper value reflects the fact that in the future, a partial shift could occur from current landspreading of sewage sludge to incineration. This could be triggered by concerns on presence of organic contaminants in sludge, and increased possibilities to recover phosphorus from sewage sludge ashes.

Table 3-5 Estimation of the potential additional amount of sewage sludge that could be incinerated with energy recovery in the EU-27 and EEA-32 (in million t of dry sewage sludge)

Area	Landfill	Lower value of total mass of sewage sludge with additional energy recovery potential	50 % of sludge spread on agricultural land	50 % of sludge composted	Upper value of total mass of sewage sludge with additional energy recovery potential
EU-27	0.83	0.83	1.34	1.05	3.22
EEA-32	0.97	0.97	1.38	1.06	3.41

Source: (Eurostat, 2020a) *Sewage sludge production and disposal [env_ww_spd]*

We also estimated the total amount of sludge with energy recovery potential (including that used in the calculations above) if the UWWTD were to be fully implemented in the EU-27.

To do so, we summed the estimated population (in p.e.) connected to secondary and more stringent treatment under full compliance with UWWTD scenario (Pistocchi, et al., 2019) and divided it by the current p.e. for each country to obtain a scaling factor. We then multiplied this factor by the amount of sewage sludge currently produced by each country using Eurostat data and summed it to obtain the total amount of sewage sludge that would be produced in the EU-27 if the requirements of UWWTD were fully complied with.

We assumed the same disposal routes as detailed in Section 3.2 if the UWWTD was fully implemented. Table 3-6 shows the results of the estimation for the potential additional amount of sewage sludge that could be pre-treated by AD and Table 3-7 for the potential amount of sewage sludge incinerated.

Table 3-6 Estimation of the total potential additional amount of sewage sludge that could be pre-treated via AD in the EU-27 if the UWWTD was fully implemented (in million t of dry sewage sludge)

Area	Landfill (estimated)	Compost (estimated)	% currently pre-treated by AD (assumed)	% AD pre-treatment potential (estimated)	Total mass with additional energy recovery potential
EU-27	0.90	2.30	50%	50%	1.60
EU-27	0.90	2.30	10%	90%	2.90

Source: (Pistocchi, et al., 2019)



Table 3-7 Estimation of the total potential additional amount of sewage sludge incinerated in the EU-27 if the UWWTD was fully implemented (in million t of dry sewage sludge)

Area	Landfill (estimated)	Lower value of total mass of sewage sludge with additional energy recovery potential	Assumed 50 % Agricultural use	Assumed 50 % Compost	Upper value of total mass of sewage sludge with additional energy recovery potential
EU-27	0.90	0.90	1.50	1.20	3.60

Source: (Pistocchi, et al., 2019)

The energy consumption and production are estimated based on findings from a literature review and expressed in kWh/t of dry sewage sludge. These are summarised in Table 3-8. The sources used offered the most robust and consistent estimates of energy consumption and production, without necessitating too many variable assumptions (e.g. on factors such as calorific value of biogas and sewage sludge produced per m³ of waste water treated).

Table 3-8 Energy consumption and production (in kWh/t of dry sewage sludge)

Sources	Energy consumption*		Energy production**	
	AD pre-treatment	Incineration	AD pre-treatment	Incineration
(Palmer, et al., 2020)	338	485	1 600	819
(Mills, et al., 2013)	135	NA	1 455	NA
(Tonini, et al., 2019)***	125	NA	1 160	NA
Average	200	NA	1 400	NA

* Assumed to be electricity consumption only as sources were unclear on what type of energy is consumed.

** Assumed to be electricity and heat production for AD pre-treatment with a CHP electrical efficiency of 38 %, a CHP high grade heat efficiency of 18 % and a low grade heat efficiency of 20 % (Mills, et al., 2013). Assumed to be electricity production only for incineration as source was unclear on what type of energy is produced.

*** This study provided data for wet sewage sludge (5% dry matter content). A multiplication factor of 20 has been used to express data on a dry matter basis, based on expert stakeholder liaison.



Estimated additional energy recovery potential

Based on the assumptions and calculations detailed above, Table 3-9 shows the estimated additional energy recovery potential for the EU-27 and EEA-32.

Table 3-9 Estimation of the potential additional energy recovery by using AD pre-treatment or incineration (in gigawatt-hour (GWh))

Treatment method	Area	% AD pre-treatment potential (estimated)	Total potential energy consumption	Total potential energy production	Total net energy recovery potential (production – consumption)
AD (Heat and Electricity)	EU-27	50%	290	2 060	1 770
	EU-27	90%	525	3 700	3 175
	EEA-32	50%	310	2 175	1 865
	EEA-32	90%	555	3 920	3 365
Incineration (Electricity)	EU-27 Lower	NA	400	680	280
	EU-27 Upper	NA	1 560	2 640	1 080
	EEA-32 Lower	NA	470	795	325
	EEA-32 Upper	NA	1 655	2 795	1 140

For the EU-27, our estimations show that:

- If an additional 50 % of the total amount of sewage sludge currently landfilled and composted in the EU-27 were pre-treated via AD, the potential additional energy recovery would be around 1 770 GWh.
- If an additional 90 % of the total amount of sewage sludge currently landfilled and composted in the EU-27 were pre-treated via AD, the potential additional energy recovery would be around 3 175 GWh.
- If 100 % of the total amount of sewage sludge currently landfilled were incinerated instead, the EU-27 could generate an additional 280 GWh electricity.
- If 100 % of the total amount of sewage sludge currently landfilled and 50 % of the current total amount of sewage sludge being used in agriculture and composted were incinerated instead, the potential additional electricity recovery for the EU-27 could be 1 080 GWh.

For the EEA-32, our estimations show that:

- If an additional 50 % of the total amount of sewage sludge currently landfilled and composted in the EEA-32 were pre-treated via AD, the potential additional energy recovery would be around 1 865 GWh.
- If an additional 90 % of the total amount of sewage sludge currently landfilled and composted in the EEA-32 were pre-treated via AD, the potential additional energy recovery would be around 3 365 GWh.
- If 100 % of the total amount of sewage sludge currently landfilled were incinerated instead, the EEA-32 could generate an additional 325 GWh electricity.
- If 100 % of the total amount of sewage sludge currently landfilled and 50 % of the current total amount of sewage sludge being used in agriculture and composted were incinerated instead, the potential additional electricity energy recovery for the EEA-32 could be 1 140 GWh.



This can be set in the context of the European public water sector energy needs which are estimated to be over 80 000 GWh, of which around 24 800 GWh for waste water treatment (Magagna, et al., 2019). Our EU-27 estimates represent respectively 7 %, 13 %, 1 % and 4 % of the waste water treatment sector needs²¹.

Table 3-10 shows the estimated potential additional energy recovery for the EU-27 if the UWWTD were fully implemented

Table 3-10 Estimation of the potential additional energy recovery by using AD or incineration in the EU-27 if the UWWTD was fully implemented (in GWh)

Treatment method	Area	% AD pre-treatment potential (estimated)	Total potential energy consumption	Total potential energy production	Total net energy recovery potential (production – consumption)
AD (Heat and electricity)	EU-27	50%	320	2 265	1 945
	EU-27	90%	575	4 075	3 500
Incineration (Electricity)	EU-27 Lower	NA	445	745	300
	EU-27 Upper	NA	1 720	2 905	1 185

If the UWWTD was fully implemented and the resulting additionally generated amount of sludge was included in the calculation, our estimations show that the EU-27 could produce in total:

- 1 945 GWh of energy if an additional 50 % of the total amount of sewage sludge currently landfilled and composted in the EU-27 were pre-treated via AD
- 3 500 GWh of energy if an additional 90 % of the total amount of sewage sludge currently landfilled and composted in the EU-27 were pre-treated via AD
- 305 GWh of electricity if 100 % of the total amount of sewage sludge currently landfilled were incinerated instead
- 1 184 GWh of electricity if 100 % of the total amount of sewage sludge currently landfilled and 50 % of the current total amount of sewage sludge being used in agriculture and composted were incinerated instead

This respectively represents around 8 %, 14 %, 1 % and 5 % of the total waste water sector energy needs in the EU-27 in 2018.

Limitations

- For the potential additional energy recovery by using AD pre-treatment, it is assumed that sewage sludge currently landfilled would be previously pre-treated by AD and that sewage sludge composted can be pre-treated via AD.
- It is important to note that, while it is true that the total value of electricity and heat provide the potential energy recovery from AD, electricity is much easier to recover as heat is reliant on having an offtake (e.g. district heating or industrial user), and there are variations in the amounts being used in the process depending on the technology.
- No hard data on the amount of sewage sludge currently subject to AD could be found. Therefore, calculations have been performed assuming hypothetical shares produced in Europe being subject to AD. These shares are purely theoretical to demonstrate a range and we do not believe any country has achieved this level of AD pre-treatment to date.

²¹ As with all energy related sustainability discussion, the first step should be to improve the energy efficiency of current infrastructure before considering to what extent demand can be met with self-supply.



- The assumptions made on the share of sewage sludge currently landfilled, used in agriculture and composted that would be incinerated instead were based on expert judgment and not validated by information found in literature due to a lack of relevant data.
- Combinations of various incineration techniques (such as incineration with energy recovery of sludge that has been treated through anaerobic digestion, or energy generation from landfill gas obtained from landfills where sludge was disposed) have not been considered when estimating energy recovery potential.
- We did not include the disposal method “other” in our estimations as what it covers is unclear and differs significantly between countries.
- We assumed the same proportions in disposal routes detailed in Section 3.2 for the estimation of sewage sludge produced if the UWWTD would be fully implemented.
- The estimations are based on the limited data on conversion factors from mass of sewage sludge to energy. Calculations could be improved by modelling more precisely each step of the process.
- We did not take into consideration the practical limitation to the implementation of AD like the size of the UWWTP or the economic feasibility of the solution.
- Increasing the share of sludge incinerated may require building of new facilities which are associated with environmental and economic impacts (e.g. high costs, emissions, etc.).
- Incineration residue still often becomes final waste which has to be disposed in landfill although recovery of nutrients from incineration ashes is becoming more popular.
- Public concern around waste incineration may prevent the full potential to be realised. This is being illustrated in the Italian case study in section 4.3.

3.4 Unused nutrient recovery potential

Summary

- Estimations show that the EU-27 could potentially annually recover a total of between 6 900 and 63 000 t of P and a total of between 12 400 and 87 500 t of N from sewage sludge. This represents respectively between 0.6 % and 6 % of the total amount of P fertiliser and between 0.1 % and 1 % of the total amount of N fertiliser used in EU agriculture in 2018.
- Estimations show that the EEA-32 could potentially annually recover a total of between 8 100 and 68 100 t of P and between 14 600 and 94 700 t of N.
- If the UWWTD were to be fully implemented, estimations show that the EU-27 could potentially annually recover a total of between 7 600 and 69 300 t of P and a total of between 13 700 and 96 300 t of N. Estimations are based on the total amount of sewage sludge currently landfilled and 50 % of the sludge currently reported as ‘other disposal’. In a mono-incineration scenario, the P recovery potential rises to up to 105 500 t.
- As with energy recovery estimations, the data presented have to be taken with caution given numerous assumptions made in the estimations, including:
 - The recovery rate of nutrients in sewage sludge and sewage sludge ash by landspreading
 - The amount of sewage sludge for which the disposal method is “other” that finds its way to land
 - The amount of sewage sludge that is mono-incinerated



Methodology to estimate the nutrient recovery potential of sewage sludge

The nutrient recovery potential has been estimated for P (P_2O_5) and N. The primary methods for recovering nutrient from sludge are:

- Recovery of nutrient from the landspreading of sewage sludge
- Recovery of nutrient from mono-incineration of sewage sludge ashes, either by the landspreading of the sewage sludge ash or the use of specific nutrient recovery technologies on the sewage sludge ash

The starting point for the calculation of the nutrient recovery potential is data from Eurostat:

- For the estimation of the nutrient recovery potential from landspreading: the total amount of sewage sludge landfilled and 50 % of the total amount of sewage sludge for which the disposal method is “other”. Even though what is covered by the category “other” is unclear and differs significantly between countries, it is likely that a share of it finds its way to land as it is the cheapest management route
- For the estimation of the nutrient recovery potential from mono-incineration ash: the total amount of sewage sludge currently incinerated and the potential additional amount of sewage sludge that could be incinerated in the future (calculated in Section 3.3)

We assumed that all sewage sludge for which the disposal route is “compost and other applications” would be already used as fertiliser. Eurostat is not clear on what “other applications” refer to and therefore it has been assumed that there is no *additional* nutrient recovery potential from this sludge. Flow charts describing the calculation methodology are provided in Annex D.

In the absence of hard data, the following assumptions were made:

- We assumed that 100 % of the nutrient content in the sewage sludge would be recovered, both through landspreading of sewage sludge and sewage sludge ash, and plant available²²
- Calculations have been performed assuming a hypothetical share of 50 % of sewage sludge currently incinerated is subject to mono-incineration. This share has also been applied to the potential additional amount of sewage sludge that could be incinerated in the future (i.e. that is currently disposed of to landfill).

The recovery rate of P from sewage sludge ash using recovery technologies has been assumed to be at 90 % (The German Environment Agency, 2018). No recovery of N from incinerated sewage sludge has been estimated as N is removed during the combustion process.

Not considered in these calculations, but worthy of further thought following this study, is the impact of ‘more stringent’ or advanced treatment techniques on nutrient recovery potential. These treatments can target specific constituents of sewage sludge and result in a sludge output of different N/P profile than primary or secondary sludge. P is (generally) still diverted to the resultant sludge, from which it can be recovered. N is generally transformed into nitrate NO_3 (nitrification) and then transformed in N_2 that goes to air (denitrification). This obviously has consequences for circular economy aspirations and there does not appear to be much incentive to develop processes to recover N (except some trials on urine separation).

²² This share is theoretical to demonstrate a potential of recovery



Also outside the scope of these calculations is the plant availability of recovered P in the specific soil it is applied to. For P to be effective, certain conditions need to be met. These are that P must be:

- In the right form as inorganic free ions, in solution.
- Present at the soil – root interface.
- Available at the right time as the crop demands.

The P availability for plants in fertilisers is not considered in the calculations. However, key findings from academic literature are summarised in Box 3-1.

Box 3-1 P availability for plants in fertiliser application

In the first year after P application, a significant amount of P fertiliser is not taken up by plants, and then becomes unavailable. This is due to a range of factors influencing the soil's ability to fix P, and therefore render it plant available. There is a process of in-soil conversion of inorganic P to organic P form. Phosphate ions tend to form complex compounds with other soil minerals and constituents (including iron, aluminium and cadmium), adsorb to soil solid surfaces and be taken up by soil organisms and converted to organic forms following metabolism, excretion and decay. Once these processes take place, P is no longer available for plants (Mezeli, et al., 2019). However, some studies show that in the long run, most of the P in sludge will be used by the plants (Linderholm, 2011).

Research using bacterivorous nematodes also suggests that in-soil diversity or abundance are not the only factors in plant acquisition of P, but that much more sophisticated interactions play a significant part. Certain biological interactions can play a big part in phosphorus uptake by plants and raise uncertainty about the focus on soil biodiversity as a key instrument to improve plant productivity (Mezeli, et al., 2020).

Alongside the N and P potential considered in this report, application of sewage sludge to land has also been shown to be beneficial to soil fertility through improved soil structure via increased humus content, and increased soil organic carbon (Borjesson & Katterer, 2018). This can form a positive feedback cycle, with the long-term study concluding that increased organic carbon stocks (up to 9.1 Mg/hectare at the 0 – 0.4m depth) were due to both direct retention from application of sewage sludge, and from residues from the enhanced crop yields. Increased humus levels in soil can reduce the level of nutrients required to produce crops (Someus, 2009). As such, sewage sludge can play an important role in both providing valuable nutrients and reducing related demand for them. Nevertheless it is one of many potential sources of organic carbon and humus content in soils, including biowaste-derived compost and other decayed plant and animal matter.

It has not been possible to calculate the additional increase potential in carbon stocks. This is because soil carbon levels are determined by local factors, such as frequency and duration of application of the sludge, application of other improvers, frequency of cultivation (where frequent cultivation enhances carbon loss), type of crops grown and prevailing weather conditions (Borjesson & Katterer, 2018; Chen et al, 2018; Nguyen et al., 2018).

The left side of Table 3-11 shows the amount of sewage sludge currently landfilled in the EU-27 and the EEA-32. Based on the same assumptions used for the estimations of the additional energy recovery potential, we also estimated the total amount of sewage sludge with nutrient recovery potential, including that used in the initial calculations, if the UWWTD were fully implemented. The right side of Table 3-11 shows the projected total quantity of sewage sludge that may be produced and landfilled or disposed as 'other disposal' assuming full implementation of UWWTD.



Table 3-11 Estimation of the total amount of sewage sludge that could have an additional nutrient recovery potential via landspreading in the EU-27, in the EEA-32 and in the EU-27 if the UWWTD was fully implemented (in million t of dry sewage sludge)

Area	Landfill	50 % "other"	Total
EU-27	0.83	0.97	1.80
EEA-32	0.97	0.97	1.95

Area	Landfill (estimated)	50 % Other (estimated)	Total (estimated)
EU-27	0.91	1.07	1.98

Source: (Eurostat, 2020a) *Sewage sludge production and disposal [env_ww_spd]* and: (Pistocchi, et al., 2019)

Table 3-12 shows the assumed amount of sewage sludge that could have an additional nutrient recovery potential via mono-incineration ash in the EU-27 and the EEA-32 and assuming full implementation of UWWTD.

Table 3-12 Estimation of the total amount of sewage sludge that could have an additional nutrient recovery potential via mono-incineration in the EU-27, in the EEA-32 and in the EU-27 if the UWWTD was fully implemented (in million t of dry sewage sludge)

Area	50 % of incinerated sludge to mono-incineration with P recovery	
EU-27	Current	1.13
	Current + potential additional - lower value	1.54
	Current + potential additional - upper value	2.74
EEA-32	Current	1.29
	Current + potential additional - lower value	1.77
	Current + potential additional - upper value	3.00
EU-27 if UWWTD fully implemented	Current (estimated)	1.24
	Current (estimated) + potential additional - lower value	1.70
	Current (estimated) + potential additional - upper value	3.01

Values of total P and N content of sludge generated in Europe were sourced from literature and are presented as ranges in Table 3-13.

Table 3-13 P (P₂O₅) and N content in sewage sludge (in kg/t of dry sewage sludge)

Sources	P (P ₂ O ₅)		N	
	Low	High	Low	High
Xu (2014)	15	20	20	38
Hamood & Khatib (2016)	8	28	15	40
(Canziani & Spinosa, 2019)	2	57	10	68
Average	8	35	15	49

This can be set in perspective with other recycled nutrient sources such as manure and bio-waste. The average nutrient content in dairy cow manure is 1.4 kg of P/t of dry dairy cow manure and 4 kg of N/t of dry dairy cow manure (UMass Extension and the Center for Agriculture, Food and the Environment, Unknown). In 2016, WRAP provided the following indicative nutrient content for bio-waste (WRAP, 2016):

- In food-based digestate: 0.5 kg of P/t of fresh weight and 5 kg of N/t of fresh weight
- In compost: 3.4 kg of P/t of fresh weight and 9.25 kg of N/t of fresh weight



Estimated additional nutrient recovery potential

Based on the assumptions and calculations detailed above, Table 3-14 shows the additional nutrient recovery potential via landspreading for the EU-27 and EEA-32.

Table 3-14 Estimation of the potential nutrient recovery via landspreading in the EU-27 and EEA-32 (in t of nutrient)

Area	Scope	P (P ₂ O ₅)		N	
		Low	High	Low	High
EU-27	Landfill	6 910	29 030	12 440	40 365
	Landfill + 50 % "Other"	14 990	62 950	26 980	87 530
EEA-32	Landfill	8 090	33 980	14 560	47 245
	Landfill + 50 % "Other"	16 220	68 115	29 190	94 710

For EU-27, our estimations show that:

- If 100 % of the P and N content in sewage sludge currently landfilled could be recovered via landspreading, between 6 900 and 29 000 t of P and between 12 500 and 40 400 t of N could be recovered annually.
- If 50 % of sludge managed by "other" methods was included, the EU-27 could recover a total of between 15 000 and 63 000 t of P and a total of between 27 000 and 87 500 t of N.

For EEA-32, our estimations show that:

- If 100 % of the P and N content in sewage sludge currently landfilled could be recovered via landspreading, the EEA-32 could annually produce between 8 100 and 34 000 t of P and between 14 600 and 47 200 t of N.
- If 50 % of sludge managed by "other" methods was included, the EEA-32 could recover a total of between 16 200 and 68 100 t of P and a total of between 29 200 and 94 700 t of N.

However, these figures have to be taken with caution as the data collection has highlighted that the fate of sewage sludge that is managed by "other" methods is unclear for many countries, and when it is known, varies significantly between the countries, making full accurate calculations difficult. Secondly, the sludge currently landfilled might not be of good enough quality for landspreading.

In 2018, the EU-27 used 10.2 million t of N and 1.1 million t of P in EU agriculture (Eurostat, 2018). This means that the potential additional nutrient recovery via landspreading could represent between 0.6 % and 5.7 % of the total amount of P fertiliser and between 0.1 % and 0.9 % of the total amount of N fertiliser used in the EU agriculture in 2018. This relative contribution could increase in the next decade as the EC's Farm to Fork strategy aims to reduce the fertilisers use by 20 % by 2030 (EC, 2020).

Table 3-15 shows the estimated potential additional nutrient recovery via landspreading for the EU-27 if the UWWTD was fully implemented.

Table 3-15 Estimation of the total potential additional nutrient recovery via landspreading in the EU-27 if the UWWTD was fully implemented (in t of nutrient)

Area	Scope	P (P ₂ O ₅)		N	
		Low	High	Low	High
EU-27	Landfill	7,605	31,945	13,690	44,415
	Landfill + 50 % "Other"	16,495	69,270	29,690	96,320



If the UWWTD was fully implemented, our estimations show that the EU-27 could recover in total:

- If 100 % of the P and N content in sewage sludge currently landfilled could be recovered via landspreading, between 7 600 and 32 000 t of P and between 13 700 and 44 400 t of N could be recovered annually.
- If 50 % of sludge managed by “other” methods was included, the EU-27 could recover via landspreading a total of between 16 500 and 69 300 t of P and a total of between 29 700 and 96 300 t of N.

This means that the potential additional nutrient recovery via landspreading could represent between 0.7 % and 6.3 % of the total amount of P fertiliser and between 0.1 % and 0.9 % of the total amount of N fertiliser used in the EU agriculture in 2018.

Based on the assumptions and calculations detailed above, Table 3-16 shows the additional nutrient recovery potential via mono-incineration for the EU-27 and EEA-32.

Table 3-16 Estimation of the potential P recovery via mono-incineration in the EU-27 and EEA-32 (in t of nutrient)

Sewage sludge ash landspreading			
Area	Scope	Low	High
EU-27	Incineration	9 410	39 510
	Incineration + potential additional lower value	12 865	54 030
	Incineration + potential additional upper value	22 830	95 885
EEA-32	Incineration	10 745	45 120
	Incineration + potential additional lower value	14 790	62 110
	Incineration + potential additional upper value	24 970	104 875
Use of nutrient recovery technology			
Area	Scope	Low	High
EU-27	Incineration	8 465	35 560
	Incineration + potential additional lower value	11 575	48 625
	Incineration + potential additional upper value	20 545	86 295
EEA-32	Incineration	9 670	40 610
	Incineration + potential additional lower value	13 310	55 900
	Incineration + potential additional upper value	22 475	94 385

Our estimations show that:

- In the case of sewage sludge ash landspreading, for EU-27
 - If the P content in 50 % of the sewage sludge currently incinerated could be recovered via ash landspreading, between 9 500 and 39 500 t of P could be recovered annually.
 - If the sewage sludge currently disposed of to landfill were added as additional incineration potential, between 13 000 and 96 000 t of P could be recovered annually.
- In the case of sewage sludge ash landspreading, for EEA-32
 - If the P content in 50 % of the sewage sludge currently incinerated could be recovered via ash landspreading, between 11 000 and 45 000 t of P could be recovered annually.
 - If the sewage sludge currently disposed of to landfill were added as additional incineration potential, between 15 000 and 105 000 t of P could be recovered annually.



- In the case of the use of nutrient recovery technologies on incineration ash, for EU-27
 - If the P content in 50 % of the sewage sludge currently incinerated could be recovered via ash landspreading, between 8 500 and 36 000 t of P could be recovered annually.
 - If the sewage sludge currently disposed of to landfill were added as additional incineration potential, between 11 500 and 86 000 t of P could be recovered annually.
- In the case of the use of nutrient recovery technologies on incineration ash, for EU-32
 - If the P content in 50 % of the sewage sludge currently incinerated could be recovered via ash landspreading, between 10 000 and 41 000 t of P could be recovered annually.
 - If the sewage sludge currently disposed of to landfill were added as additional incineration potential, between 13 000 and 94 000 t of P could be recovered annually.

This means that the potential additional nutrient recovery via mono-incineration could represent between 0.8 % and 8.7 % of the total amount of P fertiliser used in EU agriculture in 2018.

Table 3-17 shows the estimated potential additional nutrient recovery via mono-incineration for the EU-27 if the UWWTD was fully implemented.

Table 3-17 Estimation of the total potential additional nutrient recovery via mono-incineration in the EU-27 if the UWWTD was fully implemented (in t of nutrient)

P (P₂O₅)			
Sewage sludge ash landspreading			
Area	Scope	Low	High
EU-27	Incineration	10,352	43,480
	Incineration + potential additional lower value	14,155	59,452
	Incineration + potential additional upper value	25,122	105,513
Use of nutrient recovery technology			
Area	Scope	Low	High
EU-27	Incineration	9,317	39,132
	Incineration + potential additional lower value	12,740	53,507
	Incineration + potential additional upper value	22,610	94,961

In the case of sewage sludge ash landspreading, if the UWWTD was fully implemented, our estimations show that:

- If the P content in 50 % of the sewage sludge currently incinerated could be recovered via ash landspreading, between 10 000 and 43 000 t of P could be recovered annually.
- If the sewage sludge currently disposed of to landfill were added as additional incineration potential, between 14 000 and 106 000 t of P could be recovered annually.

In the case of the use of nutrient recovery technologies on sewage sludge ash, if the UWWTD was fully implemented, our estimations show that:

- If the P content in 50 % of the sewage sludge currently incinerated could be recovered via ash landspreading, between 9 000 and 39 000 t of P could be recovered annually.
- If the sewage sludge currently disposed of to landfill were added as additional incineration potential, between 13 000 and 95 000 t of P could be recovered annually.



This means that the potential additional nutrient recovery via mono-incineration could represent between 0.8 % and 9.6 % of the total amount of P fertiliser used in EU agriculture in 2018.

Limitations

- The unused potential of nutrient recovery from sewage sludge has been limited to P and N. However, greater controls of SO₂ emissions have been put in place meaning that the deposition of S from air into soil has decreased and that therefore lead to increased demand for S fertiliser. Literature shows that sewage sludge can contain between 5 and 15 kg of S/t of water and ash free sewage sludge and could therefore be considered as a potential source of S fertiliser (The German Environment Agency, 2018).
- Most UWWTP in Europe use more stringent treatment that reduces the amount of N and P discharged to rivers. For P, this is done via processes which transfer it to sludge but for N, this can involve nitrification/denitrification processes. If N was extracted instead of stripping to N₂, the potential recovery could be higher (Van der Hoek, et al., 2018).
- Full accurate calculation using sewage sludge that is managed by “other” methods is difficult as the fate in this category is unclear for many countries and when known, varies significantly between the countries.
- Full accurate calculation using sewage sludge that is currently mono-incinerated is difficult as the current split between the use of co-incineration and mono-incineration for management of sewage sludge is unclear.
- A full calculation of the potential for sewage sludge in Europe to contribute to increased soil quality and fertility via the addition of organic carbon and related humus content was not possible due to the wide variety of factors determining the potential.
- It is assumed that all sewage sludge for which the disposal route is “compost and other applications” would be used as fertiliser as sources were unclear on what “other applications” cover.
- We assumed the same proportions in disposal routes detailed in Section 3.2 for the estimation of sewage sludge produced if the UWWTD would be fully implemented.
- The recovery rate of the nutrients contained in sewage sludge and landspreading of sewage sludge ash is purely theoretical.
- We assumed that the nutrients recovered would be fully plant available as nutrient recovery methods obtain variable plant availability and no hard data could be found on that aspect.
- Nutrient recovery of currently landfilled sludge via landspreading would require that the sludge is of sufficient quality for agricultural use.
- The upper potential of incineration through mono-incineration cannot be added up to the potential for landspreading as both assume the diversion of currently landfilled sewage sludge to either landspreading or incineration.

4. Sewage sludge management in selected European countries

4.1 Estonia

The national policy context

Sewage sludge management in Estonia is regulated at the national level. Available options and practices are governed by a combination of legislation which, amongst other points, places restrictions on the use of sewage sludge for agricultural purposes and sets stricter metal limits for sewage sludge products (e.g. compost, digestate) than the EU SSD.

National context

- “Sustainable Estonia 2021”, the Estonian National Strategy on Sustainable Development, was established in 2005. One of the five goals of the Strategy was to maintain an ecological balance and reduce pollution (e.g. air quality, water quality, waste management).
- In 2007, the “Estonian Environmental Strategy 2030” was developed, based on “Sustainable Estonia 2021”. The objectives of this strategy included a decrease of waste disposed in landfill by 30% and of waste harmfulness, the improvement of the surface water and groundwater quality, sustainable mineral resource extraction with minimum losses and waste, sustainable soil management practices (e.g. nutrient and organic matter balanced) and the development of renewable energy generation (e.g. biogas).
- Sewage sludge is qualified as waste, and as such, the Environmental Board requires mandatory registration of all sewage sludge handlers and notification of any sewage sludge handling.
- The level of organic pollutants present in sewage sludge is not fully understood and studies have found pharmaceuticals concentrations exceeding safe levels in treated and untreated sewage sludge (Haiba, et al., 2016).
- There is an important need in Estonia for organic material for soil prepared for landscaping and recultivation to recover former mining areas, quarries, milled peat fields and closed landfills (OÜ aqua consult baltic, 2015).
- As a result of the legislation on sewage sludge products, a Quality Assurance System has been developed for them (compost, digestate, biochar, dried product, ash).



The key provisions of current national legislation are presented here, with references to relevant legislation included in brackets:

- For the use of raw sewage sludge in agriculture, the limit values for metal and pathogen concentration are set in accordance with the Council Directive 86/278/EEC. These uses are prohibited where the limits are exceeded and depend on the maximum limits of the average metal content over ten years (1).
- Prohibited actions include; the cultivation of crops, berries, herbs and aromatics for food or feed on fields where sewage sludge has been sprayed less than a year ago as well as animal harvesting or collecting animal feed where sewage sludge was sprayed less than two months ago (1, 4).
- Transportation and use of sewage sludge are qualified as waste management and as such require the waste handler to be registered and to notify any activity about two weeks in advance (1, 2).
- On average over five years, the quantity of P must not exceed 25 kg/hectare. The quantity of N must not exceed 170 kg/hectare per year and its use can be restricted in areas sensitive to nitrate and karstic areas (those with natural underground drainage systems with sinkholes and caves, due to the presence of soluble rock such as limestone, dolomite, and gypsum). These P and N limits include amounts from manure left on the land by livestock. (3).
- The use of sewage sludge can be restricted in nitrate sensitive and karstic areas to protect drinking water catchment area against water pollution and is prohibited in water protection zones (3).
- Safety requirements and quality limit values for sewage sludge products (e.g. compost, digestate, biochar, dried product, ash) follow much stricter rules than those set up for the use of raw sewage sludge in agriculture (4).

Legislation regulating sewage sludge in Estonia

1. *Regulation No. 29 (2019)*
2. *Waste Act (2004, as amended in 2014)*
3. *Water Act (2019, as amended 2020)*
4. *Regulation No. 24 (2017)*

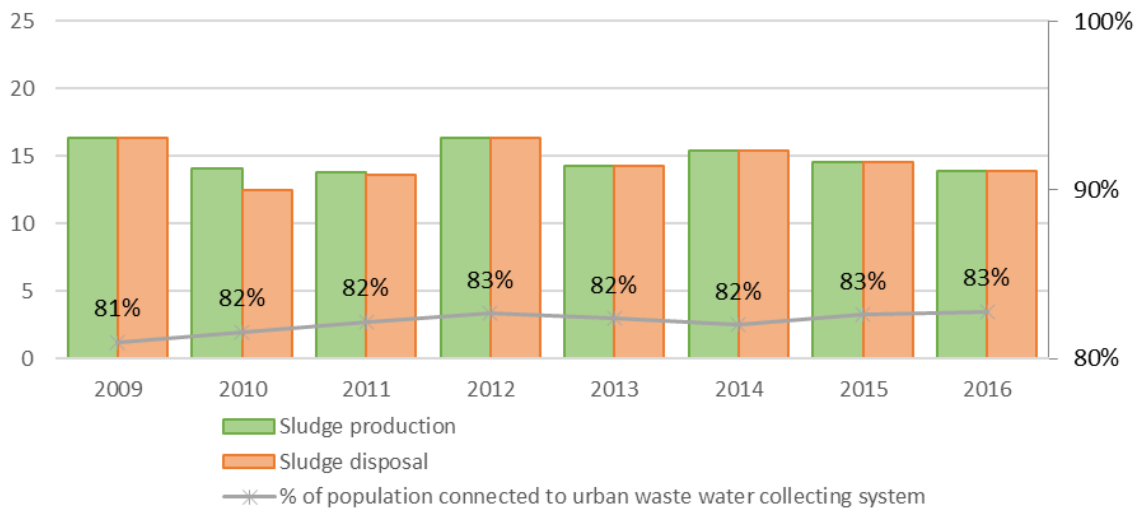
Current sewage sludge levels and management

Sewage sludge production

In 2016, 13.9 t of dry sewage sludge per thousand inhabitants were produced in Estonia and 83% of the population was connected to the urban waste water collecting system. Figure 4-1 below shows that between 2009 and 2016, the quantity of sewage sludge produced and disposed per thousand inhabitants has fluctuated year on year but overall decreased by around 15 %, from 16.3 to 13.9 t of dry sewage sludge produced and disposed per thousand inhabitants (production and disposal levels are identical for most years). While the total population remained relatively stable during this period, the percentage of population connected to urban waste water collecting systems has increased by around 2 %. The increase of sewage sludge production in 2012 could be attributed to the implementation of stricter requirements. It is believed that the overall reduction could be linked to the development of waste treatment technologies (OÜ aqua consult baltic, 2015).



Figure 4-1 Annual quantities of sewage sludge produced and disposed in Estonia (in t/thousand habitants), and population connected to urban waste water collection systems



Source: (Eurostat, 2020a) and (Eurostat, 2020b)

In Estonia, different regulations have established different limits for metals in sewage sludge, depending on how it is used. Table 4-1 below summarises these limits and includes the European limits for the use of raw sewage sludge in agriculture and the Estonian average concentrations in sewage sludge from ten UWWTPs in 2010. Of these ten UWWTPs, no single example was found of levels exceeding Estonian limits. This shows that metals levels found in sewage sludge are far below limits established at EU-level.

Table 4-1 Limits for metals contained in raw sewage sludge in EU and Estonia and in sewage sludge products in Estonia, and metals concentrations in sewage sludge in Estonia (in mg/kg of dry solids)

Limits set in EU and Estonian legislation		Cd	Cr	Cu	Hg	Ni	Pb	Zn
Directive 86/278/EEC		20 - 40	NA	1 000 - 1 750	16 - 25	300 - 400	750 - 1 200	2 500 - 4 000
Estonian limits	Sewage sludge used in agriculture	20	1 000	1 000	16	300	750	2 500
	Sewage sludge products use in agriculture and horticulture*	0.15	15	45	0.1	4	7.5	125
	Sewage sludge products use in landscaping and recultivation*	2	60	200	1	40	130	2 500
Average concentrations in Estonia		Cd	Cr	Cu	Hg	Ni	Pb	Zn
Sewage sludge used in agriculture (2019)		0.80	31.16	120.53	0.49	15.83	11.92	531.72
Sewage sludge compost (2010)*		0.42	12	43	0.2	8	9.2	220

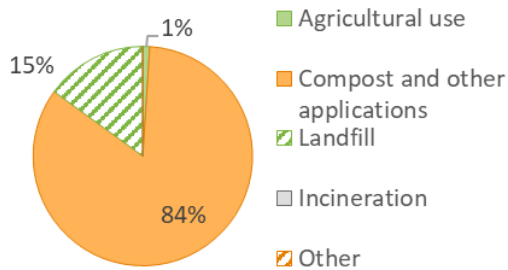
* E.g. compost, digestate, biochar, dried product, ash.

Sources: (Hudcová, et al., 2019), (Ministry of the Environment of Estonia, 2019), (Ministry of the Environment of Estonia, 2017a), (Pauklin & Iiige, 2012) and liaison with Estonian Environment Agency representatives



Sewage sludge management

Figure 4-2 Sewage sludge management practices in 2016 (in % of total sludge disposed)



Source: (Eurostat, 2020a)

In Figure 4-2, the Eurostat data show that in 2016, 84 % of the sewage sludge disposed in Estonia was used in compost and other applications, 15 % was landfilled and 1 % was used in agriculture. The Estonian Environment Agency provided a slightly different split with 79 % of the total amount of sewage sludge disposed being used in compost and other applications, 16% being landfilled and 5 % being used in agriculture.

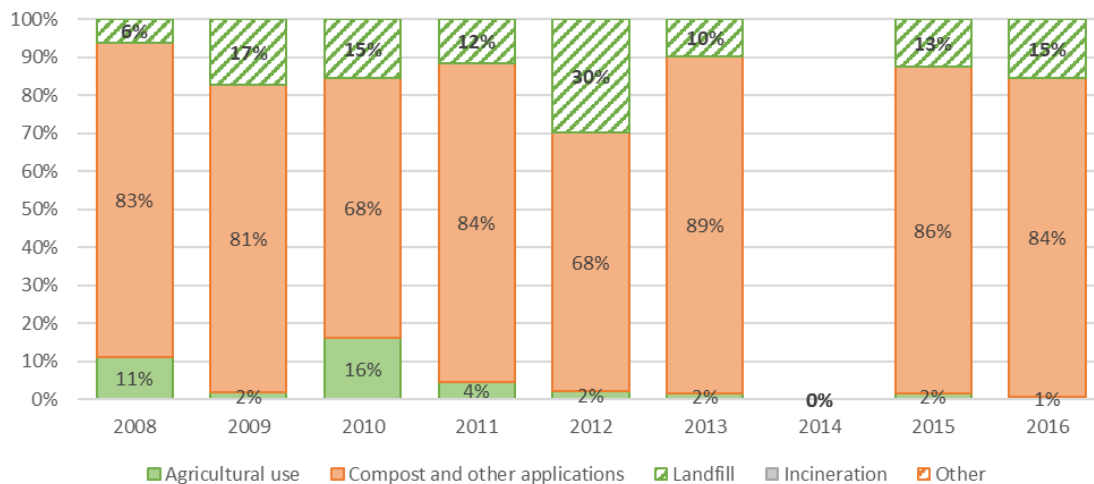
While these differences do not impact significantly on the analysis which follows, the issue does highlight potential discrepancies between data held across Estonian institutions. The large share of sewage sludge used for compost and other applications is explained by the important need in Estonia for landscaping and ‘recultivation’. Recultivation is *“the use of sludge for the reconditioning or preparation for reuse of areas spoilt by mining of natural resources or in some other manner, or for covering landfills.”* (OÜ aqua consult baltic, 2015). Compost and other applications also include landscaping and storage.

Use of sewage sludge in landscaping and recultivation is especially important in case of industrial waste landfill sites that have thin topsoil layers and a lack of organic matter (Ministry of the Environment of Estonia, 2017b). The nutrients in sewage sludge are therefore helpful in promoting sustainable plant growth via a contribution to the creation of a more productive topsoil layer. The use of stabilised sewage sludge for these purposes is seen as economically attractive (OÜ aqua consult baltic, 2015). The sources consulted could not provide a further breakdown of how much of the 84 % was used for individual activities within the ‘Compost and other applications’ category, i.e. how much for contribution to topsoil layers as described above, or how much for purely refilling or closing quarries and pits etc. It is therefore difficult to assess the overall balance between environmental benefit and simple ease of disposal currently practised.

Figure 4-3 shows that the quantity of sewage sludge landfilled in Estonia has been relatively stable in the past ten years at around 13 %, except in 2012 when it went up to 30 %. However, in the same period the quantity of sewage sludge used for agriculture has dropped by 10 percentage points, being nearly non-existent in 2016.



Figure 4-3 Annual sewage sludge management practices (in % of total sludge disposed)



Source: (Eurostat, 2020a)

Due to the lack of a national sewage sludge management strategy, the management of sewage sludge, and the sewage sludge recovery purpose differ significantly between individual regions. For example, the Ida-Viru County is particularly concerned by the need to recultivate areas due to a history of industrial waste disposal. Thus there is a potential economic benefit from sludge use in these often nutrient-poor areas (Ministry of the Environment of Estonia, 2017b). On the other hand, the Harju County, the most populated region and therefore the area generating the highest quantities of sewage sludge, mainly uses sewage sludge for landscaping purposes (OÜ aqua consult baltic, 2015).

Phosphorus recovery

P is not actively recovered in Estonia, outside of the relatively small percentage used in agriculture, but it is not entirely lost as sewage sludge is re-used in the form of compost for landscaping and the contribution to productive topsoil in the recultivation activities described above. While these activities will contribute to better plant growth in these areas, and the P will be fixed in the soil, without further detailed assessment the actual value extraction of the P in these activities is difficult to compare to, for example, use in agriculture for specific crop growing.

Impacts of the sewage sludge management method(s)

Costs of sewage sludge management

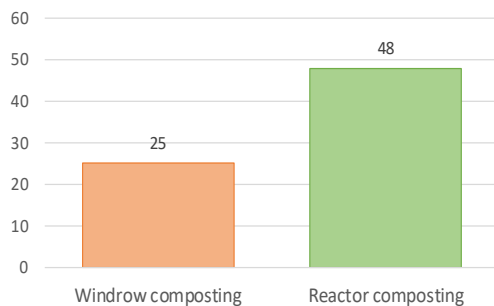
Pre-treatment

It is estimated that the average cost of AD of sewage sludge in Estonia is 82 €/m³ (OÜ aqua consult baltic, 2018). It is unclear based on the source whether this applies to wet or dewatered sludge, or whether energy revenues/savings from AD product is incorporated. It should also be considered that AD leads to reduction in quantities of sludge requiring management afterwards (30-40%) hence also a reduction of cost associated with the further management of sludge.



Sludge treatment and disposal

Figure 4-4 Costs of sewage sludge treatment methods (in €/m³ of dry sewage sludge)



Source: (OÜ aqua consult baltic, 2018)

Figure 4-4 shows the ranges of economic costs associated with different types of sludge composting in Estonia. Methods are briefly explained in Annex 1. Windrow composting, which involves the piling of material in long rows to produce compost, is the less expensive option compared to more technologically complex composting in reactors.

Other cost considerations

The rate of utilisation of sewage sludge on land might be subject to fluctuations due to the market conditions. Sewage sludge compost is in competition with biowaste compost and digestate. But, studies have shown that, on average, the P content of sewage sludge compost can be four times higher than in biowaste compost, giving sewage sludge compost a competitive advantage (OÜ aqua consult baltic, 2015). However, concerns about the use of sludge for landscaping have been raised by the Estonian Agricultural Research Centre due to the pharmaceutical content in sludge and its impacts on soil bacteria (Liiva, 2015).

Lessons learnt

- In Estonia, the limit values for metals content in stabilised and treated sewage sludge used in agriculture are set at a lower end of range specified in the SSD. However, in addition Estonia has set limit values for the use of sewage sludge products in agriculture, horticulture, landscaping and recultivation – largely due to concerns related to health and safety issues and lack of knowledge concerning long-term negative impacts from such uses.
- Estonia does not have any limit values nor regular monitoring for organic pollutants.
- The need for landscaping and recultivation to recover former mining areas, quarries, milled peat fields and closed landfills drives demand for this sewage sludge. In 2016 84 % (79% according to data held by the Estonian Environment Agency) of the sewage sludge produced was composted and used for these purposes. This demand presents a significant disposal route for the sludge, but the overall environmental benefits are unclear without more detailed assessment and further breakdown of the actual uses which made up this management route.
- The mixing of sewage sludge with sand, chopped wood and other materials is encouraged as it makes the sewage sludge more suitable for landscaping and recultivation.



4.2 Germany

The national policy context

Sewage sludge management in Germany is regulated at the national level. Practices are governed by a combination of legislation which restricts application of sludge to soil, prohibits the landfilling of untreated sludge, and sets targets for the recovery of P.

National context

- Prior to taking the current policy course, there was a long-standing debate in Germany regarding the application of sewage sludge in soil, triggered by human health and environmental considerations.
- Stakeholders such as the German Union of Landowners, the German Association of Industrial Bakeries and The German Church, all held concerns over the quality of sludge and the risk related to potential pollutants.
- The negative perception of sludge could be linked to analysis from the 1970s and 1980s, which showed high levels of Cd and dioxins in sludge (EC, 2001). Concerns about pharmaceuticals, microplastics and other micropollutants also arose as they are not regularly monitored and the knowledge around them is limited. The risk that these components could accumulate and contaminate soils is one of the main reasons why a conservative approach was taken regarding sewage sludge utilisation for agricultural purposes, following the precautionary principle
- Some German Federal States, such as Bavaria, held firmer stance on the problem and lobbied to increase the legislative constrictions on sludge application on agricultural land.

Furthermore, the German Association of Consumer Bodies considered that the risks of application to soil were too high since legislation did not set any limits for organic pollutants.

- The Circular Economy Act was adopted in 2012 to promote the circular economy and reorganise the management of waste across the country. It stipulates the employment of the waste hierarchy and includes specific provision for the management of sewage sludge. The Act empowers the Government to introduce secondary legislation to determine the management of sludge.
- Germany adopted the German Resource Efficiency Programme (ProgRess²³) in 2012, with citizen dialogue influenced updates incorporated in 2016 and 2020. The programme sets targets, guiding principles and approaches to the conservation of natural resources.
- Recognising the importance of P as a “critical raw resource”, and one for which Germany is highly dependent on imports, ProgRess commits to improve P recycling for waste water and sewage sludge and sets a target of ten years after the entry into force of the Sewage Sludge Ordinance of 2017 to show a significant increase in the P recovery from secondary sources that is economically usable.
- It provides support for the industrial-scale deployment of P recovery technologies and the creation of conditions for the use of recycled sewage sludge on soil (Federal Ministry for the Environment, Nature Conservation and Nuclear Safety, 2016)

²³ <https://www.bmu.de/en/topics/economy-products-resources-tourism/resource-efficiency/overview-of-german-resource-efficiency-programme-progress/>



Incorporating the German Government's response to this historical discussion, and the resultant commitment to the precautionary principle, the key provisions of current national legislation are presented here, with references to relevant legislation included in brackets:

- Application of sewage sludge in e.g. organic farming, forest, gardens, grassland, arable land, fruit and vegetable cultivation has been prohibited (1). Where applied on soil, Germany set stricter value limits for metals in sewage sludge compared to the SSD²⁴ (1, 3, 5, 6).
- A transitional period was set to terminate the direct sewage sludge utilisation on soil. For sewage treatment plants of more than 100 000 p.e. the period ends in 2029, for plants between 50 000 and 100 000 p.e., the period ends in 2032. For installations below 50 000 p.e., sewage sludge can still be used in agriculture or landscaping (1).
- A maximum of 5 t of sewage sludge dry solids/hectare may be applied in any three-year period (1). This maximum is further impacted by the provision that the amount of N applied via sewage sludge must be fully included in the operational upper limit (from all sources) of 170 kg N/hectare per year (6). There exists a compensation fund financed by those who are using sewage sludge in agriculture for damage caused by the agricultural utilisation of sewage sludge to people and properties (3). The contribution is paid yearly and based on the amount of dry sewage sludge used.
- Starting in 2029 at the latest, UWWTP and sewage sludge incineration (mono-incineration and co-incineration) plants must recover P from sewage sludge and sludge incineration ash when sewage sludge has a P content of 20 gram (g) or more per kg of dry solids. Sewage sludge with low P content (less than 20 g/kg of dry solids) is exempt from the recovery obligation. The P recovery process employed must ensure a P content reduction of at least 50 %, or to below 20g/kg of dry solids, and at least 80 % from sewage sludge incineration ash or carbonaceous residues (1).
- Sewage sludge and sewage sludge ash are permitted as a main fertiliser component (5).
- Emissions to air from installations drying or co- incinerating sewage sludge are regulated (2 ,4).
- Landfilling of untreated sewage sludge is prohibited (7).
- National soil limit values are established. If these are exhausted, no sewage sludge application is allowed (8).
- Limits are set for Polychlorinated dibenzo(p)dioxin and furan (PCDD/F) of 30 ng/kg for uses of all fertilizer on normal agricultural soil and 8 ng/kg for pasture or agricultural area without ploughing (6).

Legislation regulating sewage sludge in Germany

1. *Sewage Sludge Ordinance (1992, as amended 2017)*
2. *Technical Instructions on Air Quality Control (2002)*
3. *Fertiliser Act (2009, as amended 2017)*
4. *17th Ordinance of the Federal Emission Control Act (2013)*
5. *Fertiliser Ordinance (2012)*
6. *Fertiliser Application Ordinance (2017)*
7. *Landfill Ordinance (2009, as amended 2017)*
8. *Federal Soil Protection and Contaminated Sites Ordinance (1999)*

²⁴ This is because the SSD is from 1986 so limits are outdated.



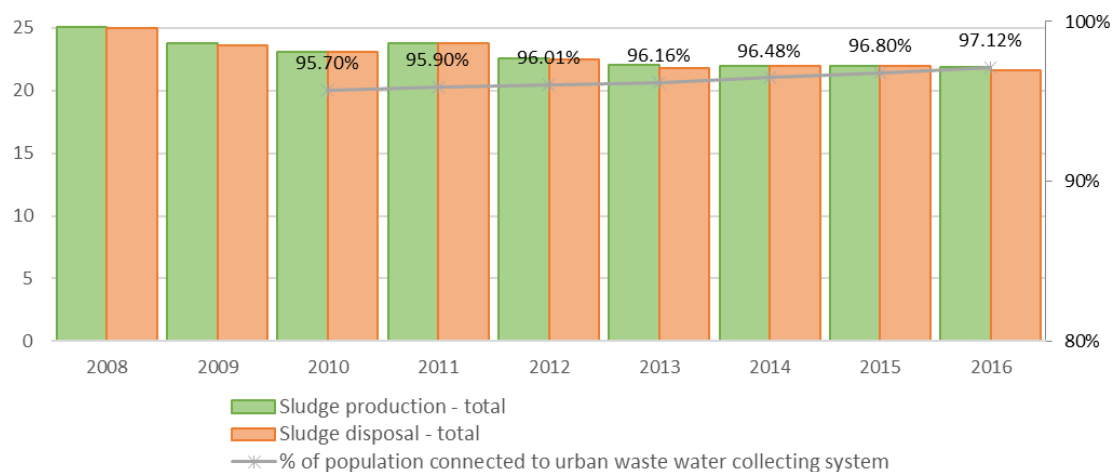
Current sewage sludge levels and management

Sewage sludge production

In 2016, 22 t of dry sewage sludge/thousand inhabitants were produced in Germany and 97 % of the population was connected to the urban waste water collecting system. According to Eurostat, Germany has the third highest level of connection to urban waste water collecting systems, after the Netherlands (99 %) and Malta (99 %).

Figure 4-5 shows that between 2008 and 2016, the quantity of sewage sludge produced and disposed/thousand inhabitants has decreased by around 13 %, from 25 to 22 t of dry sewage sludge produced/thousand inhabitants and 25 to 22 t of dry sewage sludge disposed/thousand inhabitants. During this period, the percentage of population connected to urban waste water collecting systems has increased by around 4 %. This counter-intuitive reduction in sewage sludge production could be attributed to increases in the use and efficiency of anaerobic sludge stabilisation in UWWTPs (The German Environment Agency, 2018).

Figure 4-5 Annual quantities of sewage sludge produced and disposed in Germany (in t/thousand habitants)



Source: (Eurostat, 2020a) and (Eurostat, 2020c)

Sewage sludge contamination

The German Environment Agency recognises the presence of standard contaminants in the country's sewage sludge: metals; organic compounds; pathogens; pharmaceutical residues; plastics and nanomaterials.

Nationwide data exist for concentrations of metals in sewage sludge dating back to 1977.

Table 4-2 presents the historic and current limits (for direct application of sludge in agriculture) and average concentrations of metals and nutrients in agriculturally used sewage sludge in 2015 and shows that metals levels are far below these limits. The sources available only provide average concentration figures, so no assessment of a range, including potential individual measurements outside of the limits set, was possible.



Table 4-2 Limits and concentration of seven metals, PCDD/F, N and P in agriculturally used sewage sludge in Germany (in mg/kg of dry solids)

Limits set in EU and German legislation									
	Cd	Cr	Cu	Hg	Ni	Pb	Zn	N total	P total
Directive 86/278/EEC	20 - 40	NA	1 000 - 1 750	16 - 25	300 - 400	750 - 1 200	2 500 -4 000	NA	NA
Old limits (1992)	From 2 % P ₂ O ₅ (WM*)	900	800	8	200	900	500	NA	NA
New limits (2012, except Zn set in 2017)	50 mg/kg P ₂ O ₅	NA	900	1.0	80	150	4 000	NA	NA

* 2 % of the arithmetic mean weighted by the sample length

Sources: (Hudcová, et al., 2019) (The German Federal Ministry of Justice and Consumer Protection and Federal Office of Justice), 1992), (The German Federal Ministry of the Environment, Nature Conservation and Nuclear Safety, 2017) and (The German Federal Ministry of Justice and Consumer Protection and Federal Office of Justice, 2012)

Concentration in sewage sludge in 2015									
	Cd	Cr	Cu	Hg	Ni	Pb	Zn	N total	P total
Germany	0.74	32.6	293.6	0.39	24.7	30.6	772.8	43 796	24 576

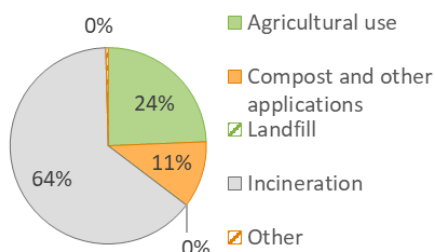
Source: (The German Environment Agency, 2018)

Note: regarding Cd limits here, there is a move to express some pollutants in terms of nutrients. Hence these limits are for levels/ mass of P₂O₅

Germany also has set limits for the levels of dioxins such as PCDD/F in sludge and fertilisers, of 30 ng/kg for uses of fertilizer on normal agricultural soil and 8 ng/kg for pasture or agricultural area without ploughing. (Werber, et al., 2018) These present particular risks for food, and food producing animals, and there is discussion at the EU level to set a similar limit (proposed for 20 ng/kg dm) (JRC, 2019). There are no specific data available for concentrations present in sludge in Germany, but it is believed via liaison with the contributors to the Werber report (Werber, et al., 2018) that most sludge in Europe would meet the limits discussed.

Sewage sludge management

Figure 4-6 Sewage sludge management practices in 2016 (in % of sludge disposal - total)



Source: (Eurostat, 2020a)

Figure 4-6 shows that in 2016, 64 % of the sewage sludge disposed in Germany was incinerated, 24 % was used in agriculture and 11 % in compost and other applications. Large uptake of incineration has been driven by the limitations on sewage sludge application on soils and the landfill ban for untreated sludge. In 2016, around 1.15 million t of dry solids were incinerated, nearly half of it in co-incineration plants. Of the total amount of sewage sludge co-incinerated, around 70 % was treated in coal-fired power plants and 22 % in cement works. The remainder was treated in waste incineration plants where the share of sewage sludge should not exceed 20 %.

While the legislation applies to the whole country, sewage sludge management practices differ between individual federal states due to local circumstances and Federal State policies. For some states, the share of land used for agricultural purposes explains this dissimilarity. For example, Schleswig-Holstein has the highest percentage of utilised agriculture area (63 %) and



also the highest quantity of sewage sludge used in agriculture (18 t/ thousand habitants). Berlin and Hamburg have the lowest percentage of utilised agriculture area (respectively 2 % and 19 %) and the sole route used for sewage sludge disposal is thermal treatment. Also, in zones containing drinking water containment facilities and in drinking water protected areas, the direct use of sewage sludge on soil is fully prohibited within a distance of ten metres. The presence of sewage sludge and the utilisation thereof as fertiliser in zone III water protection areas (i.e. the catchment area of a protected containment facility) are banned in certain cases at the regional level (The German Environment Agency, 2015).

The public acceptance of sewage sludge use also explains dissimilarities. For example in Bavaria, the population is strongly against the direct use of sewage sludge on soil, which leads to 67% of its sewage sludge being incinerated (The German Environment Agency, 2018).

Phosphorus recovery

Due to the upcoming legislative requirements for increased P recovery, various processes are being investigated, including precipitation, crystallisation, adsorption, wet-chemical digestion, thermochemical and metallurgical processes, but they offer varied results and many of these technologies are not at a level of technological readiness to allow large-scale implementation.

Wet-chemical methods (e.g. precipitation) are proving promising. They allow a level of P recovery between 5 and 30 % of the P contained in UWWTP feed, are not expensive to implement and provide a low pollutant recyclate. However, the risk of this recyclate containing unwanted residues is relatively high (e.g. organic micropollutants). The main method currently used in Germany for P recovery is the Magnesium Ammonium Phosphate (MAP) precipitation of sewage sludge using the AirPrex process.

Results from thermal processes are also encouraging since they allow a very high recovery level of P (up to more than 90% of the P contained in UWWTP feed) and eliminate all organic content including micropollutants. However, these processes are far more expensive to implement and operate. Mono-incineration required for these processes leads to the loss of carbon and N and is linked to air emissions. Major pollutants emitted are particulate matter (PM), metals, CO, NO_x, SO₂, and unburned hydrocarbons (EMEP and EEA, 2019).

Impacts of the sewage sludge management method(s)

Costs of sewage sludge management

Pre-treatment

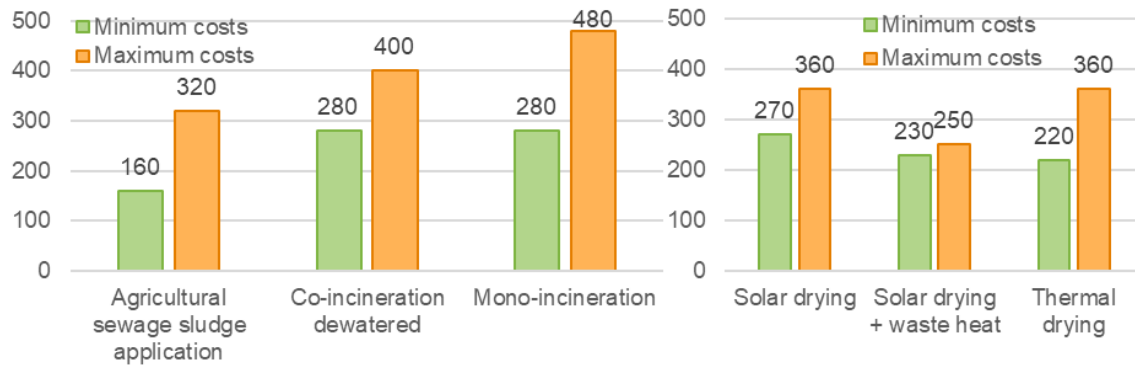
It is estimated that the costs for drying sewage sludge are between €20 - €25/t of dry solids. Drying systems that use waste heat from nearby biogas plants or power plants have a resultant cost advantage (The German Environment Agency, 2018).

Sludge treatment and disposal

Figure 4-7 shows the ranges of economic costs associated with different sewage sludge management methods applied in Germany. Methods are briefly explained in Annex A: Sewage sludge treatment techniques. Drying techniques are presented separately as they are intermediate steps in wider management methods. The use of sewage sludge in agriculture is the most economic option for disposal for operators, and mono-incineration the least, but local conditions (e.g. transport distances, sewage sludge amount, existing treatment capacities) influence the costs of different management pathways.



Figure 4-7 Costs of sewage sludge management methods and treatments (in €/t of dry residue)



Source: (The German Environment Agency, 2018)

Sewage sludge treatment costs are included by operators of UWWTPs in drinking water prices paid by households and waste water treatment charges paid by companies.

Other cost considerations

As previously mentioned, the majority of technological solutions being investigated are not at a technology readiness level to make P recovery as a revenue stream from sewage sludge viable at large scale. Indeed, these technologies involve high level investments for UWWTP to implement them that will have to be passed on P prices, making it more expensive than P currently found on the market and therefore more difficult to find buyers. On the other side, P buyers who would be willing to pay this premium would also have to pass on the price to their products, making them less attractive compared to other products on their market. Therefore, investments have to be made in improving the readiness level of technologies to help further development.

A further potential cost consideration arising from the limitation of agricultural use of sewage sludge is the probable increase in transport costs, which will impact various sectors of the German economy. Indeed, the agricultural use of sewage sludge is mainly local to UWWTP and as it is likely that not all plants would implement sewage sludge treatments in-house, transportation of sewage sludge, and consequent related costs, could increase. (The German Environment Agency, 2018).

Nutrient recovery

Germany, like other EU countries, is dependent on the import of phosphates and mineral fertilisers produced from them. For the economic year 2018/19, 201 000 t of P were sold in Germany (Federal Statistical Office Germany, 2010-2019). P therefore represents a strategic resource, particularly as a plant nutrient.

A large number of different technical processes for the recovery of P from sewage sludge and sewage sludge ash are available in Germany. Most, however, are still in the experimental stage. It is estimated that 50 000 t of P could be recovered per year from municipal sewage sludge mono-incineration (The German Environment Agency, 2018)²⁵. This would correspond to 25 % of the annual agricultural consumption of P. The P recovered from sewage sludge ash could potentially be used as a direct fertiliser, provided that it complied with the limits detailed in the Fertiliser Ordinance. The suitability of sewage sludge compost for delivering P depends on the level of P availability in the soil, which is influenced by several soil factors such as pH levels, mineral composition, moisture levels and clay and organic matter content.

²⁵ This figure based on historic sewage sludge production data. Following our methodology set out in Section 3.4, the upper limit of P recovery for the 2016 production levels would be ~43 000 t.



Lessons learnt

- A combination of various national level strategic drivers and direct legislation has eliminated the landfilling of untreated sewage sludge and reduced relatively widespread health and safety concerns, by restricting the direct use of sewage sludge in many applications, including for agricultural purposes, to protect soil from contamination.
- As a result of this legislative landscape, incineration and co-incineration are now commonly used for sewage sludge treatment and disposal.
- Germany is championing P recovery as part of its German Resource Efficiency Programme, with the potential to reduce its dependence on imports for a critical raw material.
- Germany could produce a theoretical 25 % of its annual agricultural consumption of P if the recovery technologies were fully developed. However, this potential is currently limited by lack of treatment capacity and availability of appropriate P recovery technologies.
- Therefore, any other countries looking to replicate a similar push for resource resilience and the adoption of circular economy principles in its management of sewage sludge needs to be fully aware of the technological and investment support required for research and industrial infrastructure.
- While the costs of incineration of sewage sludge are higher than those related to direct use on soil, they could be reduced to some extent by savings in energy use, and revenues from energy and P recovery.



4.3 Italy

The national policy context

Sewage sludge management in Italy is regulated at the regional level, with the only exception being the national legislation implementing the SSD on national level in Italy. **This means that there is a large variability between the sewage sludge management practices in the different Italian regions.**

National context

- Italy is comprised of 20 regions, of which five have autonomous status (Aosta Valley, Friuli Venezia Giulia, Sardinia, Sicily, Trentino- Alto Adige/South Tyrol).
- Regional governments are allocated policy-making powers following the 2001 constitutional reform introduced in Constitutional law No. 3 of 18 October 2001.
- In the case of sewage sludge management policy, the transposition of the SSD lies within the remit of the state.
- However, regional governments are responsible for the authorisation of the collection, storage, treatment and use of the sewage sludge (Waste and Chemicals, 2018). Furthermore, they can introduce additional rules about the application of sewage sludge to land (ISPRA, 2020) and introduce specific plans about sludge use on the basis of nutrient needs and soil composition (Waste and Chemicals, 2018).
- Regional governments are required to report to state governments about the quantities of sludge produced and their subsequent fate (ISPRA, 2020).
- The northern parts of Italy are characterised with a high share of the population, high population density and economic activity which lead to the production of large amounts of sewage sludge. The application of sewage sludge to land in these regions and its regulation precedes the SSD.



The key provisions of current national and regional legislation are presented here, with references to relevant legislation included in brackets:

- In Italy, National Decree 75/2010 governs the use of fertiliser “plaster”²⁶, which are produced from sewage sludge and manure and are used in agriculture (1).
- For sewage sludge applied on soil, Italy set its limit values at the lower end of the ranges specified in the SSD. Lower limit values have been set for Hg and limits have been set for As, Cr, PCB²⁷, NP/NPE, PAH, PCDD/F which are not regulated under the SSD (2).
- The regulation specifies the sludge stabilisation treatment that should be performed, as well as the maximum amount of sludge that could be applied in a specific area in the duration of three years (2).
- The regulation bans the application of sludge on flooded soil, land intended for pasture or animal feed five weeks before harvest, land intended for horticulture and fruit growing, or when crop is in progress. Furthermore, it lays down requirements for soil moisture and pH (2).
- In 2017, the Court of Cassation Criminal Section III decided that the lack of limits for hydrocarbons and phenols in soils in the regulation does not create an administrative loophole and introduced a limit for hydrocarbons of 50 mg/Kg on the basis of legislative decree 152/2006 “Environmental regulations”. The limit was seen as unattainable for any type of sludge²⁸ (2,3).
- Following the court decision, the region of Lombardy accepted a regional decree setting a higher limit for hydrocarbons at <10 000 mg/kg dry matter (7).
- This led to an appeal by 50 mayors of the neighbouring municipalities of Lodi and Pavia to the Regional Administrative Court of Lombardy on the ground that the higher limit poses serious danger to public health, since soil that contains over 500 mg/kg of hydrocarbons can only be used as landfill. The appeal was accepted but led to a “sludge emergency” since Lombardy was no longer able to dispose of 3,000 t of sludge on agricultural land per week. The situation was resolved with emergency legislation imposing a limit of <1 000 mg/kg of wet matter (8).
- In 2019, Italy updated the limits for PCDD/Fs and PCBs in agricultural topsoils at national level. The limits are more stringent than the limits for sludge in the National Decree 130/2018 which causes problems for the disposal of sludge in some regions (2, 4). The regions of Veneto, Lombardy and Emilia-Romagna have set stricter limits for some pollutants regulated under Decree 99/1992, as well as for additional pollutants not regulated on the national level (6, 9, 10).

Legislation regulating sewage sludge in Italy

1. National Decree 75/2010
 2. National Decree 99/1992 as revised by Decree 130/2018
 3. Court of Cassation Criminal Section III, sentence n.2795806/06/2017 National Decree 46/2019
 4. National Decree 75/2010
- Key regional provisions:
5. Lombardy: Decree X/2031/2014 and Decree 6665/2019
 6. Lombardy Resolution N ° X/7076 Session of 11/09/2017
 7. Lombardy: Decree 109/2018
Veneto: Decree 2241/2005 and 130/2018
 8. Emilia Romagna Decree 285/2005 and 326/2019

Another possibility for use of sludge in agriculture is the use of “plaster” – pellets produced from sludge which are applied to soil as pH improver. These are not regulated under National

²⁶ Agricultural corrective, resulting from the chemical reaction (basic hydrolysis, and possible enzymatic attack) of biological materials using lime and/or sulfuric acid, followed by a precipitation of calcium sulphate and quicklime

²⁷ Polychlorinated biphenyl

²⁸ This information was provided in consultation with the Italian authorities



Decree 99/1992 but need to comply with the metal limits specified in National Decree 75/2010 (1,4). Compost is regulated under National Decree 75/100 following the EU Fertilising Products Regulation.

Current sewage sludge levels and management

Sewage sludge production

Italy has not reported its annual production of sewage sludge to Eurostat since 2010, when it reported 1 103 000 t of dry sludge produced. According to a report by ISPRA, the annual production in Italy between 2015 and 2018 was between 3 069 302 t and 3 183 919 t of wet sludge/year. Assuming average concentration of 22.5 % of dry matter in wet sludge, Mininni et al. (2019) calculates that, in 2016, 716 382 t of dry sludge were produced. This indicates a decrease from 2010, however, a full comparison is not possible due to the limitation of the assumptions of dry content.

The majority of the sewage sludge produced in Italy (approximately 40 %) in 2018 comes from the northern regions, with Lombardy representing 14.2 %, Emilia Romagna 12.4 % and Veneto 12.2 % of the national total (Table 4-3).

Table 4-3 Sewage sludge production/region 2015-2018 (t of wet sludge/annum)

Region	2015	2016	2017	2018
Piemonte	260 453	271 945	260 890	244 636
Valle D'Aosta	4 140	4 624	4 806	4 231
Lombardy	448 155	452 204	480 138	445 245
Trentino Alto Adige	129 068	128 240	129 795	136 454
Veneto	360 643	359 020	373 218	381 215
Friuli Venezia Giulia	82 422	81 634	81 746	79 810
Liguria	41 565	43 071	46 392	39 864
Emilia Romagna	409 214	431 356	445 269	387 538
Toscana	290 931	276 453	291 673	291 196
Umbria	47 289	52 324	49 106	38 181
Marche	74 794	77 035	77 817	80 551
Lazio	312 161	304 962	271 956	370 121
Abruzzo	56 298	70 357	62 694	68 005
Molise	3 102	4 712	5 087	2 553
Campania	188 054	211 037	178 294	145 747
Puglia	221 401	256 754	265 989	280 277
Basilicata	6 572	4 688	4 419	1 754
Calabria	23 530	25 628	25 516	25 030
Sicilia	33 843	42 702	44 150	31 255
Sardegna	75 668	85 173	84 686	83 618
TOTAL	3 069 302	3 183 919	3 183 641	3 137 372

Source: (ISPRA, 2020)



Sewage sludge contamination

Table 4-4 presents the national and regional limits in Italy, as compared to the requirements of the SSD as well as the concentrations recorded between 2013 and 2014. The table shows that the national regulations set the concentration limits at the lower end of the ranges set by the SSD. Lower limit values have been set only for Hg. In addition, limits have been set for As, Cr, PCB, NP/NPE, PAH, PCDD/F which are not regulated under the Directive. On regional level, Lombardy distinguishes between two types of sludge, setting more stringent concentration limits for the second one listed in Table 4-4 (Collivignarelli, et al., 2019). These are:

1. Sludge suitable for application in agriculture, i.e. sludge that meets the national quality limits;
2. High quality sludge, which needs to meet more stringent values and could be applied to land in larger quantities;

Besides the pollutants regulated under the national decree, Lombardy also sets limits for NP/NPE, AOX, Toluene, DEHP.

Veneto's and Emilia Romagna's regional decrees set the pollutant concentration limits at the same levels as the national limits. However, Emilia Romagna also sets regional limits for NP/NPE, AOX, LAS Toluene, DEHP.

In terms of the studies which looked at the levels of compliance with the limit values in Italy, the results showed compliance with the limits for Cu, Hg, Ni and Pb. With regard to Cd, NP/NPA and DEHP, the results were mixed, with some of the measurements showing values above the limits set at national or regional level.

It is noteworthy that according to experts consulted in the process of this study, Italy does not have coordinated monitoring of sewage sludge application to land and therefore, it is impossible to assess whether the limits are met.

Table 4-4 Limits and concentration of eight metals and other parameters in agriculturally used sewage sludge in Italy (in mg/kg of dry solids)

Limits set in EU and Italian legislation																		
	Cd	Cr	Cu	Hg	Ni	Pb	Zn	As	PCB	NP/ NPE	PAH	PCDD /F	AOX	LAS	Toluene	DEHP	Hydrocarbons ²⁹	
Directive 86/278/EEC	20 - 40	NA	1 000 - 1 750	16 - 25	300 - 400	750 - 1 200	2 500 - 4 000	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	
Decree 99/1992 and 130/2018	20	200	1000	10	300	750	2 500	20	0.8	NA	6	25	NA	NA	100	NA	1 000	
Decree X/2031/2014 and 6665/2019 (Lombardy)	Suitable	20	200	1000	10	300	750	2 500	0.8	0.8	50	6	25	500	NA	100	100	1 000
	High quality	5	150	400	5	50	250	2 500	0.8	0.8	50	6	25	500	NA	100	100	1 000
Decree 2241/2005 and 130/2018 (Veneto)	20	200	1000	10	300	750	2 500	20	0.8	NA	6	25	NA	NA	100	NA	1 000	
Decree 326/2019 (Emilia Romagna)	20	200	1000	10	300	750	2 500	20	0.8	50	6	25	500	2 600	100	100	1 000	

Source: (Collivignarelli, et al., 2019)

Concentration in sewage sludge 2013-2014																	
	Cd	Cr	Cu	Hg	Ni	Pb	Zn	As	PCB	NP/ NPA	PAH	PCDD /F	AOX	LAS	Toluene	DEHP	Hydrocarbons
Italy	18–65	No data	90-206	0.2-0.9	11-15	80–126	No data	No data	No data	0-100	No data	No data	No data	No data	No data	0.6 - 149	No data

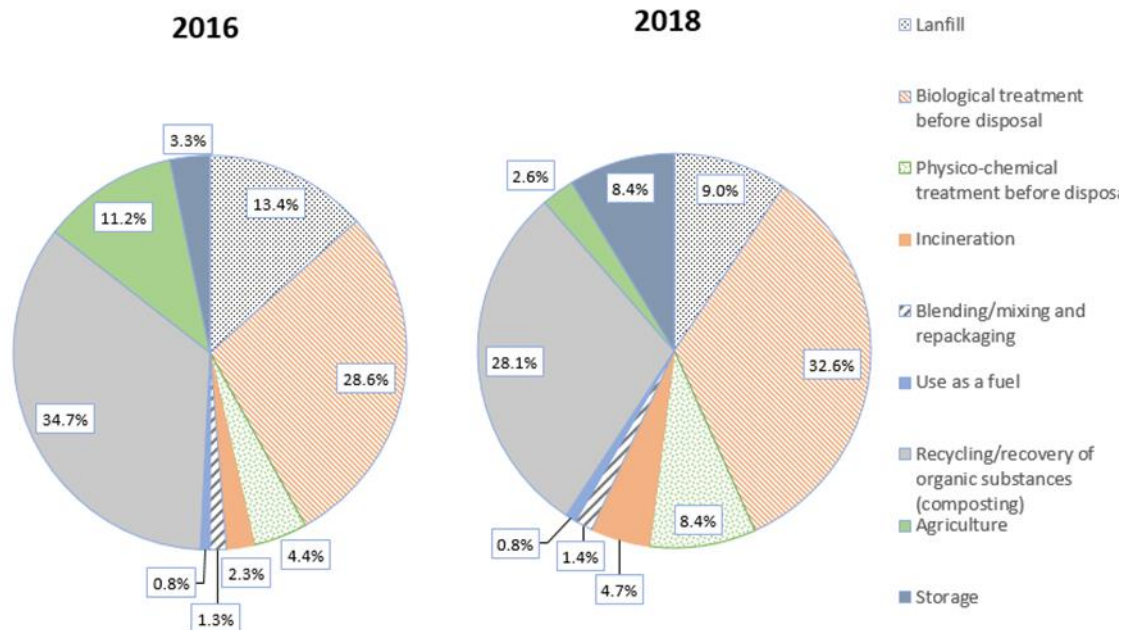
Source: (Fijalkowski, et al., 2017; Lamastra, et al., 2018)

²⁹ Expressed in mg/kg of wet solids assuming approximate 20-25 % dry content. Variations of the dry content of the sludge are possible due to seasonality and palatability of the sludge.



Sewage sludge management

Figure 4-8 Sewage sludge management practices in Italy in 2016 and 2018 (in % of sludge disposal - total)



Source: (ISPRA, 2020)

In 2016, ISPRA reported that out of the 3 183 919 t of wet sewage sludge produced in Italy, 2 924 089 t was managed (92 %). It is unclear what was the fate of the remaining 8 %. The largest share was composted (35%), followed by 29 % which was biologically treated prior to disposal, 13 % which was landfilled and 11 % which was directly used in agriculture³⁰ (Figure 4-8). In 2018, the share of sludge used in agriculture dramatically decreased to less than 3 % whereas the storage and the physio-chemical treatment had an increased share of 8 % each (ISPRA, 2020). It is unclear what is the final destination of disposal for the sludge that was biologically and physio-chemically treated, and whether any sludge was subject to both treatments. Furthermore, it is unclear what is meant by blending, mixing and repackaging. It is assumed that “use as a fuel” may refer to AD or making into a solid fuel pellet. In 2017, UTILITALIA – the federation representing the Italian water companies and cooperating with Aqua Publica Europea (APE)³¹ conducted a survey to estimate the current sewage sludge production in Italy and built up a data base on sewage sludge characteristics (Mininni, et al., 2019). UWWTPs treating 46 % of the p.e. in Italy reported data to the survey (35 million p.e. out of p.e. 72.5 million). The companies produced a total of 395 132 t of dry sludge (Mininni, et al., 2019). The survey found that water utilities deliver 65 % of sewage sludge to the final destination. The destinations include agriculture (10 %), incineration (3 %), incineration plant (3%), landfill sites (17 %), compost plants (26 %) or solid fuel (6%) (Mininni, et al., 2019). The remaining 35 % is delivered to “sludge centres” (i.e. waste management centres dealing with sludge) which are managed by private waste companies. As different types of sludges, wastes and biomass are accepted in these centres, sewage sludge traceability is lost and therefore it is not possible to ascertain which are its final destinations (Mininni, et al., 2019). The paper notes that it is likely that the sludge is subsequently recycled on agricultural land. This can explain the biological and physio-chemical treatment categories used in the ISPRA report as demonstrated in Figure 4-7.

³⁰ In Italy, sludge is usually composted prior to use in agriculture.

³¹ Non-profit organisation, European Association of Public Water Operators.



Use in agriculture

Lombardy, Emilia Romagna, Puglia, Tuscany and Veneto take up 90 % of the sludge use in agriculture in Italy (ISPRA, 2013)³². In 2015, ISPRA published a report summarising the policy requirements and practices in the three regions producing the most sludge and using the most sludge in agriculture, namely Lombardy, Emilia Romagna and Veneto (ISPRA, 2015). The report aimed to summarise best practice to encourage the use of sludge in agriculture in other regions, where this pathway is seen as bearing high levels of risk (ISPRA, 2015).

Examples of agricultural land on which sludge is applied include land producing wheat, corn, tomato, arable land, soy, barley, sorghum (ARPA Veneto, 2017). This report assumes that sludge is not applied when crop growth is in progress in line with national regulations.

Issues with the use of sewage sludge in agriculture have been experienced in some of these regions. For instance, as described above, in the Lombardy Region, in 2018, a court judgment led to the abrogation of a regional hydrocarbons limit of 10 000 mg/kg dry weight posed by the region which was higher than the national limit. As a result, the limit was set at 1 000 ppm wet weight under “emergency” conditions (AGROSISTEMI SRL, 2019). This change forced water operators to find new solutions for the non-compliant sludge, without adequate planning and time to develop sustainable solutions (Aqua Publica Europea, 2019). From the perspective of waste water operators, further sludge management challenges in the region relate to the growing public concern of agricultural sludge recovery, the decrease of agricultural land availability, and the limited capacity for co-incineration of sludge in waste incineration and cement plants, leading to increasing sludge management costs (Blazina, 2019). This also shows that sewage sludge is a small but integral part of the overall waste management system and its strategic directions.

Furthermore, localised contamination events of drinking water in Veneto also contribute to a deterioration in the public perception on sludge, albeit incorrectly, between the pollution in drinking water and the final quality of the sludge

Impacts of the sewage sludge management method(s)

Costs of sewage sludge management

Pre-treatment

The pre-treatment typically applied in Italy is sludge dewatering and thermal drying to reduce its weight (ISPRA, 2020). Furthermore, wet oxidation is also applied (ISPRA, 2020; Blazina, 2019). The Italian reports do not provide information on the costs of pre-treatment.

Treatment and disposal

Italy typically uses composting as a treatment to sludge prior to the sludge in agriculture disposing (ISPRA, 2020; Blazina, 2019). As in the case of pre-treatment, the ISPRA reports do not provide specific costs. However, one report notes that the increased treatment of sludge has led to increased costs for waste water management which are borne by customers (ISPRA, 2015).

Other specific costs

While the costs of different disposal methods are not available, the EC notes that land spreading is a popular option in Italy due to the lower economic cost for waste water management compared to other methods (EC, 2018b). However, the cost of sludge management in Italy has been rapidly increasing due to the uncertain regulatory context with frequent changes of direction (AGROSISTEMI SRL, 2019). Between 2017 and 2019, the sludge disposal cost increased from 50 EUR/t to 180 EUR/t (AGROSISTEMI SRL, 2019). During the emergency decision in Lombardy, the costs reached 245 EUR/t (AGROSISTEMI SRL, 2019).

³² The figure is unlikely to take into consideration the sludge managed by “sludge centres”.



Lessons learnt

- Italy has decentralised governance which results in different approaches being applied in different regions. While there is a central reporting requirement, there is substantially more information available about the practices in the northern regions Veneto, Lombardy and Emilia Romagna, where the most sludge is produced and application practices are better recorded. This creates an impression that most sludge in Italy is applied to land, while the reality might be different.
- Furthermore, the actors responsible for the sewage sludge management process in Italy include the regional authorities, the water companies, and in many instances private companies which deal with waste. When the sludge is managed by private actors such as sludge centres, its fate is not always traceable and therefore the final disposal route is unclear, leaving gaps in the understanding of sludge management in Italy.
- Examples exist of regulatory landlock in Italy which makes the application of sewage sludge to land difficult from the perspective of water companies. Furthermore, there is a growing public concern regarding the use of sludge in agriculture due to the possibility of environmental pollution through sludge application. There is a lack of a clear strategy on how to manage sewage sludge taking into account environmental concerns, circular economy goals as well as cost considerations for operators.
- Sewage sludge can be used in “plaster” which is not in the scope of the SSD and its corresponding national regulation. It is unclear what are the implications of the use of “plaster” on the contamination levels of soil.
- Overall, the studies investigating the impacts of land spreading of sludge have found that the pollutant concentration limits are largely met. However, there is no coordinated monitoring in place which makes it impossible to judge if all limits are met on regional level. Additional pollutants are only regulated in northern Italy where most of the industrial activity is located.



4.4 Sweden

The national policy context

In Sweden, the main applicable legislation for the use of sewage sludge in agriculture is the SSD from 1986, followed by two Swedish regulations (1994 and 1998). Since then, only policy inquiries on sewage sludge management have been conducted and many concerns remain, as current Swedish policies are considered insufficient by key stakeholders for using sewage sludge as a resource.

National context

- When the first UWWTP were developed after World War II, the use of sewage sludge for agricultural purpose was seen as the best disposal solution (i.e. low-cost and beneficial for farmers). However, questions about sewage sludge sanitisation (by cleaning or sterilisation) and concerns around metals and organic micropollutants risks started to arise in 1970s and are still going on today.
- In 2001, Sweden banned the landfill of raw sewage sludge starting in 2005 but this resolution added pressure on the government and stakeholders to find solutions as concerns never eased.
- The current Swedish sludge handling policy is, to a great extent, the result of policy initiatives from the Federation of Swedish Farmers (LRF) and the Swedish Water and Waste Water Association (Svenskt Vatten). The former has, on different occasions, recommended farmers to not apply sludge to land due to the presence of contaminants. However, in 2008 they did endorse the voluntary sludge third party audit certification system REVAQ, developed by them together with the Swedish Food Retailers Federation, the Swedish Food Federation, and the Swedish Environmental Protection Agency. In the absence of strict national regulatory measures, farmers and industry organisations of dairy, food, and waste water treatment can therefore be regarded as key actors in developing the Swedish policy in this area. Currently, REVAQ-certified UWWTP produce 45% of the total amount of sewage sludge produced in Sweden and the LRF and the Svenskt Vatten recommend that only this sewage sludge should be used for agricultural purposes. However, this has not solved concerns and some actors are still reluctant to accept crops treated with sewage sludge (e.g. flour mills, organic certification bodies).
- From a political point of view, the above also shows that there might be some potential conflicts between national and international environmental objectives and guidelines like EU's waste hierarchy and others aiming towards a "non-toxic environment", sustainable management of natural resources, and the Swedish legislative tradition of allowing a certain degree of sovereignty at the municipality and county level in the handling of these matters.
- The concerns among some actors around the use of crops fertilised with sewage sludge are leading Sweden to consider the banning of its use in agriculture. In July 2018 the Swedish government mandated an Inquiry (Ministry of the Environment of Sweden, 2020) to formulate proposals for a ban on the spreading of sewage sludge, with possible exceptions, in order to minimise hazardous substances, pharmaceutical residues and microplastics from entering the ecosystem and to steer towards a non-toxic environment. Among other things, the terms of reference did also include a need to formulate proposals for requirements on the extraction of P from sewage sludge.



- This inquiry identified large scale mono-incineration of sludge combined with technologies to recover P from the ash as one alternative route while noting that “biochar”-producing technologies like pyrolysis and hydrothermal carbonisation remain as promising competitive technologies. The inquiry also noted that Life Cycle Assessments, or similar cradle to grave studies, are lacking for the Swedish case in order to make fair comparison. Incineration of sewage sludge is not a common practice in Sweden, meaning that significant investment would be required in logistics and technology. The inquiry highlighted concerns that a sludge handling policy built around large-scale incineration that solves the sludge problem in the short term could lead to several potentially negative outcomes in the long term. Examples of such potential effects include e.g. the risk for technological lock-in effects. This could hamper innovation in the area of waste water treatment methods and slow the progress towards a more efficient use of waste water as a resource that considers values beyond recovery of P and other nutrients like N and K. The inquiry also noted risks that large scale incineration could reduce drivers for continued work with source control and biogas production (Ministry of the Environment of Sweden, 2020).
- This inquiry also stated that “Evidence for a total ban being necessary is lacking, however, research having failed to prove that crops grown with sludge have health impacts or have an adverse impact on ecosystems in agriculture. On the other hand, there is clear evidence that sludge fertiliser application supplies plant nutrients and humus that agriculture demands.”
- Based on the results of this inquiry and the remit opinions to the inquiry, the Swedish government is currently assessing two options, the complete ban of sewage sludge use in agriculture or the implementation of stricter quality requirements, with reviews conducted every five years. Both opportunities include the promotion of incineration with P recovery and a target of 60 % of P recovery in UWWTP servicing agglomerations over a certain size.

The key provisions of current national legislation are presented here, with references to relevant legislation included in brackets:

- Based on the EU Directive on the Use of Sewage Sludge in Agriculture (86/278/EEC), the use of sewage sludge is prohibited on lands used for feed and for the cultivation of crops that are in direct contact with soil and normally consumed raw (i.e. berries, potatoes, root vegetables, vegetables, fruit). In addition, these utilisations are prohibited for ten months after the spread of sewage sludge (1).
- On soils containing up to 4.0 mg of P/100 g of dry soil, 35 kg of P/hectare/year can be spread but the total amount of P spread cannot exceed 250 kg/hectare. On soils containing more than 4.1 mg of P/100 g of dry soil, 22 kg/hectare/year can be spread but the total amount of P spread cannot exceed 160 kg/hectare. The total amount of N cannot exceed 150 kg/hectare and can be spread all at once or in multiple occasions (1).
- Stricter limits than the ones set by the EU for metals contained in sewage sludge used in agriculture are in place (1).
- Further provisions for emissions of hazardous substances, including some found in sewage sludge, for collection, storage, recycling and disposal of hazardous waste including

Legislation regulating sewage sludge in Sweden

1. *Statens naturvårdsverks författningssamling 1994:2 (1994, as amended 2001)*
2. *Statens författningssamling 1998:899 (1998, as amended 2020)*



sewage sludge and for protection against harm to human health are found in Swedish legislation (2).

- Sludge use can also be regulated within the environmental/mitting process under the “General consideration rules” under the Swedish environmental code chapter 2 (1998:808) and the Swedish rules for environmental assessments (2013:251). (Government of Sweden, 2018)

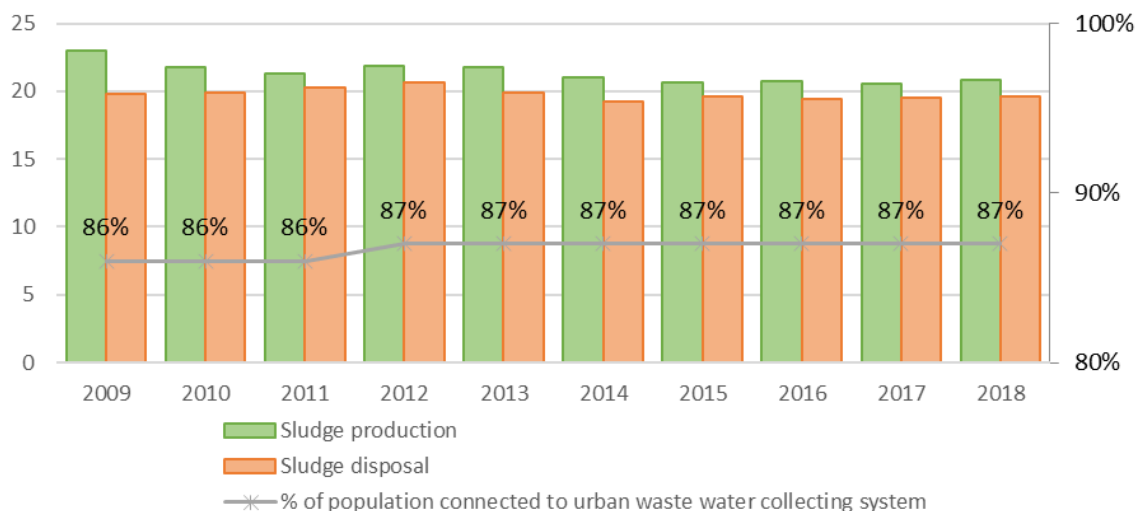
Current sewage sludge levels and management

Sewage sludge production

In 2018, 20.8 t of dry sewage sludge / thousand inhabitants were produced, 19.6 t / thousand inhabitants were treated and 87% of the population was connected to the urban waste water collecting system in Sweden (see Figure 4-9). The difference between the quantities produced and disposed can be explained by the fact that a share of the sewage sludge produced is stored which introduces a lag in the yearly comparison of the reported amounts. It is also noteworthy that a certain biological degradation occurs during the storage. (Statistics Sweden, 2018).

Figure 4-9 below shows that between 2009 and 2018, the quantity of sewage sludge produced/ thousand inhabitants has decreased by 9.2 % and the quantity disposed has remained stable while the percentage of population connected to urban waste water collecting systems remained stable.

Figure 4-9 Annual quantities of sewage sludge produced and disposed in Sweden (in t/thousand habitants) and share of population connected to urban waste water collection systems



Sources: (Eurostat, 2020a) and (Eurostat, 2020c)

Sewage sludge contamination

The *Statens naturvårdsverks författningssamling 1994:2 (SNFS 1994:2)* of 2014 set stricter limits for the metals contained in sewage sludge used on agricultural soils compared to the SSD and announced a first decrease in these levels for 1995, and another much greater one for 2000. However, the amendment of 1998 adjusted these limits to make them more achievable within the timeframe. Table 4-5 below presents the EU and Swedish limits for the direct application of sewage sludge in agriculture and the average concentrations of metals and nutrients in sewage sludge from municipal UWWTP in 2016. This shows that the average levels in sewage sludge are well below the limit values, and that the initial limits set for the year 2000 would have been met except for the concentration of copper and cadmium. In Sweden no limit values are set for substances other than metals.



Table 4-5 EU and Swedish limits for metals contained in sewage sludge used for agricultural purposes and metals and nutrients concentrations in Swedish sewage sludge (in mg/kg of dry solids)

Limits set in EU and Swedish legislation										
		Cd	Cr	Cu	Hg	Ni	Pb	Zn	P	N
Directive 86/278/EEC		20 - 40	NA	1 000 - 1 750	16 - 25	300 - 400	750 - 1 200	2 500 - 4 000	NA	NA
SNFS 1994:2	1994	4	100	600	5	50	200	800	NA	NA
	1995	1.75	100	600	2.5	50	100	800	NA	NA
	2000	0.75	40	300	1.5	25	25	600	NA	NA
SNFS 1998:4		2	100	600	2.5	50	100	800	NA	NA

Sources: (Hudcová, et al., 2019), (Swedish EPA, 2001) and (Statistics Sweden, 2018)

Average concentration in sewage sludge in 2016										
		Cd	Cr	Cu	Hg	Ni	Pb	Zn	P	N
Sweden		0.8	23.1	342.8	0.5	16.5	16.1	572	26 857	45 330

Source: (Statistics Sweden, 2018)

Sewage sludge management

Figure 4-10 Sewage sludge management practices in 2018 (in % of total sludge disposed)

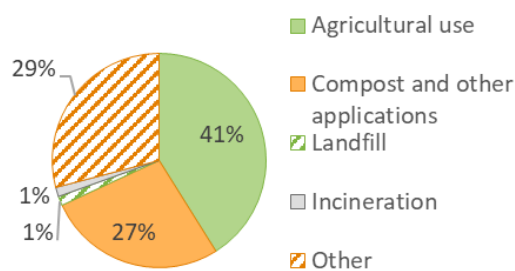


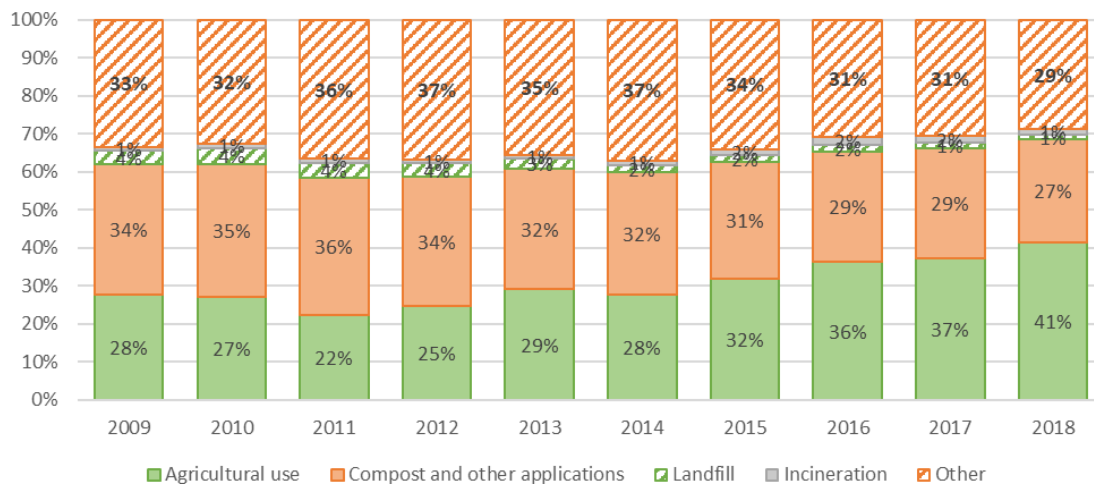
Figure 4-10 shows that in 2018, 41 % of the sewage sludge disposed in Sweden was used in agriculture, 29 % was disposed for other purposes and 27% was used for compost and other applications. The “Other” category refers to the use of sewage sludge to form a layer in order to cover landfill (Statistics Sweden, 2018). The split between agricultural use, composting and landfill covering can be explained by the historical questioning around sewage sludge management practices in Sweden.

Source: (Eurostat, 2020a)

Figure 4-11 shows that while the quantities of sewage sludge landfilled and incinerated have been relatively stable in the past ten years, the quantity of sewage sludge used for composting has been decreasing since 2011 to reach its lowest level in 2018 (27 %). The agricultural use significantly dropped in 2011 (22 %) in favour of composting (36 %) and landfill covering (36 %) before increasing significantly to attain in 2018 its highest level since 2008 (41 %). This regain in confidence towards the use of sewage sludge in agriculture can be explained by yearly improvements in the sludge quality due to successful source control as well as an overall increased trust between sludge producers and users that has developed via the cooperation enabled through the REVAQ network. REVAQ network’s work includes working with industries to help them decrease the use of specific compounds that were discharged to waste water. Also, when new activities start in certified UWWTPs’ area, the UWWTPs have the right to express their opinion in order to avoid the establishment of activities that would pollute the waste water (IEA Bioenergy Task 37, 2015).



Figure 4-11 Annual sewage sludge management practices (in % of total sludge disposed)



Source: (Eurostat, 2020a)

Phosphorus recovery

While some P is recovered through agricultural use, the actual value extraction of the P in the non-agricultural disposal paths (composting and other applications) is difficult to evaluate without further detailed assessment. No P recovery currently takes place after incineration.

Impacts of the sewage sludge management method(s)

Costs of sewage sludge management

Costs related to sludge treatment are not collected on a national level and vary considerably between regions. Proximity to and local supply/demand situation for different end uses of the sludge as well as political ambition seem to be most important driving forces for costs associated with end use of the sludge and associated/required treatment steps within the UWWTP. An estimate of UWWTP's costs for different sludge end use cases was found to range between €20 and €110/t of dewatered sludge (Ministry of the Environment of Sweden, 2020).

Lessons learnt

- Decades of debate around health and safety risks related to the use of sewage sludge in agriculture have led the Swedish government to consider strong actions such as the total ban of sewage sludge use in agriculture.
- The growing importance of resource efficiency (e.g. nutrient recovery) and the Circular Economy (e.g. finding solutions to recycle and recover waste instead of disposing of it on landfills) has led to a shift in policy direction and decisions being taken to support the search for alternative sewage sludge disposal management methods.
- Recently, increased amounts of sewage sludge have been used in agriculture owing to greater confidence in sludge quality. This can be explained by successful source control of pollutants and increased trust between sludge producers and users developed via cooperation in the sludge assurance scheme REVAQ.
- The incineration of sewage sludge with P recovery technologies reflects the existing policy ambition. However, the lack of existing infrastructure for mono-incineration in Sweden would mean a strong commitment and significant investment is required if an incineration-based policy is to be achieved. Also, there is a risk that large scale



incineration would reduce drivers for continued work with source control and biogas production.

- Alternatively, Sweden is considering putting in place further restrictions and/or quality requirements for the application of sewage sludge on soil. These actions could be carried on with the help of national sludge quality certifications and be conducted while promoting sewage sludge incineration with nutrient recovery technologies.



4.5 Horizontal assessment

The country case studies have demonstrated that different approaches to sewage sludge management are applied across Europe, with different levels of nutrient and energy recovery. Table 4-6 compares the findings for all examined countries, underlining country-specific opportunities and threats.

Regulatory context

As required by the SSD, all countries have set limits for metals in sludge used in agriculture. In Germany and Sweden, limits have also been set for N and P, whereas Italy regulates further pollutants including As, PCB, PAH, PCDD/F and Toluene. The regional governance of Italy means that some regions have also set limits for additional pollutants. In Lombardy and Emilia-Romagna, limits have been set for NP/NPE, AOX and DEHP, and Emilia-Romagna has set limits for Linear alkylbenzene sulfonate (LAS). Most countries have set their metal limits for agricultural use either at the lowest levels of the SSD or at even lower levels. Limits are also set for metals concentrations in soils.

Regulatory gaps and landlocks were identified in some regions of Italy. In Lombardy, the growing public concern regarding the environmental and health impacts of agricultural sludge spreading has led to exploration for alternative disposal methods. However, the region has limited capacity for additional incineration, and is opposed to increasing this capacity which leads to difficulties with managing sludge.

Predominant sludge management method

The case studies found that the two key approaches to final sewage sludge disposal are agricultural use and incineration. Sludge is also used for landscaping (e.g. in Estonia). AD is often used to reduce volume of sludge and produce biogas. With respect to sludge used in agriculture, there are three key options – direct application of treated sludge, composting followed by application to land, and production of “sludge gypsum”. Agricultural use is common in Sweden and Italy, though in Italy the direct application of treated sludge is in decline. Public concern is driving a shift away from agricultural use towards alternative management approaches that ensure nutrient recovery, such as P recovery from incinerating sewage sludge ash. Germany is a pioneer in the P recovery technologies; however these are not fully commercially developed yet and therefore their widespread implementation is still emergent. In addition, they require significant investments, and also have the potential to destroy the organic matter and N compounds which could be useful for soil. Sweden is currently considering rolling out similar approaches to strike a balance between circular economy ideals and public concern over soil contamination..

Some regions of Germany and Italy which are more densely populated, and generate more sludge, also have potential issues with land availability for sludge recovery. Agricultural use tends to be local or regional to the generation of the sludge, which can cause these difficulties but raises the possibility of exporting to areas where sludge use in agriculture would be possible and beneficial. Nevertheless, the transport costs of sludge may be high compared to the value of sludge as a fertiliser.

Impacts from sludge management

The measurements of metal concentrations in soils which received sludge show that these are significantly lower than the set limits. Despite this, most countries studied perceived the application of sludge to agricultural land negatively, which includes spreading of treated sludge directly and composting. This is mainly driven by health and environmental concerns and social perception of use of waste for food production. Odours are also a driving factor.



Lessons learnt

The key lessons learnt from the case studies include:

- Upstream approaches to reduce sewage sludge contamination with the aim to enable use of sludge on land do not play a big role in national strategies. The only exception is Sweden where the confidence in the use of sewage sludge in agriculture can be explained by yearly improvements in the sludge quality due to successful source control. Limit values for metals seem to have worked in driving metal contamination down.
- Public concern regarding the use of sludge in agriculture is driving policy away from agricultural land spreading in many regions. This is in line with stakeholder feedback received in the 2014 SSD evaluation which indicated that public perception was a key challenge (EC, 2014). The concern is not always in line with findings on the quality of the sludge but could be explained by concerns on emerging contaminants.
- The P recovery from sewage sludge ash technologies have a ground-breaking potential, striking a balance between environmental and health concerns and circular economy principles. Europe is dependent on P imports and therefore the recovery of P on larger scales, once the technologies are fully developed, is strategically attractive. However, incineration leads to the loss of carbon and N and is linked to air emissions. Major pollutants emitted are PM, metals, CO, NO_x, SO₂, and unburned hydrocarbons (EMEP and EEA, 2019). Currently these technologies are also too expensive to be widely applied.
- The case studies have demonstrated the need of coordinated policy on sewage sludge management to avoid situations of regulatory landlock which may inhibit the disposal of sludge in a sustainable way.
- The reporting on sewage sludge could be improved. Current Eurostat reporting requirements mix sewage sludge treatments (composting and other applications) with sewage sludge final destinations (incineration, agricultural use, landfill). Also, significant amounts of sewage sludge disposal are reported as 'other', with no further information on what this entails. This creates gaps in our understanding of how sludge is ultimately disposed of across Europe. Greater clarity on the reporting categories for sewage sludge disposal, splitting the 'other' category into useful and clearer categories, as well as requirements for greater detail on pre-treatment practices employed (e.g. AD), would bring them much more in line with Circular Economy aspirations.

Table 4-6 Summary of sewage sludge management practices in all case study countries

Member State	Governance	Stricter limits than SSD	Contaminants other than metals regulated	Sewage sludge management practices	Nutrient recovery	Energy recovery	Costs	Public perception of agricultural use	Opportunities	Risks
Estonia	Central	x	x	Composting, landscaping and 'recultivation'	x (minor reclamations through landscaping)	<i>Unclear</i>	€	Neutral	Alignment of current practices with environmental principles.	There are no clear rules for treatment prior to landscaping and 'recultivation';
Germany	Regional but central control of management	✓	✓ (P and N only)	Agriculture, compost, mono-incineration (some P recovery)	✓	✓	€€	Negative	Potential to decrease dependency on P imports.	P recovery from ash is limited by technological development.
Italy	Regional	x (Regional level only)	✓✓	Land spreading, Composting, Use as fuel	✓✓	✓✓ (low)	€	Negative in some regions	Introduction of a more consistent approach	Fate of all sludge is unclear, regulatory landlock in some regions.
Sweden	Central	✓	✓ (P and N only)	Agriculture, compost and landscaping, landfill covering	✓	✓	€	Some environmental NGOs and farmers associations are negative	Consider P recovery from incineration ash as an alternative to land spreading. The REVAQ certification system with strong control has increased use among farmers.	P recovery from ash is limited by technological development;

Legend: ✓ - Yes x – No; €- low; €€-medium-high; *Unclear* - No information was identified with respect to this Member State, however, the relevance of this indicator is possible



5. Upstream measures to avoid contamination of sewage sludge

Contamination of sewage sludge may prevent circular economy related opportunities for it, such as a safe use on land or in fertilising products. Depending on the substance, treatment of sewage sludge can decrease the levels of contaminants in sludge yet usually not eliminate it completely. Spreading contaminated sludge or a fertiliser product derived from sludge can pose risks to the environment as well as human health.

A traditional policy approach to limit the risks related to contamination of sludge is setting quality standards or limit values content in sludge intended for land application, or the limits for a substance content in soil following sludge application. However, setting such limit values requires robust monitoring data and usually involves a risk assessment for impacts on human health and the environment. The SSD sets limit values for metals but has not acted on other contaminants present in European sludge, including a broad range of organic chemicals. In recent years, more attention in Europe and across the world has been paid to the presence in water and in the environment of contaminants of emerging concern (e.g. pharmaceuticals, personal care products, persistent organic pollutants). Some of these micropollutants may already be present in sludge if, in the process of waste water treatment, they sorb onto sludge particles. Measures aimed at further removal of such contaminants from waste water carry a risk of further accumulation of these pollutants in sludge. Setting limit values for all organic substances and contaminants of emerging concern likely to be present in sludge would require extensive monitoring of sludge composition across Europe. While it is technically possible to set up such monitoring regimes, such an approach would require extensive investment.

An alternative approach for managing risks related to use, treatment or disposal of contaminated sewage sludge is to prevent the contaminants from entering the waste water in the first place (acting “upstream” of the UWWTP). This approach is more in line with the polluter pays principle than the currently dominant end-of-pipe approach, that largely leaves contamination to be handled by waste water operators. This part of the report explores the feasibility of upstream measures to prevent the contamination of waste water, and subsequently sewage sludge, for two contaminant substances: di(2-ethylhexyl) phthalate (DEHP) and benzo(a)pyrene (BaP). The rationale for selection of these substances was described in detail in section 2.4. Both substances are well-studied water contaminants, already regulated under European legislation. In addition, they originate in waste water from different types of sources: DEHP enters waste water primarily from point sources (i.e. industrial facilities and households) while BaP originates from diffuse sources (i.e. deposition from air on to the ground and from run-off). As opposed to some of the contaminants of emerging concern, there already exists evidence on DEHP and BaP presence in water and sludge, as well as on potential upstream removal options. They have therefore been considered as suitable candidate substances to illustrate an approach to evaluate the feasibility of upstream options to reduce contamination of sewage sludge.

This chapter presents the assessment of each substance in turn. The selection of potential options for upstream reduction of the substances is driven by assessment of the information available on the sources of their release to waste water. The investigations were then focussed on measures which could act on the most significant of these sources.

Based on these assessments, conclusions are then drawn in section 6 on the key principles for the methodology to assess removal of sludge contaminants “upstream”. Subject to data availability, such principles could be applied to assessments for any contaminant, including contaminants of emerging concern.



5.1 Di(2-ethylhexyl) phthalate (DEHP)

5.1.1 Introduction to the case study

DEHP is a manufactured chemical that has been used as a plasticiser in manufacturing of polymers. Release of DEHP into waste water occurs primarily from use of polymer-based products (ECHA, 2020d). In waste water treatment works, DEHP is mostly removed from waste water and transferred into sewage sludge thus leading to its contamination.

This case study brings together literature evidence on the pathways and presence of DEHP in European sewage sludge and the associated risks to the environment and human health. It includes information on potential upstream measures to prevent DEHP from initially entering the waste water, thereby eliminating its presence in sludge. The case study is structured as follows:

1. Description of the substance and its use – provides information on general characteristics of DEHP as well its current use and trade in Europe.
2. Current regulation – summarises EU legislation applicable to the production of DEHP, use of DEHP by industry, as well as in manufacturing of products and articles. It also captures regulation applicable to limiting impacts at product end-of-life as well as limiting DEHP emissions to waste water and its presence in sludge.
3. Sources and pathways to the environment – describes the sources of DEHP to waste water and sewage sludge, and its pathway to the environment. It also provides an illustrative mass balance for DEHP from source (i.e. product use) through waste water to sewage sludge.
4. Options for upstream reduction of contamination:
 - i. describes the options available to prevent DEHP from entering the waste water (referred to in this study as “upstream” measures)
 - ii. discusses how the upstream options compare to each other in terms of environmental and economic performance
 - iii. discusses what trade-offs need to be considered when deciding on possible further “upstream” policy interventions.
5. Conclusions – draws conclusions from the case study.

5.1.2 Description of the substance and its use

DEHP is entirely synthetic and is not naturally present in the environment. DEHP has been most commonly used in manufacturing of poly vinyl chloride (PVC) (ENTEC UK Limited, 2011). Release of DEHP into the environment occurs mainly from both outdoor and indoor use of polymer-based products (ECHA, 2020d). These involve construction and building materials, flooring, and every day consumer items such as furniture, toys, curtains, footwear, leather products, paper and cardboard products, electronic equipment. DEHP is officially recognised in the EU as toxic to reproduction and as an endocrine disruptive compound (ECHA, 2020d) (the latter meaning it is a substance which can adversely interfere with endocrine or hormonal systems).

No clear information could be found on the current levels of production of DEHP in Europe. One source (DEZA A.S., n.d.)³³, stated that between 70 000 and 120 000 t of DEHP are placed on the EU market, with four to six active suppliers. However, this information could not be verified with other literature sources. Table 5-1 shows that currently in the EU-27, 14 companies have an active registration for the manufacture and/or import of DEHP, of which four are based in Germany, three in Czech Republic and two in Belgium. This table also shows that seven

³³ <https://www.echa.europa.eu/documents/10162/d075f650-7914-47e9-b727-9ef73c28fccf>



companies have ceased manufacturing of DEHP including one in 2013, one in 2014, three in 2015 and two in 2017.

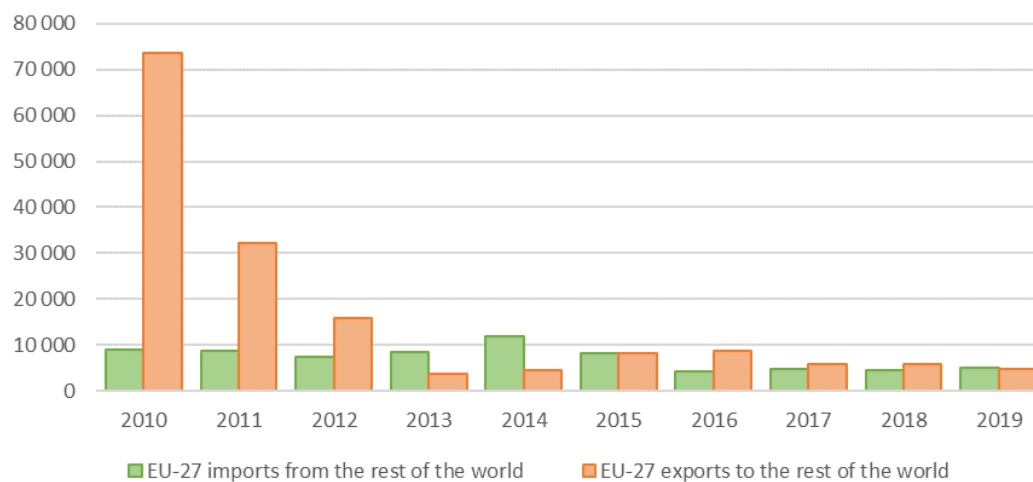
Table 5-1 Number of active registrants/suppliers and ceased manufacturing of DEHP in the EU-27 in 2020

Country	No. of active registrants/suppliers	Country	No. of inactive registrants/suppliers
Belgium	2	Belgium	2
Czech Republic	3	France	2
Germany	4	Poland	1
Italy	1	Romania	1
Luxembourg	1	Sweden	1
the Netherlands	1		
Poland	1		
Spain	1		

Source: (ECHA, 2020d)

While the information on trade in DEHP (i.e. substance, rather than products DEHP is used in) is also not available, Eurostat provides information on the trade in dioctyl ortho-phthalates (a family of chemical substances to which DEHP belongs). Figure 5-1 shows that the imports and exports of dioctyl ortho-phthalates between the EU-27 and the rest of the world have decreased significantly in the past 10 years, by 43 % and 93 % respectively. In 2019 the imports and exports of the substances were almost equal (a total of 5 110 thousand t of dioctyl ortho-phthalates was imported and 4 920 thousand t were exported).

Figure 5-1 Imports and exports of dioctyl ortho-phthalates between EU-27 and the rest of the world in the past 10 years (in thousand t of DEHP)



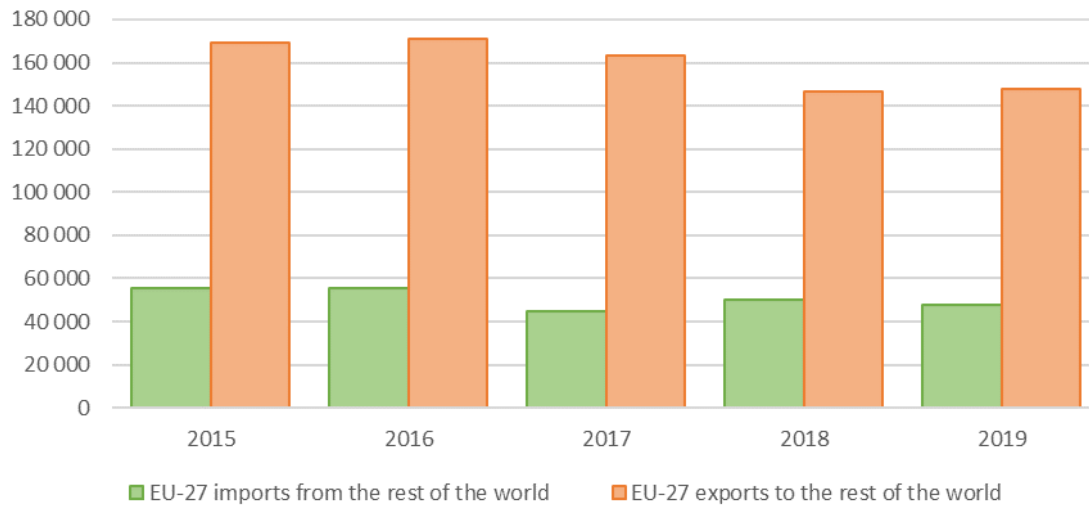
Source: (Eurostat, 2020d)

No information has been identified on the exact volumes of DEHP currently used in production of PVC. Nevertheless data published by the European Plasticisers (2017) shows that the use of low-molecular-weight ortho-phthalates as plasticiser in Europe has decreased from around 30 % of all plasticizer use in 2005 to around 10 % in 2017.

Figure 5-2 shows that the imports and exports of PVC between the EU-27 and the rest of the world have decreased in the past five years, by 14 % and 13 % respectively, with a total of 475 179 thousand t of PVC imported and 1 477 746 thousand t of PVC exported in 2019. Europe remains therefore a net exporter of PVC.



Figure 5-2 PVC imports and exports between the EU-27 and the rest of the world in the past 5 years (in thousand t of PVC)



Source: (Eurostat, 2020d)

Based on the available data it is impossible to conclude on the current level of use of DEHP in Europe, however the following is observed:

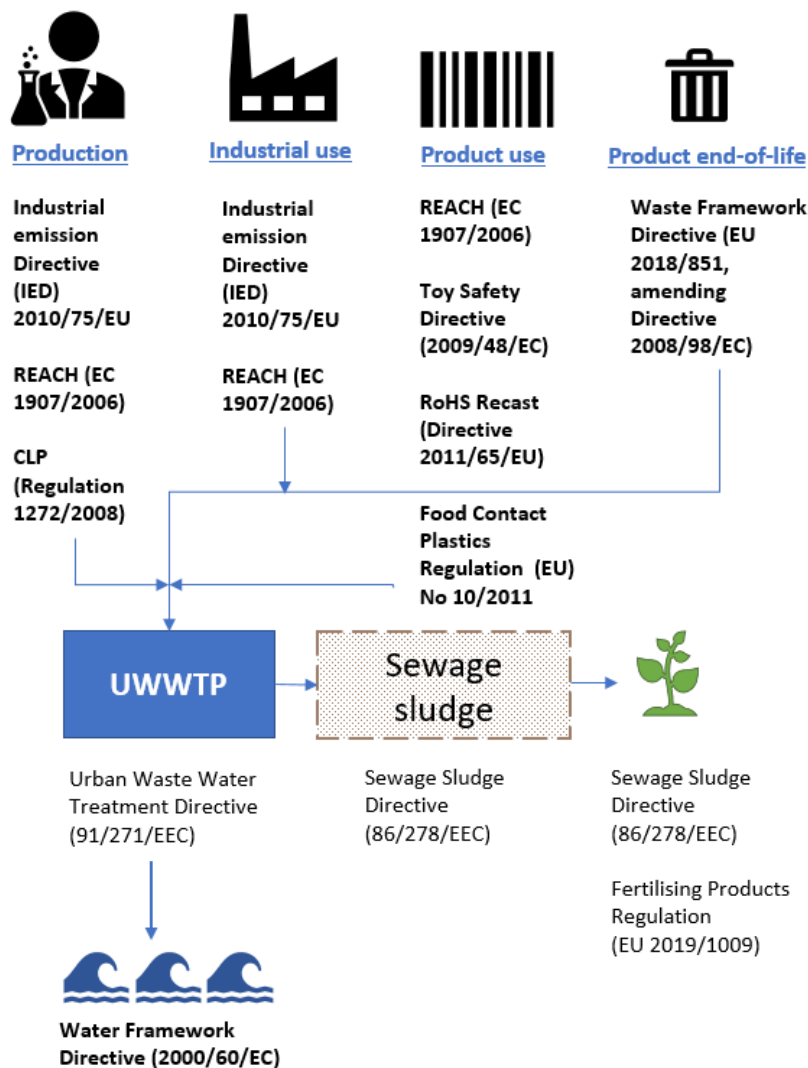
- DEHP continues to be used in Europe.
- Europe's demand for DEHP is either met by EU-based production or imports from outside of Europe
- Europe is a net exporter of PVC, however it also imports PVC from the rest of the world.
- While the use of DEHP as plasticizer has declined in Europe in the last five years, it is unclear if the PVC imported to Europe has been manufactured using DEHP or other plasticizers. Imported PVC and PVC articles and products may therefore continue to be a source of DEHP emissions in Europe.

5.1.3 Current European regulation applicable to the use and emissions of DEHP

Current EU policy aims to limit the use of DEHP as well as its emissions to the environment. Canada, the US and Japan have also reviewed the safety of DEHP in products and decided legislative measures to control its presence and use (No Harm, 2019). The Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH) presents a regulatory framework in the EU currently impacting the production of DEHP, its use in products and fate at product end-of-life, as well as removals at UWWTP and treatment and application of sludge (ECHA, 2020b).



Figure 5-3 Legislation impacting use and emissions of DEHP in Europe



Source: Own elaboration

Notes: Legislation in bold includes specific requirements for DEHP. Other legislation may have indirect impacts on DEHP releases.

Production of DEHP and industrial uses

Regulation 1272/2008 on the Classification, labelling and packaging of substances and mixtures classifies DEHP to hazard class “Toxic to reproduction, Category 1B” (i.e. may damage fertility and may damage the unborn child) and requires appropriate labelling by manufacturers. Releases of DEHP to waste water from production and industrial use are controlled through application of Best Available Techniques (BAT) in chemical industry installations under the IED (2010/75/EU).

Use of DEHP in products

Along with certain other phthalates, use of DEHP is regulated under REACH. In 2018, the European Union (EU) adopted a decision to restrict use of phthalates under Annex XVII of REACH. Entry 51 (ECHA, 2020b) applies to DEHP, Dibutyl phthalate (DBP), Benzyl butyl phthalate (BBP) and Diisobutyl phthalate (DIBP). As of 7 July 2020, the following uses, individually or in any combination, in a concentration equal to or greater than 0.1 % by weight of the plasticised material are restricted:



1. Use as substances or in mixtures in toys and childcare articles
2. Placing toys and childcare articles containing the four phthalates on the market
3. Placing articles (other than toys and childcare articles) on the market. The restriction does not apply to articles placed on the market before 7 July 2020 and several uses are exempt, including articles exclusively for industrial or agricultural use, aircrafts, motor vehicles, products covered by other dedicated legislation (see below).

The definition of placing on the market under REACH in practice means that companies in the EU cannot import articles containing DEHP and cannot manufacture articles containing DEHP. These restrictions target use of DEHP in articles and products. The uses of DEHP e.g. in industrial or agricultural applications are permitted subject to Authorisation being granted by the EC for a specific use. The list of Applications for Authorisations under REACH either decided on or pending a decision, concern formulation of DEHP in compounds, industrial use in polymer processing, production of certain electrical components and use of recycled PVC³⁴.

Recycling of PVC containing DEHP has been approved in the past (No Harm, 2019) and there are currently two applications for authorisations concerning recycled PVC awaiting decision from the Commission. One application for authorisation is for recycling between 10 to 100 t/year of PVC containing around 5% DEHP by weight (ECHA, 2020a). The second application concerns the use of the same tonnage of recycled PVC containing DEHP for manufacturing of PVC articles (used outdoors e.g. in construction, civil engineering, garden features, indoors in industrial and agricultural workspaces; footwear used in these workplaces) (ECHA, 2020a). Both applications are sought for the period of 12 years. Based on the data in the authorisation applications, annually the processes would re-introduce into market PVC and PVC manufactured articles containing between 0.5 to 5 t of DEHP.

Since 2008, DEHP has been on the Candidate List of substances of very high concern (SVHC) for Authorisation, Article 33 of REACH requires suppliers of articles containing such a SVHC in concentrations $\geq 0.1\%$ w/w to provide recipients with sufficient information to ensure safe use. There is also a requirement to notify SVHCs in articles to the European Chemicals Agency (ECHA) where it is present in concentrations $\geq 0.1\%$ w/w and in quantities greater than 1 t/producer or importer per year (Article 7(2) REACH).

The following EU legislative acts restrict DEHP uses in specific products:

- Directive 2009/48/EC³⁵ prohibits use of phthalates in toys.
- Directive 2011/65/EU³⁶ restricts DEHP use in electrical and electronic equipment (EEE).
- Regulation (EU) 10/2011³⁷ allows the use of DEHP as additives or polymer production aids and as plasticisers in repeated use food contact material (FCM) for non-fatty foods. This Regulation refers to plastic materials and articles intended to come into contact with food and stipulates limits of $\leq 0.1\%$ for the technical support agent and ≤ 1.5 mg/kg for the specific migration limit.

Regulation (EC) No 1223/2009 on cosmetic products prohibits the use of DEHP. According to ECHA (2014), despite a ban on use of phthalates in toys, 20% of inspected toys contained DEHP. Most were the products of unknown origin or imported from China.

³⁴ Opinions on Authorisations are formulated and adopted by the Committee for Socio-Economic Analysis (SEAC) and the Committee for Risk Assessment (RAC). These form recommendations for the Commission on whether or not the Authorisation should be granted. The European Commission makes a decision on whether to grant an authorisation. Whilst an application for authorisation is “pending decision” the substance can continue to be used

³⁵ <https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32009L0048>

³⁶ <https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32011L0065>

³⁷ <https://eur-lex.europa.eu/legal-content/EN/ALL/?uri=CELEX%3A32011R0010>



Product end-of-life

Directive (EU) 2018/851, amending 2008/98/EC on waste requires ECHA to establish a database for information on substances of concern in articles and in complex objects (products) (SCIP database³⁸). Suppliers³⁹ of articles containing SVHCs which are on the Candidate List (thus including DEHP), have to submit information on these articles to ECHA from 5 January 2021. The database was launched on 28 October 2020 and is expected to be publicly available to waste operators and consumers in Q1 2021. It will provide information on any articles or products containing DEHP when placed on the EU market.

DEHP emissions to water

DEHP is a priority hazardous substance (PHS) under the Water Framework Directive (as stated in Directive 2013/39/EU⁴⁰). This sets out an objective for the discharges, emissions and losses of DEHP to water to be ceased entirely. The UWWTD does not directly impact removal of DEHP from waste water, however the Environmental Quality Standard set for DEHP in inland surface waters (1.3 µg/l) was a driver for reductions in emissions of the substance to water bodies.

DEHP content in sludge

There are no EU limits for DEHP in sewage sludge used in agriculture. The SSD (86/278/EEC) sets requirements for the use of sludge in agriculture but sets limits only for selected metals. In the early 2000s, a limit for DEHP in sludge of 100 mg/kg of dry weight was considered by the European Commission under the SSD but eventually the Directive was not revised. Denmark has determined a limit for DEHP in sludge used in agriculture of 50 mg/kg of dry weight (Norwegian Water, 2018) (Collivignarelli, et al., 2019) (Hudcová, et al., 2019). The EU Fertilising Products Regulation (EU/2019/1009) prohibits sewage sludge to be part of compost and digestate used in fertiliser products placed on EU market. While the Regulation sets limit values for some substances, it does not include specific requirements on DEHP content even if DEHP is used in food contact materials.

5.1.4 Sources and pathways of DEHP to waste water and the environment via sludge route

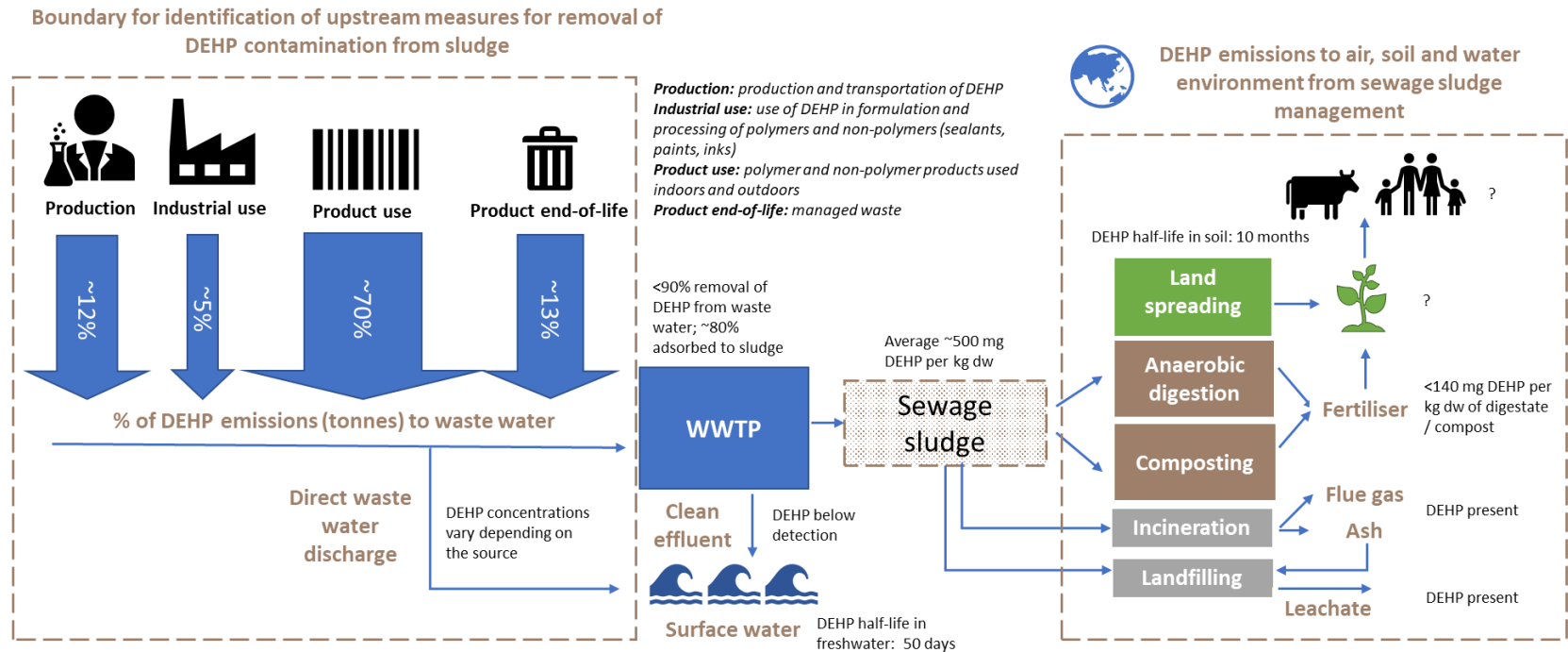
Figure 5-4 presents a simplified schematic of sources of DEHP emissions to waste water and its pathway to the environment via a sewage sludge route. Each step of this pathway is described below starting from the origins of DEHP in waste water, through the UWWTP, sludge and eventually the environment.

³⁸ <https://echa.europa.eu/scip>

³⁹ EU producers and assemblers, EU importers, EU distributors of articles and other actors who place such articles on the EU market

⁴⁰ <https://eur-lex.europa.eu/legal-content/EN/ALL/?uri=CELEX:32013L0039>

Figure 5-4 Illustration of DEHP pathways to the environment and its sources in waste water in Europe



Source: Own elaboration based on information in: (ENTEC UK Limited, 2011), (Wood, 2019), (JRC, 2012), (ECHA, 2020c), Comber et al. (2015)

Notes: dw = dry weight. Waste water from production and industrial use is likely to be released to the environment following treatment in the on-site waste water treatment plants (it is assumed that it does not enter the urban waste water collection system). However, the share of waste water being released directly to the environment versus released to urban waste water collection system is unknown. All emissions of DEHP from product use are assumed to be entering urban waste water collection systems (either with household water or through run-off).

Half-life is the time it takes for an amount of DEHP to be reduced by half through degradation.

Average concentration of ~500mg per kg dw of DEHP in sewage sludge has been calculated from European samples referenced in Table 5.3 (JRC, 2012).



Sources of DEHP emissions to waste water

Emissions of DEHP to waste water can occur from: production of DEHP in a chemical plant and its transportation for further use; its industrial use in production of polymer and non-polymer materials and subsequently polymer-based or other products; product use and product end of life (ENTEC UK Limited, 2011).

1. Production and industrial use of DEHP

As estimated in Figure 5-4, emissions from production of DEHP represent around 12 % of all DEHP emissions to waste water and emissions associated with industrial use of the substance for around 5 %.

The European Pollutant Release and Transfer Register (E-PRTR) shows reported emissions and transfers of contaminants from industrial facilities above certain size (i.e. capacity thresholds are set for individual activities). Emissions of DEHP have to be reported to the E-PRTR if a facility emits in excess of 1 kg of DEHP per annum. Only UWWTP with a capacity greater than 100 000 p.e. are required to report to the E-PRTR.

E-PRTR⁴¹ records that in 2017, six chemical industry facilities and in 2018 two facilities, transferred DEHP containing waste water for treatment in UWWTPs (excluding reporting by the UK facilities). In 2017 four, and in 2018 three, chemical industry facilities reported direct releases of DEHP to water. In those two years, transfers of DEHP with waste water to UWWTP also occurred from industrial facilities producing paper and board, landfills, incineration plants, surface treating metals, recovering hazardous wastes, pre-treating or dyeing of fibres or textiles. Total reported transfers of DEHP from these facilities in 2017 were equal to 140 kg and in 2018 to over 5.8 t (the large increase between 2017 and 2018 transfers in DEHP is associated with over 5.5 t of DEHP reported by a single plant producing surface-active agents and surfactants; if this number is excluded the total transfers of DEHP amount to 335 kg).

Industrial installations producing or using DEHP in the formulation and processing of polymers and non-polymers are most likely to have their own on-site treatment plants for treatment of industrial waste water, and as such they would not be discharging waste water to urban waste water collection systems. Instead they would be releasing treated waste water directly to the environment. In 2017, just over 8 t and in 2018 around 3.4 t of DEHP were released from industrial facilities in Europe to the water environment according to the data reported to E-PRTR (excluding emissions reported by facilities in the UK). In those two years, on average 97 % of DEHP releases to the water environment originated from UWWTPs. The remaining 3 % of DEHP was emitted from other industrial activities such as facilities for the recovery or disposal of hazardous waste, facilities for the incineration of non-hazardous waste, independently operated industrial waste-water treatment plants, installations for the disposal of non-hazardous waste, mineral oil and gas refineries, paper and wood production processing⁴². Based on the data in the E-PRTR, the scale of direct releases of DEHP to the environment from industrial production and use is of similar scale as that of DEHP transfers.

2. Product use

Product use is responsible for the largest share of DEHP emissions to waste water. Based on two sets of figures on source apportionment of emissions of DEHP to waste water provided in (ENTEC UK Limited, 2011) and Comber et al., (2015), the product use has been estimated as responsible for around 70 % of all emissions of DEHP to water:

- Entec (2011) states that the emissions of DEHP from product use are around 41% of all emissions of the substance to waste water. However, it also states that around 40% of emissions is associated with the end-of-life phase, of which the majority are from “particles/fragments abraded from end-use products during their service life and during

⁴¹ <https://www.eea.europa.eu/data-and-maps/data/industrial-reporting-under-the-industrial-1>

⁴² Other sectors discharged less than 10 kg of DEHP to water per year in either 2017 or 2018



disposal (e.g. particles abraded from car undercoating, coil coating, shoe soles and fragments of plastic bags etc.)⁴³. Those emissions may be interpreted by some to form part of a use phase rather than end-of life as they do not represent emissions from management of product waste as has been attempted in this case study. These figures are meant to be representative for Europe and originate from the European Union Risk Assessment report for DEHP (ECHA, 2008).

- Comber et al. (2015) reported average concentrations of DEHP from nine UWWTPs in the United Kingdom, apportioning them to domestic, run off, trade, light industry, and town centre flows. Trade⁴⁴ and domestic waste waters were found to have the highest concentrations of DEHP, in line with the evidence presented in Entec (2011). Concentrations of DEHP in domestic, runoff, and trade waste water jointly accounted for 67 % of DEHP concentrations in UWWTPs. Together with discharges from light industry they would account for 79 % of DEHP concentrations. On that basis, and to reflect the actual measured concentrations of DEHP in different types of waste water flows, the share of emissions associated with the product use have been estimated at around 70 % and from the end-of-life phase to around 13 %.

This estimate could not be validated with evidence found in the wider literature. Furthermore, the estimate is based on information in studies published nearly a decade ago. It may be that due to the regulatory restrictions put in place in the meantime, changes in production and product use patterns, the contribution of various sources of DEHP in waste water has changed. Nevertheless, emissions from the use of products are still expected to be a dominant source of DEHP emissions to waste water in Europe.

As stated above, DEHP is present in many plastics, especially PVC materials, which often contain around 30 % of DEHP by weight (ENTEK UK Limited, 2011). Up to date information on how much DEHP is used in different products is not available. Based on the evidence quoted in Entec (2011), 97 % of all DEHP used in products is used in polymer-based products. These are divided into products used indoors (78 % and include: sheets, film, wall- and roof covering, flooring, coatings and leather imitations, pastes for sealing and isolation, medical products) and outdoors (22 % roofing material, roofing (coil coating), cables, coated fabric, hoses and profiles, car undercoating, shoe soles).

As described in earlier sections, it can be expected that due to regulatory measures put in place in Europe in the last 10 years to limit the use of DEHP, its use in manufacturing of new products should have declined and be limited only to authorised uses. Nevertheless, emissions of DEHP to waste water continue from current use of products placed on the market and put in use pre-current restrictions. During product use, DEHP may slowly leach from the surface of the product when in contact with water (for example during washing or in wet weather). As such DEHP from the product use stage can enter the waste water with:

- Urban surface run-off: run-off from buildings, pavings and constructions
- Households waste water: losses from household pipes, polymer floors and other DEHP containing household items

With time, emissions of DEHP to waste water from product use should diminish as the products containing the substance reach the end of life. The technical lifetime of products containing DEHP differs from 1 year (printing inks) to 20-30 years (floors, sealants, adhesives, roofing materials, cables) (ECHA, 2008). Products with DEHP such as hoses and profiles, coated products and fabrics, film and sheets, lacquers and paints, shoe soles have

⁴³ No more detailed information could be found on different levels of abrasion or leaching from different types of product.

⁴⁴ It is unclear how DEHP enters waste water from trade activities as this pathway is not described in the underlying study. Overall DEHP leaches to water when a product comes into contact with water (e.g. during washing, wet weather).



an average lifetime of between 5 to 10 years. Assuming that as a result of the 2020 use restrictions under REACH no new products used by households will contain DEHP, the emissions of DEHP to domestic waste water should significantly decrease by 2050.

3. **Product end-of-life**

Emissions from **product end-of-life** occur from landfills and other waste management and end-of-life activities such as paper recycling and car shredding (ENTEC UK Limited, 2011). Overall, the product end-of-life emissions represent a smaller share of total DEHP emissions to waste water (following adjustment of figures in Entec (2011) estimated at around 13 %). According to the 2017 E-PRTR data, transfers of DEHP to UWWTPs from waste and waste water management facilities represented 17 % of DEHP transferred to waste water treatment plants. In 2018, the share was much smaller (around 4%⁴⁵), however, the 2018 dataset is still incomplete.

Fate of DEHP at waste water treatment plants

When waste water enters UWWTP, it is subject to several treatment processes which affect DEHP's fate. An understanding of DEHP behaviour within the treatment system is key to understanding its fate in terms of emissions or transport to the wider environment. DEHP has a high sorption potential ($\log K_{ow} = 5.37$) and a low volatilisation ($K_h = 3.52 \cdot 10^{-6}$) (Seriki, et al., 2008). This means that the majority of DEHP is expected to partition to sediment during treatment. In conventional activated sludge UWWTP, the main removal process for DEHP from waste water is sorption to primary and secondary sludges, and to a lower extent biodegradation. However, based on information identified in literature, behaviour of DEHP during waste water treatment varies depending on the exact treatment type deployed. As shown in Table 5-2, some studies have reported much greater share of DEHP being biodegraded. Overall DEHP is fully biodegradable in freshwater with half-life of 50 days (ECHA, 2020c). DEHP biodegrades to form monoethylhexyl phthalate (MEHP) which is moderately toxic.

Table 5-2 Reported removal of DEHP from waste water during treatment

Source	Total DEHP removal from waste water (%)	Share of DEHP biodegraded (%)	Share of DEHP removed with sludge (%)
(Marttinen et al. (2003).	97%	23%	65%
Clara et al. (2010)	<98%	<17%	78-81%
Roslev et al. (2007)	96%	81%	15%
Tan et al. (2007)	96%	74%	22%
Fausser et al. (2003)	98%	70%	28%

DEHP presence in sludge and behaviour during treatment

Several studies have highlighted that DEHP can occur in high concentrations in sludges, predominantly a result of its preference to partition to sediments during waste water treatment. Net et al. (2019) highlight that DEHP is a predominant phthalate in sludges and compost. Net et al. (2015) and (Salaudeen, et al., 2018), both note in their studies that of the phthalates observed in the sludge samples, DEHP could account for 58% of the total phthalate concentration.

Literature suggests that DEHP concentrations in sludge are widely variable, for example in one study across 16 treatment works, concentrations ranged from 0.63 mg/kg to 84 mg/kg (Gardner, et al., 2013). This was understood to arise from the wide variation in catchment usages, such as those which are heavily urbanised versus those which are agricultural, for each of the WWTWs.

⁴⁵ Calculated when 5 510 t of DEHP reported as transferred from one facility is excluded from the total.



This pattern is also reflected at the European level with ranges detected in sludge varying considerably as shown in Table 5-3.

Table 5-3 Reported DEHP concentrations in European sludge

Unit	Reported ranges
mg/kg	100
mg/kg	122.09 - 1 651.9
mg/kg	16.5
mg/kg	27 - 55
mg/kg	0.028 – 0.122
mg/kg dw	0.012 – 1.11
mg/kg dw	47.1 – 1,651.9
mg/kg dw	1.5 – 3 513.8

Source: Reproduced from Table 2 (JRC, 2012) “Organic pollutants range concentration comparison reported in the literature including the EU Working Document on Sludge”.

Notes: Average concentration across all samples was 506 mg/kg of dry weight.

DEHP in sludge is understood to have a half-life ranging from 30 to a few hundred days (Cartwright, et al., 2000) however storage conditions can alter this. For example, the half-lives of DEHP for lagooning sludge were 45.4 days whereas for activated sludge this was 28.9 days (Net, et al., 2019)

In terms of degradation of DEHP within sludge, evidence suggests that oxygen conditions available during treatment can influence the rate of biodegradation of DEHP. In the study by Armstrong et al. (2018) it was found that DEHP in sludge was readily degraded during aerobic treatments, whereas during anaerobic treatments, no significant change in concentrations was found. For advanced anaerobic processes an increase in concentration was noted in some cases. In a study for the European Commission it has been suggested that typical DEHP ranges within the EU in sewage sludge digestates range from 0.6 – 140 mg/kg as dry weight (Wood, 2019).

Under aerobic conditions, studies have shown that DEHP is ‘readily biodegradable’ (82% after 28 days) however in anaerobic conditions, it is resistant (ECHA, 2020c)

Risks to the environment and human health

Once sludge with DEHP is spread on land it is subject to degradation but can also accumulate in amended soils (SEPA, 2014). ECHA (2020d) reports DEHP half-time in soil as 10 months. A study by SEPA (2014) determined that after 10 years of repeated application of sludge to soil, DEHP half-life was up to 100 days. Pedersen et al. (2019) evaluated phthalates as posing a risk to soil living organisms in the month immediately after application of sewage sludge.

A study undertaken in Norway (Norwegian Water, 2018) quotes results of the risk assessment on human health for phthalates present in sewage sludge spread on soils. The tolerable human intake of DEHP is provided as 50 µg/kg body weight/day, while the calculated maximum intake through “the sludge” route in Norway was estimated at 10.5 µg/kg body weight /day. The risk assessment concluded that presence of phthalates in sludge were of a low risk to the environment and public health, even when the highest application rate of sludge on soils was assumed (i.e. 60 t of sludge/hectare over 10 years).

With regards to application to land of digestate or compost containing DEHP, Wood (2019) found that the main exposure to humans is through consumption of root vegetables. It also



concluded that a long-term gradual accumulation is not anticipated from exposure via this source. The study projected that measures under REACH (see below) could further reduce the inputs of DEHP to compost and digestate and that DEHP concentrations in compost and digestate should be further monitored to inform future risk management.

DEHP is an endocrine disruptor, with the potential to cause cancerous tumours and birth defects and is toxic to reproduction (ECHA, 2020d). Levels observed in humans, however, are low and the tolerable daily intake⁴⁶ of DEHP is 0.05 mg/kg body weight per day.

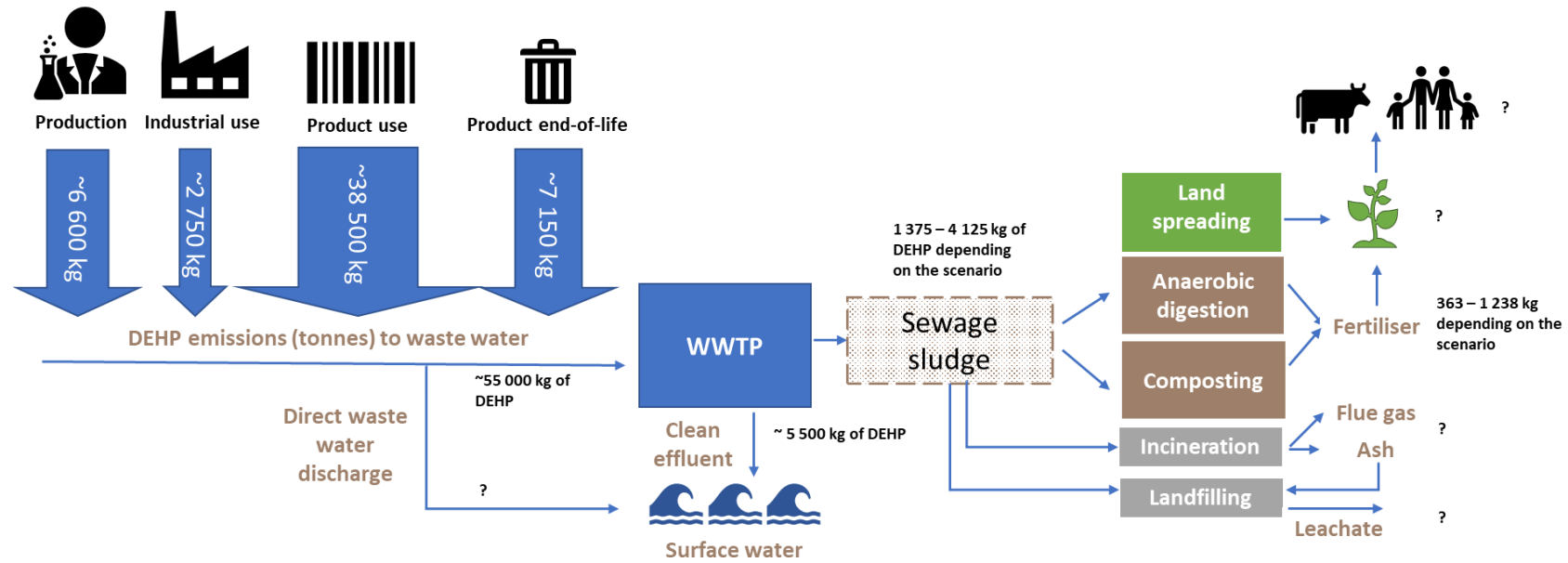
DEHP Mass balance: from product use to sewage sludge

In the absence of coherent and comprehensive data on mass of DEHP present in products currently in use, releases of DEHP from industrial facilities, households and end-of-life disposal, an illustrative mass balance for DEHP presented below in Figure 5-5 is based on the following data and assumptions:

- Emissions of DEHP to freshwater from UWWTP in EU-27 are on average 5.5 t/year (average calculated using 2017 and 2018 data from the E-PRTR). This is the starting point for the calculation.
- Around 90% of DEHP is removed from waste water during waste water treatment through sorption on sludge and biodegradation.
- Two scenarios reflect different levels of DEHP biodegradation during waste water treatment as reported in literature: Scenario 1) 25% of DEHP biodegrades and 75% of DEHP sorbs onto sludge, Scenario 2) 78% of DEHP biodegrades and 22% sorbs onto sludge.
- Treatment of sludge (AD, composting) is assumed to reduce content of DEHP by around 70% (based on the average content of DEHP in sludge of 500 mg/kg dw (See table 5.3) and in sewage sludge digestate/compost of 140 mg/kg). The average DEHP concentration in sludge is the average of figures quoted in (JRC, 2012). It should be noted that the study provides wide ranges of DEHP concentrations in sewage sludge, and thus the average does not reflect this variety. This average concentration may therefore be overestimating the actual presence of DEHP in sludge ; especially given that since the publication of the study, policy measures resulting in substitution of DEHP in some uses would have led to reduced concentrations in waste water and consequently sewage sludge.

⁴⁶ An estimate of the amount of a substance in air, food or drinking water that can be taken in daily over a lifetime without appreciable health risk.

Figure 5-5 Illustrative mass balance for annual releases of DEHP



Source: own elaboration



5.1.5 Options for upstream reduction of contamination

The rationale employed for selecting options to investigate upstream reduction of DEHP is based on the proportional release of DEHP from different sources, as presented in Figure 5-4. The product use phase accounts for ~ 70 % of emissions, and within that polymer-based products, predominantly PVC, hold by far the largest share of DEHP placed in products (estimated at 97 %). (ENTEC UK Limited, 2011).

Therefore, two options for upstream reduction have been considered:

1. Substituting DEHP in PVC with alternative chemicals
2. Substituting PVC with alternatives

Option 1 Substituting DEHP in PVC with alternative chemicals

The first option being considered to prevent harm from DEHP in PVC is to replace the DEHP with an alternative chemical. A careful risk assessment of the alternatives is required to avoid “regrettable substitutions”, i.e. replacing one harmful chemical with another equally or more harmful chemical. The data collected through the implementation of the REACH, CLP and Biocidal Products regulations, as well as data collected by foreign agencies and academia can be of great help in the assessment of hazard and risk of alternatives⁴⁷.

DEHP could be substituted with other safer chemicals from the same group (i.e. phthalates) which have similar properties, such as DINP and DIDP. These have been a popular replacement in the past (Nagorka & Koschorreck, 2020). However, the growing regulation of the use of phthalates in the EU has led PVC manufacturers to opt for non-phthalate alternatives (Nagorka & Koschorreck, 2020). For instance, the use of DINP and DIDP in toys is restricted in the EU. Since the replacement of DEHP with phthalate alternatives could be considered “regrettable substitution”, this option focuses on the replacement of DEHP with non-phthalate alternatives. Several widely available non-phthalate DEHP replacements exist on the market and are used by companies which commit to go DEHP-free. These include di-2-ethylhexyl-terephthalate (DEHT), diethylhydroxylamine (DEHA), 1,2-Cyclohexane dicarboxylic acid diisononyl ester (DINCH), epoxidized soybean oil (ESBO), trioctyl trimellitate (TOTM), and citrates such as acetyl tributyl citrate (ATBC) (Gall, 2014; ChemSec, 2019). The levels of Terephthalates (including DEHT), Trimellitates (including TOTM) and Cyclohexanoates (including DINCH) have been increasing in the past 15 years (Gall, 2014). Extensive risk assessments have been performed for most of these chemicals as a part of REACH and under similar initiatives in the US and Canada, the key outcomes of which are summarised in Table 5-4. Limited risk assessments have been identified for ESBO which is the only organic compound based on renewable feedstock in the group.

An assessment of the non-phthalate alternatives, their suitability for PVC uses and the associated costs and benefits have been summarised in Table 5-4. DEHP is regarded as a baseline to the assessment. The substitution option would be applied at PVC manufacturing facilities. The information presented in the table suggests that while these alternatives represent an improvement compared to DEHP, some non-phthalates should be treated with caution since their Greenscreen benchmark⁴⁸ is ‘Use – but search for safer substitutes’.

⁴⁷ <https://echa.europa.eu/data-to-prevent-regrettable-substitution>

⁴⁸The GreenScreen for Safer Chemicals is a transparent, open standard for assessing chemical hazard that supports alternatives assessment for toxics use reduction through identifying chemicals of concern and safer alternatives. See scores at: <https://tcocertified.com/accepted-substance-list/>

Table 5-4 DEHP non-phthalate alternatives in PVC

Alternative chemical	Greenscreen benchmark	Environmental risk	Health impacts ⁴⁹	Market availability	Application
DEHT/DOTP	Use – but there is still opportunity for improvement	No evidence; Bioaccumulation is unlikely due to virtually complete excretion;	Decrease in appetite in high doses >2 154 mg/kg-day (based on a study with rats)	Widely available in the United States	PVC toys, childcare articles, consumer products, beverage closures and other polymer materials including cellulose acetate-butyrate, cellulose nitrate, and chloroprene rubbers
DEHA	Use – but search for safer substitutes	Negative impacts on aquatic species and soil dwelling organisms	Liver tumours in female rodents at mid-high doses at 617 mg/kg- day, reproductive toxicity at 800 mg/kg a day	Widely available	Toys, vinyl flooring, wire and cable, stationery, wood veneer, coated fabrics, gloves, tubing, artificial leather, shoes, sealants, and carpet backing. films employed in food packaging materials, fillers, paint and lacquers, adhesives, plastic in concrete, and rubber products.
DINCH	Use – but search for safer substitutes	Negative impacts on aquatic species and soil dwelling organisms	Kidney toxicity, reproductive toxicity at 1,000 mg/kg-day	Widely available in Europe	Medical devices, toys, food packaging, cosmetics products, shoes, exercise mats and cushions, textile coatings, printing inks
ESBO	Use – but there is still opportunity for improvement	Negative impacts on aquatic species	Suspected to cause some effects on the kidney, liver and uterus by repeated oral administration.	Widely available Market experience on use in medical devices	Seal glass jars, and as a stabilizer to minimise the ultraviolet degradation of PVC resins baby food jars, fillers, paint and lacquers, adhesives, printing inks, and packaging

⁴⁹ Repeat toxicity

Alternative chemical	Greenscreen benchmark	Environmental risk	Health impacts ⁴⁹	Market availability	Application
TOTM	Use – but search for safer substitutes	Negative impacts on aquatic species and soil dwelling organisms	Negative impacts on reproduction	Widely available	Vinyl automotive interior, dishwasher gaskets, seals, and telephone wire.
ATBC	Use – but there is still opportunity for improvement	Negative impacts on aquatic species	Decreased body weight, reproductive toxicity at 100 mg/kg-day	Widely available	Cosmetic products, toys, vinyl, adhesives, medical devices, pharmaceutical tablet coatings, food packaging, flavouring substance in foods, printing inks and plastics in concrete

Source: ChemSec (2019); University of Cincinnati (2018); SCENIHR (2016); Environment Canada (2011); Danish Ministry of Environment (2014); Lowell Centre for Sustainable Production (2011), Viet et al. (2011)



Impacts on sludge and soils

Table 5-5 summarises the impacts of Option 1 on the emissions from different sectors. The table shows that the key sources which will continue releasing DEHP emissions relate to product use, for products which have not reached their end of life (see Table 5-6), recycling of PVC and landfills.

Table 5-5 Impacts from Option 1 on DEHP emissions in the environment

Source	Reduction of emissions?	Considerations
Production of DEHP	✓	DEHP emissions to water from DEHP production will be reduced. If the rest of the world continues using DEHP, the emissions from production for export will not be reduced. As a result of increasing regulation of DEHP since 1999 and it being included as a priority substance under the Water Framework Directive in 2008, the market share of DEHP in plasticisers in Europe decreased to 10% in 2014 compared to 42% in 1999 (Nagorka & Koschorreck, 2020). <i>It is noteworthy that emissions from alternative chemicals will increase.</i>
Recycled PVC	✓	Recycling of PVC from buildings has been rapidly increasing since 2003, with 514 913 t being recycled in the EU in 2015 (VinylPlus, 2017). Emissions from the recycling of existing PVC and use of recycled PVC will continue.
Industrial use	✓	Emissions from products which are still in use will continue.
Product use	✓	Emissions from products which are still in use will continue. The products with the longest lifespan and associated impacts are those used in buildings, e.g. pipes.
End life	✓	Transfer to waste management facilities, incineration, leachate from landfilled products containing PVC will continue. Further emissions will arise at the end of life of products currently in use.

Table 5-6 summarises the lifespan of different PVC products, showing that the products used in buildings, electronics, furniture and automotive have the longest lifespans. Of these product categories, products used in buildings and furniture are currently recycled (approximately 155 000 t in 2015) and a likely source of DEHP in the future (VinylPlus, 2017).



Table 5-6 Lifespan in years of PVC products used in different sub-sectors

Product use sub-sector	% of EU PVC	Lifespan
Building	57	10 to 50
Packaging	9	1
Furniture	1	17
Other household appliances	18	11
Electric/Electronic	7	21
Automotive	7	12
Others	7	2 to10

Source: European Commission (2004b) VinylPlus (2017)

It was not possible to estimate what the implication of replacing DEHP with alternative chemicals will be for sludge contamination levels and the application of sludge in agriculture. This is because no information has been identified regarding what proportion of the DEHP in products enters waste water and subsequently sludge. However, Nagorka & Koschorreck (2020) showed that the mean concentration of DEHP in German rivers was reduced by 60 % between 2006 and 2017 as a result of existing restrictions at the EU level and companies' initiatives to substitute DEHP. This indicates that policies restricting the use of DEHP have had significant impact on its presence in the environment, though the impacts could be also linked to improved waste water treatment and higher application of AD which degrades DEHP (Hariklia N, et al., 2003).

Qualitative analysis on feasibility

Several companies have moved away from the use of PVC that contains DEHP due to the existing policies restricting some uses of DEHP and to public concerns about the toxicity of DEHP (ChemSec, 2019). This has led to a decline in DEHP manufacturing in Europe and a general decline of DEHP in the environment (ECHA, 2020d). Since non-phthalates have similar properties and are widely available, the substitution of DEHP in all PVC products is feasible. While the cost of alternatives has been found to be slightly higher than cost of DEHP, the difference is small and may be reduced as the market for alternatives grows. Furthermore, as discussed in sub-section 5.1.3, the 2020 REACH restrictions on DEHP prevent placing PVC products containing DEHP on the EU market. This further encourages the replacement of DEHP with non-phthalates.

It is noteworthy that the use of alternative chemicals is also associated with health hazards and hazards for aquatic species and soil dwelling organisms, however these are substantially lower when compared to DEHP. The safer chemicals of the ones discussed in Table 5-4 are DEHT, ESBO and ATBC, as showed by the Greenscreen benchmark. DEHT has a particularly low oral toxicity due to the fact that it is poorly absorbed when consumed orally (SCENIHR, 2016). EBSO and ATBC are toxic at substantially higher repeat concentrations when compared to DEHP (Danish Ministry of Environment, 2014). Overall, substitute chemicals represent a lower risk compared to the DEHP. However, since their presence in the environment is rapidly



increasing, further research is required to establish their impacts (Nagorka & Koschorreck, 2020).

Option 2 Substituting PVC with alternatives

The second option with potential to reduce the levels of DEHP entering the environment from product use, is to encourage or legislate for the use of alternative materials to PVC, thus fully removing the need for phthalate or non-phthalate plasticisers.

Several studies have discussed and compared potential alternatives to PVC products, with the most recent relevant to our study being (ENTEC UK Limited, 2011). Other relevant studies are:

- Five Chemicals – Alternatives Assessment Study Prepared by The Massachusetts Toxics Use Reduction Institute, University of Massachusetts Lowell June 2006 (TURI, 2006)
- Life Cycle Assessment of PVC and of principal competing materials, commissioned by the EC, July 2004 (EC, 2004b)

None of the studies conducted on key products include consideration of DEHP in particular, or environmental toxicity in general. It is also noted that the studies have been produced more than a decade ago and as such may not reflect the latest developments in PVC alternatives. Yet no recent studies on the topic have been identified in the literature research. The EC (2004b) report is a comparative summary of LCAs carried out on various PVC uses. The report contains an ‘application matrix’, replicated in Annex C, which gives proportions, based on extensive industry consultation, of the distribution of the PVC market in Europe across key industry sectors and their products. In the matrix, ‘Share of used PVC mass’ refers to the percentage of the total PVC used in Europe for each entry, and ‘market share of PVC’ refers to where PVC sits in each relevant market, compared to the competing materials. The matrix provides a list of example alternative products/materials which could be substituted for PVC for all uses listed.

The matrix shows that building and construction applications were the major uses of PVC in Europe, making up 49.5 % of the total PVC mass considered. This was then followed by packaging application, making up 8 %, and electric and electronic cables, along with their associated casings and duct, making up 11.5 %.

Within the building and construction sector, the largest proportionate use was for waste water and rainwater pipes, with 12 % of modelled PVC use. However, the report highlights that these do not generally contain plasticisers, and therefore do not represent a significant phthalate emission source.

TURI (2006) looks at DEHP containing products in particular, focussed on use in Massachusetts, USA, and give a good structure to how alternative product options may be investigated. The key criteria for selection of priority uses are volume of use, the geographic, potential environmental and occupational exposure, and availability of viable alternatives.

Applying these in combination with the EC (2004b) estimates of PVC use in Europe, the example of replacement options for resilient flooring⁵⁰ is the most relevant for further consideration as illustrative of the process.

The consideration factors for what makes a product alternative ‘viable’ should include performance requirements, maintenance and durability, cost, and environmental credentials. The environmental considerations can include: the type of raw resource material(s) required, the requirement of use, or use-phase release of toxic or harmful substances, and end of life recovery or circularity options.

⁵⁰ “tile and sheet materials which have the ability to return to their original form after compression” (TURI, 2006)



The alternatives to PVC flooring selected for detailed study in TURI (2006) were natural linoleum⁵¹, cork and polyolefins (polyethylene (PE) and polypropylene (PP) thermoplastics). In addition to these we have considered wood laminate flooring and an innovative ‘bio-tile’ flooring option.

A number of examples of biobased flooring are coming to market. Generally, they are constructed of limestone and a natural based polyester binder made from rapidly renewable sources such as corn. These systems may require an additional coating for surface protection and may have less durability than tradition PVC flooring. Installation and maintenance costs are also likely to be higher. (Wolfe Flooring, 2019)

It should be noted that polyolefins were considered as potential substitute materials for several other PVC uses, including cables (including casings and ducting), coated fabrics, roofing, medical devices and water pipes (waste, rain and drinking water). The assessment, therefore, while specifically referred to flooring, has some valid conclusions for a wide range of uses of PVC.

Impacts on sludge and soils

As with Option 1, the DEHP emissions from products currently in use would continue but would be removed in new products. When considering the use and maintenance and end-of-life phases of flooring products, the assessed environmental and human health aspects, both positive and negative, outside of the removal of DEHP containing materials, are summarised in Table 5-7.

Table 5-7 Assessment of non DEHP related environmental aspects of PVC alternatives for flooring

Product	Lifecycle Phase	Environmental concerns	Environmental benefits
Natural linoleum	Use and maintenance	<ul style="list-style-type: none"> • Cleaning, waxing VOC off-gassing potential 	<ul style="list-style-type: none"> • Can be cleaned with mild detergent
	End of life	<ul style="list-style-type: none"> • Not recyclable 	<ul style="list-style-type: none"> • Biodegradable raw materials • Compostable • No chlorine products generated if incinerated
Cork	Use and maintenance	<ul style="list-style-type: none"> • Off-gassing of polyurethane maintenance coatings 	<ul style="list-style-type: none"> • Hypoallergenic
	End of life	<ul style="list-style-type: none"> • Not recyclable 	<ul style="list-style-type: none"> • Compostable (dependent on coating) • No chlorine products generated if incinerated
Wood laminate	Use and maintenance	<ul style="list-style-type: none"> • Wood should be sourced from sustainable forestry practices 	<ul style="list-style-type: none"> • >80 natural materials • No VOC off-gassing
	End of life	<ul style="list-style-type: none"> • N/A 	<ul style="list-style-type: none"> • Recyclable • Easy deconstruction systems readily available

⁵¹ Made up of natural substances such as pine resin, solidified linseed oil, wood flour, ground cork dust, and mineral fillers.



Product	Lifecycle Phase	Environmental concerns	Environmental benefits
Bio-based tile system (limestone and bio-polyester resin)	Use and maintenance	<ul style="list-style-type: none"> • Some VOC off-gassing • Protective coating potentially required 	<ul style="list-style-type: none"> • Can be made with recycled content
	End of life	<ul style="list-style-type: none"> • N/A 	<ul style="list-style-type: none"> • Recyclable but dependent on coating applied
Polyolefins (PE and PP thermoplastics)	Use and maintenance	<ul style="list-style-type: none"> • N/A 	<ul style="list-style-type: none"> • Can be cleaned with a mild detergent • No polishing or waxing required • Very low VOC emissions
	End of life	<ul style="list-style-type: none"> • N/A⁵² 	<ul style="list-style-type: none"> • Recyclable • No chlorine products generated if incinerated

Sources: Own composition and (TURI, 2006)

The alternatives studied offer significant environmental benefits over the use of DEHP containing PVC for flooring, with only the non-recyclability of cork and linoleum, and some aesthetic considerations comparing unfavourably. To counter this, both cork and linoleum can theoretically (if somewhat unlikely) be composted, although this does not facilitate their use in new products.

Costs are generally similar for all types considered, apart from the more expensive bio-based flooring. In terms of performance factors, PVC flooring is highly moisture resistant compared to other options.

Qualitative analysis on feasibility

The substitution of PVC products with alternative materials such as those considered here would undoubtedly reduce the level of DEHP entering sewage sludge, and as a result the environment at large. The phasing out of DEHP containing PVC products would need to be a long-term initiative. Product recalls of all DEHP containing items is unfeasible, so any interventions would need to be considered in relation to the eventual environmental benefit from reducing the levels of DEHP in the environment from new products, while DEHP emissions from products already in use would continue during their in-use phase.

Previous legislation has been successful in ending the use of DEHP in toys, and several global corporations are taking steps to reduce or eliminate DEHP containing products from their supply chains (Apple, Ikea, H&M etc. (ChemSec, 2019)). This demonstrates that a combination of regulatory and industry procurement engagement levers, and consumer awareness, has the potential to impact the level of DEHP-containing PVC products either manufactured or imported into Europe in the future. Producer responsibility schemes offer another approach to steer producers' choice of materials, especially when fee modulation is applied. For example, In

⁵² The actual TURI study details the lack of recycling infrastructure, but is from the USA in 2006. PP and PE are widely recyclable today.



France, the use of PVC bottles declined since a higher fee (malus) was introduced for PVC bottles. (Eunomia, 2020).

The potential interventions to stimulate this change could include offering incentives to manufacturers, retailers, wholesalers and end-users to use or procure alternative products, e.g.:

- Subsidising alternatives
- R&D support for industry
- Producer responsibility schemes, especially with fee modulation

In reality, a balanced combination of these will be needed to achieve the desired result, and detailed feasibility and impact assessment will be required to identify the correct balance.

5.1.6 Comparative analysis of options

Table 5-8 presents a comparison of the two options, using a Red-Amber-Green (RAG) rating. Overall, the analysis shows that while Option 1 (replacement of DEHP in PVC products) is feasible to implement and is not associated with significant additional costs, there are some risks associated with the increase of substitute chemicals in the environment. While research shows that these are hazardous to human health only in high concentrations, further evidence is required to make a conclusive judgment of the environmental and health impacts of these chemicals. Option 2 is associated with positive impacts on the environment and human health due to the reduction of DEHP and the lack of additional chemicals. However, while the measure is feasible and cost-effective with regard to the flooring example considered in detail, it is unclear whether it is feasible in other applications, e.g. pipes, for which the key alternatives would include PP or PE plastic, concrete or metal replacements. Finally, Option 2 is associated with substantial impacts for the PVC industry if DEHP containing PVC is banned in Europe⁵³.

Table 5-8 Comparison of options

	Option 1 Substituting DEHP in PVC with alternative chemicals	Option 2 Substituting PVC with alternatives
Environmental		
Final contamination levels	Decrease in DEHP but increase in substitute chemicals in environment.	Sustainable substitution options made from natural materials have better performance, and polyolefins has similar performance to PVC when considering sewage sludge contamination.
Air	No impact	As above.
Water	As above.	As above.
Soil	No impact	As above.
Biodiversity	Some substitute chemicals have negative impacts on aquatic and soil species.	No negative impacts on biodiversity have been identified explicitly, but if there were to be large scale increase in, say, cork use, the space needed would need to be considered in relation to existing natural habitats

⁵³ It is noteworthy that DEHP containing PVC products were restricted from being placed on the EU market under REACH in 2020 during the preparation of this report (ECHA, 2020b).



	Option 1 Substituting DEHP in PVC with alternative chemicals	Option 2 Substituting PVC with alternatives
Climate	No information	Full LCA assessments would need to be completed on alternatives, from resource cultivation and extraction, through manufacture and use, to end-of-life phases.
Resource recovery from sludge	The substitute chemicals are hazardous at very high concentrations which are unlikely to be reached in sludge.	All options should lead to the decrease of DEHP in sludge, and a better resultant recovery potential.
Circular economy options for products	No evidence has been identified which states that the use of substitute chemicals reduces the recyclability of PVC.	Polyolefins, wood and bio-based flooring can be recycled. Laminate flooring, if used as a 'click-lock' type system has high re-use potential. Linoleum and cork are both biodegradable if not subject to coating with non-biodegradable materials
Economics		
Cost for industry	The cost of substitute chemicals is similar to this of DEHP.	Substantial potential impacts on PVC producers. Sectors surrounding substitution options could benefit from increased demand.
Cost for end-user	The cost of substitute chemicals is similar to this of DEHP. Therefore, no significant implications are expected for end user.	All PVC substitutes have longer lifetime and therefore lower costs for end-users.
Feasibility	Widely applied and feasible	Several substitutes which are widely in use exist. Different substitutes are needed for each of the PVC uses. Further information is required on whether the replacement of DEHP containing PVC in all applications is feasible.
Social		
Health	The substitute chemicals have some negative impacts on human health at higher concentrations.	The complete phase out of phthalate containing PVC is not associated with any detrimental health impacts.

Legend: Red – Negative impacts; Amber – Positive impacts but some issues; Green – Positive impacts; Grey- No impact.

5.1.7 Conclusions

Releases of DEHP into waste water occur primarily from the use of polymer-based products. Being a priority hazardous substance under Water Framework Directive, DEHP needs to be effectively removed from waste water before the effluent is released to the environment so that the environmental quality standard set for DEHP in freshwaters is respected. In UWWTPs DEHP is most likely to partition to sludge particles. Thus, if effectively removed from water, it contaminates the resulting sewage sludge. Spreading of DEHP contaminated sewage sludge on



land is a potential exposure pathway for the environment as well as humans, although literature quoted in this case study suggests that human exposure to DEHP via sludge route does not constitute dangerous levels.

Due to being toxic for reproduction, the use of DEHP in consumer products has now been severely restricted by the EU regulations. Despite that, its production and use in industrial and agricultural settings can still be authorised by the European Commission. Taking into consideration the technical lifetime of products containing DEHP, and assuming that no new consumer products will be placed on the market from 2020 onwards, emissions of DEHP from product use in households is likely to continue until 2050. However, emissions from the authorised uses as well as imported products which may contain DEHP are likely to continue beyond that. As long as the emissions to waste water continue, DEHP will be found in sewage sludge.

The case study considered two upstream options for avoiding contamination of sewage sludge with DEHP: substitution of DEHP with other plasticizers in manufacturing of PVC and substitution of DEHP containing PVC products. The substitution of DEHP with non-phthalates alternatives is a more feasible option, however, it is associated with higher environmental and health risks. In comparison, the replacement of phthalate-containing PVC materials with alternatives can have a good environmental footprint, provided that the alternatives are sustainable throughout their full life-cycle. Different substitutes for different PVC applications would be required depending on performance requirements, and as such a natural or renewably sourced alternative may not always be feasible.

A conclusion of which is the more favourable option has not been reached due to the following uncertainties:

- Environmental/health impacts of non-phthalate alternatives, even though the alternatives have less hazardous properties than DEHP;
- Impacts of non-phthalate alternatives on sludge composition and recyclability;
- Feasibility of the replacement of PVC in flexible PVC applications.

In order for either of the two options to be widely implemented by industry, regulatory intervention is required. The costs for industry are likely to be smaller if Option 1 is selected due to the fact that some companies have already moved away from DEHP use as a result of existing regulatory measures. In addition, Option 2 will be associated with a sharp decline of the EU market share of DEHP containing PVC production.

The following limitations and gaps in evidence have been identified:

- Information on current production volumes of DEHP in Europe and its use is limited and where it has been found, it is either out of date or incomplete. It is unclear how much DEHP is contained in products imported to Europe from the rest of the world.
- The analysis of sources of DEHP emissions to waste water has relied on studies published more than a decade ago and as such it does not reflect the impacts of the Regulations put in place since to restrict the use of DEHP in consumer products. It is recognised that the proportion of DEHP emitted to waste water from product use might now be smaller compared to the values presented in the case study.
- The fate of DEHP in UWWTPs differs depending on the specifics of treatment applied. Depending on the technology choice, greater rates of biodegradation of DEHP could be achieved which in turn lead to lower quantities of DEHP entering sludge. While the optimum parameters of UWWTP to encourage greater degradation of DEHP in waste water have not been within the scope of the case study, it appears worthwhile that this is investigated further in future studies and their costs and benefits weighted against the upstream options.
- The illustrative mass balance for DEHP has been derived from emissions of DEHP from UWWTP reported in the E-PRTR and evidence found in literature on the fate of DEHP



from source of emissions to sewage sludge. The data in the E-PRTR is limited due to pollutant and capacity thresholds applied in reporting obligations. As such the mass of DEHP emitted from UWWTP and therefore the whole mass balance is likely to be underestimated.

- Additional information is required on the health hazards linked to some alternative chemicals such as ESBO. It was not possible to estimate what the implication of replacing DEHP or PVC with alternative chemicals will be for sludge contamination levels and the application of sludge in agriculture.
- Regarding alternatives to DEHP containing PVC products, full LCAs would need to be carried out for all substitution options before a definitive judgement could be made on which has the least harmful environmental impact. This body of evidence does not yet exist, particularly for recent studies. All options would eventually reduce the amount of DHEP emitted to waste water, and hence sewage sludge, but wider environmental impacts also need to be considered.
- The most recent, comprehensive listing to be found of PVC applications in Europe was from 2004. More recent data might have allowed for a more targeted selection of products to investigate replacements for.

5.2 Benzo(a)pyrene (BaP)

5.2.1 Introduction to the case study

Benzo(a)pyrene (BaP) is a polycyclic aromatic hydrocarbon (PAH), a hydrocarbon composed of multiple aromatic rings, produced and released to the air during the partial combustion of carbon-based material (e.g. wood burning, fuel combustion, tobacco smoking, food charring). BaP and other PAHs released to the atmosphere subsequently reach the ground via dry and wet deposition (Environment Agency, 2019). Once deposited on soil, PAHs accumulate for a long period of time and are subject to various partitioning, degradation, and transport processes (Environment Agency, 2019). PAHs can either enter the water bodies directly - via deposition from the air, or with surface run-off following rainfall (Environment Agency, 2019). In UWWTP, BaP is mostly removed from waste water and transferred into sewage sludge, thus leading to its contamination.

This case study brings together evidence from literature on the pathways and presence of BaP in European sewage sludge and the associated risks to the environment and human health, with information on potential upstream measures to prevent BaP from initially entering waste water, and thus eliminating its presence in sludge. The case study is structured as follows:

1. Description of the substance and its use – provides information on general characteristics of BaP as well its levels in Europe
2. Current regulation – summarises EU legislation applicable to limiting BaP emissions to waste water and its presence in sludge
3. Sources and pathways to the environment – describes the sources of BaP to waste water and thus sewage sludge, and its pathway to the environment. It also provides an illustrative mass balance for BaP from source (i.e. partial combustion) through waste water to sewage sludge.
4. Options for upstream reduction of contamination – describes the options available to prevent BaP from entering the waste water (referred in this study as “upstream” measures).
5. Comparative analysis of options – discusses how the upstream options compare to each other in terms of environmental and economic performance, and what trade-offs need to be considered when deciding on possible further “upstream” policy interventions.
6. Conclusions – draws conclusions from the case study.

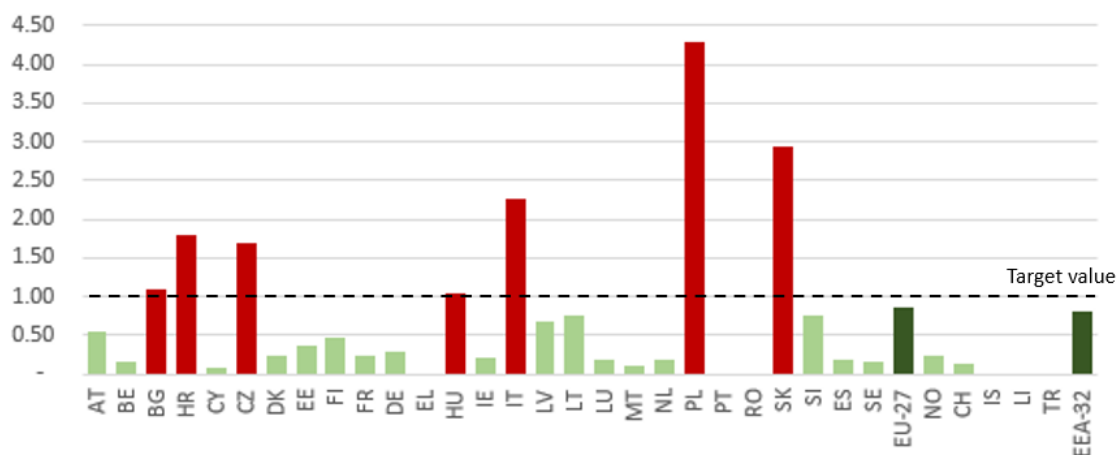


5.2.2 Description of the substance and its levels of concentration

BaP is a 5-ring hydrocarbon and is within the class of chemicals called PAHs, which occur naturally in coal, crude oil and gasoline (Environmental Protection Authority, 2009). PAHs containing four or more rings, such as BaP, are defined as 'heavy' PAHs which means they are more stable and more toxic. PAHs are synthesised during formation of fossil fuel or during the partial combustion of carbon-based material at temperatures between 300 °C and 600 °C. For example, PAHs are emitted from heating appliances using coal, wood or biomass fuels, cooking or motor vehicles exhausts (especially from diesel engines), industrial operations (from fuel combustion and process emissions) and forest fires. These chemicals are then released to the air before falling to the ground or being washed down water, soils and sediment, generally at trace levels, except near their source. BaP then makes its way to waste water, and eventually sewage sludge, via these routes. Once BaP is present in sludge, humans can be exposed to it via food grown on soils fertilised by sludge or sludge-derived products (International Agency for Research on Cancer, 2010). BaP can stay in the environment for a long period of time and has therefore been classed as a Persistent Organic Pollutant (POP) (Scottish Environmental Protection Agency, 2020).

This chapter presents sources and trends of atmospheric emissions of BaP, as they are the major pathway leading to deposition of BaP on land, and BaP entering waste water with surface run-off. In 2004, with a view to protect human health and the environment, the European Commission set a target value of 1 ng/m³ for the concentration of BaP⁵⁴ in ambient air (EC, 2004a). Figure 5-6 shows that even though the average annual mean concentration of BaP in the air across the EU-27 and EEA-32 fall below this limit in 2018, there are countries in Europe such as Poland, Slovakia and Italy where the target values for BaP are exceeded. This can be explained by some countries' reliance on domestic coal combustion and the development of biomass combustion.

Figure 5-6 Annual mean concentration of BaP in ambient air by country in the EU-27 and EEA-32 in 2018 (in ng/m³)



Source: (EEA, 2020a)

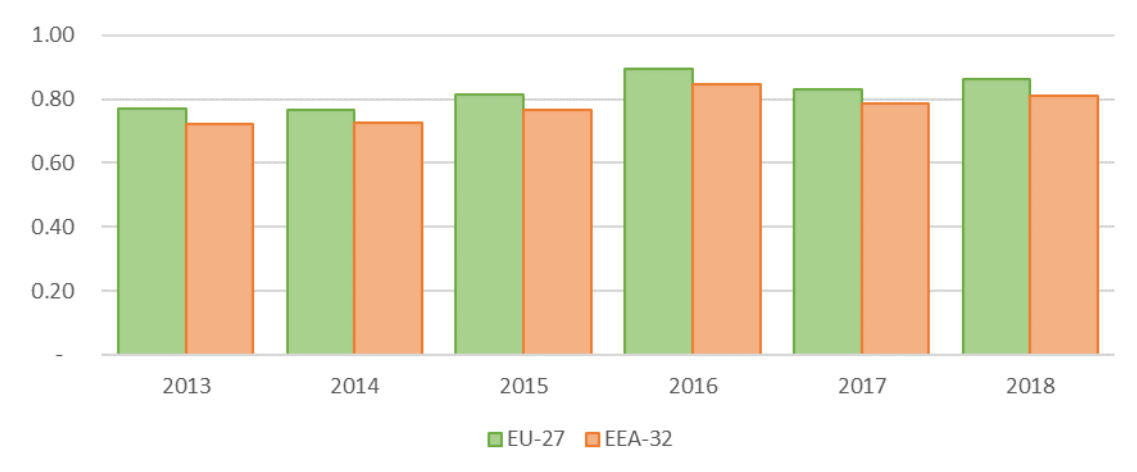
Note: Information not available for EL, IS, LI, PT, RO, and TR

⁵⁴ For the total content in PM₁₀ sample fraction in ambient air averaged over a calendar year



Figure 5-7 shows that the annual mean concentrations of BaP in the EU-27 and EEA-32 increased by around 12% between 2013 and 2018 (from 0.77 to 0.86 ng/m³ in EU-27, and 0.72 to 0.81 ng/m³ in EEA-32). According to the EEA (2019), this increase can be explained by the introduction of policy measures in some countries to encourage biomass combustion.

Figure 5-7 Trend in average annual mean concentration of BaP in ambient air in the EU-27 and EEA-32 (in ng/m³)

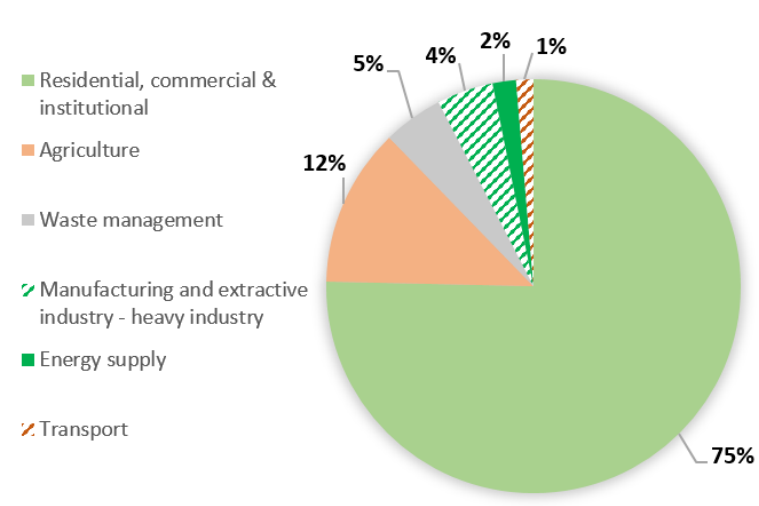


Source: (EEA, 2020a)

Note: Information not available for EL, IS, LI, PT, RO and TR

In 2018, 266 t of BaP was emitted to the air in the EU27 (EEA, 2021). Figure 5-8 shows the distribution of these emissions per sector in the EU-27. The residential, commercial and institutional sector was the primary source of BaP emissions to air, with 75 % of the total emissions, followed by agriculture with 12 % and waste management with 5 %. The figure from agriculture is dominated by the on-field burning of crop stubble residues.

Figure 5-8 Contribution to EEA-27 emissions from main source sectors in 2018 of BaP



Source: (EEA, 2020d)

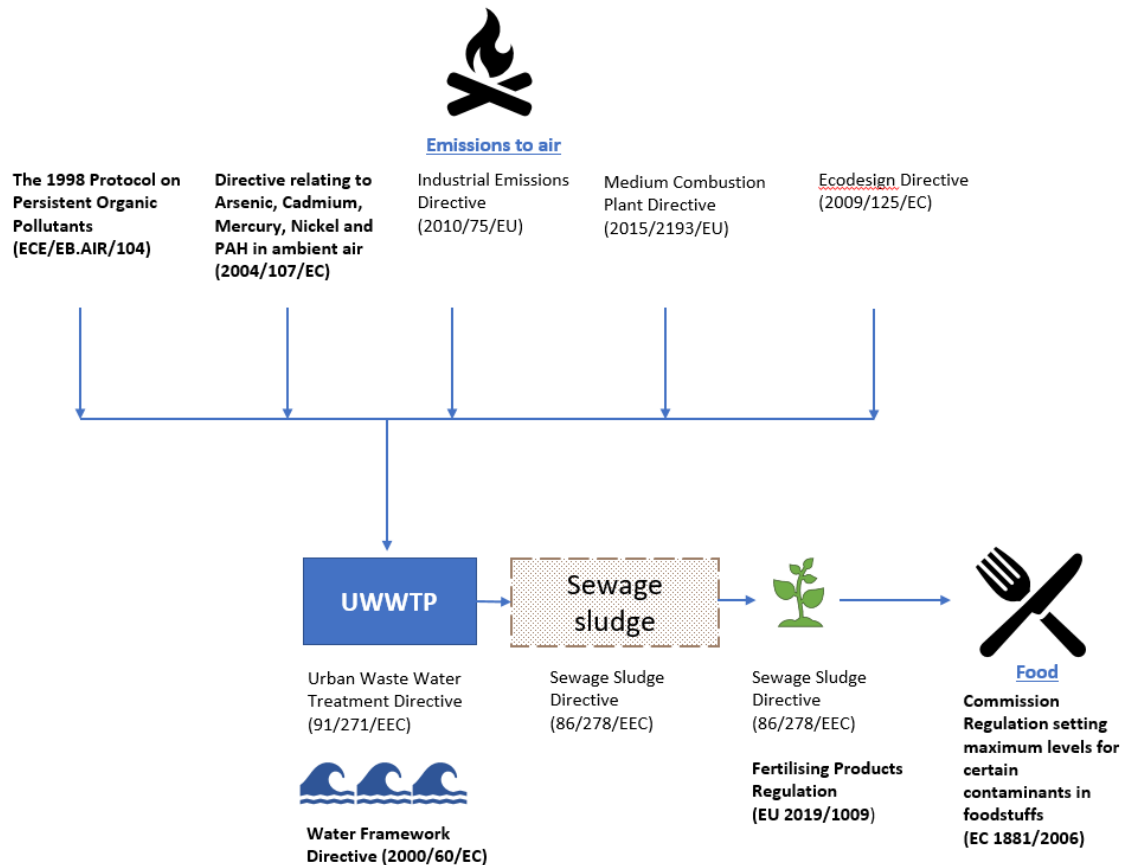


5.2.3 Current European regulation applicable to the emissions of BaP

Figure 5-9 presents a regulatory framework currently impacting EU emissions of BaP to air, removals at UWWTP, treatment and application of sludge, and resultant levels of contamination in foodstuffs.

Under REACH, BaP has been identified as a substance of very high concern (SVHC) due to its carcinogenic, mutagenic, toxic for reproduction and bioaccumulative properties. Because of this, additional regulations set limits for its concentration in certain products, such as tyres and extender oils used in their manufacture. As this is not a major pathway to waste water it is not described in detail here.

Figure 5-9 Legislation impacting emissions of BaP in Europe



Source: Own elaboration

Notes: Legislation in bold includes specific requirements for BaP/PAH. Other legislation may have indirect impacts on BaP releases.

Emissions of BaP to air

Emissions of PAHs are controlled internationally through the UNECE’s 1998 Aarhus Protocol on Persistent Organic Pollutants (POPs)⁵⁵. The Protocol obligates parties, of which the EU is one, to reduce total annual emissions of POPs, including PAHs, and employ best available techniques (BAT) to limit the emissions, and reduce levels from a set baseline year.

BaP emissions to air are regulated by a suite of European instruments falling under the umbrella of the Clean Air Policy Package⁵⁶. The Ambient Air Directive (2004) sets a target value of 1 ng/m³ for the concentration of PAH in air, using BaP as a marker for the total PAHs carcinogenic risk in ambient air, to avoid, prevent or reduce harmful effects on human health and the environment (EC, 2004a). This is complemented by indirect impacts on BaP and other emissions by the

⁵⁵ <http://www.unece.org/fileadmin/DAM/env/lrtap/full%20text/ece.eb.air.104.e.pdf>

⁵⁶ https://ec.europa.eu/environment/air/clean_air/index.htm



Industrial Emissions Directive, the Medium Combustion Plant Directive, and the Ecodesign Directive. These regulations impact emissions from varying sizes of combustion equipment, and are discussed in more detail in Section 5.2.5.

BaP emissions to water

PAHs including BaP are a priority hazardous substance (PHS) under the Water Framework Directive (as stated in Directive 2013/39/EU⁵⁷). This sets out Environmental Quality Standards for BaP in surface waters of $1.7 \times 10^{-4} \mu\text{g/l}$.

BaP content in sludge

There are no EU limits for BaP in sewage sludge used in agriculture. The SSD (86/278/EEC) sets requirements for the use of sludge in agriculture but sets limits only for selected metals. The EU Fertilising Products Regulation (EU/2019/1009) prohibits sewage sludge to be part of compost and digestate used in fertiliser products placed on EU market. The Regulation sets a limit value for PAHs, including BaP, of 6mg/kg dry matter of compost, digestate or other derived products.

Levels of BaP in food

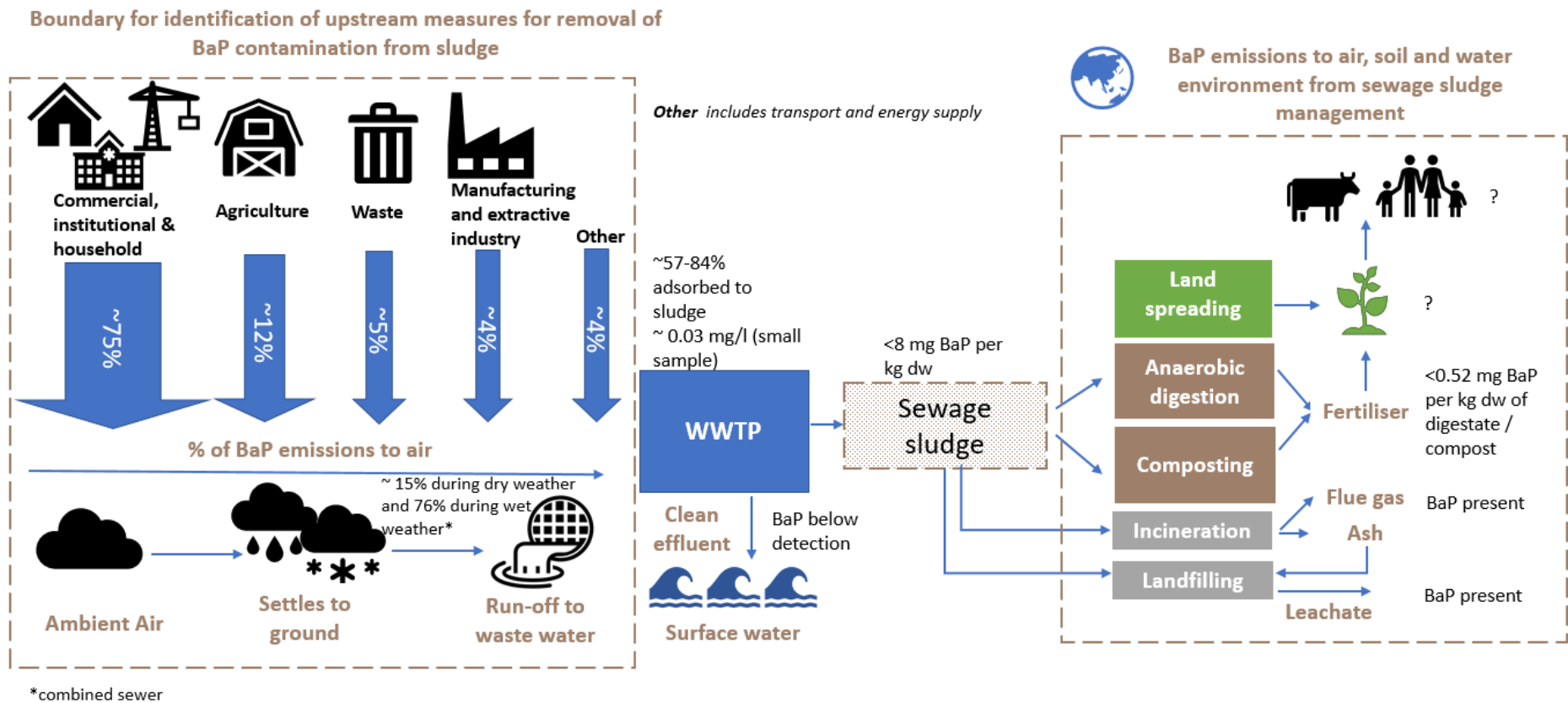
As well as reaching the human food chain via sludge-derived fertilisers, PAHs in foods are formed by cooking at high temperatures, via means such as charbroiling, grilling and frying. The 2006 Commission Regulation setting maximum levels for certain contaminants in foodstuffs (EC, 2006) sets maximum levels of PAHs including BaP, and BaP alone, for 16 foodstuff categories. These include oils and fats, meat and seafood products, processed baby foods and milk formula and dried herbs and spices. The limits range from $1 \mu\text{g/kg}$ for food specifically meant for babies and infants, to $10 \mu\text{g/kg}$ for herbs and spices (EC, 2011; EC, 2015) .

5.2.4 Sources and pathways of BaP to the environment

Figure 5-10 presents a simplified schematic of sources of BaP emissions to waste water and its pathway to the environment via a sewage sludge route.

⁵⁷ <https://eur-lex.europa.eu/legal-content/EN/ALL/?uri=CELEX:32013L0039>

Figure 5-10 Illustration of BaP pathways to the environment and its sources in waste water in Europe



Source: Own elaboration based on information in: (EEA, 2017b), (Włodarczyk-Makuła, 2008), (Wood, 2019), (Mouchel, et al., 2020), (UKWIR, 2013)

Notes: dw = dry weight. Numbers are indicative only based on studies referenced.



Sources of BaP emissions to waste water

Full mass balance analysis was not possible for BaP due to the following data gaps:

- Information on average deposition rate - the deposition of BaP depends on multiple factors including weather conditions (Mouchel, et al., 2020);
- Information on concentration in UWWTPs – while some information was identified, the data is from a limited sample (30 results from 16 UWWTPs) (UKWIR, 2013).

BaP is emitted to the air from the following natural and/or anthropogenic sources:

- Thermal sources: partial combustion of carbon-based materials (e.g. fuel, char, wood)
- Evaporating sources: use of products containing PAH (e.g. creosote, coal tar)

During the 1990s, the releases from the latter have been significantly reduced by targeted legislation on use and PAH content of certain products. However, emissions from partial combustion remain and have shown gradual increases in Europe.

As stated above, the key source of atmospheric emissions is combustion in the domestic and commercial sector (EEA, 2020c). Concentrations of BaP in the ambient air in urban areas can range from 1 to 30 ng/m³, however this can increase to several tens of ng/m³ in road tunnels or large cities which rely heavily on coal or biomass heating. For industrial combustion, the highest levels of emissions are observed in aluminium production, mid-range levels are observed in roofing and paving and the lowest concentrations are observed in coal liquefaction, coal-tar distillation, wood impregnation, chimney sweeping and power plants (International Agency for Research on Cancer, 2010).

PAHs are semi-volatile and stable, meaning that they are highly mobile throughout the environment and can persist and travel for a long period of time. Once in the atmosphere, depending on their physical and chemical properties, PAHs are distributed between gas, particle and droplet phase and are disseminated mainly via the air. In the course of time, they will turn into a dry form due to sunlight or be caught into rainfall and fall to the ground where they will slowly break down in soil or run off into surface water, ground water and tap water and eventually end up in waste water (DHSS, 2015).

Fate of BaP at waste water treatment plants

BaP does not dissolve in water; however, it would dissolve in organic (carbon-containing) solvents. As a result of their strong partitioning to the solid phase, during waste water treatment PAHs are adsorbed on to solid particle surfaces and retained in sludge. Practical studies confirm that most BaP follows this predicted behaviour (Włodarczyk-Makuła, 2008). The research suggests that considerable share of PAHs (between 83-85%) are removed from waste water in conventional UWWTP (Włodarczyk-Makuła, 2008). Other studies identified similar trends within UWWTPs, noting most PAHs concentrate within sludges (Alawi, et al., 2017) (Wang, et al., 2002). In a study by UKWIR, the average concentration of BaP in 16 UWWTPs across 30 samples was found to be 0.03 mg/l (UKWIR, 2013).

BaP in sludge

The level of BaP in sludge from various UWWTPs has been found to range from 0 to 8 mg/kg of dry sewage sludge. However up to 18 mg/kg of dry sewage sludge has been detected in samples from plants treating waste water from oil-related industries (petrochemical sludge) (Włodarczyk-Makuła, 2008).

Presence of BaP can impact the composting process of the sludge, as well as the quality of the compost produced (Hao, et al., 2019). Composting of sewage sludge can result in degradation of PAH leading to reduction in PAH concentrations of between 16% and 47% (Oleszczuk, 2007).



In another study, it was found that the removal efficiency of BaP at 5 and 20 mg/kg of dry sewage sludge after composting was 51 % and 74 %, respectively (Hao, et al., 2019).

Risk to the environment and human health

The general population can be exposed to BaP through ambient air, water, soils, foods, and tobacco smoke.

The main concern arising from BaP content in compost and fertiliser derived from sewage sludge, is the potential for high cumulative load over time (Wood, 2019). The biggest contribution to human exposure to PAH from sludge-derived fertiliser and compost would be consumption of root vegetables, but the human exposure levels studied did not exceed the study's benchmark dose level for risk of adverse effect (International Agency for Research on Cancer, 2010). (Wood, 2019).

BaP can be present in liquids and presents a threat to the environment, as in this form it can easily penetrate soil and contaminate groundwater or nearby waterways, becoming a micro-pollutant in an aquatic environment (Gourlay, 2004). BaP tends to accumulate to soils and other solid matter. It has low mobility and degradability in the soil (its half-life in soil under aerobic conditions vary from 57 to 530 days depending on the nature of compounds accompanying it, as well as the nature and history of the soil) (Bisson, et al., 2019). BaP presence can also accelerate the decomposition of silicates (i.e. minerals made of silicon-oxygen compounds) and other minerals in sludge (Hao, et al., 2019).

With regard to human health, BaP is considered a DNA adduct, meaning it is able to damage DNA and cause teratogenic effects (Bisson, et al., 2019) and mutations which can lead to cancer. This cancer-creating capability means that BaP is also classed as a genotoxic carcinogen by the EU, i.e. a chemical able to produce cancer by directly altering the genetic material of target cells (Science Direct, 2011).

BaP Mass balance

Lots of reported data and literature exist covering the volumes and source apportionment of emissions of BaP to air, and there is a significant body of research on its behaviour in UWWTP and eventual limit values in sludge derived compost and fertiliser. Data are limited, however, on BaP deposition rates from air and how much BaP actually flows *into* UWWTP.

Therefore, although it is commonly accepted that atmospheric deposition is the predominant source of BaP, and all PAHs, in waste water (Lofrano, et al., 2012), it has not been possible to complete a mass balance model for this substance.

This highlights a potential difficulty in applying the options identification and analysis methodology in other diffuse pollution sources, which is captured in the conclusions section below.

5.2.5 Options for upstream reduction of contamination

The selection of upstream BaP reduction options for discussion is driven by the pathway of the pollutant to waste water, and measures already in place to manage it. The options described below for upstream reduction of sludge contamination with BaP are:

1. Regulation of common sources of BaP emissions to air
2. Capture of BaP from waste water before entering UWWTP



Option 1 Further reduce common sources of BaP emissions to air

As discussed above, the two major contributors to BaP levels in the environment are commercial institutional and household fuel combustion (75 %), and the on-field burning of agricultural residues (12 %) (EEA, 2017b). In tackling the first, legislative controls on combustion emissions and efficiencies have already brought about significant reductions.

Reduction of BaP emissions from combustion

There are a number of measures that indirectly control emissions of PAHs, and thus limit the potential for BaP entrance to waterways in the first instance (Environment Agency, 2019).

In addition to the Directive 2004/107/EC which sets target values for PAHs concentrations in ambient air, the EU policy landscape covers combustion plants at all levels as part of the Clean Air Policy Package⁵⁸:

- Heaters and boiler of capacity less than 1 Megawatt Thermal (MWth) used for domestic heating are regulated by the Ecodesign Directive (2009/125), which stipulates that emissions to air must be included in design parameters and reduced where possible. The standards apply only to new appliances (AIRUSE, 2016). New ecodesign rules for domestic stoves will come into force in 2022⁵⁹.
- Plants between 1MWth and 50 MWth fall under the MCP Directive (2015/2193/EU), which places limit values on emissions and permitting requirements on plant operators. While no direct requirements are placed for PAHs, the Directive requires MCPs to be designed and operated in an energy efficient manner.
- Plants over 50MWth are covered by the IED (2010/75/EU) which requires permitting for emissions from major industrial sources, placing a compliance obligation on emitters to reduce PAHs. As the MCP directive, the IED requires industrial installations to operate in an energy efficient manner.

As a result of these latter two Directives, BaP emissions from industrial combustion have steadily declined in the last decade (EEA, 2016). However, emissions from domestic combustion have increased their share (Clean Heat, 2016). These emissions are linked to the use of wood/biomass as a domestic or commercial fuel (Clean Heat, 2016).

Domestic combustion is indirectly regulated under the National Emissions reduction Commitments Directive (NECD) (2016/2284/EU). The NECD sets national emissions reduction commitments and obligatory reporting for Member States for five pollutants (SO₂, NH₃, NO_x, NMVOCs and PM_{2.5}). Under Article 6, Member States are required to develop National Air Pollution Control Programmes. While these do not directly regulate PAHs, some measures introduced by Member States to reduce PM_{2.5} emissions focus on reducing domestic combustion of firewood and other fuels such as coal (EC, 2021). These could lead to indirect reduction of PAHs.

An additional option to further reduce BaP emissions to air would be to explicitly regulate BaP emissions from domestic and commercial combustion as a part of air quality policy acquis. This could incentivise the replacement of biomass burners with more sustainable heating methods. The option could be based on the learning from PM reduction policies aimed at domestic combustion. For instance, in Austria, Denmark, Germany and Switzerland, emission limits have been set for PM emissions from domestic combustion (AIRUSE, 2016). In Germany, the approach has been combined with mandatory phased replacement of domestic boilers with deadlines for replacement being governed by when the boilers were initially installed (AIRUSE, 2016). Furthermore, many countries have provided subsidies for the installation or the use of renewable energy or heating which provide incentives for households to stop using wood-burners (AIRUSE, 2016)

⁵⁸ https://ec.europa.eu/environment/air/clean_air/index.htm

⁵⁹ [EUR-Lex - 32015R1185 - EN - EUR-Lex \(europa.eu\)](#)



Besides these approaches, emissions from domestic combustion could be reduced by selecting fuels such as pellets associated with lower BaP emissions (AIRUSE, 2016). Evidence suggests domestic abatement technologies such as catalytic convertors and electrostatic precipitators fitted to a wood stove and a pellet stove do not have significant impact on emissions from combustion (AIRUSE, 2016). In addition, improving energy efficiency of buildings would reduce the need for heating, thus reducing BaP emissions⁶⁰. Finally, besides regulation, awareness raising could contribute to shift in behaviour to reduce BaP emissions.

In summary, the most efficient ways to reduce emissions from domestic combustion would include:

- Introducing regulations on the emissions from domestic combustion,
- Improving energy efficiency of buildings;
- Incentivising the replacement of old combustion appliances with more modern and efficient appliances;
- Providing fiscal incentives to replace domestic combustion with alternative heating methods, such as solar;
- Encouraging the use of pellets instead of wood/other biomass;
- Raising awareness.

Reduction of BaP emissions from agriculture

Burning of agricultural crop residues leads to emissions of several atmospheric pollutants such as ammonia (NH₃), sulphur dioxide (SO₂) and carbon monoxide (CO) as well as BaP. The alternative to open burning of crop residues is the re-ploughing of them into the soil. This has the additional potential benefit of retaining the N and P in-situ, and replenishing soil carbon. It is suggested in Amann et al. (2017) that this practice is in effect cost neutral, as the soil benefits balance the cost of equipment, training and staff resource.

The NECD while not targeting BaP directly, encourages Member States to ban open field burning of residues to reduce emissions from other pollutants, and the practice is strongly discouraged via the Common Agricultural Policy⁶¹, but the enforcement of any bans is difficult and the practice continues, particularly in parts of eastern and southern Europe (idaea, 2016). If the bans were fully enforced, this should effectively cease BaP emissions from agriculture.

If Member States were mandated to ban open field burning of agricultural harvest residue and waste and forest residue and applied stricter enforcement, emissions of BaP from agricultural sector would likely cease. Amann et al (2018) recommends that the following mix of measures could be used for this purpose:

- Tight monitoring and enforcement of bans and restrictions (such as agricultural residue burning or use of solid fuels)
- Public engagement campaigns and knowledge sharing

⁶⁰ This is going to be addressed under the European Commissions' 2020 initiative 'A Renovation Wave for Europe - greening our buildings, creating jobs, improving lives' under the EU Green Deal, Available at: <https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A52020DC0662>

⁶¹https://ec.europa.eu/info/food-farming-fisheries/key-policies/common-agricultural-policy/cap-glance_en#documents



Impacts on sludge and soils

If successful, the control of BaP air emissions at source should produce substantial reductions of BaP in waste water and consequently sludge and soils. For instance, Shatalov et al. (2004) demonstrates that stricter controls between the 1980s and 2000s meant that PAHs emissions to air were halved. This led to the same reductions for water. On the basis of the mass balance approach presented earlier, it can be inferred that reduction to BaP emissions to air will lead to reductions of BaP in sludge.

The benefit of an approach targeting BaP emissions to air is that besides the reduction in BaP concentrations in air, the approach would also reduce deposition of BaP into waste water, water bodies and soils (Public Health England, 2017).

Qualitative analysis on feasibility

The key issue with the feasibility of this approach relates to the difficulties in enforcement. The measures relating to domestic combustion are particularly challenging. Their effective enforcement requires household inspections – this in turn requires residents' permission and is resource intensive. The replacement of combustion fuels might be associated with higher costs for households, and therefore incentives might be required to encourage households to uptake alternative fuels or other heating solutions.

It is noteworthy that the Ambient Air Quality Directive⁶² is currently under revision and tighter pollution controls are expected. These may address BaP emissions to some extent. Furthermore, the European Commission's initiative 'A Renovation Wave for Europe - greening our buildings, creating jobs, improving lives' under the EU Green Deal would aim to improve energy efficiency of households which will lead to further reductions in BaP emissions.

Option 2 Remove BaP from run-off water before it enters UWWTP using sustainable urban drainage systems

Another option to reduce the levels of BaP entering waste water is to install systems which will remove BaP from runoff water before it enters the sewers. Reduction in pollution is one known benefit of sustainable urban drainage systems (SUDS)⁶³ designed to:

- manage volumes and flow rates of runoff water,
- reduce flooding risks and impacts,
- reduce the amount of water required to be handled in UWWTPs, and
- provide valuable habitats for wildlife in urban areas (Susdrain, 2020) & (NetRegs, 2020). SUDS remove pollutants from water by means of sedimentation, photodegradation, volatilisation and adsorption (Robinson, 2020).

SUDS can consist of several different elements and stages, and can be designed into, or retrofitted to diverse settings including roads and streets, and residential, commercial, industrial or public developments. The most common elements of SUDS along with examples of related techniques and their potential to reduce pollutants in water, are presented in Table 5-9.

⁶² https://ec.europa.eu/environment/air/quality/revision_of_the_aaq_directives.htm

⁶³ Also known under terms such Best Management Practices, Blue-Green Infrastructure, Low Impact Development, and Water Sensitive Urban Design.



Table 5-9 Elements of SUDS and their features

SUDS element	Examples	Potential for pollution removal from water
Source control	Green roofs	Y
	Rainwater harvesting	N
	Permeable pavements	Y
	Other permeable surfaces	Y
Swales ⁶⁴ & conveyance channels	Swales	Y
	Canals and rills	Y
Filtration	Filter strips	Y
	Filter trenches	Y
	Bioretention areas	Y
Infiltration	Soakaways	Y
	Infiltration trenches	Y
	Infiltration basins	Y
	Rain gardens	Y
Retention & detention	Detention basins	Y
	Detention ponds	Y
	Geocellular drainage	N
Wetlands	Vegetated water bodies	Y

Source: (Susdrain, 2020)

Jeffries & Napier (2008a; 2008b) conducted qualitative research and field monitoring on the pollution removal rates of various SUDS techniques for a range of diffuse source pollutants, including PAHs. The key finding of the work was that the most effective removal of PAHs was at SUDS locations which are periodically wet and dry, such as soil-based detention basins, swales or infiltration basins. This characteristic is most effective in facilitating sediment collection and removal, and break down and degradation, of pollutants compared to ponds or wetlands (Jeffries & Napier, 2008a; 2008b). The studies do not make it clear how the soil used in these solutions is disposed of.

Jeffries & Napier (2008b) also provide guidance on the most effective design parameters for sedimentation and degradation:

- A wide and shallow basin shape, to maximise surface area of contact with soil;
- Periodically wet or dry base of basin;
- Loamy soil type, of recommended minimum depth 300 mm;

Some research indicates that swales are the most effective in the initial rainfall event, and subsequent events lead to continued slow release of temporarily detained sediment (Blue Green Cities, n.d.). While further research is needed, this observation could indicate a limitation to the SUDS impact on PAHs.

⁶⁴ Swales are shallow, broad and vegetated channels designed to store and/or convey runoff and remove pollutants.



Impacts on sludge and soils

Robinson (2020) studied the retention of PAHs in vegetated swales in stimulated storm water conditions. The study results show that, on average, BaP concentrations were 56 % lower in the outflow from the swale than from the inflow. The hydrophobic nature of BaP leaves it likely to rapidly adsorb to particles, such as soil particles in soil-based SUDS. This provides opportunities for the suspended particles to be removed, or for natural degradation to be engineered via the appropriate selection of vegetation used in the SUDS. The study showed that BaP degradation varied between 75.5 – 91 % depending on type of vegetation used (Robinson, 2020). While the impacts on composition of sludge have not been examined, it is likely that the similar scale of reduction would be observed. This approach would thus minimise the content of BaP in sludge that may subsequently be used in agriculture.

Feasibility analysis

Implementation of SUDS is increasing across Europe, and there are examples of projects being installed with overall positive cost-benefit ratio, compared to alternative flood protection options (Davis & Naumann, 2017). However, given the large capital costs associated with this infrastructure solution, the uptake is likely to be higher in countries with high rainfall and stormwater management problems where costs arising from flood damages are high (Defra, 2011). The implementation of SUDS solely as a pollution management solution may be too costly for many EU countries, especially in light of the significant investment costs in waste water treatment in recent years (EC, 2019a). However, if the solution is implemented as a part of a holistic approach to climate change adaptation and flood risk mitigation, the overall benefits are likely to outweigh the costs.

5.2.6 Comparative analysis of options

Table 5-10 presents a comparison of the two options, using a Red-Amber-Green (RAG) rating. Overall, the analysis shows similar results for both options. For example, both options will lead to significant environmental benefits, with Option 1 (Regulating the common sources of BaP) leading to benefits for all media, including air, water and soil and Option 2 (Remove BaP from run-off water before it enters UWWTP using sustainable urban drainage systems) leading to specific benefits for waste water. Both options will lead to a subsequent decrease of BaP content in sludge and therefore improve its recyclability in agriculture and other land uses.

However, both options are linked to feasibility problems. In the case of Option 1, the enforcement of policies which are related to domestic combustion and burning of agricultural residues is challenging and requires substantial resources. The option is linked to costs for end-users who may need to install alternative heating solutions or modern combustion appliances in the case of domestic combustion, or purchase equipment and train up staff in the case of agricultural stubble burning. Part of these costs could be mitigated through fiscal policies where subsidies are provided to help the transition. It is also worth noting that the measures involved do not only reduce BaP levels but have other environmental benefits such as energy and associated carbon and air pollutant emission savings. In the case of Option 2, SUDS are associated with high infrastructure costs. These are outweighed by the benefits from stormwater management and flood prevention. However, this could mean that the uptake of this measure in countries with lower rainfall is limited. It is noteworthy that the transfer of BaP to waste water would be lower in countries with lower rainfall.

It is noteworthy that a combination of options could be applied, with Option 1 being introduced as a mandatory measure, and Option 2 being implemented only in countries or localities with stormwater management issues. Furthermore, both options might be implemented for other reasons, and BaP reduction in sludge can occur as a co-benefit. For Option 1, energy renovation of buildings is driven by climate and energy policies. For Option 2, SUDS is implemented as part



of climate change adaptation initiatives or to reduce water pollution through stormwater overflow.

Table 5-10 Comparison of options

	Option 1 Regulation of common sources of BaP emissions	Option 2 Remove BaP from run-off water before it enters UWWTP using sustainable urban drainage systems
Environmental		
Final contamination levels	Decrease in BaP from key sources.	Decrease of BaP in waste water only.
Air	Decrease in BaP from key sources	No impact.
Water	Decrease in BaP from key sources	BaP captured before entering waste water streams.
Soil	Decrease in BaP from key sources	Decrease arising only from the lower content of BaP in sludge applied to soil.
Biodiversity	No info. Impacts linked to the decreased demand for wood-burning are possible.	No info. Indirect impacts from flood prevention are possible.
Climate	The use of more energy efficient domestic combustion appliances, demand and the change to renewable heating have positive impacts on CO _{2e} emissions.	Impacts from reducing waste water load for UWWTPs would lead to reduced energy use and subsequently reduced GHG emissions.
Resource recovery from sludge	The option will result in significant reductions of BaP in sludge and improve its recyclability.	The option will result in significant reductions of BaP in sludge and improve its recyclability.
Cost for industry	Benefits for industries specialising in renewable heating/modern domestic combustion appliances.	Benefits for industries specialising in SUDS design and development.
Cost for end-user	The costs for end-users can be minimised through fiscal policies.	The costs of SUDS are high since they are associated with building development. They are likely to be reflected in change of prices for elements such as water utilities, road building and housing developments.
Feasibility	The enforcement of policies linked to combustion and stubble burning are difficult to enforce and require a lot of resources.	Voluntary uptake in countries with no stormwater management problems is unlikely considering cost.
Health	Overall health benefits linked to the decrease of BaP in all media.	Health benefits only linked to the decrease of BaP in sludge used in agriculture. Additional health benefits in some countries may link to the mitigation of flood hazards and flood events.

Legend: Red – Negative impacts; Amber – Positive impacts but some issues; Green – Positive impacts; White- No impact; Grey- No info;



5.2.7 Conclusions

The largest sources of BaP are combustion for heating in households and commercial buildings and burning of agricultural residues. BaP has been identified as a substance of very high concern (SVHC) due to its carcinogenic, mutagenic, toxic for reproduction and bioaccumulative properties. Among other routes, humans can get exposed to BaP by consuming food grown in contaminated soil. BaP needs to be therefore effectively removed from sludge before it is spread on agricultural land.

Extensive regulations for BaP emissions to air from industry already exist, however, there is scope to further limit emissions from domestic and commercial sources.

The case study considered two upstream options for avoiding contamination of sewage sludge with BaP: 1. further reduction of common sources of BaP emissions to air; and 2. removal of BaP from run-off water before it enters UWWTP using SUDS. Both options are linked to substantial environmental benefits in addition to the reduction of BaP, but their feasibility and associated costs need to be carefully considered, with the former option involving higher costs for enforcement and the latter higher infrastructure costs. In order for either of the two options to be widely implemented, regulatory intervention is required. The costs for end-users are considerable in both instances, though with regard to the first option, subsidies for households can be provided as mitigation. Both measures will incur significant benefits for industries; those specialising in renewable heat and modern combustion boilers in the case of Option 1, and SUDS in the case of Option 2.

As such, a conclusion on the more favourable option has not been reached due to the similar profiles of both. It is noteworthy that a combination of options could be applied, with Option 1 being introduced as a mandatory measure, and Option 2 being implemented as a voluntary measure. As a measure to minimise the overall human exposure of BaP, option 1 is preferred.

The following limitations and gaps in evidence have been identified:

- While PAH, and specifically BaP emissions to air are comprehensively monitored and reported throughout Europe, and there is consensus that the most significant contribution of PAHs in waste water is from deposition from air, there is little-to-no firm data on what proportion of emissions to air end up entering UWWTPs on the European scale.
- Further evidence is required regarding the impacts of policies targeting BaP on sludge content and recyclability in agriculture.
- A more in-depth cost-benefit analysis of the two options considered here, beyond the scope of this study, is required to fully appraise the comparative benefits and challenges.
- Further information is needed about the key problems and learnings experienced from the implementation of similar measures such as those included in the National Air Pollution and Control Programmes developed by the Member States under the NECD.

5.3 Key principles of the methodology

The methodology trialled in this study for identification of the “upstream” measures to prevent contamination of sewage sludge involved five steps guided by the following principles:

- **Step 1 Understanding the substance and its properties** (e.g. physio-chemical properties, toxicological characteristics, quantities produced, used, traded, disposed of) **and current regulatory framework** in place to limit the negative impacts of the substance (i.e. regulation limiting the production, use and emissions of the substance, regulation applicable to operations of UWWTP and sludge management, regulation applicable to the end use of sludge as well as recycling). The purpose of this step is to understand the scale of the problem that may need to be tackled by new “upstream”



measures and the extent to which this problem may already be acted on as part of existing regulatory and other measures.

- **Step 2 Mass balance approach:** When designing the pollutant case studies it was useful to visualise the fate of the pollutant from its production until it reaches sewage sludge, showing the loads and/or concentrations of the substance at respective steps of the pathway. A simplified mass balance approach was thus attempted for both substances covered. The mass balance approach is a helpful tool in moving towards circularity as it shows how the materials flow through a value chain. In the context of the pollutant case studies, the preparation of the mass balance required identification of the major pathways of the pollutant to waste water – what is its origin? How does it enter waste water? This involved collection of information on:
 - Production of the substance (in case of DEHP) and source of releases to the environment (in case of BaP).
 - Use of the substance (e.g. in industry or consumer products manufacturing)
 - Disposal of the substance or products containing it (i.e. end-of-life)
 - Points at which the substance enters waste water

For each of these points, the literature research has focused on identifying information on the mass of the substance (e.g. loads released to waste water), and where this was not available shares (e.g. share of total substance produced used in certain ways) or concentrations (e.g. content of a substance within a product). Further steps applied in mass balance calculations included:

- Fate of the substance in UWWTPs – to what extent a substance degrades, sorbs to sludge or remains in treated water? How is the initial load present in waste water split across these pathways?
- Presence in sludge – what is the resulting mass of a substance per tonne of raw or dry sludge?
- Fate during treatment – how does a mass of a substance in sludge change as a result of treatment e.g. anaerobic digestion or composting or disposal method (e.g. incineration)
- Presence in treated sludge e.g. digestate, compost, incineration sludge – how much substance remains in the product of sludge treatment?
- How much of the substance is absorbed by plants grown in soil fertilised with treated sludge or sludge derived product?
- What is the likely exposure to the substance for humans through food consumption (and subsequently are the levels likely to cause harm)?

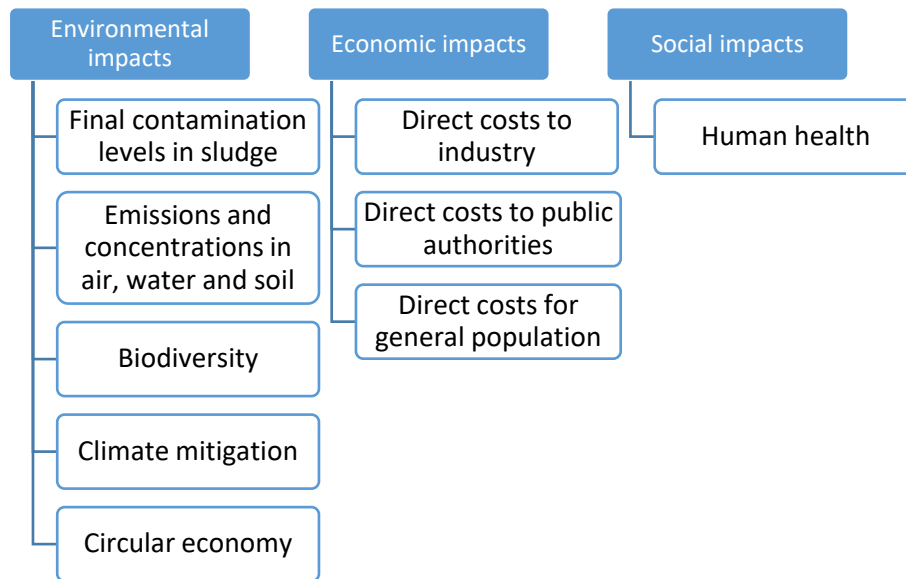
The above approach has focused on the pathway: source - sludge – human exposure. For the full mass balance of the substance additional information would be needed, for example on the direct emissions to air and soil at each step of the value chain and the associated exposures and risks. Due to limited data available, the calculations for BaP could not be conducted. The mass balance approach for DEHP was completed, however due to numerous assumptions that had to be taken to resolve data gaps, the mass balance serves only as an illustration of a possible approach.

- **Step 3 Identifying “upstream” measures for reduction:** Considering the sources of the substance in waste water, this step should concentrate on identification of options for management of the largest sources (in terms of pollution load) which are not yet restricted with existing measures. The objective is to prioritise research in areas which will make the largest impact on reduction in contamination, and which are not yet sufficiently regulated (recognising that it may take several years before the impacts of existing regulations on the levels of contaminants in waste water take effect).



- **Step 4 Comparing the options:** the identified options need to be compared via a holistic approach and considering the human exposure, environmental, economic as well as social impacts. The impacts to be considered are illustrated in Figure 5-11 below.

Figure 5-11 Typology of impacts to be assessed for comparing “upstream” measures



A tabular format supports comparison of costs, benefits and trade-offs between each option and how these compare across options. Such a comparative table should, as far as possible, include quantitative evidence in comparable units so that costs and benefits can be directly compared. In the absence of quantification, qualitative assessment identifying whether an impact is likely to be positive or negative, and indicating the likely scale (i.e. low, medium, high) could also facilitate comparisons. The extent to which the comparison of options involves detailed investigations and a full cost-benefit analysis should be determined based on the scale of the issue determined by the end of step 2 (applying proportionality principle, the larger the scale of the problem the more detailed the investigation should be), as well as resources and time available.

- **Step 5 Concluding on the preferred option(s):** based on the comparative table populated in step 4, conclusions on the best options can be reached. This needs to consider the trade-offs of each option. For example, it may be that impacts of the substitution of a substance upstream, while bringing positive impacts for water quality and sludge, lead to air quality issues. Depending on a specific country situation this may not be a trade-off that policy makers can accept. Similarly, co-benefits might make a measure much more feasible and cost-efficient. When concluding on the preferred option, it is also important to consider the robustness of the evidence identified. If large data gaps exist which prevent comparison between the options, the decision may need to address these data gaps or accommodate that uncertainty.



6. Conclusions

6.1 Existing practices in sewage sludge management in Europe

Potential for further contribution to circular economy objectives

Considerations on sewage sludge management in Europe increase in importance as countries further improve the level of treatment applied to waste water they collect, and Europe starts to fulfil its ambition to manage resources in a more circular way. Recent developments in European water policy, such as the evaluation of the SSD and its potential revision and the review of the UWWTD provide opportunity to enhance synergies between regulations, while at the same time redefining their underlying objectives.

In the context of circular economy, sewage sludge management in Europe offers further potential for energy recovery and nutrient recovery. The additional energy recovery potential is estimated in the study as being up to 3 500 GWh per year in the EU-27 if full implementation of the UWWTD is assumed. Nutrient recovery estimated potential, when considering conversion of the sludge currently reported as landfilled, is up to 32 000 t of mineral P, and 44 500 t of N/year in the EU-27. When including a speculative 50 % of sludge reported as being disposed of by 'other' means, these rise to 69 300 t of mineral P and 96 300 t of N / year. In a scenario assuming that 50 % of all incinerated sludge, all currently landfilled amounts and a speculative 50 % disposed as 'other' are mono-incinerated with landspreading of the ash, the recovery potential is estimated to be up to 105 500 t of mineral P. These estimates underline the difficulty in accurately predicting, and therefore facilitating, recovery potentials caused by the current lack of granularity in reporting requirements. While the estimates derived in the study are highly uncertain and based on a number of assumptions owing to the absence of data, they provide an illustration of the potential opportunity. Further work would be required to validate the assumptions made and the resulting estimates, as well as to fine tune the approach applied.

The 2019 Fertilising Products Regulation prohibits use of sewage sludge derivatives (digestate and compost) in CE marked fertilising products. This may prevent greater use of sewage sludge in the context of circular economy. The European Commission has proposed adopting delegated acts to amend Annexes I to IV to the regulation to adapt them to technical progress in the light of new scientific evidence. These acts would allow EU fertilising products to contain precipitated salts and derivatives and thermal oxidation materials obtained from processing of sewage sludge.

Current approaches to management of sewage sludge

Based on the Eurostat data, less than one third of all raw sewage sludge produced in Europe continues to be used in agriculture after treatment. Other management methods applied are composting, incineration and landfilling, with some of the composted sludge also being applied to land. However, analysis of the data available from Eurostat demonstrates that the records are incomplete. This is because not all countries have reported the data and because it is unclear how the reported data should be interpreted. The latter concerns the reporting on composting, which is a treatment rather than a disposal method, and what is covered by the category "other".

To illustrate how varied are countries' approaches to the management of sewage sludge, the study explored in more detail the existing practises in Estonia, Germany, Italy, and Sweden. Based on these four case studies it can be concluded that the countries have gone beyond the requirements of the SSD regarding the quality of sludge applied to land: this is achieved by setting stricter limit values for heavy metals or for additional pollutants. Use of sludge on land can be a contentious issue, with public concern about health and environmental impacts likely to drive policy makers away from that management option and consider alternatives. Evidence has also been identified on the lack of land suitable for spreading, particularly in the more



densely populated areas. Alternative disposal routes such as incineration are more costly compared to land spreading and require specialist infrastructure to be available in the proximity of where the sludge is produced. Furthermore, incineration also attracts public concern, especially with regard to emissions to air. Absence of appropriate waste management infrastructure that could be used as an alternative to land spreading can lead to significant problems in managing sewage sludge.

The case studies demonstrated that AD is deployed to reduce the quantity of sludge and produce biogas. In the investigations made in this study on the further potential for energy recovery from sludge, lack of information on the mass of sludge currently subject to AD has been identified as a major gap in evidence. P recovery from sewage sludge ash has a strong potential to strike a balance between moving away from land spreading while supporting circular economy principles. Europe is dependent on P imports and therefore the recovery of P on larger scale, once the technologies are fully developed, is strategically attractive. However, the estimates for recovered P are significantly lower than the current use of P in Europe and the markets for recovered P are not yet established.

6.2 Decisions on “upstream” measures to prevent contamination of sewage sludge

Spreading treated sewage sludge or sludge-derived fertiliser products can lead to real or perceived risk to the environment and human health. While treatment of sludge (e.g. lime treatment) can reduce the contamination, the study explored the potential for identifying “upstream” measures for prevention of sludge contamination. The case studies developed focused on two priority substances under the Water Framework Directive: DEHP and BaP. Having been regulated at EU level for many years, both substances have been well studied although evidence gaps still exist, and evidence identified has often been dated. This was particularly the case for DEHP for which there was little information published recently, perhaps due to the regulatory restrictions on the use of the substance meaning this is no longer seen as a research priority.

In both case studies, evidence on the properties of the substance, pathways of human exposure, how it enters the waste water and how it behaves at UWWTP have been readily available. For DEHP, resources available on the ECHA website, including evidence accompanying authorisation requests, have been a useful resource on these topics.

Existing regulations were mapped for each relevant part of the pathway – from source to end use and/or disposal of sewage sludge. These were set in the context of historic emissions trends of the substances. When documenting the results, it has been useful to distinguish between policy regulating a given substance directly, versus policies that provide a wider framework but may have indirect impacts. For instance, while the SSD does not regulate either substance directly, implementation of its sludge treatment provisions may reduce presence of the studied substances in sludge. The evidence reviewed suggests that some treatment methods that are applied in European countries ahead of spreading on agricultural land (i.e. anaerobic digestion) are effective in removing DEHP.

The mass balance approach illustrates the largest sources of the substances to waste water and the changes in substances mass along the pathway. The data to underpin the mass balance has been limited despite both DEHP and BaP being well-studied chemicals. Thus, if the approach was to be replicated for less well-studied chemicals, similar challenges would be encountered.

It was possible to identify upstream options for both chemicals. However, limited quantification of impacts has been possible due to data availability and the time available to conduct the case studies. It was also apparent that data gaps limit the scope for reaching a conclusion on the most favourable options. In both case studies, key information gaps identified related to the quantification of impacts on sewage sludge and its recyclability, for which further studies are

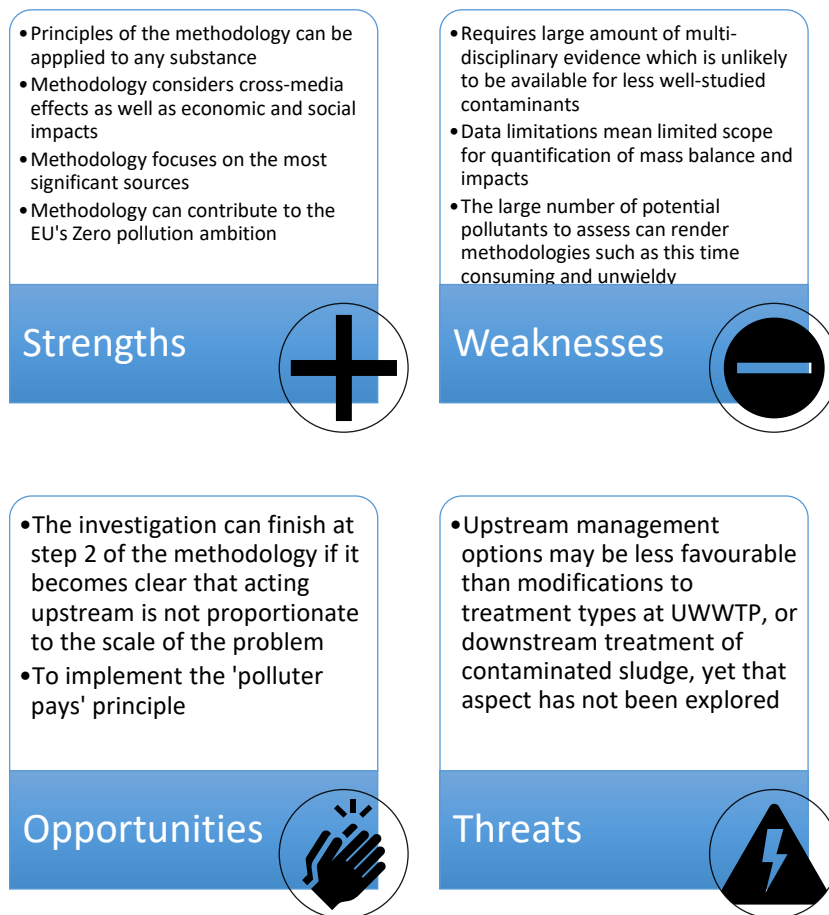


required. In the case of DEHP, data gaps also concerned environmental and human health impacts of some alternative substances; impacts of these on sludge composition, recyclability and overall feasibility of implementation. It is expected that similar gaps would be identified when the methodology is applied to other substances, especially contaminants of emerging concern for which, at this stage, less evidence is available. Assessment of upstream options should also take account of potential benefits other than just the removal of the contaminant in questions, such as climate change adaptation, energy reduction and reduction of other pollutants.

Furthermore, the case studies have not considered the options related to the removal of these chemicals in UWWTD or sludge treatment. In practice, treatment options should also be considered alongside the upstream options, as they might be more appropriate for some pollutants.

A summary of the strengths, weaknesses, opportunities and threats related to the application of the methodology for identification of “upstream” measures to other pollutants, including pollutants of emerging concern is presented in Figure 6-1.

Figure 6-1 Strength, Weaknesses, Opportunities and Threats (SWOT) of the proposed methodology when applied to other substances including pollutants of emerging concern



Source: Own elaboration



6.3 Future considerations

Based on the work undertaken, the following suggestions would improve future studies of this type:

- **Improvements of the data on current sludge management in Europe:** There is a need to develop a more comprehensive source of information on current management of sewage sludge in Europe. This could be achieved by making changes to the countries' reporting requirements towards Eurostat or reporting on implementation of the SSD and/or UWWTD. Regarding the Eurostat data, mandatory reporting would improve data availability and should be extended to explicitly capture two separate aspects: a) pre-treatment methods applied to sludge (e.g. drying, stabilisation, anaerobic digestion, composting), and b) final disposal routes (in addition to existing categories, reporting could distinguish between incineration, incineration with energy recovery, incineration with nutrient recovery, land uses other than for agriculture). Ideally the reporting should also allow for correlation of the two data streams to understand the flow of the sludge through the treatment system, e.g. what share of sludge spread on land has been subject to anaerobic digestion. Reporting further detail of the meaning of the category 'other' should also be provided, or the category be split up into more meaningful final disposal routes. Under the revised SSD and UWWTD such reporting requirements could also be introduced, although it is recognised that reporting would apply only to EU-27 and the geographical scope of Eurostat is broader.
- **Further elaboration of the energy and nutrient recovery potential estimates:** Whilst the study attempted to estimate further energy and nutrient recovery potential from sewage sludge, the exercise here was limited. Further work on these aspects could involve consultation with countries on the existing levels of energy generation from sludge, techniques used and their energy yields, the share of sludge produced which could be deployed for energy generation and nutrient recovery, and the levels of investment needed to maximise the potential recovery taking into account realistic practical constraints like the size of the UWWTP. Based on the evidence gathered, the calculations performed could be further fine-tuned to reflect national circumstances.
- **Application of the "upstream" methodology to a pollutant of emerging concern:** While the core principles of the methodology worked for the well-established contaminants, a further step would be to apply it to pollutants of emerging concern. This would require understanding of the significance of the quantities or concentrations and of the chemical behaviour of the pollutant in the influent to UWWTPs. The scope should be expanded by considering the modifications to treatment at UWWTP and treatment of sludge to ensure that out of all options available, the most favourable one(s) is selected. At the same time, measures addressing different pollutants need to be seen in an integrated way as they are often connected, generating co-benefits or trade-offs. Finally, the 'polluter pays' principle should be considered.



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Annex A: Sewage sludge treatment techniques

This annex presents information about the most common sewage sludge treatment techniques referred to in the report.

Technique	Description
Lime treatment	Lime treatment is a sludge stabilisation technique which reduces the pathogenic content of sludge, availability of metals, and the relevant environmental risks, as well as enhancing its agricultural benefits (Farzadkia & Bazrafshan, 2014). The lime stabilisation process consists of adding a lime slurry to the liquid sludge to achieve a pH higher than 12 (Farzadkia & Bazrafshan, 2014). The necessary lime dose varies with the kind of sludge and also concentration of solids (Farzadkia & Bazrafshan, 2014). High pH produces an environment that stops or considerably decelerates the reactions of microorganisms that can otherwise lead to production of odour (Farzadkia & Bazrafshan, 2014).
Solar/Thermal drying	Drying constitutes an important process for waste water sludge management, as it can reduce the mass and the volume of the product and consequently the cost of storage, handling and transport (Bennamoun, 2012). The key difference between solar and thermal dryers relates to the type of energy used in the process (Bennamoun, 2012).
Wet oxidation	Hydrothermal process in which organic and oxidizable inorganic components are degraded in the liquid phase at high temperature and pressure with the use of air or oxygen (Menonia & Bertanzab, 2016)



Annex B: Energy and nutrient recovery techniques

Technique	Description
Anaerobic digestion	Anaerobic digestion is the bacterial breakdown of organic materials in the absence of oxygen resulting in the production of biogas (Eubia, 2020). The process has the capacity to reduce the amount of organic matter up to 50 %, however it leads to accumulation of metals in the digestate (SSWB, 2020).
Composting	Composting is a biological process that uses naturally occurring microorganisms to convert biodegradable organic matter into a humus-like product (Uçaroğlu & Alkan, 2016). The composting process destroys pathogens, converts nitrogen from unstable ammonia to stable, organic forms of nitrogen, and reduces the volume of sludge (Uçaroğlu & Alkan, 2016). A variation of composting is the windrow composting. This is the production of compost by piling organic matter or biodegradable waste, such as animal manure and crop residues, in long rows. This method is suited to producing large volumes of compost. it is difficult to compost sewage sludge as sole feedstock – it is best mixed with other materials, e.g. green waste, when composted.
Lime treatment	Lime treatment is a sludge stabilisation technique which reduces the pathogenic content of sludge, accessibility of metals, and the relevant environmental risks, as well as enhancing its agricultural benefits (Farzadkia & Bazrafshan, 2014). The lime stabilisation process consists of adding a lime slurry to the liquid sludge to achieve a pH higher than 12 (Farzadkia & Bazrafshan, 2014). The needed lime dose varies with the kind of sludge and also concentration of solids (Farzadkia & Bazrafshan, 2014). High pH produces an environment that stops or considerably decelerates the reactions of microorganisms that can otherwise lead to production of odour (Farzadkia & Bazrafshan, 2014).
P recovery from sewage sludge ash	Ashes from sewage sludge mono-incineration, mainly due to significant reduction (70 to 90%) in the volume of the incinerated materials, contain much higher amounts of phosphorus (Günther, et al., 2018). P recovery can be facilitated via chemical precipitation, adsorption and biological process (Günther, et al., 2018). Plants in Germany use biological processes with chemical back-up to extract P from the sewage sludge ash (Günther, et al., 2018).
Sludge use as fuel pellet	This concerns the pelletisation of sewage sludge through drying and using it as a fuel (Ghiocel & Panaitescu, 2018). A common example is its use as a fuel in cement production (Ghiocel & Panaitescu, 2018).



Annex C: PVC Application Matrix

PVC Application		Example Competing Materials	Share of Used PVC Mass	Market Share of PVC
Building and construction				
profiles and sheets (rigid)	window frames	wood, aluminium, steel, polyurethane, polyolefins	9%	major
	cladding	aluminium, steel, fibre cement, wood	9%	major
	sheets	acrylics, glass, polystyrene (non transparent), polycarbonate (transparent)		
	conduits, shutter, rails, skirts	Other plastics including polystyrene, Styrene-butadiene, polypropelene, polycarbonate, wood, metals		
sheets and foils (flexible)	flooring (incl. Sport)	carpet, wooden flooring, laminate, linoleum (soft), polyolefins, tiles (hard), natural stone, rubber, cork	8%	major (soft) medium (total)
	roofing sheets	bituminous sheets, synthetic rubbers, other plastics	1%	major
	membranes	polyolefins, polyester (coated)/ glass fibre (coated)/ polyamide (coated)	1%	major
	wallpaper	paper, mineral plaster, acrylics	2%	medium
pipes and fittings	waste-/rain-water	polyolefins, cast iron, concrete, stoneware/clay, Cu, Al, Zn	12%	major
	drinking water	(stainless) steel, copper, polyolefins, other plastics	4.5%	medium
	gas pipes	polyolefines	< 0,5%	small
	irrigation and draining pipes	polyolefines	2.5%	major
Toys/sports goods				
	dolls, bath ducks, snorkles	polyolefins, wood	< 0,1%	small
	inflatable beach toys, balls, paddling pools	rubber, polyolefins	3%	major



PVC Application		Example Competing Materials	Share of Used PVC Mass	Market Share of PVC
	rubber boats, rafts	composites, polyurethane, rubber		
	building blocks, play figures	polystyrene, polyolefins	negl.	Negl.
	Camping/tents	rubber, thermoplastic elastomers	< 0,5%	medium
	luggage	leather, cotton, polyester, polyurethane	1%	medium
Consumer goods				
furniture	edge protection, furniture profiles	wood, other plastics	1.5%	medium
	coating	melamine paper, veneer, varnish (epoxy, acrylic, alkyd, polyurethane)	0.5%	major
	garden hoses		1.5%	major
	garden furniture	wood, steel, aluminium, polyolefins, polyamides	negl.	Negl.
Clothing	wellingtons, ski boots	leather, polyamides, ethylene vinyl acetate, rubber	2%	medium
	soles/bottoms	leather, polyurethane		
	rain covers	cotton, polyurethane, polyester, polyamides (all coated)	< 0,5%	small
	fashion ware	wool, polyester, cotton, polyamides, silk, latex		
office equipment	transparencies, trays, folders	polyolefins, polystyrene, cardboard	2%	medium
	credit cards	Other plastics including Polyethylene terephthalate	< 0,5%	major
household goods	shower curtains	cotton, polyester	< 0,5%	major
	packaging-/electrical tapes	Polyethylene terephthalate, polypropylene	1%	major
	sealants	silicones, polyurethane, thermoplastic elastomers	< 0,5%	medium
Packaging				
container	bottles	Polyethylene terephthalate, glass, polyolefins, ceramics	1.5%	small



PVC Application		Example Competing Materials	Share of Used PVC Mass	Market Share of PVC	
	food packs	Polyethylene terephthalate, aluminium, paper, polystyrene, polyolefine, polyamides	6.5%	medium	
	shrink foils	polyolefins,		small	
	blister packs	Cyclic olefin copolymers, polypropylene, paper		medium	
Transport					
cars and trucks	plastisol (sealing, underbody protection)	polymer bitumen, PB rubber, Styrene maleic anhydride, Zn	2.5%	major	
	parts	Other plastics, including	1%	small	
	tarpaulins	acrylics, polyurethane (all coated)	0.5%	major	
	dashboard	Other plastics, leather	1.5%	medium	
	artificial leather	polyurethane, leather, cotton, wool, polyester		major	
	cable harness	Other plastics	1%	major	
yachting	foams, fenders, ...	wood, polyurethane	< 0,5%	major	
trains	seat covering	polyurethane, polyester	< 0,5%	small	
Electric- and electronic equipment					
	cables	Rubber, other plastics	11%	major	
	casings	Other plastics, including polyolefins, metals	0.5%	small	
	cable ducts	polyolefins		major	
Medical applications					
	blood and infusion bags and medical devices	multilayer polyolefins, glass	0.5%	major	
	gloves	rubber, polyurethane			
Agriculture					
	green houses sheets	polyethylene, glass, PMMA	negl.	Negl.	
	Foils	polyolefines	< 0,5%	small	
Others					
		undefined		~ 7,5%	
			sum	100%	

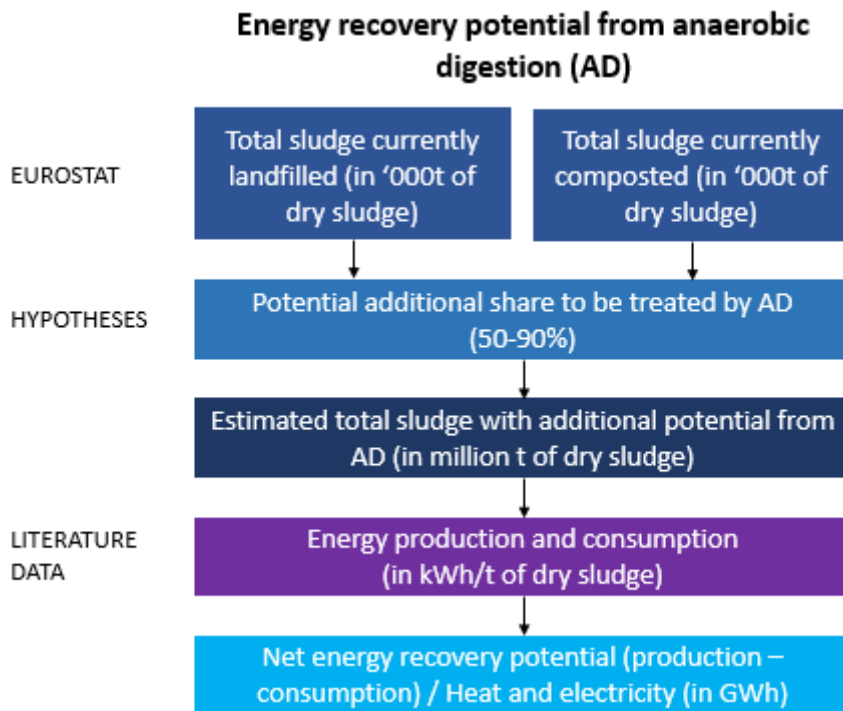
Source: (EC, 2004b)



Annex D: Energy and nutrient recovery calculation methodologies

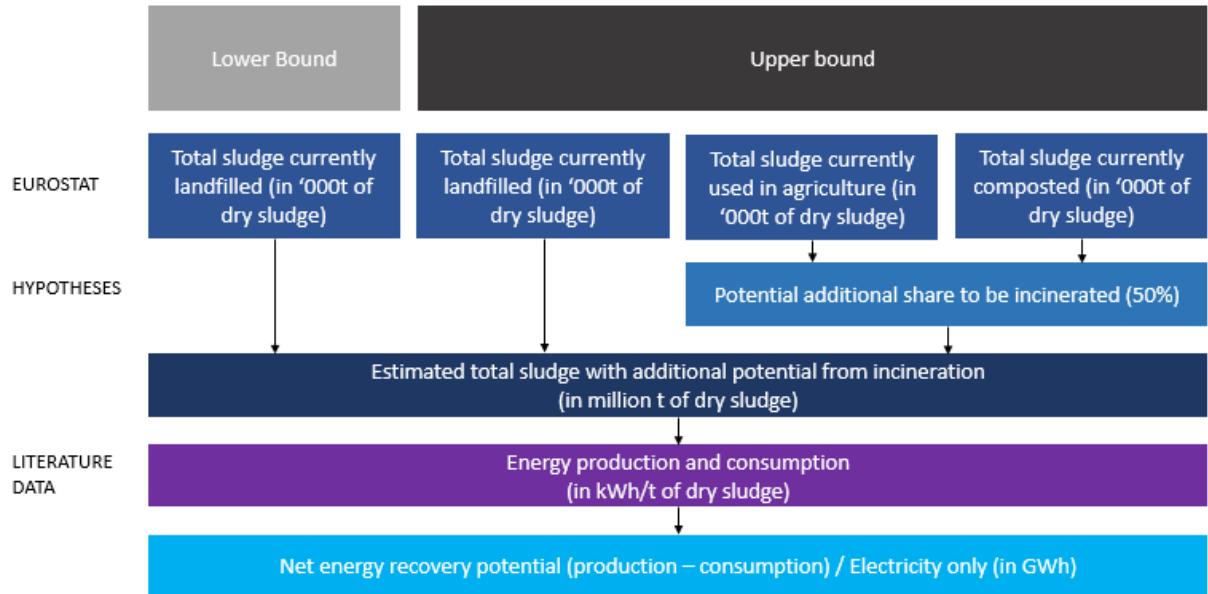
LEGEND

- Input data
- Weighting factors
- Estimated data
- Conversion factors
- Estimated potential additional recovery

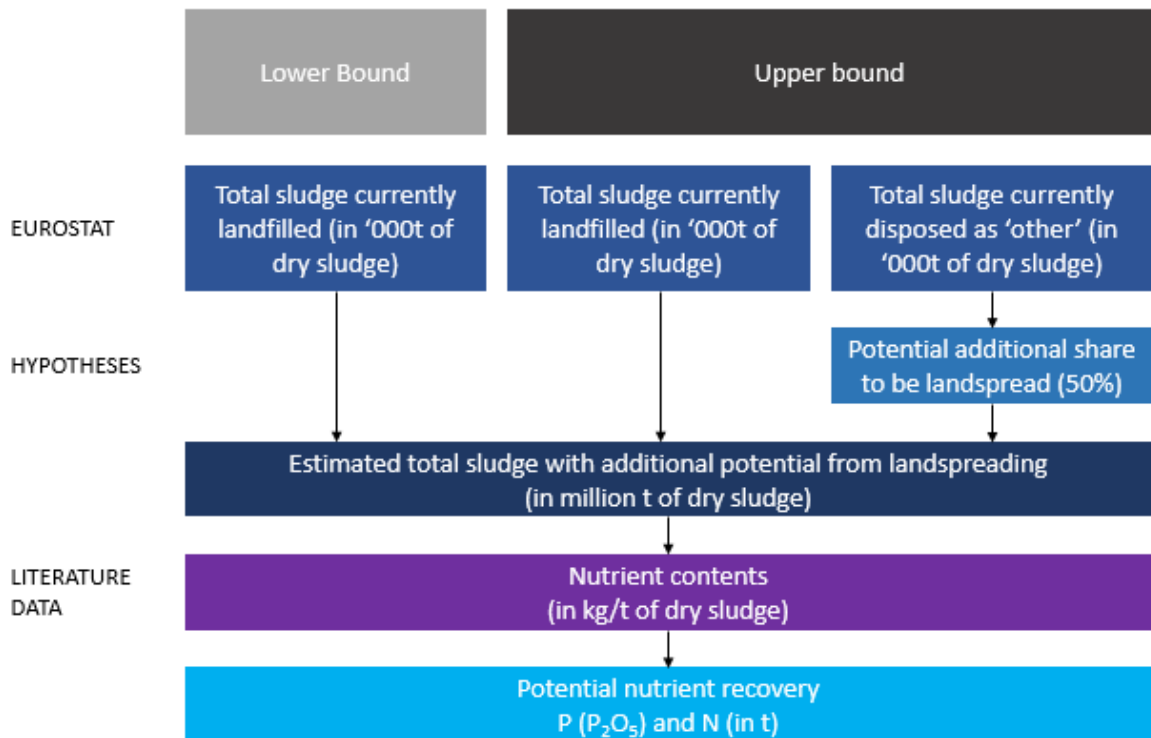




Energy recovery potential from incineration



Nutrient recovery potential from landspreading



Nutrient recovery potential from landspreading of mono-incinerated sludge

