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Feasibility study in support of future policy developments of the Sewage Sludge Directive (86/278/EEC)

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Contents

Abstract

This study aims to bring forward a preliminary assessment of policy options for a possible review of the Sewage Sludge Directive. The main problem identified is that current sludge management is not fully aligned with current EU policy objectives, particularly on environmental and human health protection and circular economy. Two policy options were evaluated in detail:

(i) monitoring and control of sludge, or derived materials thereof, returned to agricultural land complemented by targets for the return of the critical raw material phosphorus to agricultural land; and

(ii) environmental and health protection through the mandatory transformation of sewage sludge into EU fertilising products that classify as phosphorus fertilisers, presently mostly following sludge incineration.

The report quantifies and discusses costs and benefits for both policy options, including aspects related to human health and environmental protection, nutrient recycling potential, methane emissions, potential to stimulate innovation, and social and distributional impacts. In addition, further information is provided on information gaps and research needs. The report may help to further increase the knowledge base on sustainable sewage sludge management and stimulate informed discussions amongst all relevant stakeholders.

1 Executive summary

This exploratory study aims to bring forward a preliminary assessment of policy issues and potential responses for a possible future review of the Sewage Sludge Directive. The main problem identified is that currently sewage sludge management practices may not fully be aligned with policy objectives aimed at achieving a high level of environmental and human health protection, and a more circular management where resources, particularly phosphorus, are retained in the nutrient cycle. Consequently, pollution of soils and their agricultural products may occur, possibly threatening the EUs natural capital and citizens' health. Furthermore, the management of waste materials exploits only to a limited extent the potential to replace primary raw materials, including rock phosphate as a finite commodity. In the absence of any legislative actions, it is likely the problems will persist. This may put at the risk the achievement of the policy objectives of environmental and health protection, as well as increased material circularity, that were flagged as relevant in the recent evaluation of the sewage sludge Directive. Main problem drivers that can be addressed in a revised framework include the lack of monitoring and risk evaluation of sewage sludge returned to agricultural land as well as a lack of consideration of externalities from current sewage sludge management.

At the outset of the study, four policy options were considered, but the following two were discarded due to uncertainties regarding their effectiveness to address the problems observed: (i) setting up a monitoring framework, without the associated setting of EU-wide limits for contaminants of greatest concern measured in sewage sludge as part of the SSD and (ii) repealing the SSD and self-regulation based on voluntary standards. Two policy options that include a set of measures to achieve the objectives of increased environmental and health protection and reduced resource loss are retained and examined. The first policy option (PO1) focuses on the monitoring and control of sewage sludge returned to agricultural land, and includes targets for recycling or recovering phosphorus (P) from sewage sludge, either by landspreading sewage sludge or recovering phosphorus from incineration ashes. The second policy option (PO2) ensures environmental and health protection as well as phosphorus recovery by making sewage sludge incineration followed by phosphorus recovery mandatory. This policy option largely circumvents the monitoring and control of sewage sludge by imposing incineration as an effective treatment to eliminate organic contaminants, biological pathogens and, microplastics. Both policy options have variants that have lower requirements for waste water treatment plants below a specific size. Costs and benefits from the different policy options are discussed, including aspects related to human health and environmental protection, potential to recover phosphorus from sewage sludge, methane emissions, compliance costs, potential to stimulate innovation, employment, social and distributional impacts, and others.

Due to monitoring and control of sewage sludge quality (PO1) or sewage sludge incineration with a subsequent P recovery (PO2) both policy options perform better compared to the baseline with regard to environment & health, but a greater removal of contaminants is feasible under PO2. Phosphorus fertilisers produced out of sewage sludge ash have a greater agronomic efficiency than sewage sludge and therefore P recovery and P loss can be improved compared to PO1. The production of mineral fertilisers out of sewage sludge ash allows the transport of nutrients over long distances, allowing the existing nutrient surpluses and deficiencies at EU levels to be better balanced. With regard to other nutrients (e.g. nitrogen (N)) or carbon, PO1 performs significantly better, since both elements are no longer available for application to the soil when incinerated under PO2. Due to the necessary investments in sewage sludge incineration and P-recycling capacity, job creation for PO2 is higher, but this comes with increased investment and annual costs for waste (water) treatment plants and thus the society in general.

Overall, this study provides a starting point for a potential revision of the policy framework for sewage sludge management in the EU. The information provided in the report confirms that feasible options exist to revise EU policies on sewage sludge in line with the EU's ambition on zero pollution and circular economy. This report has not proceeded with the selection of a preferred policy option because (i) uncertainties continue to exist in relation to the magnitude and amounts of contaminants (e.g. organic compounds, metals and microplastics) that pose risks, and ensuing risk levels accepted by stakeholders and legislators, and (ii) information and data inputs from stakeholders on this document may further enrich the information base and the policy recommendations overall. Hence, this study overall aims to provide a starting point for further evidence-based discussions amongst stakeholders.

2 Study objectives

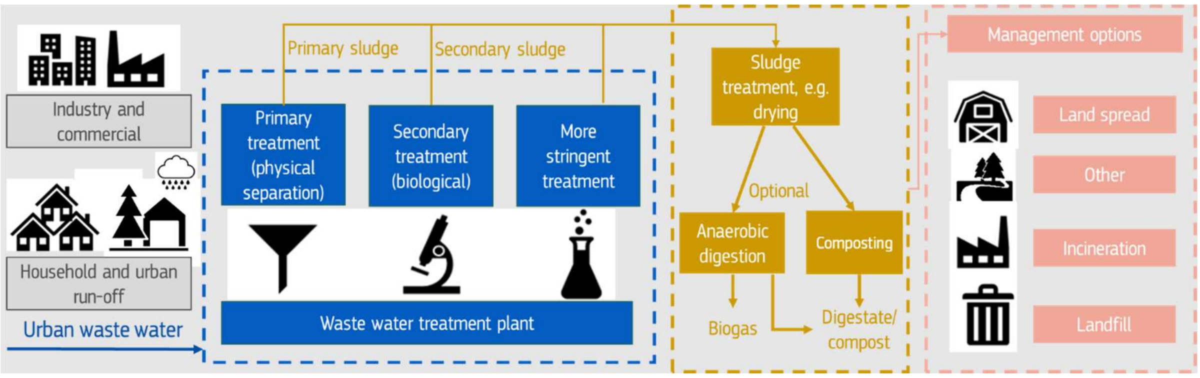
The study aims to explore and assess the necessity and feasibility for a revised policy framework on sewage sludge, starting from a detailed assessment of the problem definition. In contrast to the evaluation study of the Sewage Sludge Directive (86/278/EEC), this study has a forward-looking perspective and assesses the future evolution of the problems, drivers, and their magnitude ("baseline" without further policy intervention). Considering a possible need for further regulatory actions at EU level, the study will also explore the feasibility of certain policy options and report on next steps that would be required for the achievement of the objectives of the Directive in light of the current political vision and priorities on the EU Green Deal.

The analysis is based on a number of assumptions and modelling hypotheses which are presented within the report, or stem from existing or proposed regulations. The results presented should in no way be interpreted as pre-empting or anticipating the formulation of future regulatory proposals, which will be developed, proposed, discussed and adopted in the future, in particular regarding the scope, timeline and mandates of the potential measures envisaged herein.

3 Introduction: technical and legal context

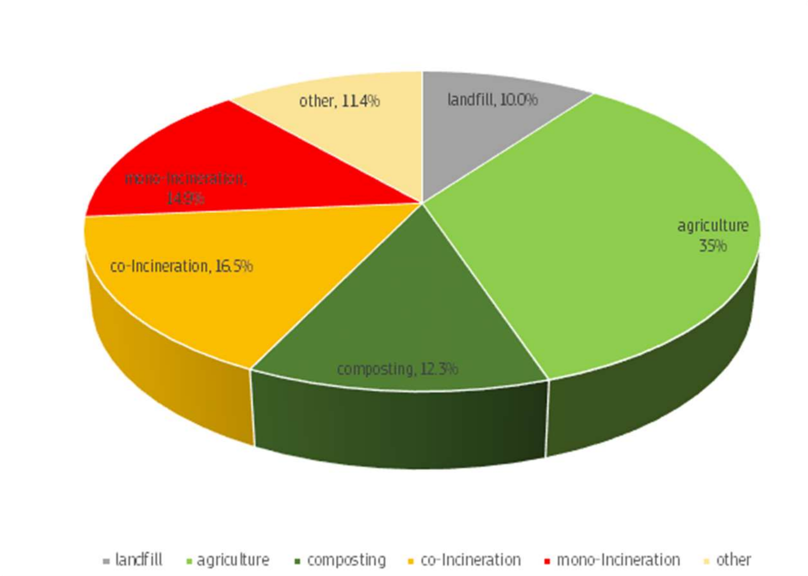
Sewage sludge is the matter resulting from the treatment of domestic or urban waste waters and from other sewage plants treating waste waters of a similar composition. After a pre-screening step filtering out larger contaminants primary treatment of wastewater involves sedimentation of solid waste within the water. Secondary treatment of waste water makes use of biological processes to further purify wastewater. Additional optional steps in the wastewater management system are mostly comprised of removing phosphates and nitrates from the water supply (tertiary treatment), as well as other contaminants and micropollutants (quaternary treatment). The sewage sludge from primary, secondary and/or tertiary treatment is then separated from the effluent, and enters the sewage sludge management line at the waste water treatment plant. Sewage sludge is rich in organic matter, nitrogen, phosphorus, and other macro and microelements, which makes it a useful fertilising material. However, contaminants can be introduced through a variety of direct and indirect pathways. Direct pathways are inflows into urban waste water treatment plants from households, urban run-off and industrial sites that discharge waste waters to the municipal water treatment systems. The inputs from household can include both intentionally used chemicals (e.g. pharmaceuticals and personal care products), as well as unintentionally released chemicals through their use phase (e.g. microplastics and per- and polyfluoroalkyl substances (PFAS) released during the washing of clothes).

Figure 1. Schematic overview of urban waste water and sewage sludge life cycle (Source: own modified, modified from Anderson et al. (2021)).



Information on sewage sludge generation and management pathways vary across sources and reference years. The 2019 Eurostat data (Eurostat 2021), complemented with data from Member State reporting for missing entries indicate that in total about 8.1 million tonnes of sewage sludge are annually produced in the EU-27. Management and use routes vary among EU Member States, but generally speaking use in agriculture and incineration are the most common use routes (Figure *2*). Some Member States (e.g. CZ, ES, FI, FR, IE) use most of the generated sewage sludge in agriculture, whereas other Member States (e.g. NL, BE, DE, GR) incinerate the majority of the sludge generated.

Figure 2. Observed distribution of sewage sludge production to the current sewage sludge management options (EUROSTAT 2021, sewage sludge production and disposal from urban wastewater). It should be noted that uncertainties apply to the distribution of use routes.

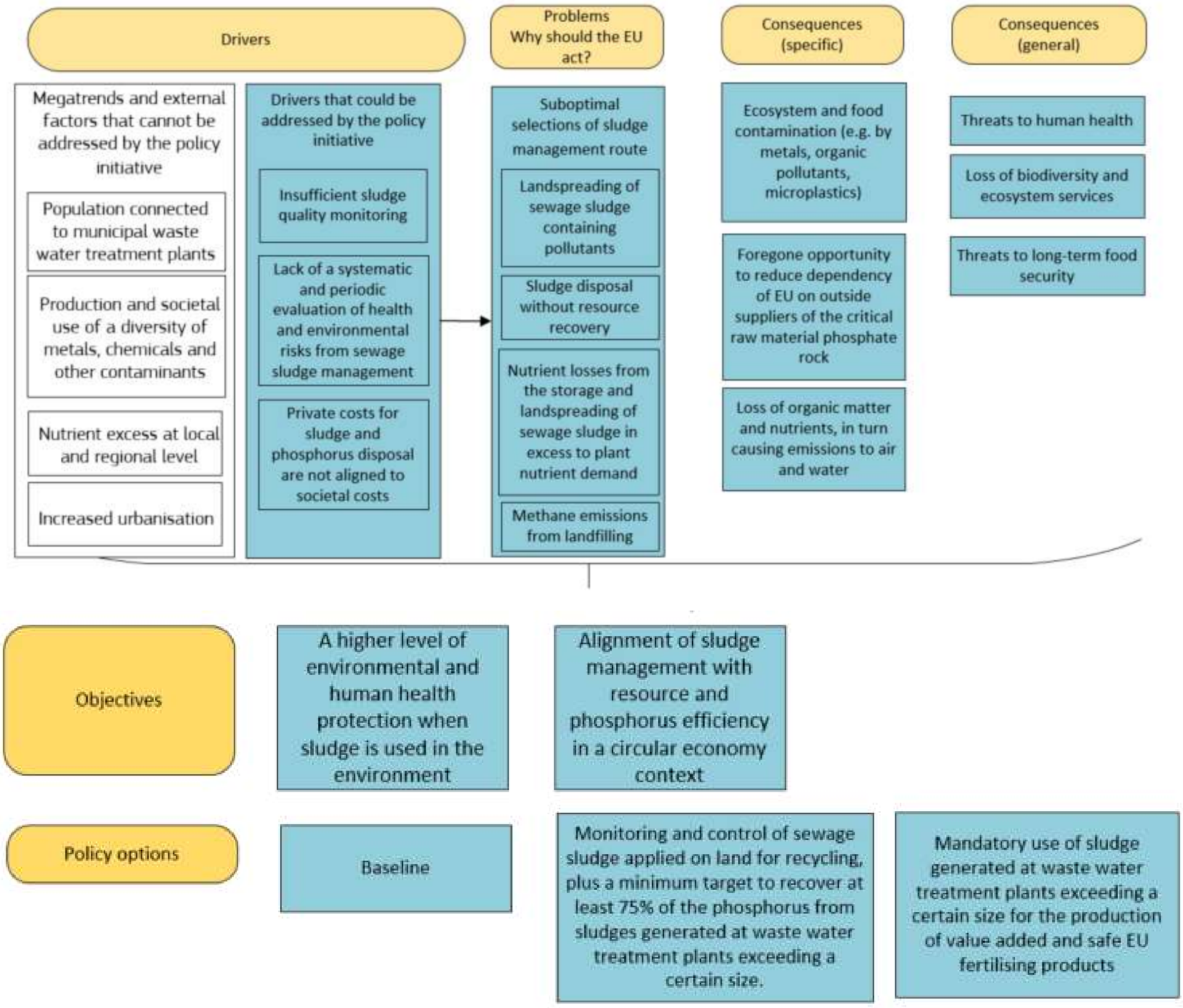


The current EU legal framework for sewage sludge management (Directive 86/278/EEC) sets rules on how farmers can use sewage sludge as a fertilising material on agricultural soils to prevent harm to the environment and human health, by ensuring that the nutrient needs of the plants are considered and that the quality of the soil and of the surface and ground water is not impaired. To this end, it sets amongst others limit values on the concentrations allowed in soil of 7 metals that may be toxic to plants and humans, and imposes sewage sludge treatment for particular agricultural settings to prevent food contamination and the protection of animals from biological pathogens. In addition to the EU legislation, EU Member States may undertake supplementary measures (e.g. set stricter limit values for certain contaminants) for sewage sludge used on agricultural land.

4 Problem definition

A well-elaborated identification, description and understanding of the problems is key to assess and compare policy proposals meant to address them. Problems can be characterised by their drivers and their (negative) consequences. Figure 3 provides a general overview of how the main problems related to sewage sludge management are conceptualised in this report. The problems and consequences are described in more detail in section 4.1, whereas the drivers and expected future evolution of the problems are outlined in section 4.2. A so-called problem tree can be employed to bring forward the objectives of the policy intervention (section 4.3) and the set of different policy options (section 6).

Figure 3. Conceptual overview of the identified problems, underlying drivers and problem consequences, as well as policy objectives and options to achieve them (the white boxes refer to drivers that cannot be addressed in this policy initiative, whereas the blue boxes are directly relevant and will be covered in this report).



4.1 Problems and consequences

Sound management of sewage sludge allows to reap the benefits of recycling organic matter and nutrients available in the sewage sludge as an agricultural resource, while controlling the environmental and health risks from contaminants in sewage sludge. The evaluation report of the SSD (European Commission, 2022) as well as supplementary techno-scientific assessments indicate sub-optimal choices of routes for present-day sewage sludge management as a core problem that could lead to the contamination of the environment and food, and the loss of agricultural resources. More specifically, four different practices and their resulting consequences have been identified and outlined below.

4.1.1 Land spreading of sewage sludge containing pollutants

Based on the available techno-scientific evidence it can be inferred that untreated and stabilised (following lime treatment, composting, and/or anaerobic digested) sewage sludge land spreading leads to contamination of soils and foodstuffs by organic compounds, metals and microplastics. At least some of these contaminants are likely to cause risks for the environment and human health (European Commission, 2022; Huygens et al., 2022). Also antimicrobial resistance genes may be present in sewage sludge, but the available evidence suggests that the contribution of (stabilised) sewage sludge to the problem are likely minor relative to other sources (Rahube et al., 2014; Nõlvak et al., 2016; Rutgersson et al., 2020; European Commission, 2022).

A recent JRC risk screening and modelling study (Huygens et al., 2022) suggested potential risks for human health and the environment from certain persistent, bioaccumulative and toxic **organic compounds** after repeated sewage sludge applications on agricultural land. The set of pollutants identified included short and mid-chain polychlorinated paraffins, long-chain per- and polyfluoroalkyl substances (PFAS), polyaromatic hydrocarbons (PAHs), polychlorinated dibenzofuran and dioxins (PCDD/Fs), and to a lesser extent, alkylphenols, polychlorinated naphthalenes and phthalate acid esters. Additional substances (e.g. benzalkonium chloride and its degradation products) are potentially causing risks to soil organisms under a reasonable worst-case application scenario. Sewage sludge may be a main source of pollution for certain agricultural fields that are subject to repeated sewage sludge applications over time. Therefore, risks for local soil organisms and humans that repeatedly consume food products from such fields and regions are possibly indicated. These modelling results are confirmed by experimental data for specific organic pollutants. For instance, Brusseau et al. (2020) and Ghisi et al. (2019) indicated that the use of PFAS-contaminated media such as sewage sludge and irrigation water can result in soil and food contamination. For PAH, it was indicated that repeated applications of sewage sludge increases PAH concentrations in soils (Lichtfouse et al., 2005; Li and Ma, 2016).

Whereas the JRC study identified possible risks from particular pollutants, further limitations were observed with respect to the evaluation of numerous contaminants and the total risk from mixtures of chemical compounds. The existing database on physicochemical and toxicological properties was, for instance, insufficient to evaluate risks from several hundreds of compounds that have been found in sewage sludge. Therefore, it cannot be excluded that the JRC study underestimated the overall risks from organic compounds in sewage sludge.

Overall, the study on organic contaminants indicates that present-day sewage sludge applications on agricultural lands insufficiently take into account and control associated environmental and health risks, and that sewage sludge can be a main source of local soil pollution that may pose risks for soil quality and humans consuming foodstuff grown on sewage sludge amended soils. Most of the priority pollutants for sewage sludge identified by Huygens et al. (2022) are already subject to use restrictions and release reduction provisions under the chemicals legislation, including the POPs[1] and REACH[2] regulation. Due to these regulations and public concerns associated with the identified chemicals, over time decreasing contaminant concentrations in sewage sludge of e.g. PFAS, PAH, nonylphenols have already been observed (Statistiska Meddelanden, 2012; Statistisches Bundesamt „Statistik" and KEK-1.2, 2014; Fredriksson et al., 2022). Nonetheless, it is unknown if such measures have led to decreases of these compounds to safe concentration levels in sewage. In addition, new and emerging chemicals are continuously placed on the market and may be used in greater volumes than before, potentially causing new risks following their accumulation in sewage sludge.

Up-to-date risk assessment studies for **metals** present in sewage sludge are not available. An overall reduction in metal emissions to the environment from industrial production and waste management processes during the last decades has been observed (EEA, 2019a). As a result, the concentrations of most metals (e.g. Cu, Cd, Hg, Pb, Ni, and Zn) in sewage sludge have decreased substantially during the last decades in samples taken in Germany, Sweden, the Netherlands and other EU Member States (Wiechmann et al., 2013; Kirchmann et al., 2017; The Phos4You partnership, 2021). Metal levels in sewage sludge can be compared to the limit values for fertilising materials in the EU Fertilising Products Regulation[3] (FPR) and national legislation of EU Member States (Tavazzi et al., 2012; Ehlert et al., 2013; Hudcova et al., 2019) to preliminary assess safety aspects. The spectrum of existing regulatory limit values has partly been based on (national) risk assessment models with differing assumptions and approaches. In addition, political and pragmatic considerations can explain observed differences in limit values across national legislation. Such comparison thus provides only preliminary evidence

[1] Regulation (EU) 2019/1021 of the European Parliament and the Council of 20 June 2019 on persistent organic pollutants.

[2] Regulation (EC) No 1907/2006 of the European Parliament and of the Council of 18 December 2006 concerning the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH), establishing a European Chemicals Agency.

[3] Regulation (EU) 2019/1009 of the European Parliament and of the Council of 5 June 2019 laying down rules on the making available on the market of EU fertilising products. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex%3A32019R1009

to assess the level of risk to the environment and human health from metals contained in sewage sludge application on agricultural land.

Table 1. Regulatory limit values for metals present in sewage sludge (mg kg^{-1} dry matter) in the EU Sewage Sludge Directive (SSD), national legislation across from different EU Member States, and the EU Fertilising Products Regulation (FPR; Product Function Categories Organic Fertilisers and Soil Improvers). The distribution of metals observed in sewage sludge has been inventoried from 61 samples taken in different EU Member States, as recorded in the JRC FATE sampling campaign (Tavazzi et al., 2012).

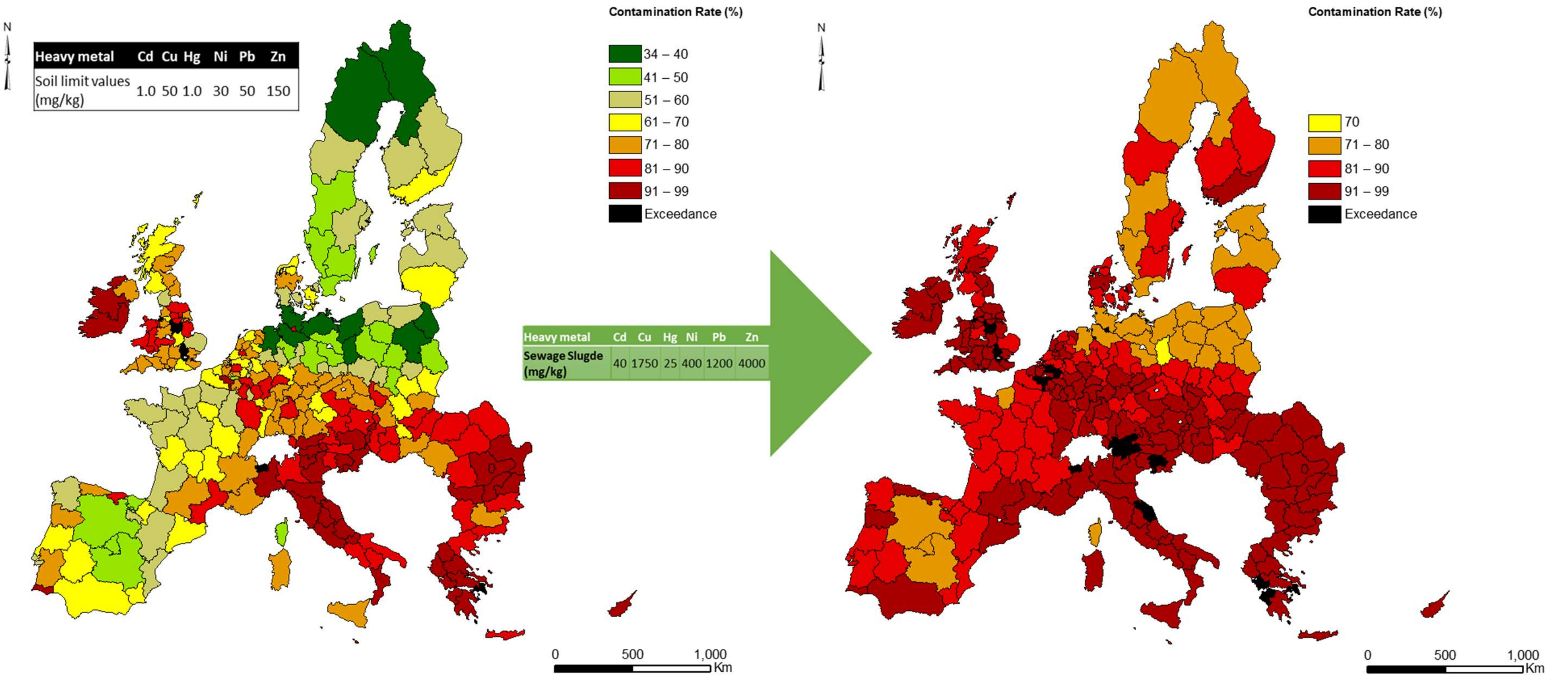
	SSD		national legislation		FPR	observed in sewage sludge		
	min	max	Min	max	max	min	max	average
As	-	-	15	25	40	<d.l.	56.1	-
Cd	20	40	0.8	10	1.5-2	<d.l.	5.1	0.9
Cr	-	-	25	900	-	11	1 542	80
Cu	1 000	1 750	75	1 000	300	27	578	257
Hg	16	25	0.75	8	1.0	0.1	1.1	0.4
Ni	300	400	30	200	50	9	310	29
Pb	750	1 200	100	900	120	4	430	48
Zn	2 500	4 000	300	4 000	800	213	1 218	663

It can be observed that (i) maximum values for metals in sewage sludge are consistently greater than the metal limit values established in the EU FPR and certain EU Member States, and (ii) that average observed values for some metals (Cu, Zn) in sewage sludge are less than 20% below the limit values of the FPR. This suggests that metals in certain sewage sludge applied on agricultural land may be present in concentrations of concern to legislators in the EU and Member States. Experimental evidence further confirms that repeated applications of sewage sludge on agricultural land increase metal contents in soils and crops growing on sewage sludge amended soils, at times to levels that are considered causing environmental risks (Charlton et al., 2016; Zaragüeta et al., 2021).

Using modelling based on current “background” concentrations of metals in soils and estimated metal inputs from long-term sewage sludge applications, our JRC assessment (Figure 4) supports concerns in relation to potential metal contamination of soils by sewage sludge meeting metal limit values established in the SSD. It is concluded that the land spreading of sewage sludge with content of heavy metals aligned to maximum limits laid down in the SSD over 10 consecutive years at 5 Mg ha^{-1} rate will largely effect on soil quality. Following repeaed applications, all agricultural soils will show average overall contamination rates[4] above 70% only taking into account inputs from sewage sludge (right map in Figure 4). Seven new NUTS regions would be labelled as contaminated and 57% of all NUTS regions would show average overall contamination rates above 90%.

[4] Soil contamination rate was calculated for each of the 6 tested metals (Cadmium, Copper, Nickel, Lead, Mercury and Zinc) as the ratio between the modelled total metal concentration in soil and the limit value for the corresponding metal in sewage sludge amended soils as per Annex I A of the SSD. Overall contamination rate is defined as the highest contamination rate among the 6 metals modelled for each LUCAS 2009 point. Thus, a point will be labelled as contaminated when the concentration of at least one heavy metal exceeds the soil quality standards of the SSD.

Figure 4. Results of the modelling of the accumulation and contamination of soils by metals following sewage sludge application. The right-hand Figure shows the effect of the application of sewage sludge with concentration values aligned to maximal permitted application rates per Annex I B and I C of Directive (86/278 /EEC; as indicated in the green arrow) over 10 consecutive years at a sludge application rate of 5 Mg ha-1, starting from the actual metal concentrations observed in EU soils (left-hand Figure, based on LUCAS 2009 database). Soil contamination rate was calculated for each of the 6 tested metals as the ratio between the measured metal concentration in soil and the (minimum) limit value for the corresponding metal in sewage sludge amended soils as per Annex I A of the SSD (Table at the top-left of the Figure). Overall contamination rate is defined by the highest contamination rate among the 6 metals modelled (Cadmium, Copper, Nickel, Lead, Mercury and Zinc).



Additional results are provided in the Annexes to this report (section 13.5).

Finally, sewage sludge contains **microplastics** as emerging pollutants of concern. Evidence suggests that regular application of microplastics leads to significant accumulation in soils, and that sewage sludge is a main entry route for microplastics into the terrestrial environment (Büks and Kaupenjohann, 2020; van den Berg et al., 2020; Azeem et al., 2021; UNEP, 2022). Lofty et al. (2022) and Nizzetto et al. (2016) estimate that sewage sludge applications on agricultural land introduce between 31 000–430 000 tonnes of microplastics to European soils annually potentially mirroring the concentration of microplastics routed towards ocean surface waters. Microplastics adversely influence physical soil properties such as water holding capacity, soil aggregation, the performance and composition of the soil microbial community and soil fauna (Büks and Kaupenjohann, 2020), and can adhere to the surface of seeds and roots and can be taken up by plants causing negative effects (e.g. oxidative stress, cytotoxicity, and genotoxicity; (Li et al., 2020; Azeem et al., 2021; Mateos-Cárdenas et al., 2021; Wang et al., 2022). Research efforts on microplastics contamination of the environment and their consequences are being scaled up, but at present there is little knowledge on the magnitude of adverse effects (Büks and Kaupenjohann, 2020).

4.1.2 Sewage sludge disposal without resource recovery

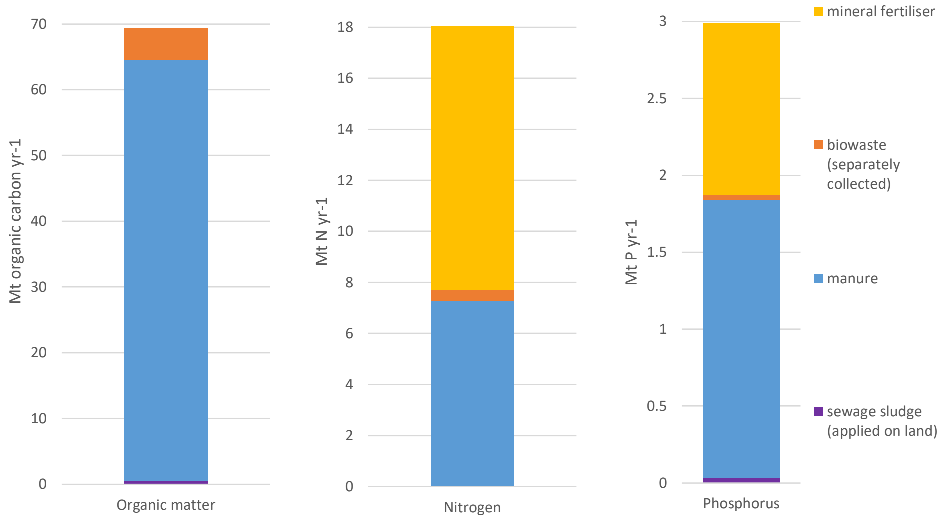
Sewage sludge contains resources that provide plants with nutrients or improving their nutrition efficiency, such as nitrogen (N), phosphorus (P) and organic matter. Phosphorus is an element that is listed as a critical raw material for the EU because of its high economic importance for agriculture and a wide range of industries, and supply risk due to its finiteness, non-substitutability, and the EU's dependence on imports, particularly from Morocco/Western Sahara and Russia (Huygens et al., 2019). Nitrogen is a main plant macronutrient that is produced in primary form via the Haber-Bosch process using air and natural gas a main feedstock, and consuming more than 1% of the global net energy demand. Organic matter is added to the soil to improve plant nutrition; it is considered an important indicator of soil fertility, and improves physical (structure, aeration, water- and nutrient retention) and biological (biomass, biodiversity, nutrient mineralisation, disease suppression) soil properties (Hijbeek et al., 2017).

Agricultural and biological resources contained in sewage sludge are typically lost when sewage sludge is disposed. Sewage sludge disposal options include the landfilling of sewage sludge, or sewage sludge (co-)incineration followed by the disposal of ashes or their use as construction materials. During landfilling, nutrients are lost to air with negative impacts on climate (see section 4.1.4), retained in the landfill, or end up in landfill contaminant-rich leachates. Incineration transform nitrogen and organic carbon into volatile compounds (mostly N_2, NO_x, and CO_2) released to air, whereas phosphorus is retained in the incineration ashes. The phosphorus retained in the ash is mostly removed from the biogeochemical phosphorus cycle when the resulting ashes are either disposed in landfills, or used as construction materials (including cement matrices). Innovative techniques are able to extract and transform phosphorus from mono-incineration ashes among others into phosphorus fertilisers with low contaminant levels, but the implementation of such techniques is currently still relatively limited (Table 52). In addition, it has been documented that sewage sludge is permanently stored outdoors close to the waste water treatment plants, thus not contributing to plant nutrition (Suchkova et al., 2010). Finally, nutrients landspreaded in excess to plant phosphorus demands does not seize its full potential to contribute to plant nutrition as it leads to long-term accumulation of nutrients in soils (van Dijk et al., 2016).

Sewage sludge generated in the EU-27 is estimated to contain about 2.5 Mt organic C yr^{-1}, 0.26 Mt N^{-1}, and 0.15 Mt P yr^{-1}. Currently around 10% of the produced sewage sludge is directly landfilled (0.83 Mt dry matter in total) and 32% is incinerated (16.4% co-incineration, 15.3% mono-incineration: 2.57 Mt dry matter in total) (see baseline as developed in section 6). Additional sewage sludge volumes subject to "composting" and "other uses" may not seize the full potential of sewage sludge as an agricultural resource (e.g. compost used as landfill cover, permanent storage of sewage sludge close to production site, sewage sludge used for the backfilling of underground mines). In total, it is estimated that 1.5 Mt organic C yr^{-1}, 0.15 Mt N^{-1}, and 0.08 Mt P yr^{-1} is not used for food and feed production purpose (see Annexes, section 11.1.1.2)

To better understand the current overall contribution of sewage sludge as an agricultural resource, organic carbon and nutrient inputs from different sources are presented in Figure 5.

Figure 5. Annual amounts of organic carbon and nutrients applied to EU agricultural land from sewage sludge, manure, bio-waste, and mineral fertilisers (source: own elaboration).



With regard to P, the EU imports annually about 1.1 million tonnes (Mt) of P as mineral fertiliser. The total content of P in landspreaded sewage sludge amounts to about 5% of the P mineral fertiliser inputs to EU agricultural land. The share of N (relative to mineral fertilisers) and organic carbon (relative to manure) contained in sewage sludge is even lower than for phosphorus (Figure 5).

While all three resources (organic matter, N, and P) are important for agriculture, phosphorus is likely the most significant resource due to its classification as an EU Critical Raw Material and the higher relative contribution of sewage sludge to the phosphorus input balance than to nitrogen and organic carbon balance.

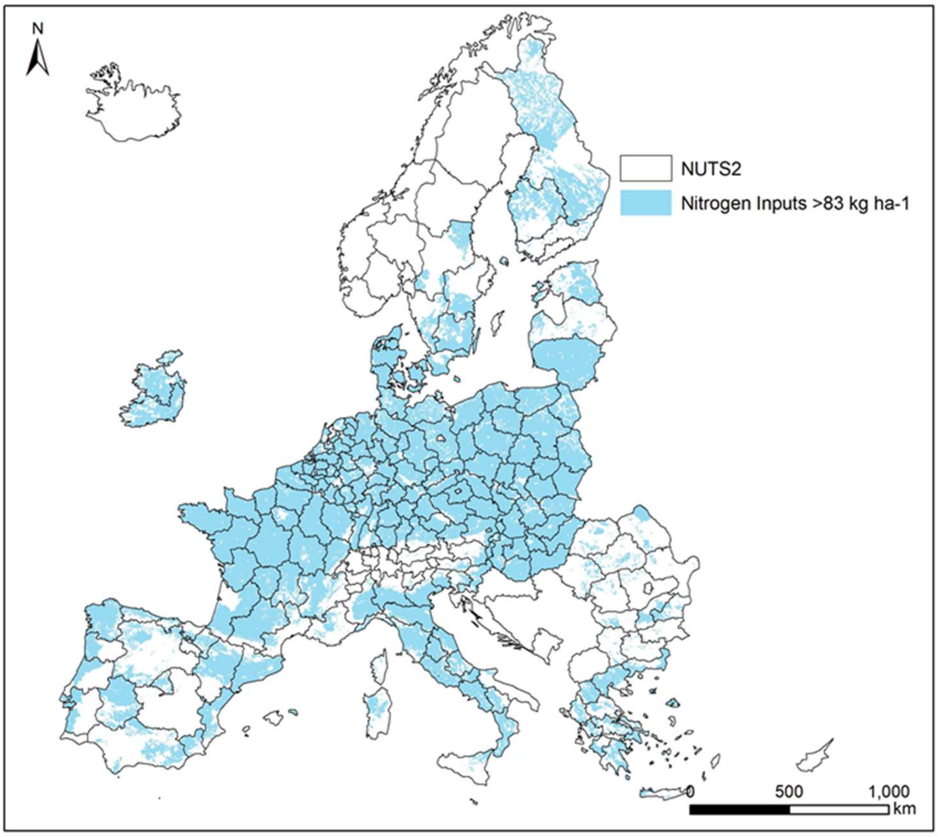
The loss of organic matter, nitrogen, and phosphorus means that sewage sludge management misses the opportunity to contribute to a more circular economy, where secondary raw materials can substitute primary raw materials. Present-day sewage sludge management is thus not aligned with the EU waste management hierarchy that promotes the recycling of waste over (energy) recovery and disposal. Concrete impacts from the lack of sewage sludge recycling are (i) a greater natural resource depletion, mainly the depletion of the finite raw material rock phosphate, (ii) an increased EU dependency on imports of finished mineral fertilisers and their precursors such as rock phosphate, ammonia and natural gas from outside the EU (EU Agricultural Markets Brief, 2019 - currently 44% and 66% of the nitrogen and phosphorus fertilisers, respectively, are imported; JRC research based on 2019 Eurostat Comext and Prodcom data; EPRS, 2022); and to a lesser extent (iii) reduced soil fertility and adverse climate change impacts from mineral fertiliser production. The magnitude of these impacts is further quantified in the section on the baseline scenario (section 6).

4.1.3 Nutrient losses from the storage and land spreading of sewage sludge in excess to plant nutrient demands

In some EU regions, the joint nutrient supply from mineral and organic sources, including sewage sludge, exceeds plant nutrient demands (de Vries et al., 2021). Nutrients applied in excess of plant demands are a source of pollution due to losses of nitrates, phosphates, greenhouse gas emissions and ammonia, to water bodies and the atmosphere (Schoumans, 2015; Zhao et al., 2019). Although the contribution of sewage sludge to the total nutrient inputs is minor, it contributes in some EU regions to aggravating nutrient losses. Sewage sludge contains a significant amount of water (e.g. dewatered sewage sludge: 65-80% water). Therefore, large quantities have to be transported in relation to the active substance, which results in high transport costs. Because of the high transportation costs, organic materials may at times be perceived as waste and disposed of on nearby available land rather than used as a value-added nutrient resource.

It is observed that in many EU regions, nitrogen inputs exceed 83 kg ha^{-1} yr^{-1}, defined as a safe planetary boundary to limit nitrogen losses to air and waters (de Vries et al., 2021), in NUT2 regions concentrated in central and northern EU (Figure 6). At least in some of these regions, sewage sludge applications contribute to the observed problem of nutrient application in excess to sustainable input values.

Figure 6. Nitrogen inputs from natural processes (biological nitrogen fixation and soil mineralisation) and applications of organic materials (manure and sewage sludge) at NUTS2 level by De Vries et al. (2021), with sites receiving inputs greater than 83 kg ha^{-1} yr^{-1} indicated in blue. The chosen thresholds of 83 kg ha^{-1} yr^{-1} can be defined as a limit values to limit nitrogen losses to air and waters and is related to the average nitrogen crop nitrogen demand and nitrogen use efficiency (De Vries et al .,2021).



Additionally, some characteristics and limitations of organic materials compared to mineral fertilisers play a role for the excessive nutrient application, in particular the timing of application, which is often not ideal due to storage limitations, the nitrogen to phosphorus ratio, and the chemical composition. Only a fraction of the nitrogen and phosphorus contained in sewage sludge is immediately present in plant-available forms (Xu et al., 2012; Kirchmann et al., 2017). With a view to fertilising efficiency and minimising nutrient losses from agriculture, it is therefore advisable to preferentially apply other mineral or organic fertilising materials than sewage sludge as organic nutrient sources, when available.

Projections for 2030 show that at EU level, approximately 30% of the nitrogen surplus from all combined sources is released into the atmosphere (European Commission, 2021a). Gaseous emissions are mainly greenhouse gases and ammonia. These cause problems such as climate change, air pollution and, acidification. Approximately 40% of the nitrogen surplus is leached into water bodies. This causes health hazards (e.g. nitrate particularly to babies, infants, and young animals), harm to living resources and to aquatic ecosystems, and/or interferes with other uses of water (including freshwater and marine coastal areas) (European Commission, 2021a).

Conversely, phosphorus is not lost to the air, only to water, because of runoff/erosion. This causes mainly eutrophication problems in surface waters. A large share of the phosphorus that enters the soil gets fixed or absorbed in the soil. It is estimated that only around 20% of the soil phosphorus is inorganic phosphorus dissolved in a water/soil solution that is readily available for plant uptake (Prasad and Chakraborty, 2019). Therefore, leaching is low under most conditions. However, particularly phosphorus saturation in soils decrease the absorption capacity of the soil, and, therefore, increase the leached amount (Schoumans, 2015). Phosphorus leaching is particularly relevant under such conditions as the nitrogen to phosphorus ratio in sewage sludge is not aligned to plant needs; the low ratio in sewage sludge leads to phosphorus excess and loss when sewage sludge is used a nitrogen fertiliser.

4.1.4 Methane emissions from the landfilling of sewage sludge

Sewage sludge contains carbon and nitrogen and thus causes global warming when greenhouse gases (methane, nitrous oxide) are released. At the same time, sewage sludge can also help to mitigate climate change impacts through the production of renewable energy from sewage sludge (following incineration and/or anaerobic digestion) or by sequestrating carbon in agricultural soils. A recent study by the JRC (Huygens et al., 2022) indicated, however, that (i) with the exception of landfilling, different sewage sludge management routes (landspreading of untreated or treated (composted, digested lime stabilised) sewage sludge, co-incineration, mono-incineration with phosphorus recovery) do not differ significantly in their global warming potential, and (ii) that the overall contribution of sewage sludge management to the overall global warming potential is negligible as positive and negative emissions largely neutralise each other. Therefore, global warming is not considered as a problem in this study. However, methane emissions from sewage sludge management is set apart as a problem. Methane is a greenhouse gas that has a global warming potential 28 times higher than that of CO_2 over a 100 year time period.

The landfilling of sewage sludge has been identified as a major source of methane emissions from waste management, a sector that emits 4.1 Mt CH_4 or 27% of the total annual EU methane emissions (European Commission, 2020; van der Veen et al., 2022). At a global level, reducing methane emissions associated with human (anthropogenic) activity by 50% over the next 30 years could reduce global temperature change increase by 0.18 degrees Celsius by 2050, a main step towards with limiting global warming from all greenhouse gases to 1.5°C.

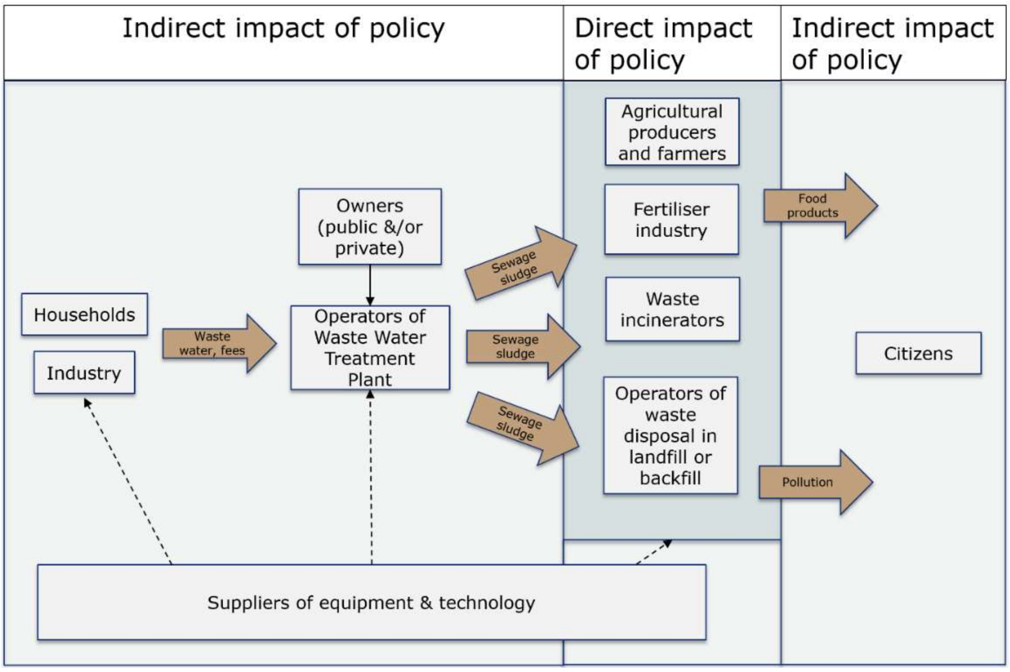
Countries as e.g. Austria, Germany, Sweden and the Netherlands banned landfilling of untreated waste or implemented measures that have similar effects as a landfill ban (e.g. taxes). However, landfilling of sewage sludge continues to be a common practice in EU Member States. Croatia, Romania, Italy, Malta, and Greece landfill 30%-90% of the total sewage sludge volumes generated. In some Member States EU landfill regulations are not applied to a satisfactory degree, especially as regards controlling the accumulation and migration of landfill gases (European Parliament, 2022).

4.1.5 Identification of stakeholders and actors affected by the problem

4.1.5.1 Identification of stakeholders

The problems related to sewage sludge management affect, directly and indirectly, a number of actors. Directly affected parties include the “users” of sewage sludge, mostly farmers, the fertilising industry, and waste treatment and disposal operators. Indirectly affected parties are the operators of waste water treatment plants, industry and households discharging and paying for waste water treatment services, involving sewage sludge management and citizens affected by food quality and pollution for sewage sludge management (Figure 7).

Figure 7. Schematic representation of the different actors affected by the observed problems.



Agri-food companies and farmers are key actors in the food production chain and the placement of food products on the internal market, thus responsible to ensure the quality and safety of their products. Some food industry companies do not buy products from farmers who used treated sewage sludge on their fields. Similarly, farmers growing organic agricultural products in line with the EU-organic production regulation (EU 2018/848) are not allowed to use sewage sludge as fertiliser. Different eco-labels established in EU MS have similar requirements and prohibit the use of sewage sludge for farmers wanting to obtain certification. Farmers are also affected because sewage sludge is a possible nutrient source, and its utilisation may reduce demand for fertilisers produced from primary raw materials. Fertilisation costs represent about 15-45% of the total production costs at farm level, depending on the year and the crop (European Commission, 2016; EU Agricultural Markets Brief, 2019). Sewage sludge may be used as a feedstock material to produce commercial fertilisers (mineral fertilisers, compost, digestate) or it may partially replace mined and synthetic fertilisers. Hence, the fertilisers industry is affected by this initiative. Waste management companies, including operators of waste disposal and recovery plants following the landfilling, incineration, and backfilling of sewage sludge are directly impacted by this initiative that sets out obligations and limitations for sewage sludge and waste management.

Waste water treatment operators are utility providers of water treatment services producing sewage sludge. The choice of their management model is subject to subsidiarity and it is a competence of Member States. As a way of simplification, four management models may be distinguished across Europe (Eureau, 2020). In the most common system, the local government is entirely in charge of service provision and their management system (*direct public management*). However, a general trend observed in the last decades involves in some Member States a shift towards public or private delegated management, where the management is delegated towards a management entity that is appointed by the responsible public (*delegated public management,* BE, BG, FR, NL, PL, SE) or private (*delegated private management* on the basis of a time-bound contract; partially applicable in e.g. FR, ES, CZ, EL) entity, but the ownership of the infrastructure remains mostly at public level. As an exception, some MS rely at times on a system where all management tasks, responsibilities and ownership of water utilities are placed in the hands of private operators (*private management* operating >30% of the waste water treatment plants in DK and EE). The total costs for sewage sludge management and disposal may reach 40–60% of the total operational costs for managing the entire waste water treatment plant, including personnel, maintenance, energy and sewage sludge treatment and disposal (Foladori et al., 2010; Scrinzi et al., 2022). Water consumers and thus EU citizens are paying water tariffs in the EU for waste water treatment and access to clean water. The total operational costs of a waste water treatment plant, including sewage sludge management, is on average €106 per inhabitant per year, of which roughly equally divided between waste and waste water treatment services (Eureau, 2021). Hence, revised policies on sewage sludge may translate into altered fees for households and potentially industries discharging to waste water treatment plants. Finally, industries of a varying nature discharge their waste waters to waste water treatment plants, and affect as such the sewage sludge quality and possible downstream management routes.

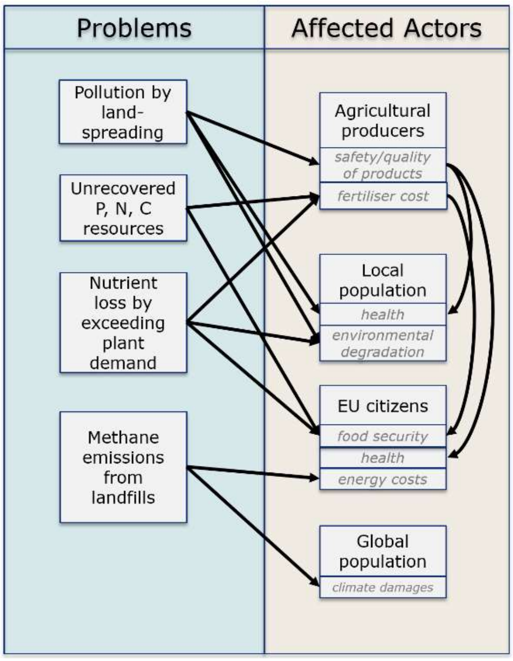
EU citizens and food consumers are affected because sewage sludge applications on agricultural lands have the potential to alter the quality of the product, with possible health impacts for food consumers. Citizens also enjoy the benefits of the EU's natural capital (including soils and biodiversity) and the ecosystem services they provide, that may be impacted by pollution from sewage sludge.

Finally, technology providers involved in different stages of the management of waste waters (upstream) and sewage sludge (downstream) may be affected because technological solutions may help to address the problems observed.

4.1.5.2 Actors causing and being affected by the problem

The actors that are affected by the problem and the resulting consequences of contamination, resource depletion and methane emissions include farmers, local population, EU citizens, and the global population, as illustrated in Figure 8.

Figure 8. Actors affected by the problem and its consequences



The problems are mainly caused by authorities and waste management treatment operators. At least partly, some national authorities fail to set up a well-functioning legal framework for waste management and the protection of agricultural soils. As a result, the selection of sewage sludge management routes by waste water treatment operators is mainly driven by economic considerations that do not take into account externalities and without having a full information base on the quality aspects of the sewage sludge (see section 4.2.2).

4.1.6 General consequences

4.1.6.1 Pollution of soils, environment and food

Pollution is a key driver of biodiversity loss and has a harmful impact on our health and environment. Biodiversity is, amongst others, suffering from the release of nutrients, chemicals, and other waste including litter. The general consequences of the considered problems are harmful effects on the environment and ensuing pollution impacts on citizen's health, by creating a potentially toxic environment for soil organisms, animals and humans. Uncontrolled methane emissions contribute to global climate change.

4.1.6.2 Increased dependency on foreign imports of rock phosphate as a critical raw material

Enhancing resilience by reducing the dependency of European agriculture on raw materials, energy, and energy intensive imports is now more of a necessity than ever before. The Russian invasion on Ukraine and a global commodity price boom are driving up prices of agricultural products and exposing the vulnerabilities of our food system. Without further actions to recycle the phosphorus contained in the food, the EU will perpetuate its dependency on foreign nutrient imports and associated shocks in fertiliser's prices, in a worst case scenario contributing to food insecurity in the EU but also worldwide.

4.1.7 Misalignment to EU policies

These general consequences are not consistent with EU policies on pollution and health impacts thereof, circular economy, and food security. The general consequences thus put a pressure on the EU goals outlined in, amongst others, the EU Zero Pollution Action Plan, the recently launched 8th Environment Action Programme, and the EU's ambition as a global leader while ensuring the highest standards of climate and environmental protection.

4.2 Identification and persistence of problem drivers

To solve the problem, its underlying causes ("drivers") should be identified. This is important for two reasons. First, it is impossible to design policy responses and study how these would mitigate the problem in the future without knowing which the underlying drivers are and how they evolve. Second, the nature and evolution of the problem (in terms of size, geographic scale, and the market actors) plays a key role in determining whether public policy action at EU level is justified. The identified drivers of the problem are split into, on one hand, those that could be targeted by policy actions under the SSD and, on the other hand, those aspects which contribute to the problem but are outside the scope of this policy framework.

4.2.1 Megatrends and external factors that cannot be addressed by the policy initiative

A first set of drivers includes megatrends (e.g. demographic considerations, use of chemicals in our society) and general economic aspects relevant for private operators involved in sewage sludge management. Such issues and drivers cannot be addressed by policies directly related to sewage sludge, but are important for assessing the likely persistence of the problem in the future.

4.2.1.1 Population connected to municipal waste water treatment plants

The amount of sewage sludge generated as well as their nutrient contents are directly related to the number of citizens connected to centralised waste water treatment. Assuming a largely stable population in the EU-27 in the coming decades[5], a continued enforcement of the urban waste water treatment directive 91/271/EEC with respect to the population share that is connected to centralised waste water treatment, and envisaged policies to reduce nutrients in waste water treatment plant effluents ending up in sewage sludge, it can be projected that raw sewage sludge volumes remain approximately stable, and the resources and nutrients contained therein (organic matter, nitrogen, and phosphorus), will increase in the future (see development of the baseline; section 6). The Urban Waste Water Treatment Directive 91/271/EEC is currently under revision and a proposal was published (EC, 2022a). With the current proposal, a certain reduction of sludge volume is to be expected as WWTP are obliged to perform an energy audit that could lead to additional anaerobic digestion installations (see Figure 10). The proposal does not contain any direct measures that lead to negative impacts on the sludge composition, but the increased focus on micropollutants removal from the effluent may alter the sewage sludge composition. Certain pollution abatement technologies (e.g. use of powdered activated carbon) transfer micropollutants to the sludge phase, making it ultimately unsuitable for posterior use on agricultural land.

Given that sewage sludge volumes are not expected to decline in the future, and that, conversely, nutrient loads might even somewhat increase, the described problems are likely to persist. Proper management of sewage sludge will therefore continue to be relevant in the future.

4.2.1.2 Production and societal use of a diversity of metals, chemicals and other contaminants

Contaminants ending up in sewage sludge are introduced through a variety of direct and indirect pathways. Direct pathways are inflows into urban waste water treatment plants from households, stormwater flows and industrial sites that discharge waste waters to the municipal water treatment systems. Contaminants from households can stem from both intentionally used chemicals (e.g. pharmaceuticals and personal care products), unintentionally released chemicals (e.g. microplastics and PFAS released during the washing of clothes), and from faeces (e.g. metals that were contained in consumed food products).

5 Eurostat PROJ_19NP (https://ec.europa.eu/eurostat/databrowser/view/proj_19np/default/line?lang=en).

Metal, metalloids and some persistent organic contaminants in sewage sludge are mostly the result of industrial emissions to air that are afterwards deposited on agricultural and urban areas, as well as waste waters from certain industrial processes. The combustion of fuels in stationary sources is the main emission source for metals such as As, Cd, Cr, and Ni (Pacyna et al., 2007), but also iron and steel production, non-ferrous metal manufacturing and waste incineration are considered to be major sources of metal emissions to air. Effluents from textile, leather, tannery, electroplating, galvanizing, pigment and dyes, metallurgical and paint industries and other metal processing and refining operations at small and large-scale contain considerable amounts of metal ions. The metals may end up in sewage, either indirectly following consumption of metal-containing foodstuff by humans or directly following industrial releases to waste water treatment plants or storm water runoff from urban and industrial areas. Polyaromatic hydrocarbons (PAH) and polychlorinated dibenzo-p-dioxins and furans (PCDD/Fs) are unintentionally released persistent, bioaccumulative and toxic (PBT) compounds from combustion and incineration processes.

With respect to intentionally manufactured organic chemicals, a comprehensive overview of the current state of play on the use of chemicals and observed trends is provided in the EEA report "The European environment – state and outlook 2020" (EEA, 2019a). The report indicates that two aspects create particular concern: the sheer volume of chemicals in use and the potential combined toxicity of these diverse chemicals. The consumption of industrial chemicals in the EU in 2017 was 304 million tonnes. Of these, 22% were hazardous to the environment and 71% were hazardous to health, similar proportions to those for chemical production. Between 2000 and 2017, the production capacity of the global chemical industry increased from 1.2 to 2.3 billion tonnes (UNEP, 2019). The proportion of consumed chemicals hazardous to the environment and health declined by 5% and 6%, respectively, between 2008 and 2017 (Eurostat, 2019). Hence, the total use of chemicals, including of substances classified as hazardous, in locally produced and imported products has substantially increased in the last decades. In terms of diversity, the EEA report indicates that 22 600 chemical registrations were registered under the REACH legislation in August 2019. This number omits chemicals on the market at volumes of below 1 tonne, as well as polymers, and those already regulated under existing regulation such as pesticides and pharmaceuticals. The total number of synthetic chemicals on the market has been estimated at 100 000 substances (Milieu Ltd et al., 2017). There are also an unknown number of transformation products from chemicals during their life cycles. Actual accumulation in sewage sludge is determined by emissions during the chemical's life cycle, including use and waste phases and possible reuse. Certain very hazardous chemicals are used in closed systems, reducing opportunities for exposure. Looking ahead, society's reliance on chemicals is projected to grow globally and in the EU (EIA, 2016; CEFIC, 2018; OECD, 2019).

4.2.1.3 Nutrient excess at local and regional level

Nutrients applied in excess to plant needs may be a source of pollution due to losses of nitrates, phosphates, greenhouse gas emissions and ammonia, to water bodies and the atmosphere (section 4.1.3). Nutrient losses exponentially increase with nutrient surplus applied to land. Soils and agricultural fields suitable for receiving sewage sludge are limited because of the fierce competition with other organic materials, particularly manure and to a minor extent bio-waste. Therefore, areas where sewage sludge can be applied within the safe planetary boundaries to avoid a deterioration of water and air quality are limited and geographically distributed throughout the EU. It can be observed that sewage sludge is not commonly applied to lands characterised by a gross nutrient excess (e.g. NL, BE, DE) due to inputs from manure and other organic materials (see section 4.1.3).

Forward-looking reports from the European Commission (until 2030) and FAO project further increases in livestock density in the EU, e.g. FAO between +31% and +48% by 2050 for livestock units under a "towards sustainability" and "business as usual" scenario for the region Europe plus Central Asia (European Commission, 2021a; FAO, 2022). With the established targets and ambitions to separately collect and process bio-waste in the EU, it is also expected that increased amounts of composted and digested bio-waste will be applied on land. Hence, altogether, the competition amongst organic nutrient sources for agricultural land spread will continue and potentially further increase in the future.

4.2.1.4 Increased urbanisation

Trends in the total population of EU-27 show a decline in the share of population living in rural areas over the total population during the last decades, while towns and cities experienced a smooth and constant increase (Competence Centre on Foresight, 2022). As a result, sewage sludge is increasingly being generated at larger waste water treatment plants located in urban areas. At present, 55% of the population is connected to large waste water treatment plants with a capacity of >100k population equivalents (p.e.) in the EU, in line with the trend of urbanisation and resource optimisation for water treatment in line with the economy of scales principle (section 6). The corresponding waste water volumes are treated by large waste water treatment plants (WWTP) representing only 6% of the total number of such plants in the EU.

However, recycling sewage sludge through landspreading requires suitable agricultural land nearby the waste water treatment plant at a short transport distance (typically <10-20 km). With the generation of sewage sludge being concentrated at specific locations, finding suitable land becomes thus more challenging and waste water treatment operators are required to find alternative solutions (Di Giacomo and Romano, 2022), often involving disposal. Europe's level of urbanisation is expected to further increase to approximately 84% in 2050 (Competence Centre on Foresight, 2022), indicating the persistence of this driver.

4.2.2 Drivers that can be addressed by the policy initiative

4.2.2.1 Insufficient sewage sludge quality monitoring data

Up-to-date data on sewage sludge quality characteristics and more particularly on priority contaminants identified in sewage sludge is rare (Huygens et al., 2022). An extensive analysis of a larger number of sewage sludge samples, taken from across the EU, was carried out more than 10 years ago (Tavazzi et al., 2012). New concerns have emerged from the placing on the market of new chemicals or previously underestimated pollution sources (e.g. short-chain PFAS, pollution mixtures). The reporting of most EU Member States on sewage sludge quality characteristics is mostly limited to some metals, in line with legislative requirements. Data on sewage sludge in existing databases, such as the Information Platform for Chemical Monitoring[6] is limited and less extensive in terms of frequency and contaminants covered than data documented for water bodies.

Because we do not have adequate knowledge concerning the identity and the quantities of the chemicals in sludge spread on agricultural land, it is not possible to assess the full impact of chemicals and their mixtures on human health and the environment, nor to assess the effectiveness and impact of regulatory measures to reduce exposure ("no monitoring means no data, and no data means no regulations") (Dulio et al., 2018; Comero et al., 2020). Some waste water treatment plants, especially larger ones, have contaminant monitoring schemes in place, but the results are usually not publically available. As a result, fundamental knowledge to assess and possibly substantiate risks from spreading sewage sludge in the environment is missing, and awareness raising on the concerns related to sewage sludge is limited (Perkins, 2019; but see e.g. Monbiot, 2022).

To some extent, a positive change is expected from the increased (research) attention given to chemical pollution (e.g. EU Zero Pollution Action Plan), the recently increased availability of high-throughput technologies (Veenaas et al., 2018; to screen and quantify a large number of contaminants in media such as sewage sludge; see e.g. Castro et al., 2021), and data mining possibilities that combine existing databases. However, obstacles related to data completeness, harmonisation, accessibility, retrievability and comparability may persist. Therefore, it remains uncertain to what extent such voluntary efforts and new research will effectively document the full presence of relevant hazardous contaminants in sewage sludge that are applied on land in the EU.

4.2.2.2 Lack of a systematic and periodic evaluation of health and environmental risks from sewage sludge management

With sewage sludge acting as a sink for a huge set of contaminants released to the environment, and in view of the limited data availability on toxicological properties of chemicals, even a targeted screening would not provide a comprehensive understanding of the potential hazards because a full assessment needs to consider also contaminants dynamics and transport within exposure pathways, and (eco-) toxicological data. Levels of protection and risk perceptions of sewage sludge management may change over time. E.g. recommendations and limit values for the dietary intake of certain contaminants present in sewage sludge were revised at repeated occasions during the last decades. Likewise, external changes have helped to reduce pressures from certain contaminants (e.g. the phase-out of PCBs), calling for a re-evaluation of risks from certain contaminants to ensure that legislation does not go further than what is strictly necessary to achieve its objective. However, partly due to the lack of sewage sludge quality monitoring data, there is no systematic and periodic evaluation of the environmental and health risks from sewage sludge management routes, neither at EU level nor consistently within each of the EU Member States. In the absence of any new requirements or guidelines for periodic evaluations within the different EU Member States, this driver is likely to persist in the future.

6 IPCHEM, https://ipchem.jrc.ec.europa.eu/

4.2.2.3 *Private costs of sewage sludge management options and phosphorus disposal are not aligned with social costs*

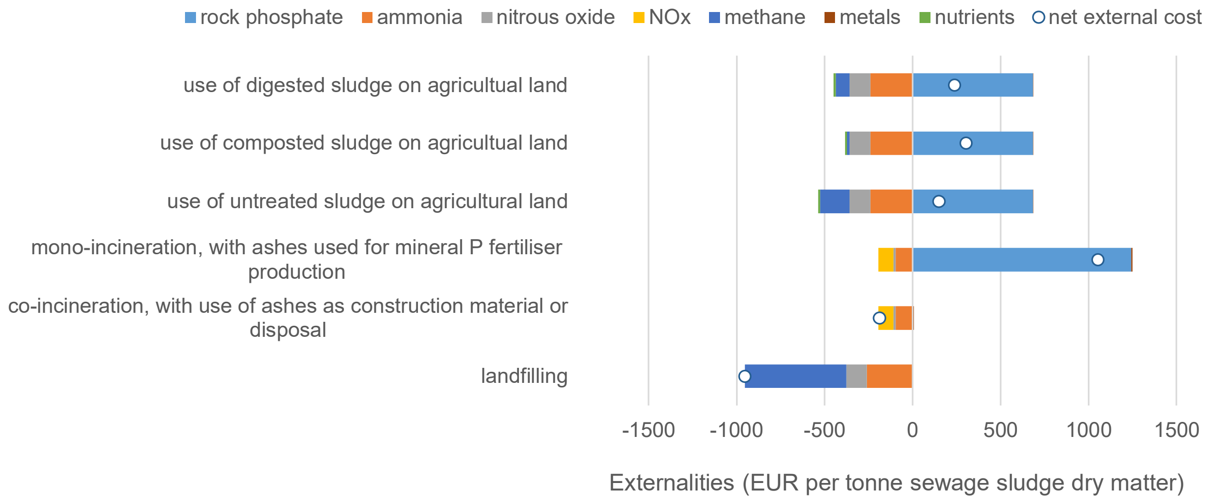
By definition, internal or – synonymously - private costs for operators involved in sewage sludge management do not include external costs borne by third parties, for instance when the latter experience adverse effects from pollution (e.g. citizens that suffer odour, noise or health-related impacts, owners of buildings that face quality loss due to air emissions). Social costs therefore consist of the sum of internal costs and external costs (Tonini et al., 2019). Budgetary considerations, i.e. the minimisation of internal costs, represent a major decision criterion for individual operators and local public entities when deciding on the treatment pathway for 'their' sewage sludge. As can be seen in the middle column of Table 2, land spreading of raw or anaerobic digested sewage sludge is one of the options with the lowest internal costs across the board, and in several EU countries landfilling is even cheaper. On the other end, mono-incineration is one of the most expensive options from the point of view of the operators. In the rightmost column, the values reported for the actual allocation of sewage sludge to the different management channels are consistent with the assumption that the direct management costs are a crucial decision driver.

Table 2. Estimates of internal (or 'private') costs of different sewage sludge management option and observed shares. *Note that what is shown here are typical costs and confidence intervals, while the full range of observed costs is even wider (see section 12.2).

Management pathway	Estimated management costs [€ per tonne of DM*]	Estimated share in total EU sewage sludge [year 2019]
Land spreading	150 (80 to 320)	34.8%
Composting of raw/digested sludge	200 (100 to 500)	12.3%
Other	200	11.4%
Landfilling	Median value of 406 in those MS with <5% landfilling, and 94 in 11 MS with >5%	35.5% in those 11 MS with landfill rates >5%; 0.9% in MS with landfill rates <5% (10.1% in total EU)
Co-incineration	250 (180 to 370)	16.5%
Mono-incineration	350 (160 to 510)	14.9%

On the other side, external costs are – by definition - not taken into account by the individual operators and can be calculated based on emission data and associated "shadow prices", which quantify the social cost of environmental emissions and resource use based on, e.g. the willingness-to-pay for preventing pollution and the systemic value of resources with absolute scarcity like fossil fuels and phosphorus (Afman et al., 2017; de Bruyn et al., 2018). Across the different sewage sludge management routes, positive and negative externalities have been identified that potentially alter the social cost and preference ranking of conventional sewage sludge disposal relative to management routes that recover and recycle resources, particularly phosphorus, from sewage sludge (Figure 9). The methodology and assumptions of the external cost assessment are explained in detail in the Annexes (section 12.5)

Figure 9. Externalities of different sewage sludge management routes (positive values are benefits to society, whereas negative values are costs).



Substitution of the valuable material rock phosphate is estimated to constitute a significant positive externality for management routes that involve the return of nutrients to agricultural or forested land (Figure 9). It is important to note that this goes beyond the direct fertilising effect of P on the land where it is applied (in fact, this is would not even qualify as externality in the narrower sense), but mainly concerns the avoided use of the globally scarce, finite and non-replaceable resource phosphorus. Through the practice of spreading sewage sludge, less rock phosphate has to be extracted and resource depletion is reduced. The global external costs of the depletion of rock phosphate have been estimated by Afman et al. 2019 under the assumption that continued use of finite stocks will in about 200 years lead to a shortage of phosphorus fertilisers, and consequently to higher food prices, famine, and loss of human lives. This yields a high worst-case shadow price for rock phosphate of 69€/kg P, more than 100 times higher than typical market prices.

This high value triggers the large social benefits associated with phosphorus recovery from secondary raw materials, including from sewage sludge. At the same time, it should be noted that the analysis of external costs is largely driven by the shadow price of rock phosphate, itself determined by the assumptions taken in the study of Afman et al. (2019) on the occurrence of "peak phosphorus" (i.e. the point in time when humanity reaches the maximum global production rate of phosphorus as an industrial and commercial raw material). The question is, however, unsettled amongst academic and industry experts and no consensus has been reached. Experts in different fields regularly publish varying estimates of the rock phosphate reserves, even calling into question the very concept of "peak phosphorus" (Edixhoven et al., 2014).

In spite of the different views amongst stakeholders on the likelihood and imminence of peak phosphorus, it can be considered a fact that there is no substitute for phosphorus in agriculture, or indeed in life. Phosphate rock is a finite resource – at some point in time the earth's supply may be exhausted. There should be a global effort to develop more effective phosphate rock mining and processing technologies and to utilize phosphorus fertiliser, other phosphate-based products and phosphorus-containing waste as efficiently as possible, while keeping unused nutrients out of watersheds and the oceans. Furthermore, the EU is largely dependent on imports as rock phosphate mines located in the EU can at maximum supply only 15% of the current phosphorus EU-demands. The EU's dependency on foreign rock phosphate is currently not reflected in its market price, and could thus also motivate a higher shadow prices of phosphorus. Also the increases in phosphorus prices following the 2022 Russian invasion of Ukraine can be seen as a proof of the vulnerability of the EU food system and the affordability of imported fertilisers. In the effort to increase the EU's strategic autonomy for phosphate demand, the recycling of phosphorus from biogenic waste has been brought forward as a promising option (Schoumans et al., 2015; MacDonald et al., 2016). Therefore, a high shadow price seems justified, and there seems to be a consensus amongst stakeholders, including industry and academics on the need to develop alternative sourcing strategies for rock phosphate (Bennett, 2020).

In addition, the European Critical Raw Materials Act[7] and the associated proposal for a Regulation on establishing a framework for ensuring a secure and sustainable supply of critical raw materials[8], lists phosphate

[7] Available at: https://ec.europa.eu/commission/presscorner/detail/en/ip_23_1661
[8] Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52023PC0160

rock and phosphorus as 'critical raw materials' (see Annex II of the proposal). The Act proposes a comprehensive set of actions, including the improvement of circularity and sustainability of waste and secondary raw materials, to ensure the EU's access to a secure, diversified, affordable and sustainable supply of critical raw materials. The Regulation establishes monitoring and national measures on circularity for critical raw materials. The Regulation also proposes to adopt and implement national programmes containing measures designed to increase the collection and recycling of waste with high critical raw materials recovery potential and ensure their introduction into the appropriate recycling system (Article 28 of the proposal). This further points towards the importance of phosphorus recovery from phosphorus-rich waste streams such as sewage sludge.

Due to this strong effect, and in spite of some negative externalities associated with the emissions of ammonia, nitrous oxide, methane and metals from sewage sludge land spreading, all routes that return sewage sludge-derived phosphorus to land show a positive external effect. The processing pathway that uses mono-incineration ashes to produce a mineral phosphorus fertiliser is associated with the highest positive externality due to the high plant-availability of the phosphorus in the sewage sludge-derived end material (Figure 9). The disposal pathways co-incineration and landfilling, the only ones which do not recover phosphorus, are associated with the highest net external costs to society, where in the latter case the main cost driver are methane emissions. In the absence of legislation and/or actions to correct for this market failure, this problem driver will persist in the future.

Taken together, these estimates point to a market failure, because the private costs and relative ranking of options as seen by the actors responsible for decisions on sewage sludge management are not aligned with the social costs and preference ranking (i.e. private costs plus external costs as provided in the Annexes, section 12.5). In particular, when adding to the reported private costs of landfilling the almost 1 000 € t^{-1} external costs, it results to be the least preferable option among all from a social point of view. Likewise, if the external benefit of more than 1 000 € t^{-1} is subtracted from the relatively high private costs of mono-incineration, it turns out to be the most beneficial option from society's point of view. This is clearly inconsistent with the rationale and financial incentives currently faced by waste water treatment plant operators who take the decisions on sewage sludge treatment.

A consequence of this market failure and the absence of tangible financial returns is that there are insufficient economic incentives for private actors to accelerate the uptake of innovative technologies that turn sewage sludge mono-incineration ashes into a mineral P fertiliser (section 13.3). In other words, private finance is undersupplied because the market revenues from the sales of the produced mineral fertiliser are generally insufficient to compensate for the expenses implied by the recovery process (e.g. purchase of supplementary chemicals and reactors) (Tonini et al., 2019). This aggravates the problem, because it inhibits innovation and technological progress and thereby induces a lock-in into current – suboptimal - practices. The evolution and persistence of this problem will depend on the total revenues and possibility of full cost recovery of P-recovery plant operators, which in turn depends on the future market prices of mineral P fertilisers derived from primary sources (rock phosphate).

4.3 EU added value

The added value of tackling the problems at EU level is in our view threefold:

First, there is a transboundary dimension because food and feed products which may contain contaminants from sewage sludge are traded and sold throughout the EU. Currently, maximum limits for certain food contaminants are set out in Commission Regulation (EC) No 1881[9]; this list includes, amongst others, metals and certain organic contaminants such as PAHs and PCDD/F. Nevertheless, given that sewage sludge contains a specific set of contaminants, some threats are not addressed by current EU food safety legislation, and additional EU legislation may be required to protect food costumers that eat food grown on sewage sludge amended soils. To preserve the integrity of the EU Single Market and to ensure that EU-wide health protection provides the same (minimum) level of safety, policies at EU level seem to be preferable. Action taken at EU level could contribute to the harmonisation of methodologies used by Member States to ensure a similar of protection of the environment, and products grown on sewage sludge amended soils placed on the internal EU market. In addition, it seems more cost-effective and proportionate for Member States to jointly undertake scientific discussions and request opinions from EU risk assessment bodies. In the interests of the Community and for a more effective regulation, it is appropriate to set up safeguards and measures for selected pollutants that are of most and joint concern.

[9] Commission Regulation (EC) No 1881/2006 of 19 December 2006 setting maximum levels for certain contaminants in foodstuffs.

Second, certain emissions (e.g. eroded microplastics from soils, ammonia and methane emissions to air) may cause direct transboundary pollution, thus causing adverse effects in a different MS than the one where the sewage sludge was originally applied. Transboundary pollution is generally recognized as a typical policy issue that is addressed most efficiently at the supranational, here EU, level.

Thirdly, sewage sludge may have a potentially irreversible impact on soil health and its capability in terms of providing ecosystem services and contributions to EU citizens in the long-term. Soil is home to an incredible diversity of organisms that regulate and control key ecosystem services such as soil fertility, nutrient cycling and climate regulation. Soil is a highly important non-renewable resource, vital for human and economic health, as well as for the production of food and new medications. Additionally, sound sewage sludge management may help to address resource depletion and dependency on critical materials mined outside the EU (e.g. rock phosphate for food production). While national policies can still be effective, given the global nature of these challenges, EU-wide harmonised policy will make the task of reaching a sustainable future a smoother operation (von Weizsäcker, 2011).

5 What should be achieved?

5.1 General objectives

The general objective is to increase the resource efficiency of sewage sludge management, and to ensure a high level of environmental and health protection from sewage sludge returned to the environment. The overall aim is to modernise the sewage sludge legislation by adapting it to current and future societal needs and to the objectives of the European Green Deal and its Circular Economy Action Plan, and the Commission's Zero Pollution objective.

5.2 Specific objectives

The specific objectives of a new policy for sewage sludge management are twofold and consider environmental and socio-economic concerns.

Firstly, the revised policy on sewage sludge would aim to ensure environmental and human health protection when sewage sludge is returned to the environment, possibly after (biological) treatment. Particularly, adverse impacts from persistent organic pollutants, metals, and microplastics should be reduced. A new policy should achieve a high level of confidence for produce grown on sewage sludge amended soils for food consumers, food producing companies, and national authorities.

Secondly, sewage sludge management should be aligned with circular economy principles. The circular economy aims to maintain the value of products, materials and resources for as long as possible by returning them into the product cycle at the end of their use, while minimising nutrient pollution. It would involve designing sewage sludge management systems with a greater nutrient use efficiency where nutrients contained in sewage sludge can replace primary raw materials, such as fertilisers derived from primary raw materials. First and foremost, sewage sludge management should consider the sustainable management of phosphorus as a critical raw material. Additional benefits of a smaller order of magnitude would be to recover other nutrients and organic matter from sewage sludge.

5.3 Operational objectives

The following two operational objectives should be met in an efficient manner, with minimal burdens to actors involved in the management of sewage sludge and respecting the principles of the single market and encouraging competitiveness and innovation:

— Identify relevant contaminants present in sewage sludge applied in the environment, and ensure that outcomes are made available and taken forward when developing guidelines for full environmental and health protection from spreading sewage sludge and/or derived materials on land;

— Correct the absence of market incentives for operators to recycle phosphorus and possibly other valuable resources (organic matter, other nutrients) that are present in sewage sludge, in line with the waste management hierarchy;

The meeting of these specific objectives also aims to strengthen coherence with policy initiatives on environmental monitoring, and the mitigation of nutrient and greenhouse gas losses from nutrient applications on agricultural land. Thus, the specific actions should support initiatives outlined in the EU Farm-to-Fork Strategy and EU Biodiversity Strategy.

6 Baseline

The baseline for this feasibility study assumes that no policy change will take place. Hence, under this baseline the Sewage Sludge Directive 86/278/EEC in its current form continues to apply. Building upon the understanding of the drivers, the starting point for the development of the baseline is 2019 (status quo). The assumed timeline for the baseline is the reference year 2050 that acts as a long term horizon. Whereas certain parameters of this reference year can be projected with a reasonable degree of certainty (e.g. demographics, connection rates to centralised waste water treatment), other parameter assumptions have clearly a more unpredictable nature or cannot be rationally projected (e.g. emergence of disruptive technology changes, political priorities in the field of environment). Therefore, the baseline should be interpreted as a best possible outlook based on the present-day knowledge and conservative assumptions on the absence of breakthrough technologies and game-changers in the field of sewage sludge management.

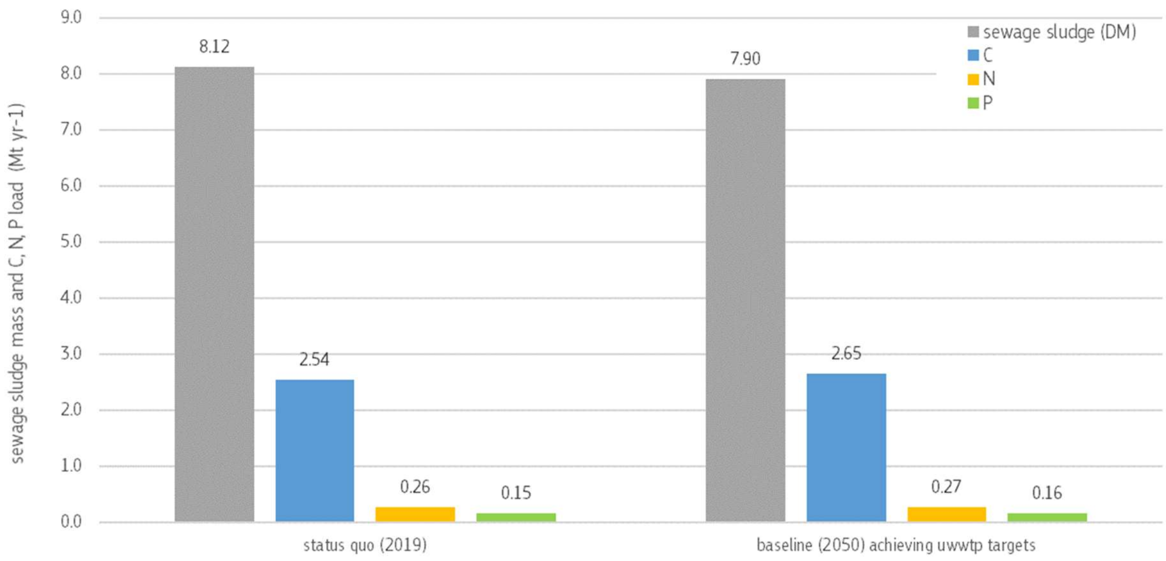
The baseline focusses on the expected evolution of the problems and consequences identified in section 4.1.

6.1 Sewage sludge mass, nutrient and organic matter content, and management routes

The outlook of the total mass of generated sewage sludge and its relative allocation to main sewage sludge processing routes is of key importance to estimate the environmental, social and economic impacts from sewage sludge management within the EU. The baseline has been developed based on the consideration of numerous drivers, including demographic evolution, national legal framework on sewage sludge in different EU Member States, and expected evolution of the main policies affecting sewage sludge generation and management (e.g. Urban Waste Water Treatment Directive, Landfill Directive, EU Biodiversity Strategy targeting nutrient loss reductions). The full details of the assumptions for the baseline, as well as the estimated shifts at MS level, are outlined in the Annexes (section 11.1.2).

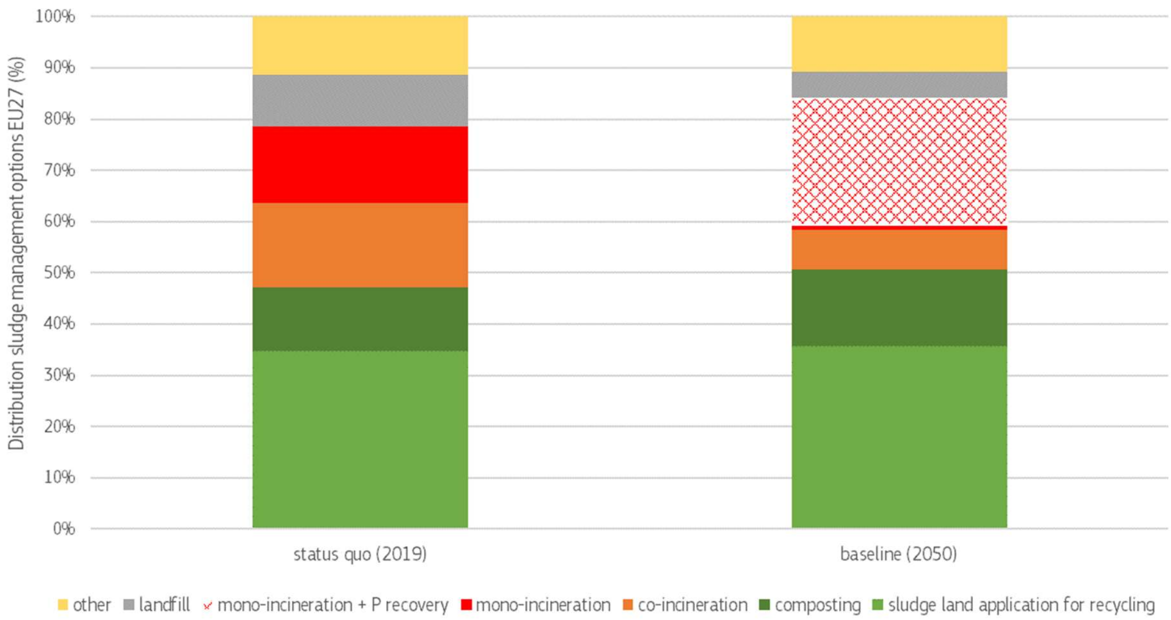
For the total sewage sludge mass, the results indicate minor changes because the different drivers have opposing impacts, leading to an overall neutral outlook. On the one hand, the increased population connected to centralised water treatment and increased efficiency to remove solids from waste waters would by itself increase sewage sludge volumes. On the other hand, increased processing of sewage sludge via anaerobic digestion and other processing techniques reduces the sewage sludge mass that is exported from the waste water treatment plant. Altogether, both effects largely compensate each other, with an estimated minor net reduction in sewage sludge mass generated of -3% (Figure 10). Also for the total organic matter content (+7%), nitrogen content (+4%), and phosphorus content (+7%), only minor positive differences compared to the 2019 status quo are expected (Figure 10).

Figure 10. Sewage sludge mass and C, N, P load in the status quo (2019) and baseline (2050) (Mt yr^{-1}) (see section 11.1.1 and section 11.1.2).



With respect to the sewage sludge management routes, we expect more substantial changes by 2050 compared to the 2019 status quo (Figure 11). First, a shift through an increased uptake of innovative technologies that recover P in mineral from sewage sludge following mono-incineration. This is due to the expected improved cost-competitiveness of such technologies relative to other sewage sludge disposal options (co-incineration, landfilling), and a resulting increased enforcement of the waste management hierarchy that promotes recycling over energy recovery. As a matter of fact, some MS have already outlined legislative proposals to mandate sewage sludge management with resource recovery for larger waste water treatment plants. A direct consequence is a reduction in co-incineration of sewage sludge. Secondly, we expect further reduction in the amounts of sewage sludge that will be landfilled, in line with a further implementation of the landfill directive and the methane strategy, continuing the historic trends of sewage sludge landfilling reductions within EU-27 Member States where such practice is still occurring (e.g. BG, ES, FR, GR, PL; with the exception of RO; Eurostat, 2022). Another observation is that the expected shares of (untreated or treated) sewage sludge that will be landspread on agricultural soils and outside the food chain (mostly comprised in the fraction 'other uses', often unidentified) is expected to remain relatively stable in the baseline, as it is assumed that possible reductions in land spreading as a result of concern on soil and water contamination in some MS would be largely neutralised by diverting the landfilling of sewage sludge to agricultural land in other MS (Figure 11). A detailed sewage sludge distribution on MS level is given in the Annexes (Figure 39).

Figure 11. Final uses for sewage sludge management in the status quo (2019) and baseline (2050) (own computations).



6.2 Environmental and health protection

At the EU-27 scale, the baseline results presented in section 6.1 indicate an overall stable amount of sewage sludge landspreaded onto agricultural soils compared to the 2019 situation (status quo). The revision of the legislative framework (REACH Regulation, and the interface with the Regulation on persistent organic pollutants (POPs), legislative initiatives on microplastics pollution), as well as the objectives set forth in the chemical products waste communication will further improve the safety of sewage sludge because of the increased information availability of physico-chemical and toxicological properties, ensuing risk assessments, and the phasing out of substances of the highest concern. The POPs Regulation will additionally restrict the further use of persistent, bioaccumulative and toxic substances, identified as being those of main concern in a recent JRC study (Huygens et al., 2022). However, even if we were to assume that the future legislation on chemicals and other pollutants will address current shortcomings, the incidence of environmental and health risks from sewage sludge will remain unknown because of two main reasons:

(i) Waste waters consist of mixtures of household and industrial discharges of different qualities. Often, waste water treatment entities do not have access to information on the composition of the (locally generated) waste waters they handle because the information either does not exist, amongst others as contamination may be dependent on local settings and the use phase of certain products (e.g. microplastics release from clothes). In addition, sewage sludge can be contaminated by substances that are unintentionally released into the environment (e.g. during combustion processes). Risk models used for the evaluation of chemicals to be placed on the market are unable to track and estimate the occurrence of pollutants throughout their life cycle to sewage sludge as a sink material. Therefore, the levels of pollutants in sewage sludge remain largely elusive without sewage sludge quality monitoring campaigns, leading to incomplete risk assessments, especially when considering local areas subject to repeated applications of sewage sludge with a particular contaminant profile. The communication up and down the supply chain on uses and necessary risk management measures lacks accuracy and clarity, which has a significant negative impact on the control of risks from sewage sludge management;

(ii) The safety assessments for chemicals placed on the market, including those present in waste as covered under the POPs Regulation, do not take combination effects of chemicals in sewage sludge into account. Individual registrants are only responsible for their own substances and do not take into account that, in reality, sewage sludge as an end-of-life sink for some contaminants contains a plethora of different substances from different sources. Thus, securing safe use of one substance is in itself not sufficient for protecting humans and the environment against combination effects.

Overall, it is unlikely that without further change in policies on sewage sludge, the drivers "Insufficient sewage sludge quality monitoring data", and "Lack of a systematic and periodic evaluation of health and environmental risks from sewage sludge management" will be addressed in the baseline. Therefore, it remains unknown if a future effective and well-functioning legislative framework on chemicals and industrial emissions, together with eco-design requirements for products, could fully address humans and environmental risks from pollutants in sewage sludge. Hence, complementary monitoring and, if necessary, control measures may be required to ensure health and environmental protection from sewage sludge applied in the environment.

6.3 Resource efficiency

Relative to the status quo, P recovery from sewage sludge is expected to increase in the baseline following the expected shifts from disposal towards mono-incineration with phosphorus recovery from the resulting ashes (see section 6.1). The latter pathway transforms the P present in sewage into a mineral nutrient source with a much greater plant nutrient availability, and therefore has a greater potential to substitute mineral fertiliser. The P contained in sewage sludge is by about 55% available to plants during the first year following application (Oenema et al., 2012; Delin, 2016). In the status quo, it is thus estimated that in absolute terms only about 0.03 Mt yr^{-1} of P is available to plants in the first year following application. In 2019, the total estimated use of mineral P fertilisers is about 1.1 Mt yr^{-1}, indicating that sewage sludge is not a main P source in the wider context. In the baseline, this number is expected to increase to 0.07 Mt yr^{-1}, because a larger share of P is returned following the transformation of sewage sludge into a mineral P fertiliser. At the same time, it is estimated that mineral P fertiliser consumption may decrease due to a tightening of the rules for the use of fertilisers in general and an increased nutrient use efficiency due to technological developments (e.g. precision farming) (Fertilizers Europe, 2021). As a matter of fact, the EU Farm-to-Fork and Biodiversity Strategy targets a reduction in mineral fertiliser use by 20% by 2030. In case mineral fertiliser reduction would be reduced by 30% by 2050 (i.e. 0.77 Mt P yr^{-1}), plant available P from sewage sludge could in the baseline account for a mineral fertiliser substitution of roughly 10%. This indicates that in the baseline, sewage sludge might play a greater relative role as a phosphorus source in EU agriculture.

The absolute amounts of nitrogen and organic matter are largely unchanged between the status quo (2019) and baseline (2050) because (i) the total sewage sludge amounts generated are similar between both (section 6.1), (ii) similar amounts of sewage sludge are returned to agricultural land (section 6.1) and (iii) thermal treatment removes organic matter and nitrogen from the sewage sludge.

6.4 Nutrient losses

Provisions on limiting nutrient loss from sewage sludge are included in the Directive (Article 8), but the recent amendment related to the reporting requirements in environmental legislation (Regulation (EU) 2019/1010) is expected to further enable the enforcement of this requirement. The 2019 regulation will ensure a higher level of transparency, whereby the required information (e.g. on spatial locations subject to sewage sludge applications, application rates) will be made available in an easily accessible manner by electronic means. In addition, reducing nutrient losses by 50% has been enshrined in the EU Biodiversity and Farm-to-Fork strategy. It is projected that these provisions will lead to further reductions in nutrient losses in the baseline compared to the 2019 status quo, particularly in certain MS (see below). A quantitative assessment of the overall contribution of sewage sludge to observed nutrient losses from agricultural fields as well as a solid outlook on the evolution of the problem is, however, challenging.

The latest implementation report of the Nitrates Directive indicates that MS with a high percentage of surface waters in eutrophic status include BE, CZ, DE, DK, EI, FI, HU, NL, and PL (European Commission, 2021b). About half of these countries already have already effectively restricted sewage sludge application in the environment during recent periods. However, CZ, DK, HU, PL and FI still apply more than 65% of their sewage sludge on agricultural soils. Although the lack of reported spatial data for sewage sludge applications on land does not enable drawing absolute conclusions, it seems plausible that a share of the nutrients contained in sewage sludge contributes to surface and groundwater pollution in these MS. With a view to align to EU targets, it is assumed that the land application of sewage sludge applied in CZ, DK, HU, PL and FI will be reduced by one third (33%) relative to the status quo in the baseline.

6.5 Methane emissions

In 2019 (status quo), 10.1% of total sewage sludge production or 0.83 Mt of sewage sludge dry matter was landfilled, contributing to the overall methane emissions associated with the waste sector (4.2 Mt CH_4 yr^{-1}) (EEA, 2021a). Methane emissions from sewage sludge landfilling in the baseline (year 2050) are expected to decrease, due to three main factors: (i) reductions in sewage sludge masses that are landfilled, in line with the targets to reduce the landfilling of biodegradable waste in the landfill directive (see section 6.1), (ii) increased stabilisation and anaerobic digestion of sewage sludge prior to landfilling, and (iii) increased enforcement to treat and use landfill gases in line with good management practices.

As outlined in the 2020 EU strategy to reduce methane emissions, the Commission will help Member States and regions to stabilise biodegradable waste prior to disposal and its increasing use for the production of climate-neutral, circular bio-based materials and chemicals, and divert this waste to biogas production. Therefore, the baseline assumes further reductions in the landfilling of sewage sludge, particularly for MS with a landfilling share above 5% (BG, CZ, EE, ES, GR, HR, IT, LT, MA, RO, SK, with an assumed reduction of 50% compared 2019 status quo). This assumption is based on the persistence of the current trends of reductions in sewage sludge landfilling observed in the last decade (Eurostat 2022). Substantial reductions in methane emissions from the landfilling of sewage sludge are also expected due to the increased sewage sludge shares that will be subjected to anaerobic digestion. Such stabilisation treatment is increasingly introduced at larger waste water treatment plants due to increased cost-efficiency and strategies on energy recovery at waste water treatment plants. In the baseline, we assumed that anaerobic digestion takes place in waste water treatment plants of sizes above 50k p.e., that generate about 70% of the sewage sludge. Finally, the landfill directive (Annex I, paragraph 4.2) indicates that "landfill gas shall be collected from all landfills receiving biodegradable waste and the landfill gas must be treated and used. If the gas collected cannot be used to produce energy, it must be flared". Gas collection systems of well monitored landfills with liners are capable to collect on average 80% of the landfill gas over a 20-year life span (Clavreul et al., 2014; Olesen and Damsgaard, 2014), and 99% of the collected methane is converted to less harmful CO_2 with subsequent flaring of the landfill gas. According to EEA (2021a), at present 84% of the EU-27 landfills are managed landfill sites, indicating that landfill gas is at least collected and flared. Caicedo-Concha et al., 2021 highlights that the collection and combustion of landfill gas in flares reduces the global warming potential by up to 60%. Hence, further compliance with the requirements of Annex I of the Landfill Directive will further reduce the impact of sewage sludge on the total methane emissions.

Altogether, the baseline assumes that the combination of these measures will reduce CH_4 emissions from the landfilling of sewage sludge from presently 0.051–0.063 Mt CH_4 yr^{-1} to 0.016–0.021 Mt yr^{-1}; the latter would correspond to only 0.11% of year 2019 total EU-27 methane emissions (15.2 Mt CH_4 yr^{-1}) (see Annexes, section 11.1.2.3).

6.6 Annual costs for sewage sludge management

Based on the assumptions presented in the Annexes (Table 44), the annual costs for sewage sludge disposal in the baseline are 1 883 M€. Here, annual costs are total EU sewage sludge volumes multiplied by the specific costs of the different assumed treatment pathways (financial costs, without externality[10]). The amount is slightly higher than the annual costs calculated for the status quo (1 829 M€ yr^{-1}), even though the sewage

10 In lack of relevant evidence and data, the specific costs of each sludge treatment channel are assumed to be the same in 2050 as in 2020

sludge volume is lower in the baseline than in the status quo. The main reason for this is that two Member States have already introduced mandatory technical P recycling by 2050 and it was also assumed for other MS that in the status quo mono-combusted ashes will be subject to P recovery by 2050. In addition, the sewage sludge that is already mono-incinerated in the status quo, will be subjected to P recovery by 2050.

6.7 Baseline implications for setting objectives for a revised policy framework

The assessment of the baseline and drivers indicates that progress will be made towards addressing all problems observed. Nonetheless, it is clear that two main problems will persist in the baseline: human health and environmental risks from sewage sludge landspreading on agricultural soils, and the disposal of resources, particularly, phosphorus from sewage sludge. Additionally, nutrient losses from sewage sludge applications on agricultural land may continue to occur when not properly addressed in a future by EU and national legislation. The fourth problem, methane emissions from the landfilling of sewage sludge, will be further reduced in the future to very low levels and no specific measures in relation to this problem seem to be required.

7 What are the measures and options to achieve the objectives?

7.1 General considerations

Policies to be considered should be closely linked to the drivers of the problems (section 4.2) and the identified specific objectives (section 4.3): a clear logic and sound principles should underpin the intervention under consideration. E.g. negative externalities should be addressed based on the polluter-pays principle, and regulation should create incentives for those actors who have the possibility to actually change their behaviour. While policies address the identified problems in their entirety, individual 'measures' only address certain aspects of the overall problem, or they are only effective when taken in combination with other measures. A policy or policy option is then a combination (or a package) of policy measures.

Based on this terminology, different measures are considered for tackling the two main individual problems identified to persist in a future when considering baseline development: (i) environmental and health risks from sewage sludge applications in the environment, and (ii) resource losses causing pollution and the further depletion of fine resources from current sewage sludge management.

The considered policy measures are based on a review of practices in the different EU Member States as well as on a techno-scientific assessment. Policies should follow the proportionality principle, which means that actions should not go beyond what is necessary to achieve the objective. Proportionality is about matching the policy intervention to the size and nature of the identified problem and its EU (subsidiarity) dimension in particular. One of the key aspects of proportionality is the right choice of policy instruments to achieve the desired policy objective. Therefore, the considered policy measures differ in their level of ambition to match the achievement of the objective, without imposing unnecessary burdens and costs. A policy option can also be extended or split into sub-options; these are very similar packages of measures that only differ in one aspect, e.g. in their level of ambition.

In a first stage, numerous potential policy measures were identified for their possible inclusion. Following the screening of options, some measures were discarded, amongst others because of a lack of technical feasibility, effectiveness to address the objectives or proportionality (see section 7.2.2).

7.2 Policy measures to address environmental and health risks from sewage sludge applications

Although developing full-fledged policy measures goes beyond the scope and mandate of this feasibility impact assessment study, this report aims to describe how policy measures would be implemented, monitored and enforced, by whom and over what timeline in order to enable a good understanding for the reader of this report. It should, however, be clear that these measures require further discussions and possible modifications based on inputs from stakeholders, including national and Commission experts, in the next stages of the policy development process.

7.2.1 Proposed measures for in-depth assessment

The policy measures that are retained to address the objective of human health protection are based on (i) the monitoring and control of sewage sludge applied on land or elsewhere in the environment, and (ii) restricting the use of sewage sludge and sewage sludge-derived materials for land spreading to EU fertilising products that are compliant with Annex I to IV of the EU Fertilising Products Regulation (Regulation (EU) 2019/1009).

7.2.1.1 Monitoring and control of sewage sludge applied in agriculture

Policy measures based on the monitoring and control of sewage sludge applied on land are inspired by similar provisions laid down in the Water Framework Directive (2000/60/EC). This Directive covers surface water pollutants in two ways – by requiring Member States to identify substances of national or local concern (included by Member States in their so-called River Basin Management Plans), and by identifying and regulating those of greatest concern across the EU ("priority substances" – listed in Annex X to the Directive). A similar requirement for the identification and regulation of contaminants in sewage sludge used in agriculture would effectively address the drivers of the problem observed (section 4.2.2; insufficient sewage sludge quality monitoring data; lack of periodic and consistent risk evaluation).

Concretely, this policy measure could include the following main elements:

- Member States have to identify pressures to the contamination of soils from sewage sludge by identifying and monitoring contaminants present in unprocessed or processed (e.g. following composting and digestion) sewage sludge applied on agricultural, based on technical guidance provided by the Commission (e.g. the nature and types of substances that are of most concern; pollutants identified as being of concern in other pieces of legislation, such as the possibly upcoming EU Soil Health Law). Member States shall encourage the active involvement of all interested parties in the monitoring campaign and discussion of the results thereof, particularly waste water treatment plant operators, companies involved in sewage sludge processing (e.g. composting plants), industries discharging to municipal waste water treatment plants whose sewage sludge are spread in the environment, academics and non-governmental organisations. Monitoring techniques will involve the targeted measurement of priority pollutants as identified by the Commission as well non-targeted screening of chemicals. Monitoring frequencies will be developed reconciling data collection and sampling/measurement costs (e.g. 1-2 samples per year for larger waste water treatment plants). The use of composite samples with materials from different (smaller) waste water treatment plants in similar settings will be encouraged to ensure cost reduction.
- The results of the monitoring results will be made available through reporting in public databases to ensure access and further usability of interested parties. Such actions will facilitate identifying priority substances in environmental legislation (e.g. SVHCs for REACH, priority substances for Water Framework Directive, POPs nomination), and thus promote actions to phase out contaminants at source. The use of digital platforms should be encouraged as a means for Member States to comply with their obligations to report chemical occurrence data and to simplify and reduce their reporting obligations. The Information Platform for Chemical Monitoring (IPCHEM) or other databases managed by the EU institutions could for instance be used for reporting purposes and as a single access point for the chemical occurrence data in all media across the EU.
- Member States shall submit summary reports of the monitoring programmes designed, every three years. The reporting shall also include reporting obligations in the field of legislation related to the environment (Regulation (EU) 2019/1010), including information on the sewage sludge quantities supplied for use in agriculture and the spatial location where the sewage sludge is used.
- Based on opinions of risk body agencies, the Commission may establish limit values for "priority pollutants" that are known to cause a significant risk to human health by defining specific limit values for sewage sludge applications on agricultural land, defined as loads (i.e. mass inputs of contaminants over a period of time, e.g. 5 of 10 years). Together with the requirement for the reporting of spatial locations of sewage sludge applications, this will enable flexibility to Member States to adapt application rates as a function of the available land according to local and regional settings. Sewage sludge containing pollutants in concentrations exceeding the limit values shall be banned for use in agriculture. At present, the identity and limit values of such pollutants remains to be defined, but it may include metals, organic pollutants, and other substances (e.g. microplastics). Additionally, conditions for receiving soils (e.g. maximum contents) and application rates could be envisaged. These controls and limit values shall periodically be reviewed (e.g. every 6 years) and, where necessary, updated. This enables a dynamic legislation to exclude pollutants that have become irrelevant for human health and environmental protection, and include new pollutants of novel concern. This will ensure that the legislation continues to be relevant over time, and may further reduce administrative burdens in case contaminants are being phased out. Member States may impose additional measures as part of their national legislation (similar to current provisions).
- Sewage sludge that will be used for recycling or recovery operations outside agriculture (e.g., for landscaping, for forestry applications, for backfilling) will not be subject to specific requirements at EU level. Member States would have to set up measures to ensure environmental and health protection from such sewage sludge uses. In our view, the added EU value (section 2.4) of regulating sewage sludge use outside agriculture is low and the proportionality of the measure could thus be challenged. These shares of sewage sludge do not enter the food chain, for which reason the human health risks of using sewage sludge outside agriculture are lower. Moreover, the JRC report suggest humans as the most sensitive end point (Huygens et al., 2022). The potential for transboundary pollution from sewage sludge used outside agriculture is not as expressed because of the relatively small shares of sewage sludge applied for such application. In addition, the uses outside agriculture are numerous and highly context-specific (e.g. at golf courses, for erosion control, forestry, restoration of degraded lands) (N. Anderson et al., 2021), and it may be challenging to develop risk assessment based on a "typical" environmental release scenario. This makes it also very challenging to set "single EU-wide limit values" for sewage sludge uses outside agriculture, without

infringing the principle of subsidiarity. It is understood that the issue could be dealt more effectively by Member States themselves at central, regional or local level on condition that extensive information obtained from sewage sludge quality monitoring is available. A requirement would be set in EU legislation that Member States would be obliged to address environmental and health risks from sewage sludge applications for uses other than agriculture.

7.2.1.2 Ban on sewage sludge land application for recycling, unless treated and processed into an EU Fertilising Product

In this policy proposal, sewage sludge generated at waste water treatment plants above a specific size would not be allowed for use as a fertilising materials. Only derived products compliant with Annex I to IV of Regulation (EU) 2019/1009 (the EU Fertilising Products Regulation) could thus be applied as a fertilising material on agricultural land. Sewage sludge use on land would be permitted for sewage sludge originating from smaller waste water treatment plants when rules (at national or EU-level, see policy options) for safe use are set.

Due to the absence of sound monitoring data on sewage sludge quality and lack of data on risk properties and evaluations for the numerous (unknown) contaminants, as well as the fact that sewage sludge is a main source of microplastics in the environment, it would not be allowed to use sewage sludge originating from larger waste water treatment plant for recycling operations outside the food chain. De facto this measure implies the mandatory transformation of sewage sludge into an EU Fertilising Product that meets end-of-waste criteria laid down at EU level for use as a fertilising material. This measure mandates that sewage sludge-derived materials should meet the strictest requirements so that they can be placed on the internal market, without further control measures (e.g. on application rates, or quality requirements for receiving soils).

At present, precipitated phosphate salts (Component Material Class 13 as per Annex II of the Fertilising Products Regulation) and thermal oxidation materials and derivates (Component Material Class 14 as per Annex II) allow the use of sewage sludge as an input material. This would, for instance, imply that sewage sludge ashes could be used in mineral phosphorus fertiliser production processes. At the same time, it is important to note that Fertilising Products Regulation has a dynamic nature, and that the Commission has a mandate to develop new Component Material Classes, to account for technological and scientific developments. Hence, if conditions for granting an End-of-Waste status (see Article 6 of Directive 2008/98/EC) are met, additional sewage sludge-derived materials could be covered in the future.

Smaller waste water treatment plants, often located in rural areas with more agricultural area and thus possibilities to spread sewage sludge at lower application rates, would be exempted from this obligation. Still, the aim would be to limit sewage sludge application on agricultural land substantially.

This measure is similar to the legislative framework proposed in some EU Member States, such as Germany and Austria.

The measure aims at ensuring a higher level of environmental (or health) protection through preventative decision-taking in the face of risk. This policy measure relies on the precautionary principle, because there is the potential for serious harm, but scientific uncertainty remains about the type or magnitude of that harm. The mandate of environmental protection as a basis for regulation provides the Commission with a relatively wide margin of discretion, which is due to the fact that it is one of the fundamental objectives of the Union and the EU policy on the environment is to aim at a high level of protection. This measure thus departs from the observation that sewage sludge may be a potential sink for the large variety of chemicals applied in society and released into the environment (see section 4.2), and the present unknown effects of other contaminants such as microplastics. This option would circumvent the need to monitor and control sewage sludge quality, and their associated administrative and cost burdens.

Concretely, this policy measure could include following main elements in a first sub-option:

- Member States shall prohibit the use of sewage sludge and sewage sludge-derived materials for agricultural use, unless treated sewage sludge and derived materials thereof shall qualify as EU Fertilising Products defined in Regulation (EU) 2019/1009. The policy measure shall apply to sewage sludge treatment plants with a capacity greater than a set threshold.
- Sewage sludge for use in or outside agriculture (e.g. for landscaping, for forestry applications, for backfilling) originating from waste water treatment plants below a threshold size will be subject to requirements, either set at national or EU level.
- A transitional period (e.g. 5 years) shall be set.

— The Commission shall carry out an evaluation of this Directive, e.g. 12 years after its entry into force.

7.2.2 Other possible policies that are not further considered in the analysis

7.2.2.1 Monitoring framework, without possibility to set EU-wide limits for contaminants of greatest concern measured in sewage sludge as part of the SSD

Description of the measure

In this policy proposal, Member States would have to set up a monitoring framework to identify contaminants of highest concern in sewage sludge. Based on the data collection, Member States could then individually draw up quality requirements and limit values for sewage sludge as part of their national legislation.

Potential advantages of the measure

Based on the current state-of-knowledge, it remains largely unknown to what extent sewage sludge quality depends on local characteristics, e.g. industries discharging to the waste water treatment plant, technological configurations at the waste water treatment plant etc. Setting Member State-specific limit values may enable to reduce administrative and cost burdens associated with sewage sludge quality compliance demonstration when certain pollutants are not equivalently relevant in all EU areas (strong role for subsidiarity).

Reason for exclusion as a policy measure: relevance, effectiveness and efficiency

Not all Member States have quality rules for fertilising materials applied on agricultural soils. In spite of concerns about organic contaminants (e.g. PFAS) returned to agricultural soils via sewage sludge land spreading, only a limited number of Member States have undertaken actions that go beyond the year 1986 SSD. Moreover, limit values set by Member States vary broadly (European Commission, 2022). This points towards uncertainties on the effectiveness of the measure at Member State level to address the needs of the policy objectives on environmental and human health protection. Therefore, this measure may not be fully aligned to the Commission's Zero Pollution Action Plan. Additionally, the benefits for the environment and human health protection might partially come as a consequence of harmonised rules for produce from sewage sludge-amended soils, particularly when sewage sludge quality criteria are developed following exchanges between EU experts and in cooperation with international risk assessment bodies. Finally, it may be more cost-efficient and provoke lower administrative burdens for Member States to undertake recurrent extensive technical assessments based on a joint approach as part of a single, EU-wide coordinated action.

Take-aways and cross-fertilisation for policy measures taken forward in the assessment

In line with the subsidiarity principle, the selected policy measure would benefit from some degree of flexibility on contaminants that should be prioritised and controlled for in the compliance scheme, and to limit them to those contaminants that present significant risks. It is noted that EU-wide limit values will only be set when these will be effective and feasible, also taking into consideration the estimated persistence of the contaminant and the time requirements for the development of international standards (see section 9.3). This could reduce unnecessary cost and administrative burdens for actors involved in sewage sludge management.

7.2.2.2 Repeal the SSD and self-regulation based on voluntary standards

Description of the measure

This measure would involve the repeal of the SSD and the introduction of self-regulation to ensure human health and environmental protection from the use and management of sewage sludge on agricultural land. Self-regulation is where business or industry sectors formulate codes of conduct or operating constraints on their own initiative for which they are responsible for enforcing. However, pure self-regulation is uncommon and at the EU level it generally involves the Commission in facilitating the drawing up of the voluntary agreement. Self-regulation for land applications of sewage sludge is relatively uncommon in the EU, and to the best of our knowledge presently limited to certification schemes in Sweden, France and Germany. Also the United Kingdom has a certification scheme. The main elements of these certification systems are based on control of the quality and nature of the discharges, sewage sludge quality (mostly metals, microbiological parameters, and at times certain organic compounds). The adherence of waste water treatment plants to the voluntary standardisation schemes is variable. The successful REVAQ system has, for instance, about 50-55% of the total Swedish population equivalents connected to a REVAQ certified waste water treatment plant (I'Ons et al., 2015; IEA Bioenergy Task 37, 2015; Ekane et al., 2021).

Potential advantages of the measure

Self-regulation by the relevant industry can in suitable cases deliver the policy objectives faster or in a more cost-effective manner compared to mandatory requirements. It also allows greater flexibility to adapt to technological change (e.g. technologies applied for the treatment of waste water) and market sensitivities.

Reason for exclusion as a policy measure: relevance, effectiveness and efficiency

The introduction of voluntary standards has likely lead to an increase in sewage sludge quality, and could thus partly address the related objectives (I'Ons et al., 2015; IEA Bioenergy Task 37, 2015; Ekane et al., 2021). Nevertheless, several issues were identified in relation to representativeness, effective implementation and monitoring. Voluntary standardisation schemes in EU Member State fail to achieve complete environmental protection due to the incomplete adherence rate of waste water treatment plants to the voluntary schemes. In addition, it remains unsure to what extent a sufficient level of environmental and health protection can be achieved as, for instance, current certification schemes do not limit organic priority substances identified in a recent risk screening assessment study (Huygens et al., 2022). Moreover, observed sampling and measurement frequencies for monitoring (e.g. every 2-3 year) for sewage sludge may not necessarily capture temporal variations in contaminant loads. Next, the elements raised above in section 7.2.2 on drawbacks resulting from a harmonised and well-developed risk assessment approach are also valid for this policy measure. The desired policy outcome may thus not be delivered in practice as the conventional additional measures on sewage sludge quality control and enforcement mechanisms associated with regulation are not available. Finally, whenever externalities lead to a misalignment between social objectives and individual incentives of operators/investors, it is unlikely that voluntary regulation can resolve this and provide a socially efficient outcome (as opposed to cases where the need for regulation does not stem from an externality but from, e.g. a coordination problem).

Take-aways and cross-fertilisation for policy measures taken forward in the assessment

The monitoring of the quality and nature of the discharges to waste water treatment plants and the implementation of innovative waste water treatment techniques could possibly impact upon the sewage sludge quality. Waste water treatment plant operators can thus be part of the solution. To incentivise such efforts, the legislation could adopt a dynamic nature to respond to such upstream actions. This could be done by periodically reviewing priority contaminant lists (e.g. via sunset clauses), based on stakeholder inputs. Such actions could reduce unnecessary cost and administrative burdens for actors involved in sewage sludge management, and ensure the continued relevance of the SSD over time in a context of an evolving spectrum of chemicals.

7.3 Policy measures to address resource efficiency

7.3.1 Proposed measures for in-depth assessment

7.3.1.1 Targets for phosphorus recovery from sewage sludge in the SSD, combined with guidance on good agricultural practices

The measure would mandate sewage sludge management routes that lead to retaining phosphorus from sewage sludge in the biogeochemical cycle, aligned to circular economy principles. The focus for this measure is on phosphorus as it is an EU critical raw material, the potential of sewage sludge-derived materials to substitute primary material, and the current state of technology that is advanced for phosphorus. Sub-options of this measure have different levels of ambition to recover phosphorus, by making it conditional on the capacity/size of the waste water treatment plant from which the sewage sludge originates. Sewage sludge management routes that maintain phosphorus and/or other resources in the cycle involve well-managed land spreading of unprocessed, composted and digested sludge on agricultural/ forested land or any other intended function that provides nutrients to plants or improves the plants' nutrition efficiency, as well as technical phosphorus recovery from sewage sludge ashes. Each of these routes recover normally >75-80% of the phosphorus contained in sludge. Alternatively, targets for a minimum share of phosphorus recovery from larger waste water treatment plants could be set at the aggregate Member State level to provide more flexibility to the Member States (not further developed in this report as impacts are expected similar to the proposed option).

The benefits of recycling are only realised in case recovered nutrients effectively contribute to plant nutrition. This is particularly important for materials that are not traded as products and placed on the market. Therefore, landspreaded sewage sludge should be applied in keeping with plant phosphorus demands, which typically are between 50–150 kg P_2O_5 ha^{-1} (Roy et al., 2006; Tóth et al., 2014), leading to maximum sewage sludge application rates of <1–3 t dry matter per hectare (DM ha^{-1}). The nitrogen to phosphorus ratio of sewage sludge (about 2-3) is lower than the nitrogen to phosphorus ratio of crops, implying that this measure will ensure that sewage sludge is not applied in excess to plant nitrogen demands under good fertilisation practices. This policy measure will request Member States to apply good agricultural practices to further limit nutrient losses in the form of nitrates, phosphates, and ammonia., with a specific reference to maximum application rates aligned to plant P demands, necessity to align the timing of sewage sludge applications to periods of plant nutrient demands, and good storage conditions for sewage sludge that cannot be applied on land during (winter) periods.

De facto, this measure will indirectly limit the occurrence of incineration without phosphorus recovery and landfilling as sewage sludge management routes, whilst promoting good nutrient management practices. Exemptions for compliance with this measure could be admitted in case upstream phosphorus recovery technologies of comparable performance are demonstrated (non-existing with the current state of technology, although technological progress on this aspect is being made).

Concretely, this policy measure could include the following main elements:

— Member States shall ensure that at least 75% of the phosphorus present in sewage sludge from waste water treatment plants is retained in the biogeochemical cycle, with the following sub-measures:

  - Sub-measure A: for waste water treatment plants exceeding a capacity of 500k p.e.;
  - Sub-measure B: for waste water treatment plants exceeding a capacity of 100k p.e.;
  - Sub-measure C: for waste water treatment plants exceeding a capacity of 50k p.e.;
  - Sub-measure D: for waste water treatment plants exceeding a capacity of 20k p.e.

— Member states shall ensure that recycled sewage sludge-derived materials shall be applied in line with plant phosphorus demands, taken into consideration also other nutrient inputs to agricultural land. Member States shall develop codes of good agricultural practice for sewage sludge applications on agricultural (and forested land) that could cover additional requirements such as closed application periods, minimum storage capacities, application methods and others (similar to requirements for manure laid down in Annex II of the Nitrates Directive 91/676/EEC).

— Member States shall report on the quantities of sewage sludge produced, the type of sewage sludge treatment carried out, the quantities supplied for use in agriculture, and the places where sewage sludge is used, in line with the obligations laid down in Regulation (EU) 2019/1010 amending the SSD. The reporting shall include a separate assessment for sewage sludge originating from waste water treatment plants exceeding the size thresholds as outlined above. The reporting frequency is every three years.

7.3.2 Other possible policies that are not further considered in the analysis

7.3.2.1 Mandatory guidelines for sewage sludge management as a function of sewage sludge properties and local environmental settings

Description of the measure

The waste hierarchy outlined in Directive 2008/98/EC promotes recycling over other recovery and disposal. Departing from the hierarchy is, however, allowed for specific waste streams when justified for reasons of, inter alia, technical feasibility, economic viability and environmental protection (Article 4(2)). This policy would narrow and restrict the justifications to depart from the hierarchy, most importantly excluding economic motives as a reason. When compliant with sewage sludge quality standards and local conditions (e.g. water and soil quality in good status and close to waste water treatment plants), sewage sludge should be applied on agricultural land with a view to recycle nutrients and organic matter contained in sewage sludge. If not possible, a cascading set of less preferable actions would be mandated, aligned with the waste management hierarchy and giving preference to the recycling (e.g. sewage sludge incineration followed by phosphorus recovery from the ashes) and recovery of sewage sludge resources.

Potential advantages of the measure

It would maximise the recycling of resources (e.g. organic matter and nitrogen) by defining conditions where land spreading could lead to an increased overall environmental performance over other sewage sludge management options, including phosphorus recovery from sewage sludge ashes.

Reason for exclusion as a policy: technical feasibility, subsidiarity, coherence with other EU policy objectives, efficiency and effectiveness

It may be technically challenging to write out clear “one-size-fits-it-all” actions and conditions that ensure the best overall environmental and social performance under all different EU settings and contexts. Likely, the issue could be dealt more effectively by Member States themselves at central, regional or local level, thus not be aligned to the subsidiarity principle. Furthermore, the waste management hierarchy, with possibilities to depart from it, are described in the Waste Framework Directive, and introducing different requirements for the hierarchy for specific waste streams may thus not be fully coherent with this Directive.

7.3.2.2 Additional targets for the recovery of nitrogen and organic matter from sewage sludge in the SSD

Description of the measure

Similar to the targets proposed for phosphorus recovery (section 7.3.1.1), targets could be introduced to recover shares of other valuable resources, mostly organic matter and nitrogen from sewage sludge. Such targets could start with a low level of ambition that is increased over time to ensure progress towards a more circular economy.

Potential advantages of the measure

There could potentially be an increased return of organic matter to soils, leading to an improvement in soil quality. The benefits of recycling nitrogen could help to a limited extent to reduce the need for mineral nitrogen fertiliser applications. However, this issue is mainly energy-related as mineral nitrogen fertilisers are produced from natural gas using an energy-intensive process.

Reason for exclusion as a policy measure: Effectiveness and efficiency, technical feasibility

The overall benefits from the implementation of such measure would be relatively small given that sewage sludge is only a minor source of organic matter and nitrogen compared to other organic materials, such as manure and bio-waste. In addition, technological and technical constraints may not allow for the implementation and enforcement of theoretical options. Current recycling technologies mostly involve the land spreading of untreated and biologically treated sewage sludge. Therefore, this measures has synergies with the options described above (section 7.3.1.1 and/or the measure discarded in the early phase outlined in section 7.3.2). Other means of recovering nitrogen and organic matter through innovative process are not available at a sufficient level of technological readiness or may occur upstream (e.g. through stripping-scrubbing of ammonia following anaerobic digestion, mostly occurring at the waste water treatment plants covered under the Urban Waste Water Treatment Directive).

Take-aways and cross-fertilisation for policy measures taken forward in the assessment

It may be highly relevant to maintain measures that are technologically neutral to enable the possible implementation of technologies that recover additional resources from sewage sludge. This is also the reason why targets are preferred over options that involve e.g. a ban on certain management options such as (co-)incinerations without P-recovery.

7.4 Scope for economic instruments

In contrast to the just discussed measures of limit values, bans or recovery targets, which represent so-called command and control instruments, economic or ‘market-based’ instruments often represent a more indirect and flexible approach to regulation. For this reason they are discussed separately.

There are prominent examples of the use of economic instruments in the field of environmental policy, e.g. the EU ETS, feed-in-tariffs for electricity from renewable sources, or deposit refund schemes to incentivise the recycling of beverage packaging. Other economic instruments include levies, subsidies, tax-breaks, and tradable standards or quotas, among others.

The use of economic instruments and their greater flexibility is advantageous when the regulating entity does not have full information on all relevant parameters of the problem or when the objectives can vary over time (a ‘moving target’). For example, the regulator may not know exactly how much it costs different firms and industries to reduce CO_2 emissions by a given amount. At the same time it is desirable that the cost burden of CO_2 emissions reductions is distributed in a cost-effective manner across firms. With the economic instrument of emissions trading, the regulator can set the total amount of emissions, distribute the corresponding amount of emission permits to firms (e.g. by auctioning), and let firms trade with each other, allowing those with higher reduction costs to reduce less and those with lower reduction costs to reduce more (and be compensated by payments for doing so). Similarly, the regulator might be able to quantify a positive externality – e.g. the external benefits from producing energy with renewable resources – but does not know firms' production costs. With a subsidy that corresponds to the external benefit, an efficient outcome can be achieved even without more detailed knowledge.

The type of situation in which economic instruments can be very efficient – i.e. when pollution levels or external benefits are driven by firms' costs and the regulator does not have this information – mostly occur in market settings or settings with heterogeneous actors that respond to financial incentives. With regard to sewage sludge management, assessing the potential of economic instruments must be done separately for each of the identified problems, in view of their idiosyncrasy:

— Health and environmental impacts from land spreading of sewage sludge with contaminants

The negative impacts of the different contaminants are not all well understood and quantified, but are deemed to be potentially high. This means that the costs and consequences of having an excessive contaminant exposure are possibly severe. In this case flexibility is not warranted, since what needs to be achieved is certainty that established safety limits are not surpassed. If, in addition, the monitoring of contaminant concentrations is not possible or very costly, a ban or restrictions on land spreading practices might be considered. This is a typical case of an issue where serious health concerns preclude the use of economic instruments – e.g. car manufacturers would not be paid a subsidy for equipping theirs cars with safety devices, they simply have to comply with certain standards.

— Loss of nutrients due to excessive land spreading beyond biological absorption capacity of plants and soils

Essentially, this is a problem of a local oversupply of nutrients. Sewage sludge competes with other nutrient-rich applications like manure for land on which it can be spread. In some EU regions, owners of the land may receive a payment for accepting and spreading the sewage sludge, and the fact that nutrients may be lost due to excessive applications does not matter to the owners as long as their land and cultivation is not damaged and the payments provide a sufficient economic incentive. From the social point of view, the concern is about the negative externality arising from the run-off of excess nutrients into the environment.[11] Land owners receive payments for accepting sewage sludge, but the costs of sewage sludge management are borne partially by society. According to standard economics, the land owners should pay for the environmental pollution they incur on common resources, e.g. with a charge. But this charge would in theory need to depend on how much excess nutrients effectively leak into the environment, perhaps even conditioned on the specific sensitivity of local ecosystems. For practical reasons this is not a workable solution, which means that eventually command and control measures, e.g. a limit on the frequency of land spreading or partial bans, are more reasonable approaches.

— Loss of nutrients due to sewage sludge disposal without resource recovery

Economic instruments can provide the needed incentives when resources are not recovered, even though this would be beneficial from a social point of view. However, proving the latter and determining the *size* of the incentive, e.g. the value of the optimal subsidy, requires careful analysis and detailed information. Currently, phosphorus and other valuable resources are not recovered for two main reasons[12]: First, the treatment of sewage sludge by mono-incineration with subsequent sale of recovered resources and energy at market prices is by itself not competitive, i.e. it is not economically self-sustained. Second, even when viewing sewage sludge management as a public service that aims at cost minimisation (not profitability), there are other management options which generally tend to be cheaper, especially land spreading and – where allowed - landfilling. An economic instrument in support of resource recovery could be warranted if recovery generates additional benefits for society that are not reflected in the market prices of the outputs of mono-incineration. For example, it could be argued that phosphorus recovered within the EU delays the global depletion of this scarce resource and makes a contribution to the EU's strategic autonomy by reducing the need to import phosphorus. Under this perspective, a subsidy for phosphorus recovery corresponding to the monetized value of this contribution would be justified, possibly financed by a levy on imported phosphorus sourced from primary raw materials, which would reflect the incurred external costs of creating and sustaining the EU import dependency on this

[11] The fact that phosphorus and other resources are lost is not *per se* a sign of inefficiency, as there might be a lack of suitable and available land nearby, while transporting sludge to further away locations can change the equation to an extent that it becomes socially preferable to lose nutrients on saturated lands.

[12] Additional sources of market inefficiencies may be present, including information failures related to the quality sewage sludge-derived fertilisers, consumption externalities and risk aversions (perceived costs associated with the quality of final goods derived from secondary materials relative to those derived from virgin materials), technological externalities (complexity of recycling due to technical characteristics of sewage sludge (ash) as a feedstock), and market power in primary and secondary markets.

critical raw material. However, given that such an economic instrument would fall outside of the policy scope analysed here, it is not further considered.

— Methane emissions from landfilling of sewage sludge

Methane emissions caused by decomposing sewage sludge in landfill or backfill sites contribute to global climate change and use up the limited greenhouse gas emissions budget to which the EU has committed itself. Being a diffuse source of emissions, landfill and backfill sites are not integrated in the EU emissions trading scheme (EU ETS), and hence there is no CO2eq price for methane emissions stemming from sewage sludge. The standard 'first-best' policy solution for this externality would be an integration in the EU ETS, but such a proposal might face technical obstacles (and is out of scope of the present analysis). In the absence of a first-best regulation, it could be analysed whether a second-best solution might be to mandate a (minimum) climate charge on sewage sludge that is landfilled or disposed in any other way that is expected to generate methane emissions. Such an analysis would need to take into account the secondary impact of such a policy on other sewage sludge management pathways (e.g. sewage sludge is re-directed from landfilling to where?), but also the impact on landfill sites (I.e. what would be landfilled instead of sewage sludge?). If the quantification of the expected methane emissions is not possible, or the application of the charge infeasible for legal reasons, then bans or restrictions of landfilling/backfilling might be considered.

Coming back to the overall potential of economic instruments, sewage sludge management is part of an essential public service (water provision), with limited flexibility in meeting demand. As such, it is best described as a problem of risk management and cost minimisation, rather than the maximisation of a market surplus. Therefore, health and environmental concerns spurred by sewage sludge management should be addressed by appropriate rules and limit values rather than by flexible market-based instruments.

On the other side, in principle there would be more scope for economic instruments to address concerns about resource efficiency/recovery and methane emissions. However, in the policy context of this study, such measures, such as climate charges on methane emissions or subsidies on phosphorus recovery to address the EU's external trade dependency, are likely out of scope of this Directive.

7.5 Policy options

A policy option is a combination (or a package) of policy measures that address all policy objectives, thus merging the measures proposed in sections 7.2.1 and 7.3.1.17.3.1. Synergies between policy measures can be sought to address more than one objective simultaneously. The following policy options are proposed that combine the measures (section 7.2 and 7.3). Finally, two different policy options are defined and proposed for in-depth analysis

— Policy option 1: Monitoring and control of sewage sludge quality, plus a minimum target to recover at least 75% of the phosphorus from sewage sludge generated at waste water treatment plants exceeding a certain size. Recycling routes that are considered include use of sewage sludge on agricultural/forested land or elsewhere in the environment where it contributes to (improving) plant nutrition (all assumed to have a P-recovery rate of 100% of the sewage sludge mass that used for these applications)[13],[14], as well as phosphorus recovery from incineration ashes (with an assumed P-recovery aligned to the P that is recovered from the sewage sludge used as feedstock)[15]. The quality of landspreaded sewage sludge for recycling would have to be monitored on contaminants by MS, unless all sewage sludge generated at waste water treatment plants exceeding a certain size is incinerated and P is recovered from the ashes. Sewage sludge applied in agriculture would be required to meet minimum quality requirements and have contaminants below limit values established in the Directive. Good agricultural practices should be applied to further limit nutrient losses in the form of nitrates, phosphates, and ammonia from sewage sludge landspreading. Member States have to develop and enforce measures to apply good agricultural practices, with a specific reference to maximum application rates aligned to plant nutrient and P demands, the timing of sewage sludge applications to periods of plant nutrient demands, and good storage conditions for sewage sludge that cannot be applied on land during (winter) periods. Member States shall develop guidelines to ensure environmental and health protection from sewage sludge used for other recycling

13 Sludge applied on agricultural/forested land or elsewhere in the environment where it contributes to (improving) plant nutrition (all assumed to have a P-recovery rate of 100% of the sludge mass that used for these applications) will hereafter be defined as 'recycled sludge', and the management pathway as 'sludge land application for recycling'. It excludes backfilling operations that are defined as a recovery process because this operation does not valorise the resources present in sludge for the purpose of crop or plant nutrition.

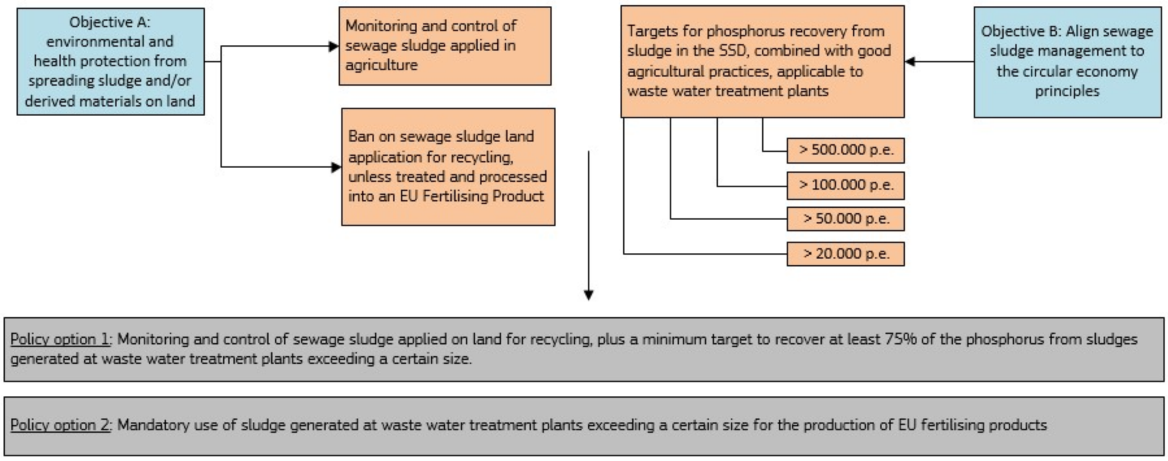
14 Hereafter also referred to as non-technical P-recovery.

15 Hereafter also defined as technical P-recovery

applications other than use on agricultural land, taking into account the results of the sewage sludge quality monitoring campaigns.

— Policy option 2: Mandatory use of sewage sludge generated at waste water treatment plants exceeding a certain size for the production EU fertilising products. This policy option implies a transformation of sewage sludge into EU Fertilising Products classified as phosphorus inorganic macronutrient fertiliser (Product Function Category 1(C)(I))[16] that contain at least 75% of the P contained in the sewage sludge used as feedstock for the recovery process. Sewage sludge originating from waste water treatment plants below the size threshold are then subject to use restrictions. In a first sub-option, Member States shall develop guidelines to ensure environmental and health protection from sewage sludge used for recycling applications, including agricultural use, originating from waste water treatment plants below a certain size threshold[17]. In a second sub-option, minimum sewage sludge quality standards for sewage sludge used in agriculture shall be set in the Directive, and Member States set requirements for other recycling options outside agriculture.

Figure 12. Schematic overview of the policy measures and options for in-depth assessment



[16] Regulation (EU) 2019/1009 of the European Parliament and of the Council laying down rules on the making available on the market of EU fertilising products and amending Regulations (EC) No 1069/2009 and (EC) No 1107/2009 and repealing Regulation (EC) No 2003/2003.

[17] As outlined in section 7.2.1.2, sewage sludge originating from waste water treatment plants exceeding the size thresholds shall not be used for other recycling operations outside agriculture because, in this strictest policy option, it is considered that the inherent nature of sewage sludge, containing numerous unknown contaminants with undetermined toxicological properties and other pollutants including microplastics cannot be applied in a safe manner in the environment.

8 Impacts and comparison of policy options

8.1 Use of quantitative and qualitative assessments

Significant impacts should be assessed qualitatively and, whenever possible, also quantitatively. The analysis will be quantified as far as possible (though in a proportionate manner), but when data are lacking, the assessment will be complemented with or replaced by a qualitative analysis. On particularly important impacts, such as the potential to contribute to phosphorus circularity and economic costs, a quantitative assessment has been prioritised.

Severe limitations to a full quantitative assessment have been observed for other impact categories, due to the limited availability of quantitative data on

- sewage sludge quality;
- sewage sludge quality standards for sewage sludge applications on agricultural and other lands that ensure environmental and health protection;
- sewage sludge management routes in the EU Member States.

Therefore, this feasibility study combines quantitative and qualitative techniques to provide a sensible assessment that can be completed in later stages of the policy development process.

8.2 Selection of main impacts to compare policy options

A policy option should deal with the identified problem(s) by inducing direct and indirect changes to the behaviour of those influencing it (i.e. addressing the problem drivers). These changes are also likely to have a bearing on the achievement of other policy goals. The first step in impact analysis is the identification of this chain of impacts.

All key parameters of an option that will directly contribute to the achievement of the policy objectives should be retained for further analysis to assess the effectiveness and efficiency of this option. Concretely, this implies the following environmental, economic, and social impacts:

Efficient use of resources (renewable & non-renewable), open strategic autonomy, and security of supply (cross-cutting impacts, on farmers, and the society as a whole): This point relates to the extent of recycling of the valuable resources contained in sewage sludge. The focus will principally be on the potential of sewage sludge management routes to contribute to plant phosphorus nutrition, but also on organic matter and other nutrients (cross-cutting with the impacts on the quality of natural (soil) resources).

Public health and food safety (social impact on food consumers): The spreading of sewage sludge may prompt (second-order) changes when contaminants in soil and water are transferred to foodstuffs (including vegetables, meat and dairy products). Therefore, impacts on public health and the possible contamination of food will be assessed.

Quality of natural resources, including soil, water and air (environmental impacts on farmers and the society as a whole): The application of sewage sludge on the environment may impact upon the quality of agricultural soils and other land areas that are treated with sewage sludge as a fertiliser or soil amendment. This is mainly the result of nutrient losses as the contribution of sewage sludge to increase soil organic matter is reduced (see section 4.1.2).

Based on the principle of proportionate analysis, the analysis should also focus on changes that may provide an important link in the chain of actions that affect the problem, as well as other cumulative impacts that new obligations may have on the main actors subject to regulatory compliance and implementation and enforcement of the obligations.

Innovation and research will be considered as further technological developments on sewage sludge treatment can be considered as a potential driver that may further contribute to the achievement of the policy objectives in the mid-to long-term.

Economic impacts on Member States and waste water treatment plant operators that may be subject to supplementary administrative burdens, changes in their conduct of business, and *costs and fees* charged for public services to citizens (waste water treatment, including sewage sludge management) are to be considered:

Economic costs on waste water treatment plants and Member States: these are on one hand the so called *substantive compliance costs*, i.e. the total costs for sewage sludge management as implied by the policy options, including Capital Expenditures (CapEx) and Operational Expenditures (OpEx), variable and fixed, which include the following elements:

- CapEx: site acquisition, planning, construction & civil work, process equipment
- OpEx: annualised investment costs and operational costs (maintenance, insurance, personal cost, material, energy, disposal).

In addition, *administrative costs* are those related to complying with information obligations stemming from the considered policies, e.g. costs for sampling and reporting. Both types of economic costs will eventually be passed on to citizens as part of the general charges for water sanitation services. Note that all economic costs reported in this study are expressed in constant Euros (€) of today.

Furthermore, it is important to consider possible social- and distributional *impacts for specific regions and Member States* that could be most affected by the introduction of certain policy options: is the *distribution of costs and benefits* commensurate? Are there relevant *employment impacts,* e.g. a relocation of jobs from the EU to external countries, or employment gains in recycling industries?

Finally, methane emissions will be briefly discussed, even though it was indicated in the baseline that this problem will diminish in the future (see section 6.5).

8.3 Policy option 1 (PO1)

Policy option 1 consists of:

- The monitoring and reporting of sewage sludge quality in terms of contaminants by Member States;
- The obligation to recover at least 75% of the phosphorus contained in sewage sludge generated at waste water treatment plants exceeding a certain size. Recovery can take place through recycling of untreated or treated sewage sludge on land (agriculture, forestry, or other uses where sewage sludge contributes to plant nutrition or improving the efficiency of plant nutrition) or through phosphorus recovery following sewage sludge incineration. *Sub-options* of this policy requirement focus on WWTP above 500k p.e.; above 100k p.e.; above 50k p.e. and above 20k p.e.;
- The requirement for sewage sludge landspread on agricultural land to meet quality requirements and limit values for contaminants set in the Directive. If sewage sludge does not meet the quality requirement, alternative possibilities should be used (e.g. mono-incineration followed by phosphorus recovery). For this analysis, the share of sewage sludge that would meet quality requirements for agricultural use remains unknown, and therefore an uncertainty analysis with three different *scenario*s of sewage sludge compliant with requirements for agricultural use have been evaluated (30%, 60% and 90%, respectively);
- Good practices on agricultural use of sewage sludge laid down in the future Directive should be met;
- Where sewage sludge is landspread elsewhere in the environment (e.g. for forestry, landscaping), it should meet any quality requirements set by Member States, based on the outcome of the sewage sludge quality monitoring.

8.3.1 Sewage sludge mass, nutrient and organic matter content, and management routes

Policy option 1 does not have any effect on the sewage sludge mass produced or the nutrients or organic matter within. With the implementation of mandatory non-technical and technical P-recovery, we expect substantial changes compared to 2019 and 2050 baseline (Figure 13; Figure 24; Figure 11). Mandatory P-recovery for WWTP with a treatment capacity of >50k p.e. seems to be a sound compromise, as it addresses a considerable amount of EU-27 wastewater (~70%) while at the same time keeping the number of affected treatment plants reasonable (12% of WWTP have treatment capacities greater than 50k p.e.; see Annexes, section 13.1).

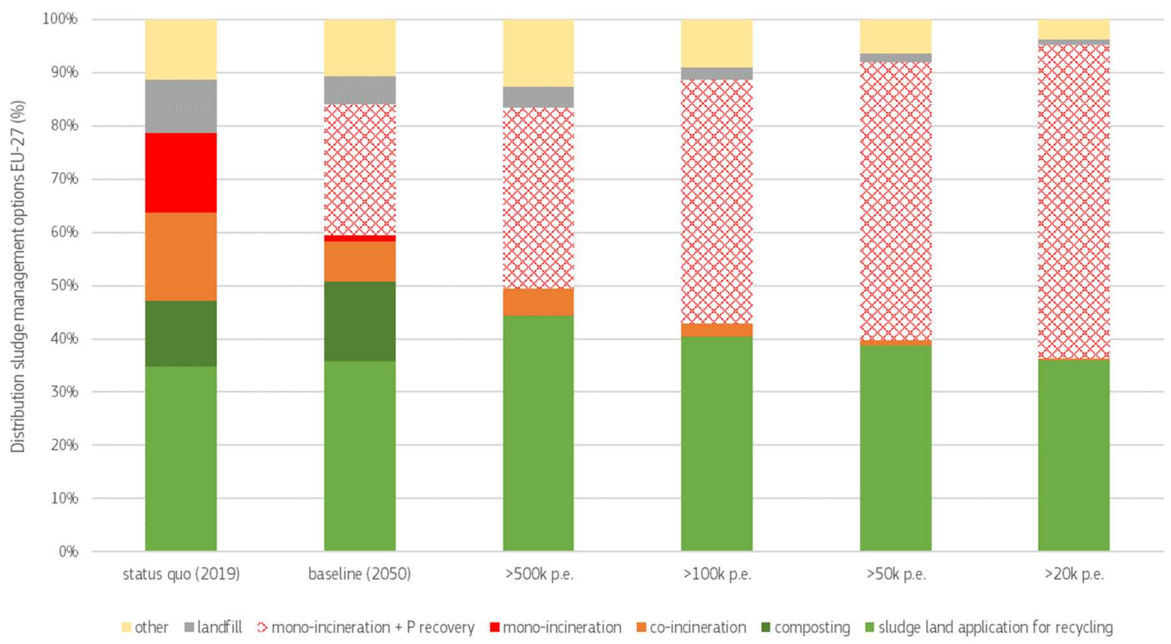
Considering this, compared to the baseline, direct sewage sludge land application for recycling[18] is expected to reduce to ~39% and co-incineration and landfill together are expected to reduce to 3%. The expected reduction

[18] The category 'sewage sludge land application for recycling' covers the landspreading of untreated or treated (composted, digested, lime stabilised, etc.) sewage sludge on agricultural land, forested land or other lands where sewage sludge contributes to plant nutrition. It is expected that application on agricultural land will continue to be dominant route across the EU-27, and that landspreading on other land

in the amounts of sewage sludge that will be landfilled is in line with a further implementation of the landfill directive and the methane strategy. On the other hand, sewage sludge incineration followed by P-recovery is expected to increase from 25% (baseline) to 34-59% upon implementation of this policy option.

Figure 13 also reveals, that still around 7% of sewage sludge (treated or untreated) will be used for other applications that do not contribute to plant nutrition (comprised in the fraction 'other').

Figure 13. Projected sewage sludge management option for status quo (2019), baseline (2050) and the PO1 sub-options with mandatory requirement to recycle at least 75% of the P contained in sludge originating from waste water treatment plants exceeding a certain size (> 500k p.e., >100k p.e., >50k p.e. and >20k p.e., respectively for policy sub-options of a different level of ambition). For these policy sub-options, average values are depicted across different scenarios that assume a varying degree of compliance of sewage sludge with future quality requirements for sludge recycling. 'Sludge land application for recycling' mostly involves agricultural use of untreated or treated (e.g. composted, digested, lime stabilised) sludge, but may also include other recycling operations where sludge contributes to plant nutrition (e.g. forestry applications). 'Other' includes sludge uses that are unidentified or cannot be classified as a recycling operation (e.g. backfilling and storage).



8.3.2 Environmental and health impacts

Environmental and health protection is an objective of the policy option, and compliance with this requirement will be enforced through a combination of measures (e.g. quality requirements for sludge used on land). The policy option enforces a much higher of environmental and health protection than the baseline due to increased monitoring of contaminants and ensuing risk mitigation levels in the form of limit values for contaminants of concern present in sewage sludge for landspreading in agriculture (including e.g. also organic contaminants) and thus reduce their input on land. However, it is noted that overall environmental and health risks decrease proportional to the amount of sludge returned to the environment, particularly on agricultural land, as contaminants of a more reduced concern may still be present in the sludge, e.g. when recently emerging (Figure 14). Thermal treatment of sewage sludge is the only way to reliably destroy organic pollutants. Metals remain fully in the sewage sludge ash with the exception of those elements that have evaporation temperature lower than the incineration temperature of 850 ºC (e.g. mercury or cadmium). If the ashes are then used directly or without targeted depollution steps as fertiliser, the metal load contained in the sewage sludge ash is incorporated into the soil, as is already the case with sewage sludge. This is the case if the sewage sludge ash

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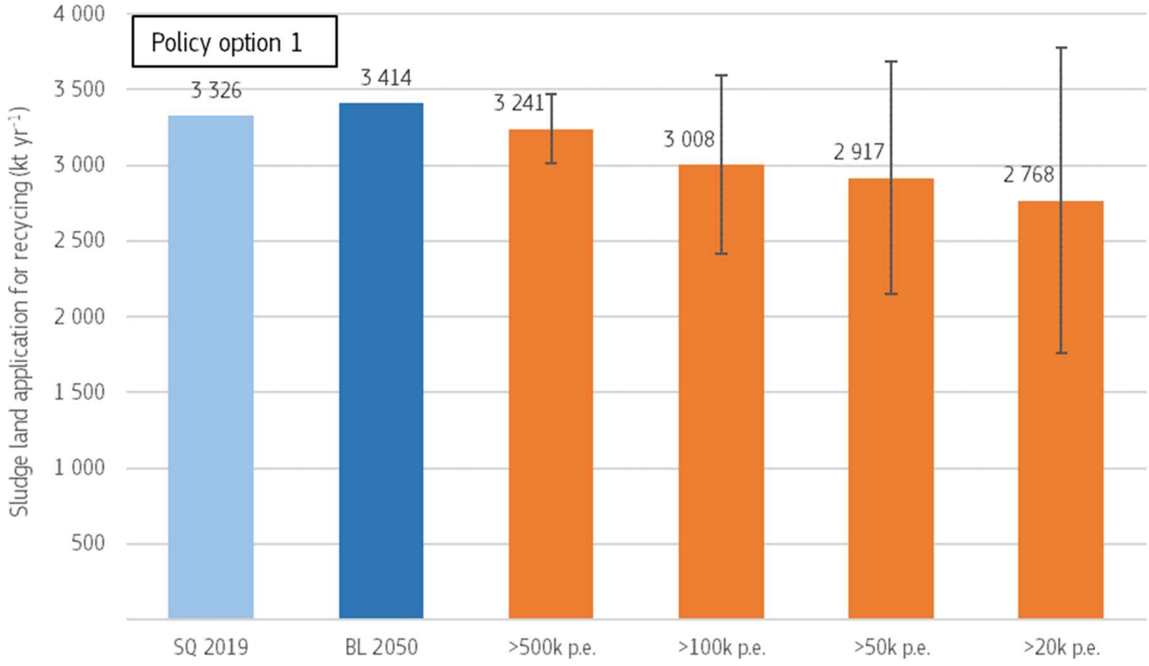
where sewage sludge contributes to plant nutrition is marginal in most EU Member States. The class 'other' involves uses of sewage sludge that are unknown or waste management operations other than recycling and disposal (e.g. backfilling, storage, and use as landfill cover). It is recognised, that uncertainties and minor inconsistencies may exist amongst the classification of such operations for the comparison of status quo, baseline and policy options. At the same time, it seems unlikely that this is occurring to the extent that is will have a significant impact on the results and conclusions presented in this report.

(SSA) is used for example in the fertiliser industry to directly substitute raw phosphate rock and no depollution step is installed. However, it is possible to subject the sewage sludge ash to special treatment processes and, in addition to recovering agronomical efficient P fertilisers, metals are removed specifically and to a great extent (>95%). This technologies are capable of lowering the transfer of metals on agricultural areas significantly and landfilled as waste accordingly. Further advantage of these technologies are that some metals as e.g. iron or aluminium can be recycled as well and re-used in the WWTP as precipitants to remove again P from the wastewater.

Figure 14. Amount of sewage sludge spread on land for recycling operations for status quo, baseline and the PO1 sub-options considering minimum and maximum values (kt yr^{-1}). The return of contaminants, including organic contaminants, microplastics and metals (depending on treatment applied following sludge incineration, see text) is proportional to the amount of sludge recycled on agricultural land. Hence, human health and environmental risks are proportional to the height of the bars for the different sub-options depicted. The error bars represent the minimum and maximum values for the scenarios that assume a varying degree of compliance of sludge with future quality requirements for sludge recycling.



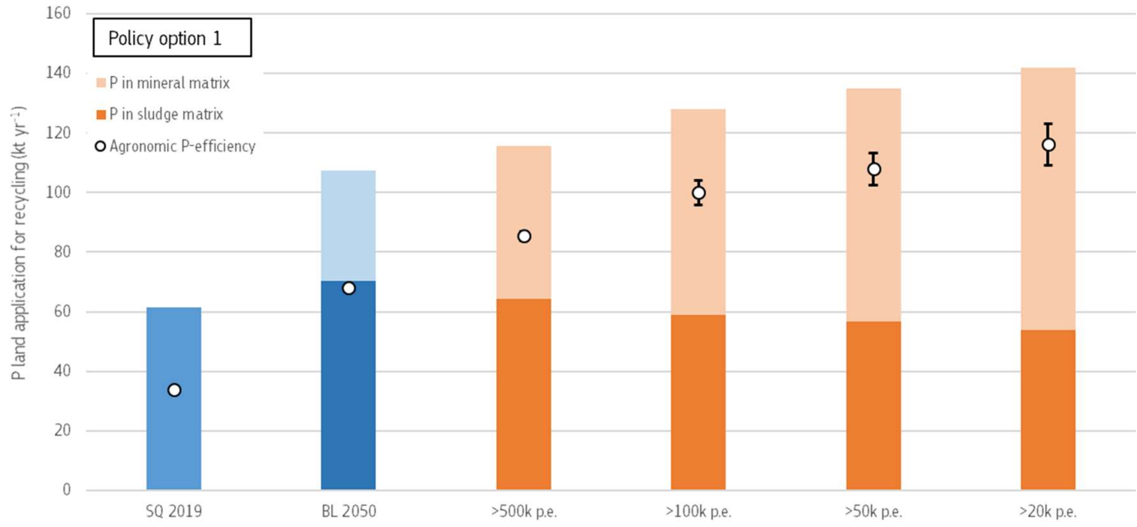
8.3.3 Resource recovery

Depending on the sub-options, the amount of total P applied to land for recycling can be increased by 7%, 19%, 26%, or 32%, compared to the baseline. One of the main advantages of technical P recovery compared to direct sewage sludge application is the production of secondary P fertilisers with agronomic efficiencies comparable to mineral fertilisers. Figure **15** illustrates the share of P brought to land within the sewage sludge matrix and P brought to land within a mineral fertiliser matrix. Considering the improved agricultural efficiency of mineral fertilisers, sub-options with a high proportion of mono-incineration and technical P-recovery perform best.

With regard to the annual EU mineral P fertiliser use of 0.77 Mt P yr^{-1} (see section 4.1.2), the P in sewage sludge applied in the baseline is equal to about 13.9% of the net imports. For PO1, a maximum additional amount of 0.04 Mt P yr^{-1} can be estimated relative to the baseline (>50-500k p.e.; **Figure 15**). This corresponds to a total potential to substitute about 17.5% of the P imported in mineral fertilisers for the year 2050 considered in this assessment.

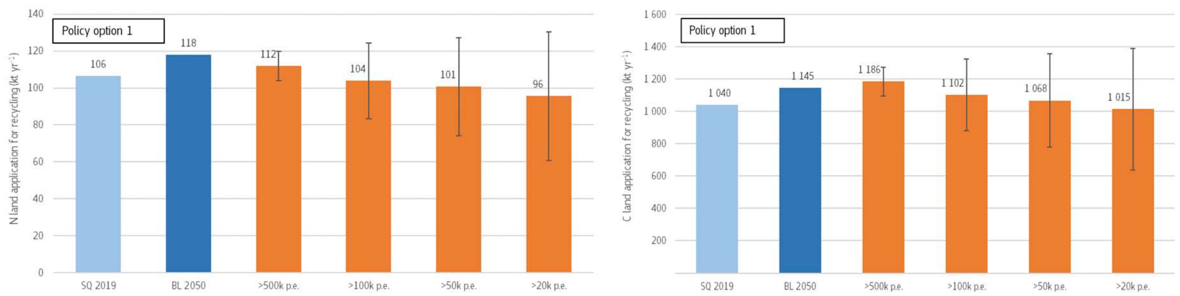
Figure 15. Phosphorus recycling (kt P yr^{-1}) for status quo (2019), baseline (2050) and the PO1 sub-options with mandatory requirement to recycle at least 75% of the P contained in sludge originating from waste water treatment plants exceeding a certain size (>500k p.e., >100k p.e., >50k p.e. and >20k p.e., respectively for policy sub-options of a different level of ambition). For these policy sub-options, average values are depicted across different scenarios that assume a varying degree of compliance of sewage sludge with future quality requirements for sludge recycling. The dots

'agronomic P-efficiency' refer to short-term bio-available P by taking into consideration the agronomic P-efficiency of sewage sludge and sewage sludge derived P-fertilisers returned to land.



Compared to P, carbon and nitrogen are volatile elements that transfer into the gaseous phase during incineration processes. As current flue gas cleaning technology aims to remove nitrogen compounds without recovery, nitrogen but also carbon from sewage sludge must be considered lost for recycling. Only through the direct agricultural use of treated or untreated sewage sludge, additional amounts of C, N and P can be returned to the cycle. In the case of incineration with technical P recovery, a trade-off between increased recycling of P with increased agronomic efficiency and the loss of N and C must be accepted (**Figure 16**).

Figure 16. Nitrogen (left) and carbon (right) land application for recycling for status quo, baseline and PO2 sub-options including uncertainties (kt yr^{-1}).



8.3.4 Nutrient losses

A reduction in nutrient losses from sewage sludge relative to the baseline is expected for this policy option because of the (i) the implementation and enforcement of additional measures that enforce a higher level of environmental sound agricultural practices for sewage sludge applications, and (ii) the increased shares of sludge that will be transformed into mineral P fertilisers with a higher nutrient efficiency than sewage sludge.

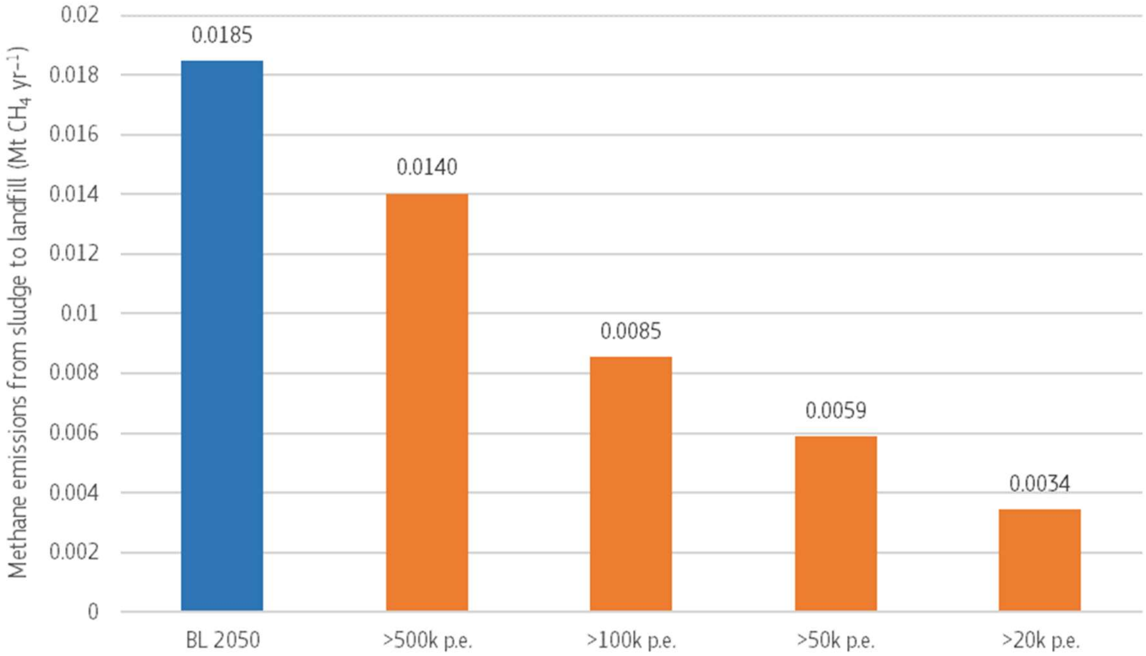
Sewage sludge does not have the optimal nutrient ratio, and nutrients are present in an organo-mineral matrix with a release efficiency that is inferior to that of mineral fertilisers. The policy option requests Member States to take additional measures to target some of the problems and their drivers observed. Specifically, Member States would be requested to develop and enforce measures to apply good agricultural practices, with a specific reference to maximum application rates aligned to plant nutrient and P demands, the timing of sewage sludge applications to periods of plant nutrient demands, and good storage conditions for sludge that cannot be applied on land during (winter) periods.

The environmental risk for nutrient losses is higher for sewage sludge than for mineral fertilisers. Plant available nutrients may be released from organic sources at a time when there is little crop uptake, and consequently gives rise to increased opportunities for nutrient losses. Gaseous emissions (e.g. ammonia) are also higher for sewage sludge than for (nitrate based) mineral fertilisers. In some soils of low organic matter content, sewage sludge can help to mitigate nutrient losses by increasing the soil organic matter content of the soil, and increase as such the ion exchange and nutrient adsorption capacity of the soil. The conversion of P from a wet organic sewage sludge matrix into a dry mineral matrix also offers the advantage that the nutrients can be stored long time and transported over long distances to areas with nutrient deficit. This reduces the potential contribution of sewage sludge to surface and groundwater pollution. Another major advantage of technical P recovery is that the P is converted into a plant-available form and then has the same agronomic efficiency as conventional mineral fertilisers. In addition, other nutrients can be added in any ratio, thus providing the plants with nutrients according to their needs (in terms of time and quantity).

8.3.5 Methane emission

Compulsory P-recycling, either through the application of sewage sludge in agriculture or through mono-incineration and technical P-recycling, will reduce the amount of sewage sludge landfilled. This in turn produces less methane. Compared to the baseline, methane emissions from landfill can be reduced by around 70% in the case of mandatory P recovery from sewage sludge from WWTP >50k p.e. (Figure 17). As already outlined in Section 6.5, the contribution from landfilled sewage sludge to the total methane emission is expected to be already minor. For this best performing sub-option, the CH_4 emissions from landfilling of sewage sludge will be reduced to 0.003 Mt yr^{-1} corresponding to only 0.02% of the 2019 total EU-27 methane emissions (15.2 Mt CH_4 yr^{-1}).

Figure 17. CH_4 emissions related to sewage sludge disposed in landfill for baseline and the PO1 sub-options (Mt yr^{-1}).



8.3.6 Compliance costs

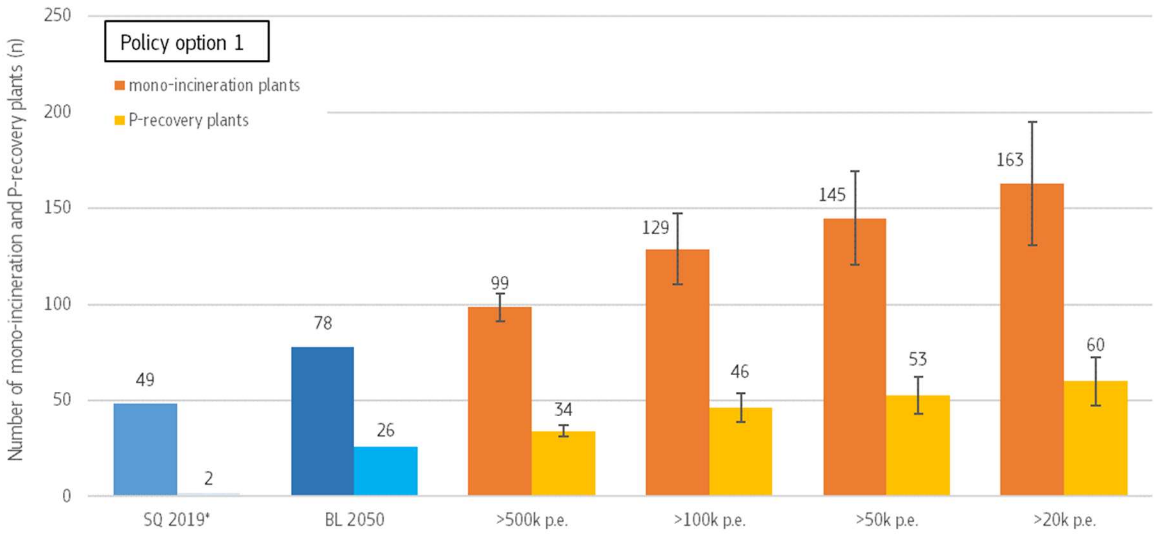
8.3.6.1 Investment cost

For the baseline, it was predicted that the numbers of mono-incinerators and P-recovery units already increase compared to the status quo due to the fact that AT and DE will have national mandatory P-recovery in place by then. Furthermore, it was assumed, that for countries with existing mono-incineration infrastructure, the generated SSA will undergo treatment for P recovery due to the availability of suitable technologies. With the implementation of the mandatory P-recovery at EU-27 level, additional mono-incineration and P-recovery plants need to be installed to enable a shift away from landfilling and sludge co-incineration without P recovery (Figure 18,

Table 10). For comparability reason, the number of mono-incineration plants were calculated by dividing the amount of sewage sludge treated in mono-incineration plant with an assumed average incineration capacity of 30kt DM yr^{-1}. For P-recovery plants, the average capacity is assumed with 30kt SSA yr^{-1}.

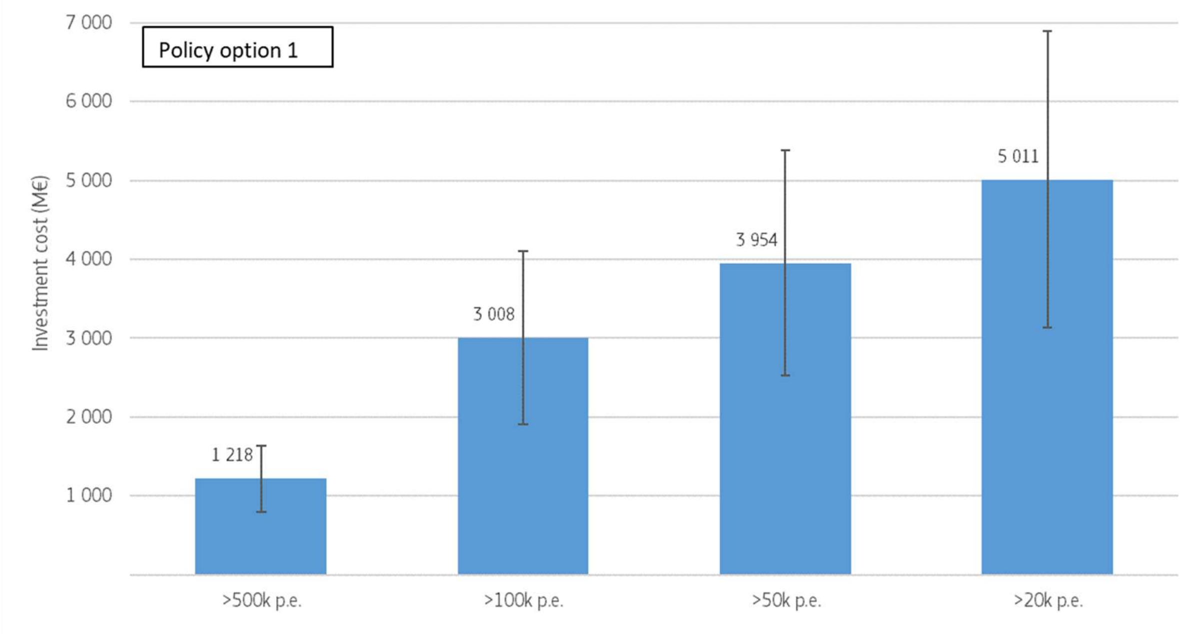
As an example, both incineration and P-recovery treatment plants have to double by number compared to the baseline in order to be able to mono-incinerate the additional sludge re-directed from landfilling and co-incineration under Policy Option 1 with 50k p.e. capacity threshold (Figure 18).

Figure 18. Number of mono-incineration and P-recovery installations for the PO1 sub-options considering minimum and maximum values.



It is assumed that each mono-incineration plant of the assumed size of 30 kt DM yr^{-1} has an investment costs of 51 M€ (or 1 700 € t DM^{-1} input), and each P-recovery plant of the assume size of 30 kt SSA yr^{-1} investment costs of 20 M€ (see Annexes, section 12.2.5 and 12.2.6). As consequence, the total required investment cost for the installation of the new mono-incineration and P-recovery infrastructure for the sub-options in which WWTP exceeding 50k p.e. are addressed, are in the range of 2 500–5 400 M€ until 2050. This investment cost analysis does not consider necessary reinvestment cost until 2050 for mono-incineration plants already in operation.

Figure 19. Total investment cost for the installation of new mono-incineration and P-recovery plants for PO1 sub-options considering minimum and maximum values (M€)

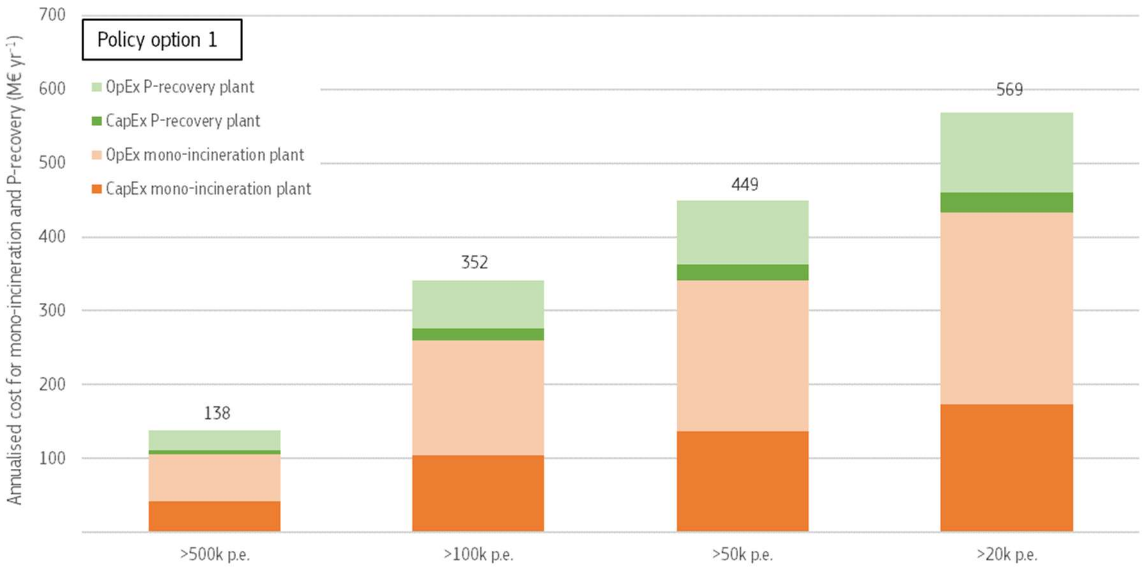


8.3.6.2 Annual costs for sewage sludge management

Annualisation of investment cost for mono-incineration and P-recovery plants

Figure 20 shows specifically the annualised cost of implementing a mono-incineration and P-recovery infrastructure based on the data from Figure 19, distinguishing between capital and operational expenditures. These costs are additional costs to the baseline. The costs are mainly determined by the required mono-incineration plants (75%). P-recovery plant accounts for 25% of the annual costs, and these facilities are dominated by operational costs due to the high energy and/or resource requirements. To annualize the investment cost, a lifetime of 25 years was considered for both types of plants. Share of capital cost on total cost: mono-incineration plant: 40%; P-recovery plant: 20% (see Annexes, section 12.2.5 and 12.2.6).

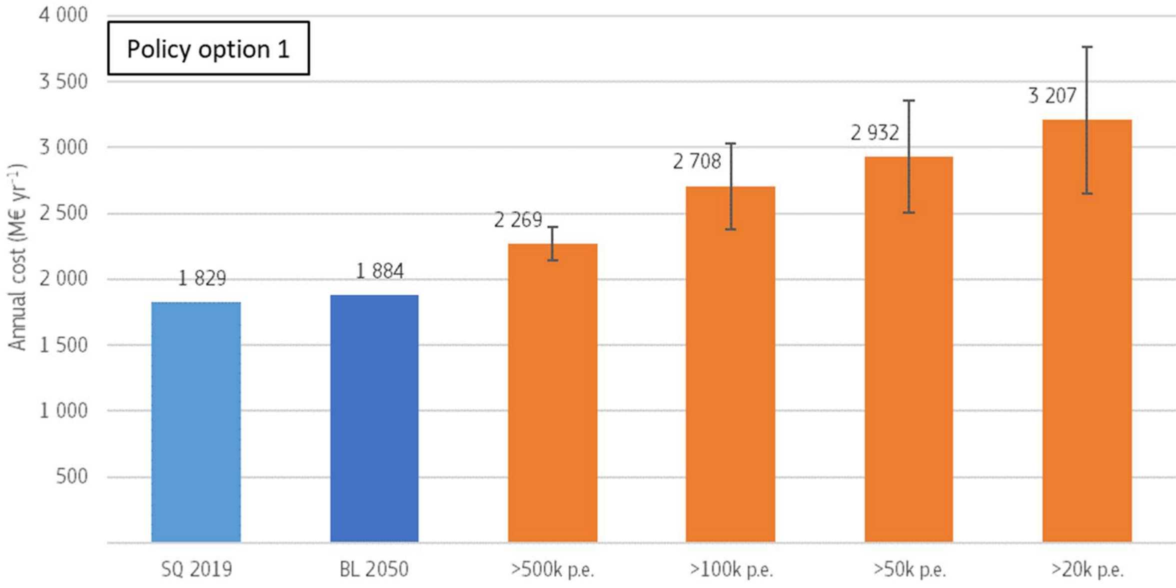
Figure 20. Annualised cost for mono-incineration and P-recovery (divided into CapEx and OpEx) for PO1 sub-options considering the average value (M€ yr^{-1}).



Annual cost of sludge management

The shift from one sludge management option to another, can result in savings (e.g. for some MS in case of a change from landfill to agricultural use) or additional cost (e.g. change from agricultural use to mono-incineration and P-recovery) (see Annexes, Table 44). Figure 21 shows the impact on annual costs due to the change in sludge management options. Implementing mandatory P-recovery for WWTP exceeding 50k p.e., the additional annual cost are in the range 620–1 480 M€ yr^{-1}. This corresponds to additional annual costs per inhabitant in the order of 1.4–3.3 €. Considering, that expenditures for wastewater treatment are on average 130 € per person and year (data for AT, DE, FR, NL, PL (BDEW, 2015)) an increase in annual cost of 1–3% can be expected.

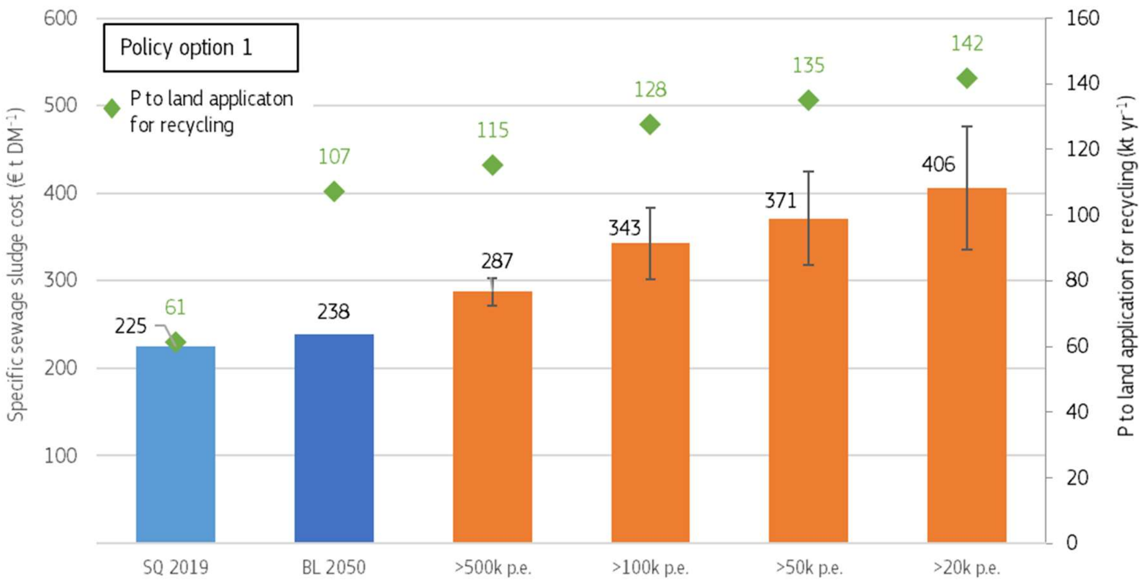
Figure 21. Annual cost for status quo, baseline and the PO1 sub-options considering minimum and maximum values (M€).



8.3.6.3 *Sludge specific cost*

Figure 22 shows that a change in sludge management options, cost per ton of sewage sludge dry matter can increase significantly. Implementing mandatory P-recovery for WWTP exceeding 50k p.e., the cost are in the range of 317–425 € t DM^{-1}. The average cost of 371 € t DM^{-1} corresponds to an increase of 56% compared to the baseline. At the same time, the amount of P applied to land for recycling increases by only 26% compared to the baseline.

Figure 22. Specific cost for sewage sludge recycling, treatment and disposal (€ t DM^{-1}) and P applied to land for recycling (kt yr^{-1}) for status quo, baseline and the PO1 sub-options considering minimum and maximum values.



8.3.6.4 Administrative costs

It can be assumed that WWTP applying sludge on agricultural land already have their sludge tested regularly on those metals required by EU or national sewage sludge legislation. In addition, it can be taken into account that already today extensive analyses on metals are carried out when sewage sludge is composted, used for 'other' purposes or even landfilled. Therefore, no additional cost are expected to monitor metals.

In comparison to metals, only a few MS demand the analysis of e.g. organic pollutants. However, due to the rising concern on organic pollutants but also microplastics, monitoring techniques will involve the targeted measurement of priority pollutants as identified by the EU Commission as well non-targeted screening of chemicals with e.g. once or twice a year if sludge is used on land (see section 7.2.1.1.). This will be associated with additional cost of around 800 € per sewage sludge sample (Agrolab, 2022).

For the amount of sludge shifted to mono-incineration with subsequent P-recovery, is assumed, that organic pollutants or microplastics are fully destroyed. On the basis of this arguments, it is assumed that organic pollutants not need to be analysed and no additional costs will be incurred.

However, the incinerators could implement acceptance criteria for sewage sludge to prevent a dilution of the sewage sludge ash with respect to the P content and to avoid excessive pollution with metals which could negatively impact the direct use of sewage sludge ash in the fertiliser industry but also downstream recycling processes. The parameters of such acceptance criteria could be similar or equal to those parameters that are routinely tested by wastewater treatment plants (e.g. total P content, ash content, set of metals demanded by EU sludge directives as e.g. Cd, Cu, Ni, Hg, Pb, and Zn, but also additional metals set by national sludge directives as e.g. As, Co, Cr, Mo, Se; (Hudcova et al., 2019).

In addition to the mentioned metals above, Tl and V need to be considered in case sewage sludge ash is incorporated into a EU fertilising product (Huygens et al., 2019). Furthermore, for the final product e.g. an inorganic macronutrient fertiliser, Cr (VI) would be another parameter which need to be considered (EU Fertiliser Regulation 2019/1009).

In summary, the following assumptions are made in order to calculate the cost for compliance:

- sewage sludge used on agriculture: 2 samples per year are monitored for priority pollutants (800 € per sample)
- sewage sludge mono-incinerated: 2 samples per year are monitored for 5 additional metals (25 € per metal parameter and sample)

To meet compliance, additional sewage sludge analytical costs of around 1.7–3.9 M€ yr^{-1} are required for WWTP exceeding 50k p.e. for the entre EU-27 (Table 48).

A report containing relevant information on the amount of sewage sludge disposed together with the sewage sludge analysis results must be prepared once a year and submitted to the competent authority. Assuming 3 h for the report preparation with an average labour cost of 24.6 € h^{-1} (EUROSTAT, 2021), additional cost of around 0.2 M€ yr^{-1} considering a WWTP exceeding 50k p.e. are to be expected (Table 48).

8.3.6.5 Competitiveness impacts

Despite the reported cost implications of Policy Option 1, we do not expect significant competitiveness impacts. This is the case because the regulatory costs are in a first instance borne by WWTP. These are public or semi-public (with concessions) entities, which do not operate and compete in open markets, but rather in local monopolies and under tight public regulation. Their objective is to provide predefined services with high reliability and at the lowest costs, and not profit maximisation. Moreover, the cost burden on these entities implied by Policy Option 1 can be widely distributed over the service users, i.e., households.

Competition takes place between different sewage sludge management options, but the costs of these options are not impacted by the envisaged policy.

The indirect impact of higher costs for water services could affect EU companies with a high share of costs related to waste water services. Some specific industries (e.g. fashion sector, energy sector) consume large amounts of water. However, it remains unknown how many companies could be affected, to what extent the additional costs for water services would have a significant impact on their total costs, and whether they would be in direct competition with outside EU companies not affected by the regulation and its indirect effects..

8.3.6.6 Innovation

The mandated control of sludge quality and the requirement to recover at least 75% of the phosphorous contained in the sludge dry mass of WWTPs larger than the capacity threshold does not – nor explicitly nor implicitly – favour any particular technology. All regulatory obligations could be fulfilled with currently existing and employed technologies. Hence it may be concluded that the considered policy would not distort innovation, but neither would it provide new incentives for innovation in the area of sewage sludge.

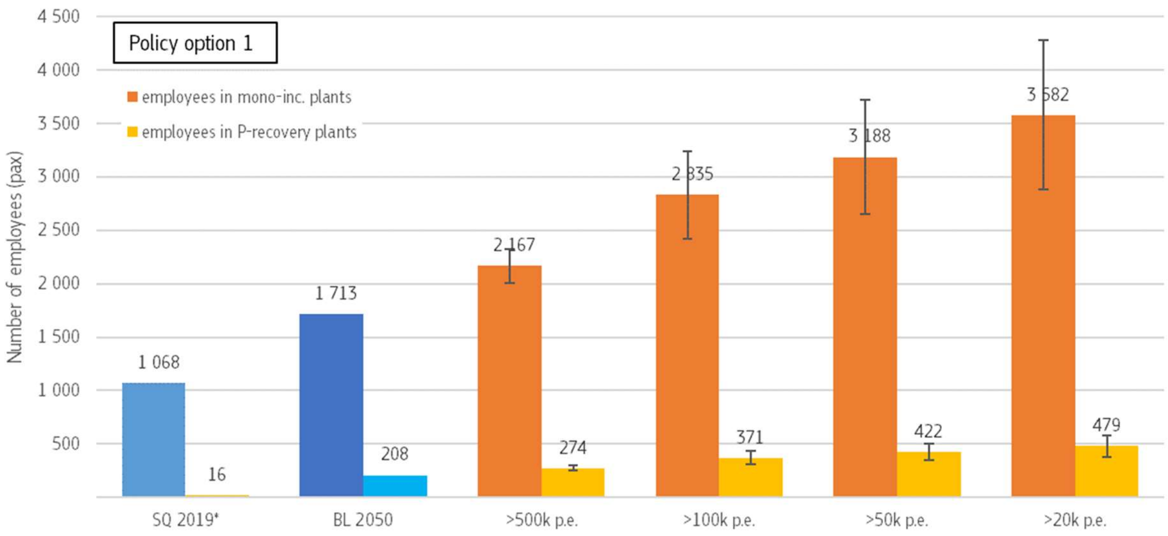
8.3.7 Social and distributional impacts

In this section the potential social impacts of Policy Option 1, in particular employment and distributional impacts, are briefly explored.

8.3.8 Employment

As explained above, Policy Option 1 implies a re-direction of sewage sludge from co-incineration and landfilling towards mono-incineration with P recovery. In view of the number of needed new mono-incineration and P-recovery plants (Figure 18) under each sub-option, and assuming 22 and 8 employees, respectively, for the considered plant sizes, a positive employment impact in the range between 3 000 to 4 220 additional jobs can be expected for WWTP exceeding 50k p.e.; (see Figure 23[19]). Note that this estimate only includes the direct impact related to the operation of mono-incineration and P-recovery facilities, and does not include additional indirect jobs, e.g. in transport services. These new jobs would be supported by the sales of the phosphorus derived from sewage sludge, but - due to lacking economic viability of this process as of today - also by higher sludge management costs borne by WWTP, which in turn would have to pass on these additional costs to the users of water services, i.e., households and industry.

Figure 23. Number of additional direct jobs due to mandatory P recycling for the different sub-options for policy option 1. Source: own estimate.



These employment gains have to be weighed against potential losses in other areas. However, it does not seem likely that the re-direction of sewage sludge from co-incineration and landfill towards mono-incineration would be significant enough to create a negative employment impact in the former.

8.3.9 Distributional impacts

Distributional impacts should include negative implications of the policy option that weigh especially on more vulnerable stakeholders, fairness aspects, or a general increase of inequality. In the present case, the projected cost increase for WWTP is likely to be financed by higher user fees. Although the absolute increase might be very small, it has to be acknowledged that expenditures for energy and utility services can represent a disproportionately large share of total expenditures for low-income households. This means that even small

[19] It also is assumed that all sewage sludge ash is treated by P-recovery facilities, instead of directly going to the fertilizer industry.

increases can have significant impacts on the available income of this stakeholder group. On the other side, many Member States have put into place social policies to cushion the impact of utility fees and their volatility on low-income households. It is recommended to verify the existence of such mechanisms to deal with the potential policy-induced increase of user fees.

8.4 Policy option 2 (PO2)

Policy option 2 consists of:

- Mandatory use of sewage sludge generated at waste water treatment plants exceeding a certain size for the production EU fertilising products. Sub-options involve here setting the requirements for sewage sludge generated at WWTP above 500k p.e., 100k p.e., 50k p.e. and 20k p.e., respectively. The EU fertilising products should classify as inorganic macronutrient P fertilisers (Product Function Category 1(C)(I))[20], and a minimum P recovery efficiency of 75% of the P contained in the sewage sludge used as feedstock for the recovery process should be achieved.
- Sewage sludge originating from waste water treatment plants below the size threshold are not subject to further use restrictions, other than the general requirements and waste management hierarchy set out in the Waste Framework Directive.
- In a first sub-option, Member States shall develop guidelines to ensure environmental and health protection from sewage sludge used for recycling operations (use in agriculture or forestry, other recycling operations) originating for waste water treatment plants below a certain size threshold[21].
- In a second sub-option, minimum sewage sludge quality standards for sewage sludge used in agriculture shall be set in the Directive, and include limit values for metals and organic contaminants of concern.

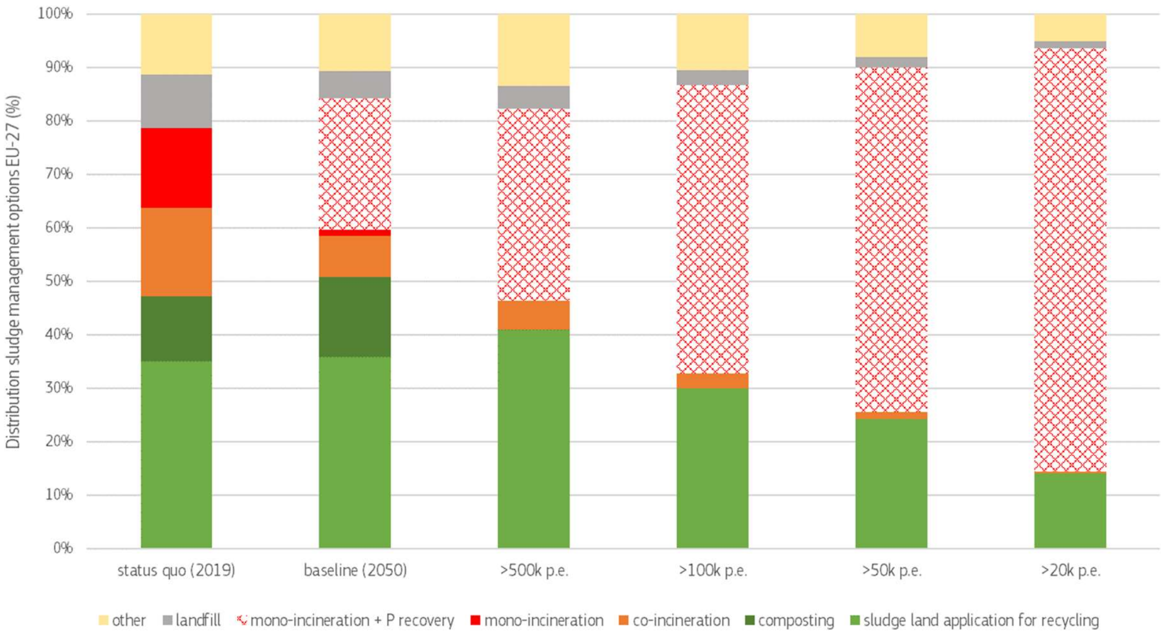
8.4.1 Sewage sludge mass, nutrient and organic matter content, and management routes

Policy option 2 (PO2) does not have any effect on the sewage sludge mass produced or the nutrients or organic matter within. However, PO2 has significant impact on the sewage sludge management routes. With the implementation of mandatory mono-incineration and mandatory technical P-recovery, we expect substantial changes compared to 2019 and 2050 (Figure 11, Figure 24). Considering all WWTP exceeding 50k p.e. shift sewage sludge to mono-incineration and innovative technologies that recover P in a mineral from, a consequence is the reduction of all other sewage sludge management options. The share of co-incineration and landfill together is expected to fall below 3%. The expected reduction in the amounts of sewage sludge that will be landfilled is in line with a further implementation of the landfill directive and the methane strategy. This table also indicates that still around 7% of sewage sludge (composted or untreated) will be landspread outside the food chain (comprised in the fraction 'other').

[20] Regulation (EU) 2019/1009 of the European Parliament and of the Council laying down rules on the making available on the market of EU fertilising products and amending Regulations (EC) No 1069/2009 and (EC) No 1107/2009 and repealing Regulation (EC) No 2003/2003.

[21] As outlined in section 7.2.1.2, sewage sludge originating from waste water treatment plants exceeding the size thresholds shall not be used for other recycling operations outside agriculture because, in this strictest policy option, it is considered that the inherent nature of sewage sludge, containing numerous unknown contaminants with undetermined toxicological properties and other pollutants including microplastics cannot be applied in a safe manner in the environment.

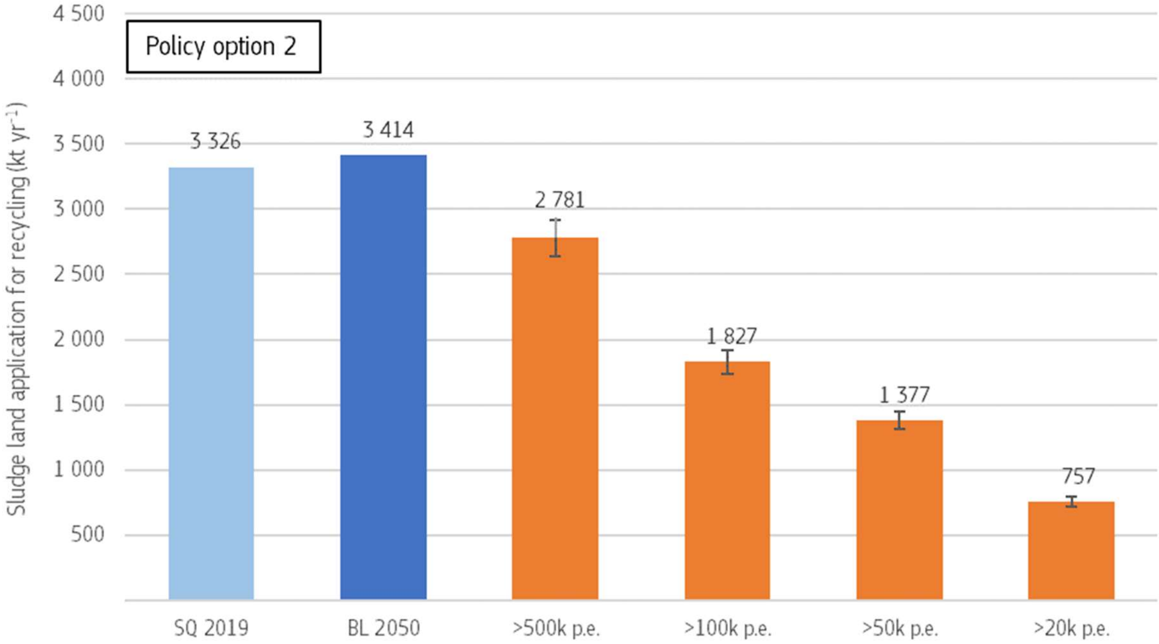
Figure 24. Final uses for sewage sludge management in the status quo (2019), baseline (2050) and with mandatory technical P-recovery for the PO2 sub-option



8.4.2 Environmental and health protection

With an implementation of mandatory transformation of sewage sludge into an EU Fertilising Product at WWTP exceeding a certain threshold size, sewage sludge landspreading would be greatly reduced. For instance, in the policy sub-options that have effect on any WWTP exceeding 50k p.e., the direct agricultural use of sewage sludge would be reduced by around 60% compared to the baseline, effectively limiting the contamination of soils and crops with contaminants contained in untreated, lime stabilised, composted, or digested sewage sludge. On the other hand, sewage sludge with the disposal route mono-incineration with subsequent P-recovery would account for around 5 600 kt per yr^{-1} (71% of total annual sewage sludge production of 7 900 kt DM yr^{-1}). EU Fertilising Products comply with strict quality requirements, implying that environmental and health risks are effectively controlled for. At present, phosphorus macronutrient EU fertilising products can only be derived from sewage sludge following a precipitation or, more likely, a thermal oxidation step. Besides the removal of organic contaminants and microplastics during incineration, the application of appropriate technologies offers the possibility to depollute the produced sewage sludge and promote a clean circular economy. Such thermal processing thus effectively controls for risks from organic contaminants that may be present in sewage sludge, but have (so far) not been identified as being of concern. This policy option also mitigates potentially increased risks from mixtures of organic contaminants present in the sewage sludge. Environmental and health risks from landspreaded sewage sludge generated at WWTP below the threshold could either be controlled by Member States or EU legislation, depending on the sub-option selected. Having EU-wide requirements could be considered the option that offers a higher level of protection because Member States have a different stance towards setting limit values for contaminants in sewage sludge. The Evaluation report of the Directive indicated that several EU Member States have, for instance, updated limit values for metals in line with scientific progress, whereas others have failed to do so. It is, however, noted that an extensive monitoring of sewage sludge quality at EU level is not included in this policy option, possibly limiting the knowledge on sewage sludge contaminant profiles that forms the basis for the setting of limit values.

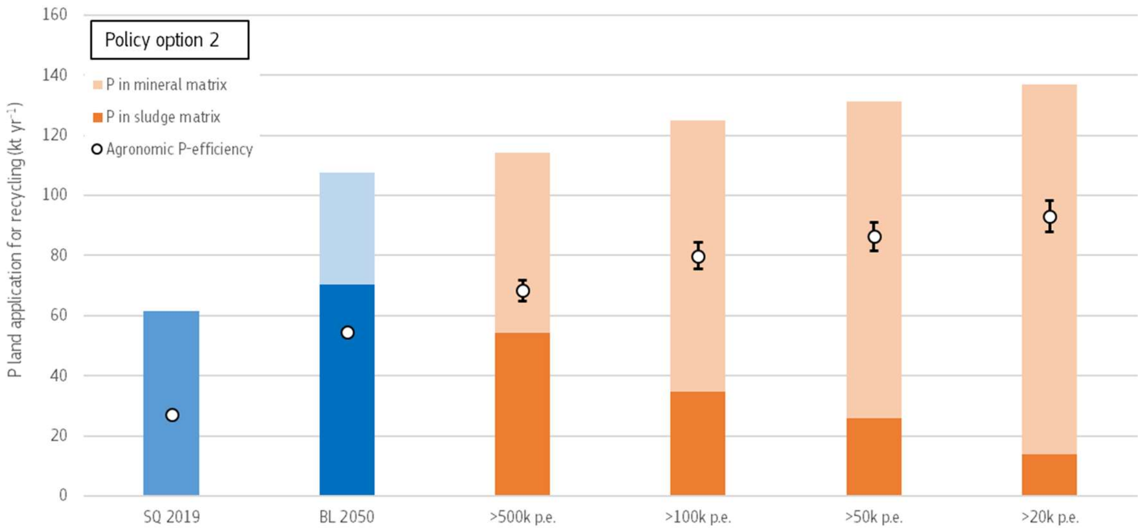
Figure 25. Amount of sewage sludge applied to land for recycling for status quo, baseline and the PO2 sub-options considering the minimum and maximum values (kt yr^{-1})



8.4.3 Resource recovery

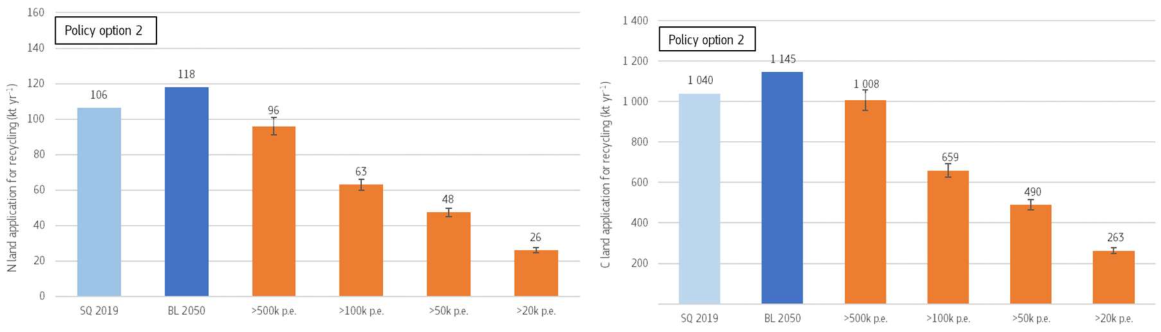
The amount of P applied to agricultural land will be increased by the same order of magnitude as the different sub-options in PO1 (section 8.3.3). For the case of mandatory technical P recovery from sewage sludge from WWTP >50k p.e., 80% (from 131 kt P yr^{-1}) of the P brought to agricultural land will be in form of a mineral matrix with high agronomic efficiency (Figure 26). With projected annual EU import of 0.77 Mt P yr^{-1} in mineral fertilisers, the P in sewage sludge applied in the baseline equals about 13.9% of the net imports. For the PO2 sub-option recovering P from WWTP >50k p.e., a maximum 0.024 Mt P yr^{-1} could additionally be recovered compared to the baseline. This corresponds to a potential to substitute about 17% of the P imported in mineral fertilisers.

Figure 26. P applied on agriculture for status quo, baseline and the PO2 sub-options in sewage sludge and mineral matrix considering the agronomic efficiency including minimum and maximum values (kt yr^{-1}). The dots 'agronomic P-efficiency' refer to short-term bio-available P by taking into consideration the agronomic P-efficiency of sewage sludge and sewage sludge derived P-fertilisers returned to land.



As a result of the incineration of the sewage sludge, carbon and nitrogen transfer into the gaseous phase and must be considered lost for recycling. In the case that all WWTP with a capacity >50k p.e. have to incinerate the sewage sludge, the application of C and N in the EU-27 will be reduced more than 50% (Figure 27).

Figure 27. Nitrogen (left) and carbon (right) land application for recycling for status quo, baseline and PO2 sub-options including uncertainties (kt yr^{-1})



8.4.4 Nutrient losses

A reduction in nutrient losses from sewage sludge relative to the baseline is expected for this policy option, because of the increased shares of sewage sludge that will be transformed into mineral P fertilisers with a higher nutrient efficiency than sewage sludge. The higher nutrient efficiency of mineral P fertilisers relative to sewage sludge is caused by the differences in the intrinsic properties of the materials as well as through differences in the ability to handle and store the materials.

Mineral fertilisers offer potential to better align nutrient release to plant nutrient demand than organic materials, especially when complexed in a Fe- or Al-rich organo-mineral matrix. Policy option 2 increases the shares of P converted into plant-available forms, with a similar agronomic efficiency than conventional mined mineral P fertilisers. Other nutrients such as nitrogen and organic matter can be added in any ratio, thus providing soils and plants with nutrients and organic matter according to their needs in terms of time and quantity. In comparison, sewage sludge does not have the optimal nutrient ratio and the bio-availability of nutrients contained in this matrix is also inferior to that of commercial mineral fertilisers.

The conversion of P from a wet organic sewage sludge matrix into a dry mineral matrix offers the advantage that the nutrients can be stored during a long time and transported over long distances to areas with nutrient deficit. This reduces the potential contribution of sewage sludge to surface and groundwater pollution.

8.4.5 Methane emission

For PO2 the reduction of methane emissions is equal to PO1 (see section 6.5), as the sewage sludge currently brought to landfill is either applied on land or incinerated in the two policy options. Therefore, also for PO2 the methane emissions will be reduced significantly although, the contribution from landfilled sewage sludge to the total methane emission is already minor.

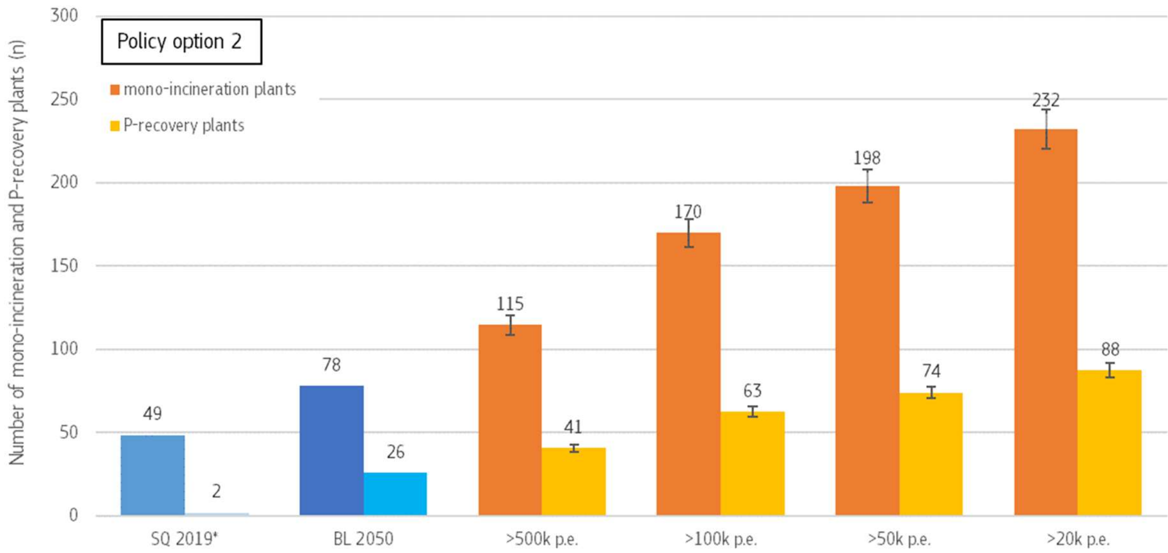
8.4.6 Compliance costs

8.4.6.1 Investment cost

For the baseline it was predicted that by 2050 the numbers of mono-incinerators and P-recovery units already increase compared to the status quo due to the fact that AT and DE will have national mandatory P-recovery in place by then. Furthermore, it was assumed that for countries with existing mono-incineration infrastructure, the generated SSA will undergo treatment for P recovery due to the availability of suitable technologies. For comparability reason, the number of mono-incineration plants each sub-option were calculated by dividing the amount of sewage sludge treated in mono-incineration plant with an assumed average incineration capacity of 30kt DM yr^{-1}. For P-recovery plants, the average capacity is assumed with 30kt SSA yr^{-1}.

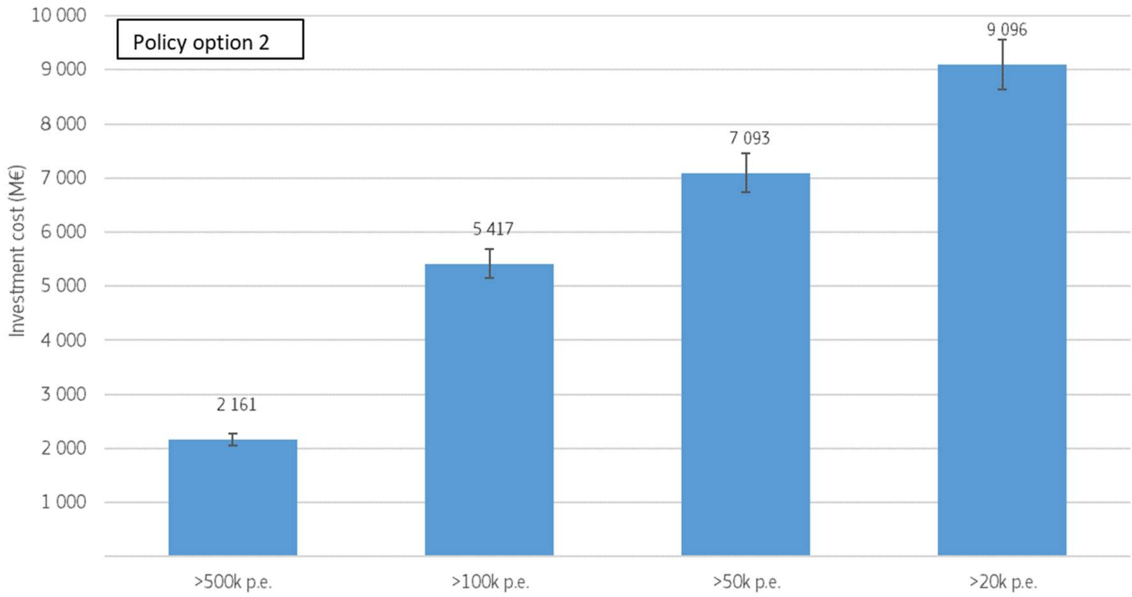
With implementation of the mandatory technical P-recovery for WWTP >50k p.e. on EU-27 level, around 115–232 additional mono-incineration and 41–88 P-recovery plants will need to be installed (Figure 28).

Figure 28. Number of mono-incineration and P-recovery installations for status quo (SQ 2019), baseline (BL 2050), and the policy option 2, including sub-options.



Due to the greater amount of sewage sludge that is assumed to be shifted to mono-incinerators with P-recovery, the investment cost are expected to be higher compared to PO1. The required investment cost for the installation of the new mono-incineration and P-recovery infrastructure for the sub-option for WWTP >50k p.e. range from 2 161 to 9 096 M€ until 2050 (Figure 29). In comparison to the PO2 investment cost of 7 093 M€ for the >50 k p.e. sub-option, the cost for the sub-option addressing the WWTP >50k p.e. in PO1, are in the range of 2 500–5 400 M€ depending on the percentage of sewage sludge shifted to mono-incineration (see chapter 8.3.6). Again, this investment cost analysis does not consider any necessary reinvestment cost until 2050 for mono-incineration plants already in operation.

Figure 29. Investment cost for mono-incineration and P-recovery installation for PO2 (M€)



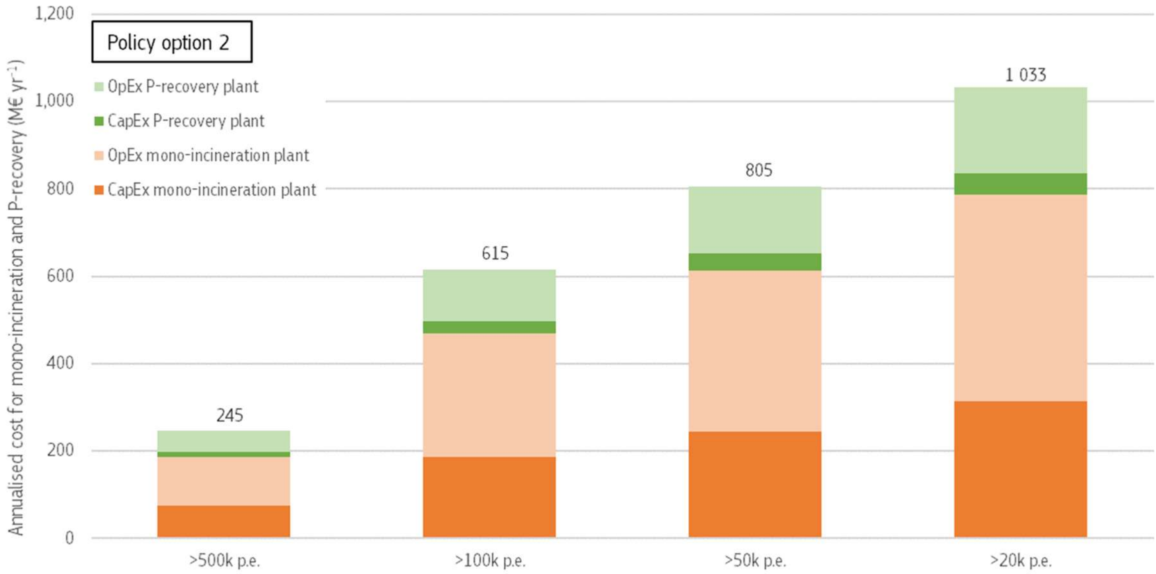
8.4.6.2 *Annual costs for sewage sludge management*

Annualisation of investment cost for mono-incineration and P-recovery plants

To fulfil the requirements for 100% mono-incineration and technical P-recovery, a greater number of plants are necessary, resulting in additional annual cost in relation the baseline of around 800 M€ (sub-option:

>50k p.e.). This represents nearly double the annual cost compared to the mean value of the >50k p.e. sub-option of PO1 (see section 0). To annualize the investment cost, a lifetime of 25 years was considered for both types of plants (Share of capital cost on total cost: mono-incineration plant: 40%; P-recovery plant: 20% (see Annexes, section 12.2.5 and 12.2.6)).

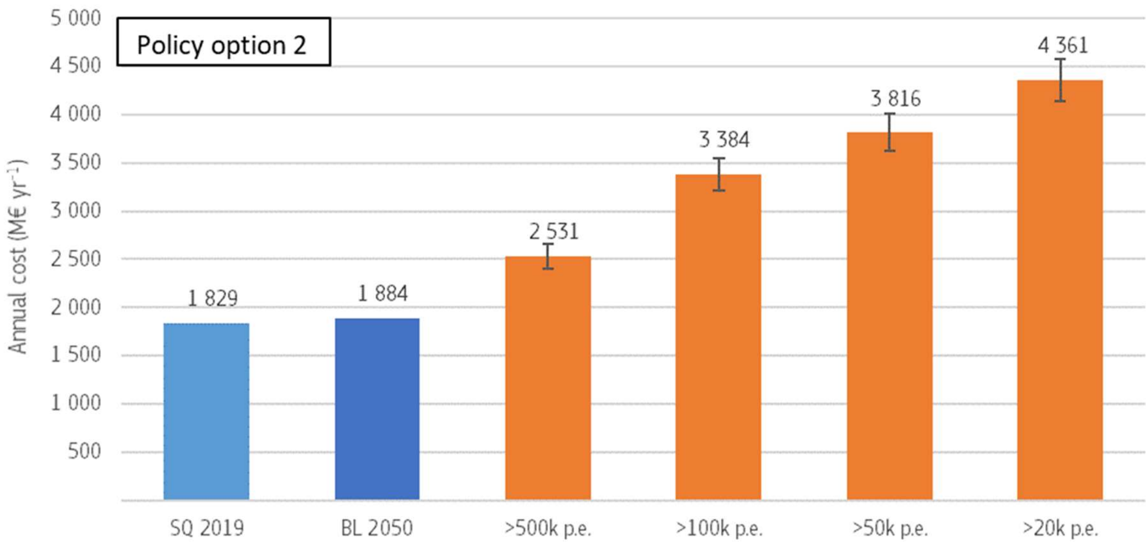
Figure 30. Annual cost for mono-incineration and P-recovery (divided into CapEx and OpEx)



Annual cost of sewage sludge management

The shift from one sewage sludge management option to another can result in savings (e.g. for some MS in case of a change from landfill to agricultural use) or additional cost (e.g. change from agricultural use to mono-incineration and P-recovery). Figure 31 shows the impact on annual costs due to the change in sewage sludge management options for each sub-option. Upon implementing mandatory technical P-recovery for WWTP exceeding 50k p.e., the additional annual cost are in the order of magnitude of 1 900 M€ yr^{-1} (in comparison PO1: 620–1 480 M€ yr^{-1}). This corresponds to additional annual costs per inhabitant in the order of 4.4 €. Considering that expenditures for wastewater treatment are on average 130 € per person and year (data for AT, DE, FR, NL, PL (BDEW, 2015)), this would correspond to an increase in annual cost by 3.5%.

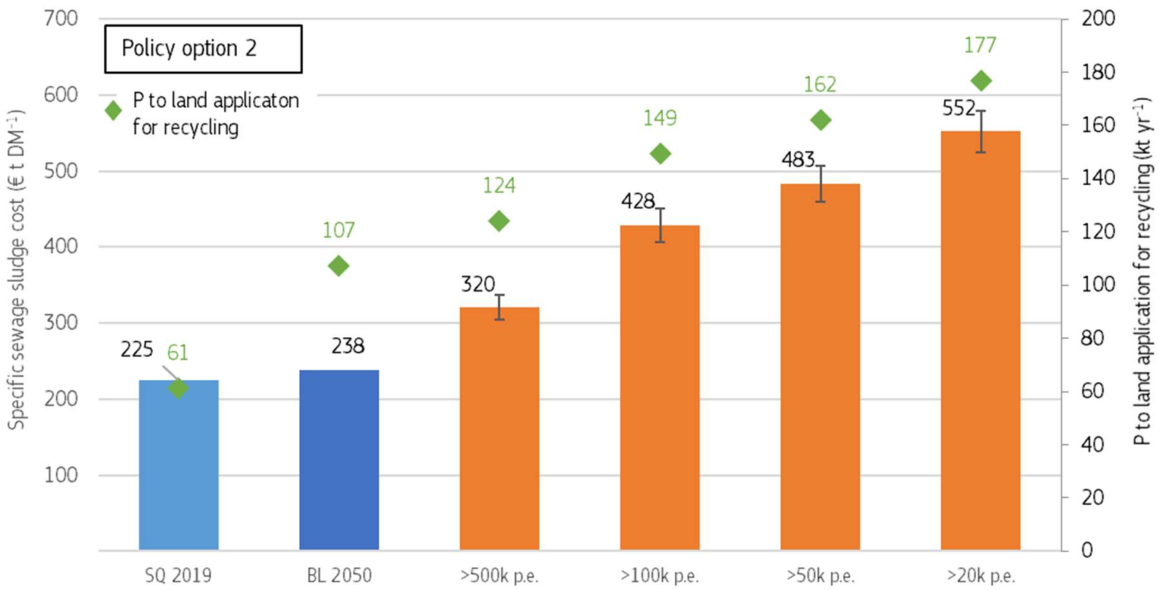
Figure 31. Annual cost for status quo, baseline and the PO2 considering uncertainty for each scenario (M€)



8.4.6.3 *Sewage sludge specific cost*

Figure 32 shows that a change in sewage sludge management options, cost per tonne sewage sludge dry matter can increase significantly. Implementing mandatory P-recovery for WWTP exceeding 50k p.e., the cost are in the range of 459–507 € t DM^{-1}. The average cost of 483 € t DM^{-1} corresponds to an increase of 103% compared to the baseline. At the same time, the amount of P applied to land for recycling increases by only 51% compared to the baseline.

Figure 32. Specific cost for sewage sludge recycling, treatment and disposal (€ t DM^{-1}) and P applied to land for recycling (kt yr^{-1}) for status quo (SQ 2019), baseline (BL 2050) and the PO2 sub-options considering minimum and maximum values.



8.4.6.4 *Overview on investment and additional annual sewage sludge management costs on MS level*

— Figure 33 (for all Member States, except ES, FR and IT) and Figure 34 (for ES, FR, IT) offer a comprehensive overview on the necessary investment cost (M€) but also the additional annual sewage sludge management cost (M€ yr^{-1}) for each MS.

— As AT and DE will or already have mandatory P-recycling in place, the necessary investments will already have been undertaken, and the higher annual costs will already have been factored into the baseline.

Countries with already high share of sewage sludge incineration (e.g. BE, GR, NL, PL) will have lower cost to comply with mandatory incineration and P-recovery. Countries with high sewage sludge production and a high share of direct sewage sludge application to land for recycling in the baseline will face high investment costs as well as higher annual costs.

Figure 33. Investment cost (M€) in relation to the additional annual cost (M€ yr^{-1}) of MS excluding ES, FR, and IT for PO2 (bubble size indicates the annual sewage sludge production for the MS; kt yr^{-1}).

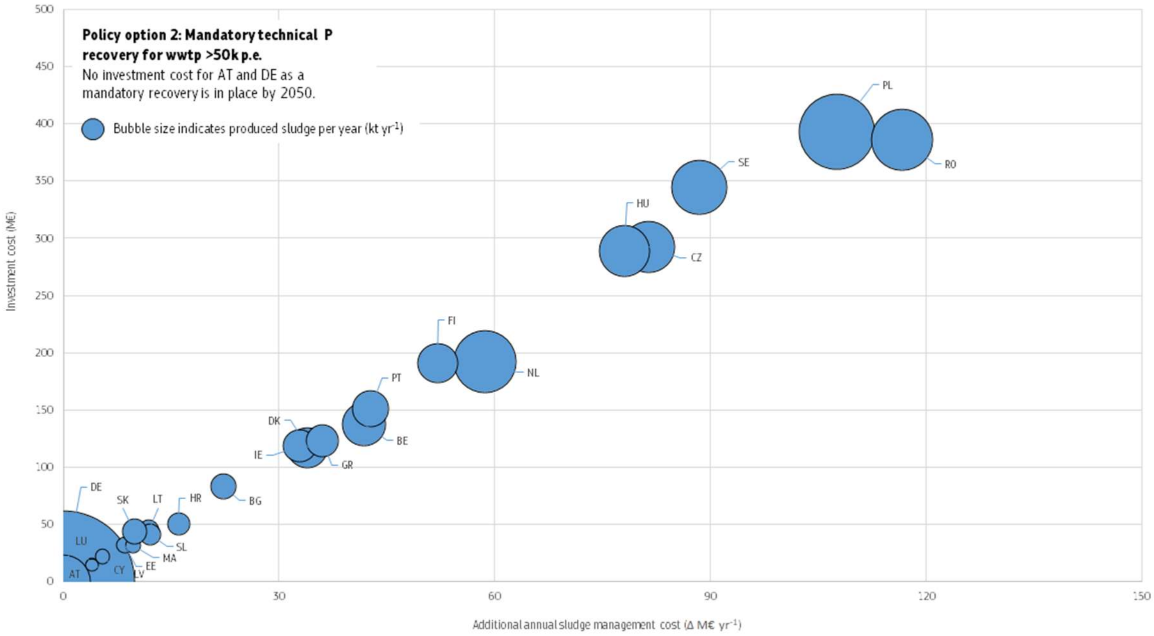
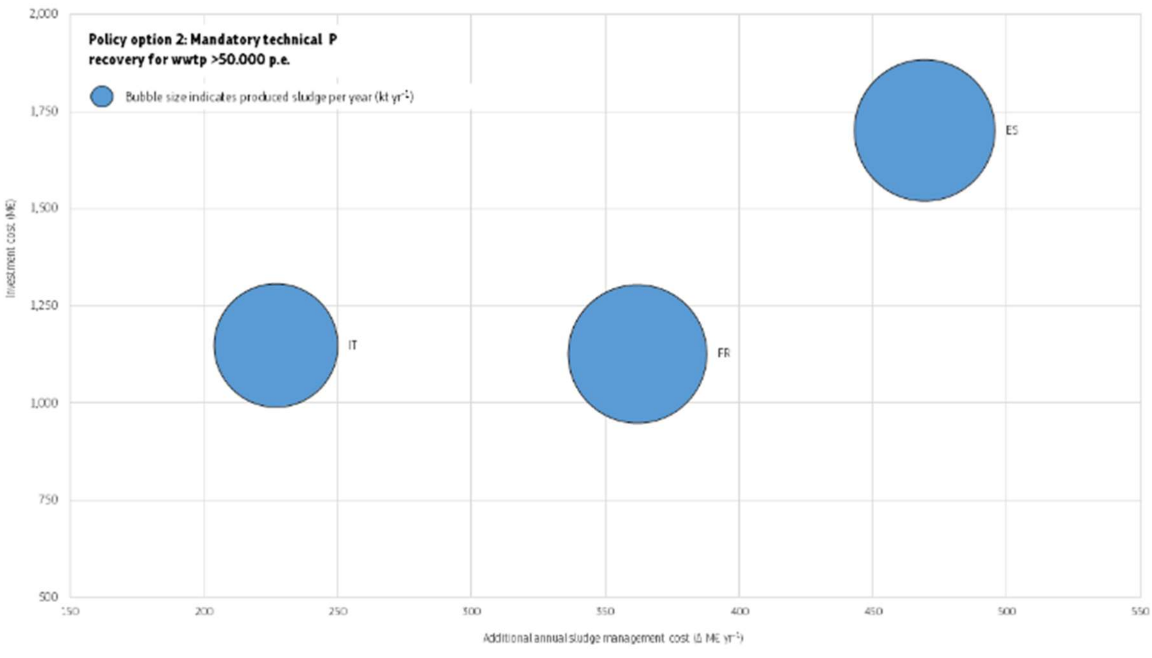


Figure 34. Investment cost (M€) in relation to the additional annual cost (M€ yr^{-1}) of MS exclusively for ES, FR, and IT for PO2 (bubble size indicates the annual sewage sludge production for the MS; kt yr^{-1}).



8.4.6.5 *Administrative costs*

It is assumed that organic pollutants are fully removed from the ashes in the event of incineration. Therefore, it follows that organic pollutants do not need to be analysed and no additional costs will be incurred.

However, it is assumed that the incinerators could impose acceptance criteria for sewage sludge to prevent a dilution of the sewage sludge ash with respect to the P content and to avoid excessive pollution with metals which could negatively impact the direct use of sewage sludge ash in the fertiliser industry but also downstream recycling processes. The parameters of such acceptance criteria could be similar or equal to those parameters that are routinely tested by wastewater treatment plants (e.g. total P content, ash content, set of metals demanded by EU Sewage Sludge Directives as e.g. Cd, Cu, Ni, Hg, Pb, and Zn, but also additional metals set by national Sewage Sludge Directives as e.g. As, Co, Cr, Mo, Se; (Hudcova et al., 2019).

In addition to the mentioned metals above, Tl and V need to be considered in case sewage sludge ash is incorporated into a EU fertilising product (Huygens et al., 2019). Furthermore, for the final product e.g. an inorganic macronutrient fertiliser, Cr (VI) would be another parameter which needs to be considered (EU Fertiliser Regulation 2019/1009., it is assumed that in addition to the standard metal parameters, a maximum of 5 additional would need to be analysed. Considering two samples per year with a price of 25€ per tested metal parameter, additional cost of around 650 000 € yr^{-1} would be required for WWTP exceeding 50k p.e. (see Annexes, section 0).

In summary, it is assumed that in addition to the standard metal parameters, a maximum of 5 additional metal parameters would need to be analysed. Considering two samples per year with a price of 25€ per tested metal parameter, additional annual cost of around 660 000 € yr^{-1} would be required for all WWTP exceeding 50k p.e. (Table 49). An annual report containing relevant information on the amount of sewage sludge disposed together with the sewage sludge analysis must be prepared and submitted by the WWTP operators to the competent authority. Assuming 3 h for the report preparation with an average labour cost of 24.6 € h^{-1} (EUROSTAT, 2021), additional cost of around 0.2 M€ yr^{-1} considering a WWTP exceeding 50k p.e. are to be expected (Table 49).

8.4.6.6 Competitiveness impacts

As in Policy Option 1, the first direct impact of the envisaged mandate to transform sewage sludge into an EU fertiliser product affects all WWTP beyond the capacity threshold, and in particular their internal costs of sewage sludge management. As argued before, the activities of these public or semi-public entities are not subject to open (international) competition, they operate in a monopolistic and highly regulated environment, and can distribute the additional costs among a broad base of users.

However, and in contrast with the previous policy option, the implied interruption of sewage sludge spreading on land could have a negative competitiveness impact on agricultural activities. Especially for farmers in the drier regions of the EU (e.g. Spain), receiving and spreading sewage sludge is an important source of organic matter and phosphorus intake for agricultural lands, and in addition generates some income. Under the here considered policy option, farmers would instead have to buy fertilising and soil improvers at market prices.

Fertilisers purchase already makes up a significant share of overall production costs in agriculture (15%-40% for arable crop farmers), especially after the recent price increases (caused by higher energy prices). Although a more quantitative assessment is not possible at this point – mainly due to a lack of comprehensive economic data on implied fertiliser needs and costs - it can be assumed that the considered policy would lead to a competitiveness loss of agricultural producers in regions where landspreading is currently practiced and integral part of the soil nutrient supply. On the other side, the large-scale employment of sewage sludge to produce certified fertiliser products can also be expected to reduce the costs of this process (economies of scale, learning by doing) and bring down the market price of the derived fertiliser products.

8.4.6.7 Innovation incentives

In contrast to Policy Option 1, a requirement to transform sewage sludge into a fertilising product can only be met with the relatively young technology of P recovery from incineration ashes that is still not widely employed (2 facilities in operation in the EU, see Table 52). Since the demand for such facilities would increase substantially if the policy were implemented, it would create a strong incentive for further diffusion of and innovation in this technology. While by itself a positive effect, this would likely be accompanied by a less desirable path dependency effect, i.e., it would induce a bias towards this incineration-based and hence relatively destructive approach and against other phosphorus recovery approaches and technologies that could be able to retain phosphorus in the cycle without incinerating sewage sludge (which accidently destroys valuable organic matter and nitrogen contained in sewage sludge). This is particularly relevant considering the typical lifetime of incineration plants (20-25 years), thus possibly creating a lock-in effect into this technology.

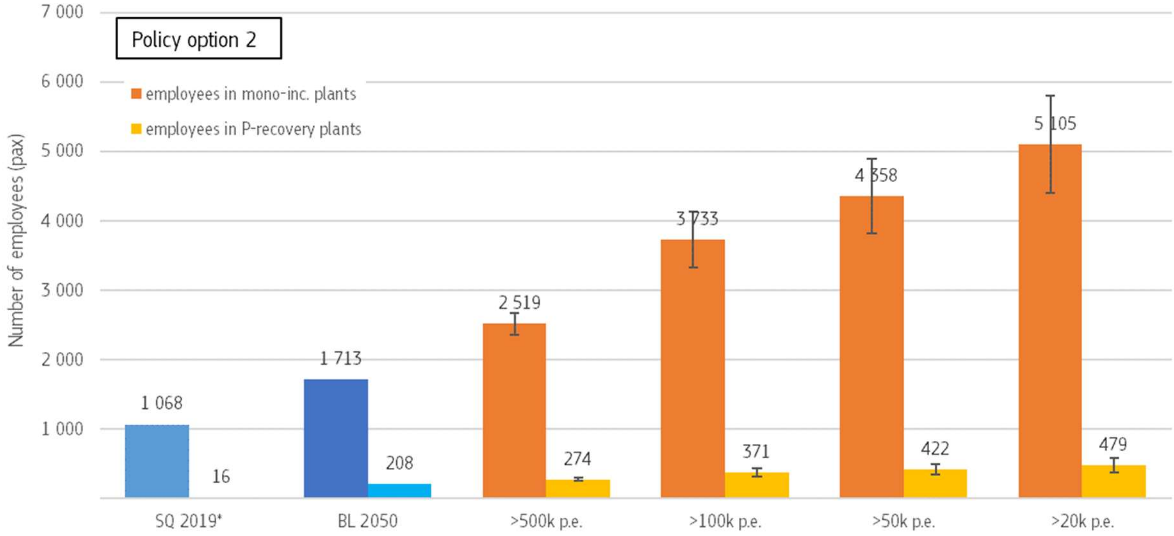
8.4.7 Social and distributional impacts

In this section, potential policy impacts on employment and distributional aspects are examined.

8.4.8 Employment

Under Policy Option 2, the positive employment impact on mono-incineration and P-recovery activities is significantly stronger than under Option 1, given that this treatment pathway is now mandated for all sewage sludge generated in WWTP larger than the threshold capacity. Based on the number of necessary new mono-incineration and P-recovery plants (Figure 35) under each sub-option, and assuming again 22 and, respectively, 8 employees for each type of plant, the estimate of additional jobs is in the range of 4 700 to 5 200 (for WWTP > 50k p.e.). Given that this is only the direct employment impact, further indirect employment gains, e.g. in the transport sector, would need to be added.

Figure 35. Number of additional jobs due to mandatory (technical) P recycling under PO2 for different sub-options considering minimum and maximum value (own computations).



Employment gains in sectors that benefit from the regulation must be checked against potential losses in other sectors that are negatively affected. In the present case, the redirection of sewage sludge towards co-incineration implies the loss of sewage sludge available for landfills, co-incinerators, agricultural businesses and owners/managers of other lands that regularly apply sewage sludge, and composters. Since sewage sludge does not represent a significant share of total input for landfills, co-incinerators, and composters[22], negative employment impacts are not expected for these activities.

For agricultural businesses and farmers, the use and application of sewage sludge is likely not requiring a significant share of the total labour effort. Hence, the impact from banning the direct application of sewage sludge is financial rather than on the workload or capacity utilisation. However, the previously described negative economic impact and potential competitiveness loss could by itself lead to a loss of employment in this sector. A more quantitative estimate cannot be provided, also because this effect is expected to strongly depend on local circumstances.

8.4.9 Distributional impacts

The implied increase in sewage sludge management costs is higher for this policy option than for Policy Option 2. As a consequence, the potential increase of the user fees for water services, and its potential distributional impact on low-income households might be more pronounced.

In addition, the policy would lead to a large-scale redistribution of economic benefits that can be derived from sewage sludge from the agricultural sector towards the fertiliser value chain (mono-incinerators, P-recovery facilities, and fertiliser producers). As described before, this implies a loss of income for the group of farmers

[22] 2–3% according to Compost Network (https://www.compostnetwork.info/policy/biowaste-in-europe/)

that traditionally rely on sewage sludge spreading (which is the case in some EU regions more than in others). The negative impact could be more severe in rural areas where livelihoods already stand on a strained economic base. Due to the local specificity of the issue and the lack of suitable economic data, a quantitative estimate cannot be provided.

At the macro level it should be acknowledged that those Member States with already high shares of sludge incineration (e.g. BE, GR, NL) and existing facilities for such processes will face lower compliance costs than those EU countries where this share is low or very low. A special case is given by AT and DE, which already have legislation in place that mandates incineration and P-recovery, and which therefore would not face any additional costs from implementing Policy Option 2.

8.4.10 Other impacts

8.4.10.1 Mono-incineration

Decreasing acceptance for direct agricultural sewage sludge use

It is quite possible, that direct agricultural use of sewage sludge will lose acceptance compared to incineration and P-recovery as soon as mandatory requirements for P-recovery from SSA are implemented. Especially the discussion about (organic) pollutants and the established incineration capacity could lead to a situation that WWTP smaller than 50k p.e. will then also canalize their sewage sludge to incineration, although they would not be obligated from the legal point of view.

Cost for mono-incineration

Currently, around 60 mono-incinerators operate already in the EU-27 MS (Table 51), which have incineration capacities ranging from 1 500 to 85kt DM yr^{-1}. In particular, incineration in small plants can lead to higher treatment costs. By planning the incineration capacities accordingly, based on the quantities of sewage sludge to be treated in the individual MS, larger standardised central plants can be built, which in turn can reduce the treatment costs per t of DM (economy of scale). Same is valid for the P-recovery plants.

Fluidized bed reactors are the most commonly used technology for the incineration of sewage sludge and paste waste. Due to their technical complexity, this technology is more expensive compared to for example grate furnaces for the incineration of municipal solid waste. The investment in up to 230 new mono-incinerators until 2050 could also lead to lower investment cost, as this technology becomes more and more a standard product.

Production of energy

In principle, the incineration of sewage sludge is a net energy positive process. The surplus the surplus electricity can be fed into the power grid and the surplus heat into the district heating grid.

Substitution of energy sources for co-incineration plants

The diversion of biogenic sewage sludge from co-incineration could lead to co-incineration plants having to substitute the loss of energy with primary energy sources or other types of waste with high fossil carbon content (e.g. RDF, waste oil). This might lead to additional fossil CO_2 emissions with a corresponding negative impact on the climate. In addition, co-incineration plants lose revenue due to not receiving sewage sludge. Furthermore, they may be confronted with additional costs if, for example, primary energy sources or wastes with an economic value have to be purchased to substitute for the loss of sewage sludge.

Transport

For reasons of economies of scale, sewage sludge mono-incineration plants with large incineration capacities should be built. This leads to the construction of a few mono-incineration plants at central locations. In turn, sewage sludge especially from small regional WWTP, has to be transported over longer distances resulting in additional emissions and noise. However, it must also be taken into account that rock phosphate or its derived products has to be imported from abroad. These transport distances can be considerable, depending on the origin of the raw phosphate rock or fertilisers (Morocco, Russia, and China).

Waste

The direct sewage sludge application is a waste free approach. In comparison, incineration and subsequent P-recovery produce different types of solid wastes (e.g. fly ash, filter cake from incineration but also P-recovery processes, depolluted SSA) which need further treatment before or can be landfilled without further treatment.

9 Next steps and timelines to develop policy options

The evaluation report of the SSD showed that the Directive is outdated on several aspects. Particularly, environmental and health risks from sewage sludge application on agricultural land should be addressed, as well as a better alignment to a circular economy. This report explores impacts from policy options that could potentially be envisaged as part of a revised Directive. Where possible, it may be appropriate to anticipate certain tasks to further develop the scientific and technical knowledge base on sewage sludge management or ensuing policy options. Such activities may help to refine the expected impacts from policy options, better inform stakeholders, and speed up the revision of the Directive to address at rapid pace the problems observed. This section lists a set of actions for attention at this stage of the policy cycle.

9.1 Issue guidance for the voluntary monitoring of sewage sludge

Even in the absence of a legal obligation, the Commission could share guidance on contaminants of possible concern for which monitoring data could be collected. The absence of a recent and extensive dataset on concentrations of organic contaminants and microplastics of potential concern has been identified in a recent JRC report (Huygens et al., 2022). Also, current data for metals are to smaller extent missing. A well-developed dataset on present-day concentrations in sewage sludge is critical to assess risks, and to evaluate the impact of chemicals legislation on sewage sludge quality. Often, parties involved in such measurements campaign are unaware of the identity of the most pressing contaminants of concern in sewage sludge, or do not succeed in accessing an available guidance. The Commission could issue guidance on data gaps that have been observed so they can be filled by actions and measurement campaigns undertaken by Member States, economic operators or research organisations active in the field of work.

9.2 Collect scientific opinions on potential contaminants of concern

One of the shortcomings observed is that the Directive has not been successful in responding to new scientific evidence on the environmental and health risks from contaminants in sewage sludge. One main reason for this observation relates to the absence of specific scientific knowledge in this field of work. At present, scientific support to the policies on sewage sludge is provided based on contractors involved, internal administrative agreements within the Commission Directorate-Generals (ENV/JRC), and contributions from Member States present in expert groups. The scientific support is limited as the scientific background of these partners is not specific on risk assessments, exposure modelling and/or (eco-)toxicology. Both policy options continue to a variable extent the landspreading of sewage sludge, and aim at turning the Directive in a more dynamic EU legislation that enables to respond to new scientific evidence on the risks from contaminants in sewage sludge. This seems a policy need due to continuing innovation in specialty and diversifications of chemicals that are placed on the market and phased out. Therefore, steps could be taken to collect more information on the actual risks from contaminants of concern to better understand the magnitude of the problem identified in relation to contaminants in sewage sludge, and correspondingly adjust the policy options if needed. At present, a risk screening study developed by JRC (Huygens et al., 2022) provides preliminary evidence for risks from organic compounds, but the authors pointed to a need to confirm the findings in a more detailed risk assessment. Moreover, present-day risks from other contaminants, including metals and/or microplastics, are not available.

9.2.1 Liaise to organisations to agree on robust, systematic and independent scientific support

In order to ensure that scientific evidence is processed and assessed in a systematic, independent and robust manner, it is appropriate to rely on the continuous support of a single organisation or working groups over time. This will ensure the robustness, continuity and accountability of the scientific support. Organisations active in this field of work are, for instance, the Scientific Committee on Health, Environmental and Emerging Risks (SCHEER) or the European Chemicals Agency (ECHA). The SCHEER, on request of Commission services, provides opinions on questions concerning health, environmental and emerging risks. The SCHEER reviews and evaluates relevant scientific data and assesses potential risks based on the opinions of top independent scientists from all over the world who are committed to working in the public interest. In particular, the SCHEER provides opinions on questions concerning emerging or newly identified health and environmental risks and on broad, complex or multidisciplinary issues that require a comprehensive assessment of risks to consumer safety or public health and related issues not covered by other European Union risk assessment bodies. Alternatively, a mandate could be given to ECHA. Under the Sustainable Chemicals Strategy and the 'one substance one assessment', ECHA will also be made responsible for all scientific aspects of all other chemicals legislation. By doing so, the EU can bring in added value to coordinate scientific support on sewage sludge management for the benefit of all EU-27 Member States.

9.2.2 Develop a technical guidance document

Risk assessments use existing physico-chemical and toxicological data of contaminants based on an agreed set of equations, assumptions and decisions that “transform” data into risk conclusions and mitigation measures. Therefore, choices of equations to model contaminants transfer in soils and exposure pathways (e.g. from sewage sludge to human organs), decisions on end-points for consideration (e.g. soil organisms, humans), scale of the assessment (e.g. local versus regional scale), and acceptable risk levels will largely impact upon the conclusions of any risk assessment. Before proceeding to any risk assessment, a clear framework and technical guidance that describes the methodology of the risk assessment should be established. Mostly, this involves a set of technical issues that need to be agreed amongst experts in risk assessment. Still, also decisions that are more political and intertwined with the policy option that will be taken forward need to be made. Examples of such decisions involve, for instance, a decision on the consideration of combined effects from chemical mixtures present in sewage sludge, degree of acceptable harm to the environment (e.g. absence of any risks versus no accumulation of contaminants in soils to levels above current background soil concentrations observed), or the consideration of temporal (thus no persistent) effects in soils. The development of such technical guidance could develop starting from guidance from the ECHA or tools available for the assessment of plant protection products (European Crop Protection, 2018; REACH R.16, 2016). Such guidance document could be developed in technical working groups that include the responsible risk assessment body, as well as Member States and relevant stakeholders. A best estimate for the development time for such guidance document would be 1–2 years, including consultations. Possibly, this process could run in parallel with the development of a first scientific opinion on environmental and health risks from a specific candidate material (see section 9.2.3).

9.2.3 Request scientific opinions based on risk assessment

The 2022 JRC risk screening assessment study identified a short list of organic contaminants that could be prioritised for an in-depth risk assessment (e.g. PFAS, PAH, polychlorinated paraffins). In addition, metals and microplastics are contaminants that remain highly relevant for assessment when aiming at the protection of soils, the environment and human health. Scientific opinions could be already be requested from a collaborating risk assessment body on the risks from these contaminants following an in-depth risk assessment, using the agreed technical guidance document (see section 9.2.2). The SCHEER produces, for instance, reports in response to a specific request. The timeframe to evaluate environmental and health risks from substances varies depending on the available data (e.g. physico-chemical and toxicological properties) from the substance, and other variables, but will likely take 6 months to 1 year of time per contaminant including a consultation period. In case additional data were to be required, a surplus period (with time dependent on the supplementary testing requirements) is needed. Under the Environmental Quality Standards Directive (2008/105/EC) SCHEER adopted more than 50 opinions in the period 2011–2022 (~5 opinions per year).

9.3 Develop international standards for contaminants of concern

In case a risk assessment indicates a need for risk mitigation measures in the form of limit values for certain pollutants in sewage sludge applied to land, harmonised standards to measure the concentration of these contaminants in sludge and receiving soils could be developed, if not yet available. The availability of standards may help to show compliance with law (e.g. revised SSD). In the EU, only Standards developed by CEN (European Committee for Standardization), CENELEC (European Committee for Electrotechnical Standardization) and ETSI (the European Telecommunications Standards Institute) are recognised as 'European Standards'. *Harmonised Standards*' are ‘*European Standards*’ adopted, upon a request made by the Commission, for the application of Union harmonisation legislation. Manufacturers, other economic operators, or conformity assessment bodies can use harmonised standards to demonstrate that products, services, or processes comply with relevant EU legislation. Still, harmonised standards maintain their status of voluntary application. At present, sewage sludge-specific harmonised standards exist for dry matter content (EN 15934), metals (EN 16170, EN 16171, EN 16173), biological pathogens (CEN/TR 16193 for *Escherichia coli*, CEN/TR 15214 for *Escherichia coli*, CEN/TR 15215 for *Salmonella* spp.), macroscopic impurities (CEN/TS 16202), and selected organic contaminants (polyaromatic hydrocarbons (CEN/TS 16181, EN 15527), dioxins and furans and dioxin-like polychlorinated biphenyls (CEN/TS 16190)). However, no harmonised standards are available to measure other organic contaminants in sewage sludge (e.g. per-and polyfluoroalkyl substances, polychlorinated paraffins, microplastics). If necessary, the Commission could initiate a standardisation request to develop harmonised standards. Based on the mandate to develop harmonised standards for contaminants in EU Fertilising Products, the timeframe from request to the publication of the standards typically involves 1.5 to 5 years.

9.4 Collect stakeholder feedback on the evidence base laid down in this JRC report

Consulting stakeholders is an important instrument to collect information for evidence-based policymaking. Their views, practical experience and data will help deliver higher quality and more credible policy initiatives and evaluations. Further inputs on the outlined problem definition, policy objectives and alternative policy options in this JRC report could be collected by the Commission. The relevant stakeholders to be consulted are in first instance all stakeholder affected by the problems identified and involved in the management of sewage sludge (see section 4.1.5).

10 Conclusion and preliminary comparison of policies

This feasibility study indicated that, without any further regulatory action, main problems in relation to environmental and health protection and resource inefficiency will continue to persist. Two policy options have been proposed to tackle the core drivers of this problem that mainly rely on monitoring and control of sewage sludge recycled on agricultural land (PO1) and the transformation of sewage sludge into EU fertilising products (PO2). Both options involve different sub-options that impose less stringent requirements for sewage sludge originating from waste water treatment plants of a different size.

In Table 3, an overview of the different impacts of the two considered policy options is provided, grouped by environmental (including health), economic, and social impact categories. All values and indications are relative to the baseline (positive value mean more emissions, more costs etc. than in the baseline) and refer to the EU-27 and the policy specification with a 50k p.e. WWTP capacity threshold. Whenever possible, quantitative values are provided, and otherwise qualitative evaluation. Environmental impacts are shown in their natural units and in monetary terms (conversion by using shadow prices).

As can be seen from the table, in terms of environmental and health impacts, the two options do not differ significantly. While PO2 would achieve a higher reduction of contaminants and a higher potential for the recovery of elements such as Fe, Al, Ca, Mg, this comes at the price of a higher loss of nitrogen and organic matter, as well as an overall lower amount of recovered phosphorus due to some losses in the technical recovery process. On the other hand, the P returned to agricultural is more available to plants and crops in the short-term, enabling a more controlled use that may further reduce nutrient losses from sewage sludge.

In terms of economic costs, the higher ambition of PO2 also implies sensibly higher investment and operational cost impacts, almost by a factor two. Only for administrative costs, PO1 shows a higher financial burden.

For the other economic and socio-economic impacts, the comparison results to be again rather balanced, since none of the two options is consistently superior to the other.

Table 3. Comparison of the different impacts of policy options 1 and 2, expressed relative to baseline.

Impacts		Policy Option 1: Control and monitoring of agricultural use		Policy Option 2: mandatory transformation into fertiliser according to the Fertiliser Product Regulation.	
Category	Impact	Value (quantitative or qualitative)	Comment	Value (quantitative or qualitative)	Comment
Environ-mental & Health	Contamination	Improvement [+]	Monitoring and control of sewage sludge quality returned to the environment improve environmental and health protection. The total volumes of sewage sludge used for landspreading and recycling is in general lower than the baseline (uncertainties apply)	Strong improvement [++]	A higher share of the sewage sludge is incinerated, resulting in strong reductions in contaminants present in sewage sludge-derived materials used as fertilisers. Organic pollutants and microplastics: Complete destruction through incineration. Metals: Innovative P-recovery technologies allow the targeted depletion of metals

Environ-mental	P recovery (total P load)	+28 kt yr^{-1}r	P-recovery is similar to higher than for PO2, as sewage sludge is applied on land for recycling direct, without losses from technical P-recovery.	+24 kt yr^{-1}	Slightly lower to similar total amounts of P-recovered than in PO1. However, higher P use efficiency is indicated due to the transformation of P from sewage sludge matrix into mineral fertilisers (higher potential to substitute mineral P-fertilisers).
	Nutrient losses	Certain improvement [+]	The partial transformation and substitution of sewage sludge into mineral fertilisers offers possibilities to better align nutrient supply with plant demand. Additional requirements to comply with good agricultural practices will further reduce nutrient losses relative to the baseline.	Strong improvement [++]	A greater degree of transformation and substitution of sewage sludge into mineral fertilisers compared to PO1.
	N, C recovery	Neutral to negative [-]	The possible partial transformation and substitution of sewage sludge into mineral fertilisers removes organic and nitrogen. The outcome is dependent on sludge quality and share of sludge compliant with quality criteria for sludge landspreading	Strongly negative [--]	Compared to PO1, a greater amount of N and C is lost for recycling, as a results of additional sewage sludge is incinerated.
	Recovery of other elements (e.g. micronutrients)	Positive [+]		Very positive [++]	Innovative P-recovery technologies allow the recovery of further elements from SSA as e.g. Fe, Al, Ca, Mg

	Methane emissions	-0.0126 Mt CH_4 yr^{-1}	Methane emissions from sewage sludge management are generally low in both POs.	-0.0126 Mt CH_4 yr^{-1}	Methane emissions from sewage sludge management are generally low in both POs.
Economic	Total investment costs	3 954 M€		7 093 M€	Distinct higher investment cost for PO2, due to the greater capacity of incinerators and P-recovery plants needed.
	Annual sewage sludge management costs	+1 049 M€ yr^{-1}	+156% relative to baseline	+1 933 M€ yr^{-1}	+203% relative to baseline
	Sewage sludge disposal costs (per ton) for WWTP	+133 € t DM^{-1}	+56% relative to baseline	+245 € t DM^{-1}	+103% relative to baseline
	Administrative costs	1.9–4.1 M€ yr^{-1}	Taking and analysing samples, reporting	0.85 M€ yr^{-1}	Taking and analysing samples, reporting
	Competitiveness	Neutral [0]		Somewhat negative [-]	Higher cost for fertilisers for farmers deprived of sludge application
	Innovation incentives	Neutral [0]		Slightly positive [0+]	Strong for incineration-based P-recovery, risk of negligence of other P recycling approaches
	P import dependence	-0.04 Mt yr^{-1}	- 3.6% of all P mineral fertiliser imports	-0.02 Mt yr^{-1}	- 2.9% of all P mineral fertiliser imports
Social	Employment	3 610 jobs	Direct employment gain in P-recovery value chain	4 780 jobs	Direct employment gain in P-recovery value chain. Additional negative impact from higher fertiliser costs cannot be excluded.
	Distribution of costs	Neutral [0]		Some negative impacts possible [0-]	Member States with already high share of sewage sludge incineration (e.g. BE,

					GR, NL) will have comparatively a lower cost to comply with mandatory incineration and P-recovery. No compliance costs for AT and DE, which already have legislation that mandates incineration and P-recovery.

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List of abbreviations and definitions

ºC	degree celsius
AD	Anaerobic digestion
C	carbon
CapEx	capital expenditures
COD	chemical oxygen demand
DAP	Di-ammonium-phosphate
DCP	Di-calcium-phosphate
d.l.	detection limit
DM	dry matter
FGT	flue gas treatment residues
FPR	Product Function Categories
ha	hectare
HTC	Hydrothermal carbonisation
JRC	Joint Research Center
KD	partition coefficients
kg	kilogram
Kleach	leaching factor (Kleach)
Ksw	soil-water adsorption coefficient
kt	kilo ton
kWh	kilowatt hour
L	liter
m^2	square meter
m^3	cubic meter
MAP	Mono-ammonium-phosphate
man*h	man-hour
M€	million euro
mg	milligram
Mg	megagramm
MJ	megajoule
Mt	million t
N	nitrogen
NAC	Calcium-ammonium-nitrate
NPK	Multicomponent fertiliser with different composition of the nutrients nitrogen, phosphorus and potassium
UWWTD	Urban Waste Water Treatment Directive (91/271/EEC)
UWWTD	Urban Waste Water Treatment Plant
OPEX	operational expenditures
P	phosphorus

P_4	phosphorus in its purest form (e.g. white phosphorus)
P_2O_5	phosphorpentoxide
PO	policy option
P-acid	phosphoric acid
PAH	polyaromatic hydrocarbons
PCDD/F	polychlorinated dibenzofuran and dioxins
PCP	Precipitated calcium phosphate,
PFAS	long-chain per- and polyfluoroalkyl substances
p.e.	population equivalent
PR	phosphate rock
SC	size category (WWTP)
SS	sewage sludge
SSA	sewage sludge ash
SSD	Sewage Sludge Directive
SSP	single-superphosphate
t	ton
TRL	technology readiness level
TSP	triple-superphosphate
WWTP	wastewater treatment plant
WtE	Waste to Energy
yr	year

Co-incineration plant means any incineration plant whose main purpose is the generation of energy or production of material products and which uses wastes as for example sewage sludge as a regular or additional fuel (e.g. coal- or cement industry) or in which waste is thermally treated for the purpose of disposal (e.g. waste to energy plant).

Mono-incineration means that the sewage sludge is incinerated separately, not mixed with e.g. municipal solid waste or other waste. With regard to P-recycling the goal is to produce sewage sludge ash contains high phosphorus levels and low content of impurities.

P recovery means that the nutrients within the sewage sludge are brought to agriculture or other land either directly or after e.g. composting.

Technical P-recovery means that technological approaches are applied on sewage sludge and/or sewage sludge ash to convert the input material into a mineral fertiliser considered within the Fertilising Products Regulation (Regulation 2019/1009).

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11 Annex: Supplementary information – status quo, baseline and policy options

11.1 Methodology

11.1.1 Status quo

11.1.1.1 Sewage sludge quantity

According to Eurostat (Eurostat, 2022), in total 8.1 Mt of sludge were produced in the EU-27 in 2019. Although many countries have improved the connection rate of the population to WWTP and improved wastewater treatment through additional treatment steps since 2007 significantly, the sewage sludge volume in the EU-27 decreased by 4% until 2019 (from 8.5 to 8.1 Mt).

The reasons are different and for some countries can be the results of a demographic change in population since 1995 (e.g. -10-30% in the Baltic States; -up to 20% in RO and BG (Eurostat, 2022). Exemplarily in DE, where the population grew by +2%, the change from lime to polymer for dewatering of the sewage sludge but also the improved industrial waste water treatment with reduced organic load to the sewer resulted in a steady decrease of sludge volume in Germany from 1995 on (-20%). Another factor have been the ongoing implementation of anaerobic digesters (+40% biogas since 2007 (DESTATIS, 2014)). This is quite remarkable, as these factor compensated the increasing connection rate as well as the increasing cleaning performance of the WWTP which increased in the same time (from 72% to 93% tertiary treatment, (EEA, 2021b)). Decreases in sludge volume since 2007 in the same order of magnitude can be seen for countries as e.g. AT, EE, IR, GR, HR, IT, LI, LU and PL. However, the reported sludge volumes stagnate or only slightly decreased in the recent years for these countries.

11.1.1.2 Nutrient content

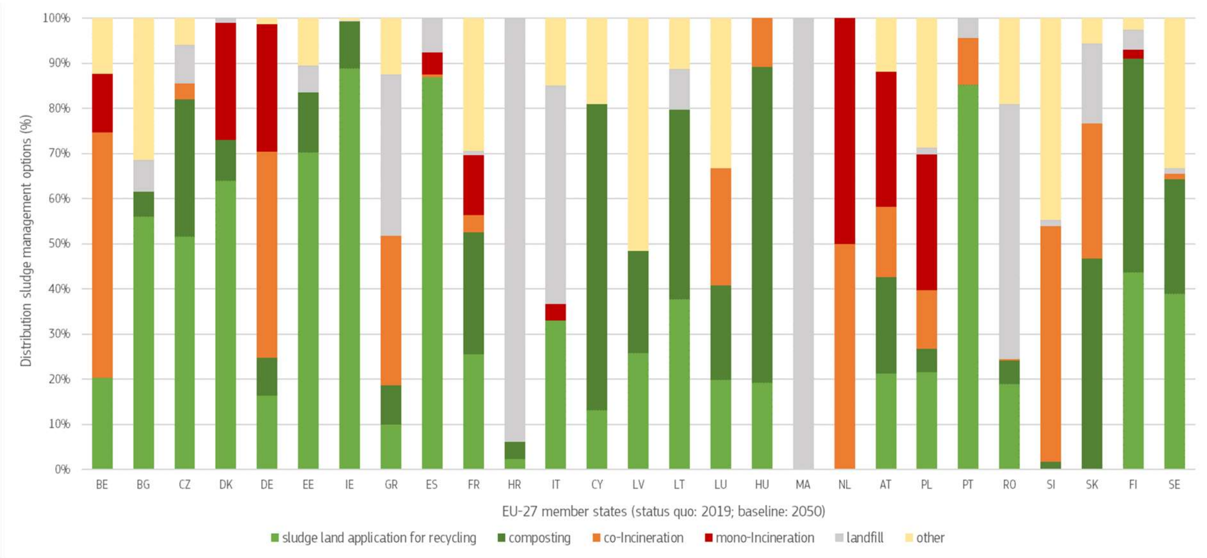
Based on nutrient contents documented by MS in their implementation report and an assumed carbon content of 30%, sewage sludge contains about 2.5 Mt C, 0.26 Mt N, and 0.15 Mt P.

Over the last years, nutrient content in sewage sludge generally increased. The German UBA reports (Roskosch et al., 2018) both, increasing N (+13%) and increasing P (+12%) concentration in German sludge since 2002. In The Netherlands, SNB, the operator of a mono-incineration plant incineration ¼ of the Dutch sewage sludge reports increasing N (+31%) and increasing organic (OM) matter content (+12 %) since 2004. The second mono-incinerator HVC observed a 13 % increase in OM since 1992. The P content, analysed in the SSA of both incineration remained constant within the last 12 years (Gerritsen et al., 2021)). The Swedish EPA reports a slight decrease in P (-4%) and a clear increase in N (+34%) since 1995. The data also reveal, that the nutrient concentration in sewage sludge from WWPT with greater treatment capacity (>100k p.e.) is significantly higher than from smaller WWTP (SCB, 2020). For Italy, (Mininni et al., 2019) describe similar findings with a clear increase in N (+14%) and a slight decrease of P (-3%) since 2000.

11.1.1.3 Distribution of sludge management options 2019

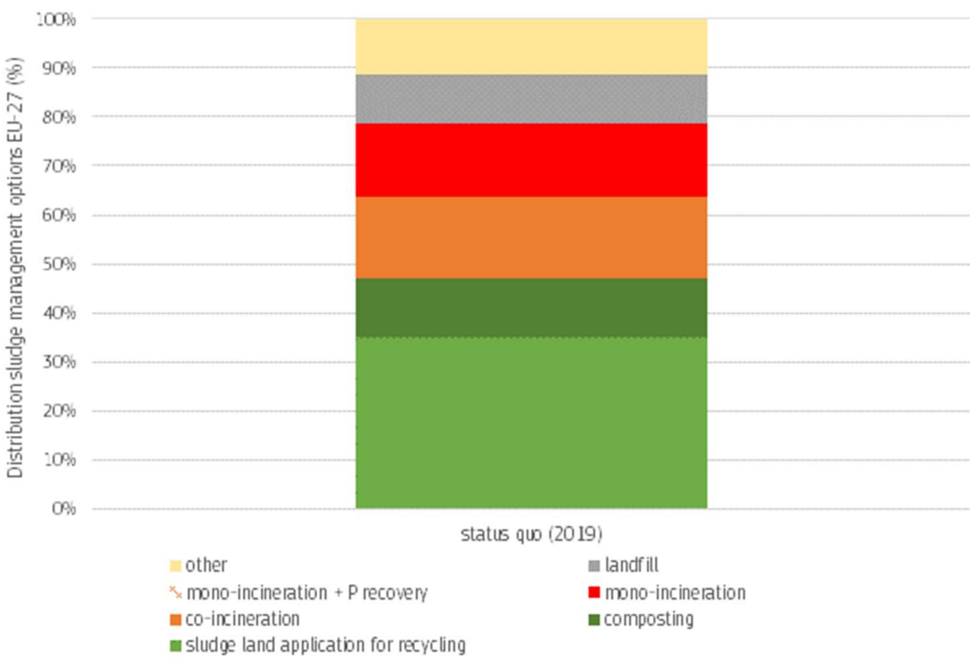
The percentual distribution of sewage sludge for the different sewage sludge management options is based on the EUROSTAT data on sewage sludge disposal and disposal ((Eurostat, 2022); Figure 36). An in-depth market analysis of the current mono-incineration capacities allows a detailed distinction between co-incineration (16.4%) and mono-incineration (15.3%, Table 39).

Figure 36. Percentual distribution of sewage sludge application, utilisation and disposal for 2019 in the EU-27.



In the absence of hard data on the amount of sewage sludge currently used following composting and classified by Member States as “other use”, it is assumed that 50% of composted sewage sludge is used in agriculture. The other 50% of composted sewage sludge is used for “other” purposes. Overall, it is estimated that around 41% of the sewage sludge, including untreated, lime stabilised, composted and anaerobically digested sewage sludge, is utilized on agricultural land, recycling about 1.04 Mt C, 0.11 Mt N, and 0.06 Mt P. From the remaining sewage sludge production, around 10% was landfilled and 31.4% was incinerated (**Figure 37**). 17.5% of sewage sludge was used for other purposes and the use of this is unclear, with landscaping and (temporary storage) being important routes in many MS (N. Anderson et al., 2021).

Figure 37. Percentual distribution of sewage sludge use 2019 in the EU-27.



11.1.1.4 Methane emissions

In 2019, 10.1% of total sewage sludge production or 0.83 Mt of sewage sludge dry matter was landfilled. Under anaerobic conditions, the methane formation potential of untreated raw sewage sludge is about 200-250 m³ CH_4 t DM^{-1} (Alberici et al., 2022; Grosser, 2018; Nielfa et al., 2015); Grosser, 2018; Nielfa et al., 2015), meaning that the landfilled sewage sludge has a **theoretical methane formation potential of 0.078-0.097 Mt CH_4 yr^{-1}**.

In paragraph 4.2, the landfill directive indicates that “landfill gas shall be collected from all landfills receiving biodegradable waste and the landfill gas must be treated and used. If the gas collected cannot be used to produce energy, it must be flared”. According to (EEA, 2021), 84% of the EU-27 landfills are managed landfill sites, indicating that landfill gas is at least collected and flared. Gas collection systems of well monitored landfills with liners are capable to collect in average 80 % of the landfill gas (over a 20 year life span (EASETECH, (Clavreul et al., 2014). 99 % of the collected methane is converted with subsequent flaring of the landfill gas. Caicedo-Concha et al., 2021 highlights, that the collection and combustion of landfill gas in flares reduces the global warming potential by up to 60%. This again significantly reduces the impact of sewage sludge on the total methane emissions.

Considering the current status of managed and unmanaged landfills results in CH_4 emissions of 0.051–0.063 Mt yr^{-1}. That corresponds to only 0.3–0.4% of the total EU-27 methane emissions (15.2 Mt CH_4 yr^{-1}). In comparison, the total methane emissions from the EU-27 were 15.2 Mt in 2019, whereas 4.2 Mt originate from the waste sector (EEA, 2021).

To understand the dimension right: In 2019, 52 Mt of municipal solid waste was landfilled, containing around 17.7 Mt of biodegradable organic waste (34% of bio-waste in MSW; (EEA, 2020)), compared to 0.83 Mt sewage sludge. The biogenic fraction of MSW has a similar methane gas building potential as sewage sludge (Tonini et al., 2016).

11.1.1.5 Annual cost for sewage sludge management

Annual cost are calculated by multiplying the total amount of sewage sludge associated to the different sewage sludge management options for each MS country (**Figure 36**) with the cost presented in Table 44. Possible savings for e.g. nutrients applied on land or revenues from energy recovery from sewage sludge incineration or revenues for the recovered materials from P recovery plants are not taken into account. Main argument for that is, that waste water treatment plant (WWTP) operators have to pay the cost given in Table 44. The beneficiaries of savings and revenues are the farmers applying the sewage sludge or the operators of the incineration- and/or recovery plants, however, these savings and revenues are already included in the cost the WWTP operator has to pay, that the sewage sludge is used on land or incinerated. Especially for P-recovery plants, revenues for the outputs are associated with high uncertainties, as these technologies have hardly been implemented on a large scale and no market price for the outputs is known yet.

Based on these assumptions, the current cost for the sewage sludge management in the EU-27 are estimated at **1 829 M€ yr^{-1}.**

To make costs of different sludge treatment options comparable, the costs are usually expressed per ton of sewage sludge dry matter. In order to facilitate the comparison of the status quo, the baseline and the policy options at a glance, the average costs for the treatment of one ton of dry sewage sludge are calculated. For this purpose, the annual cost (1 829 M€) are divided by the total amount of sewage sludge produced in the year 2019 (8 124 Mt). As a results, for the status quo, the average cost to recycle, treat or dispose 1 t of sewage sludge dry matter is **225 €.**

11.1.2 Baseline

11.1.2.1 Sewage sludge quantity and nutrient content

The assessment to estimate future sewage sludge management until 2050, the year of projection used in this assessment, considers demographic evolution, impacts of main sewage sludge treatment routes, implications from the implementation of the current and future urban waste water treatment directive (91/271/EEC; under review), EU goals and strategies (e.g. methane strategy, landfill directive), and impacts from national legislation and policies on sewage sludge. The following assumptions were taken to predict future sewage sludge volume, nutrient load in sewage sludge and cost.

Demographic evolution

Eurostat's baseline projections suggest that the EU-27 will remain largely steady until the reference year 2050 (from 447 million in 2020 to 441 million in 2050; (EUROSTAT, 2022)), however with strong fluctuations in particular countries. Population growth is expected to occur in predominantly urban region, and such regions will become the almost half (46%) of the EU population (Eurostat, 2016).

Implications from current and future Urban Waste Water Treatment Directive (UWWTD, under revision)

It is assumed that the UWWTD requirements are fulfilled and all generated urban waste water is collected and receives treatment in line with the UWWTD provisions. (WISE, 2022) indicates the distance to the UWWTD targets for each MS for connection rate, biological treatment and biological treatment with additional N and P removal in million p.e. (see MS country profile Table 4). The distance to target in p.e. is multiplied with waste water treatment specific parameters to evaluate additional sewage sludge volume and the transfer of carbon and nutrients into the sewage sludge (

Table **5** and Table 6).

Moving forward to EU energy neutrality for the sector until 2040, the revision of the UWWTD includes energy audits for all WWTP >10k p.e. to help them to better understand the potential savings achievable (EC, 2022b). The economic limit for a process conversion from simultaneous aerobic to anaerobic sewage sludge stabilisation is approximately 20–30k p.e. Disproportionate high investment costs are the main factor that anaerobic digestion (AD) at smaller WWTP cannot be operated economically. Studies from AT (Arabel Amann et al., 2021) and DE indicate, that a major part of WWTP >50k p.e. operate with AD, but also smaller WWTP use anaerobic digestion for energy recovery and sewage sludge stabilisation. Taking into account the fact that AD is not state of the art in many MS, the conservative assumption is, that EU-27 WWTP >50k p.e. will have anaerobic digestion implemented by 2050 (Table 4). Anaerobic digestion reduces raw sludge volume by 45%, thus to a greater extent than aerobic stabilisation (with approx. sewage sludge volume reductions of 20%). Besides volume reduction, anaerobic digestion has relevant impacts on the transfer of C and N into the sewage sludge, as organic C partially transforms into a gaseous phase (CO_2 and CH_4) and organic N transforms into ammonium, which is transferred into the sewage sludge water after dewatering and removed during the main aeration process. Anaerobic digestion has no impact on the P load in the sewage sludge. Transfer coefficient for C, N and P due to AD are given in Table 6 (Wett and Alex, 2003; Ghimire et al., 2021; Saud et al., 2021).

The following calculations steps are performed to determine the expected sewage sludge volume (Mt) in 2050:

$$\begin{aligned}
&\text{Step 1: Sewage sludge volume 2050 (Mt; without AD)}\\
&\qquad = \big(\text{sewage sludge volume 2019 (Mt)} * \text{population development (\%)}\\
&\qquad * \text{distance to target connection rate (\%)}\big)\\
&\qquad + \big(\text{distances to target} - \text{biological treatment (million PE)}\\
&\qquad * \text{add. sewage sludge volume with biological step (\%)}\big) + (\text{distance to target}\\
&\qquad - \text{biological treatment} + \text{NP removal (million PE)}\\
&\qquad * \text{add. sewage sludge volume with biological step and NP removal (\%)})
\end{aligned}$$

$$\begin{aligned}
&\text{Step 2: Sewage sludge volume reduction (Mt) with AD for WWTP} > \text{50k p.e.}\\
&\qquad = \big(\text{sewage sludge volume 2050 (without AD)(Mt)} * \text{sewage sludge treated in WWTP}\\
&\qquad > \text{50k p.e. (\%)} * \text{share of WWTP}\\
&\qquad > \text{50k p.e. without AD} * \text{reduction sewage sludge volume (\%)}\big)
\end{aligned}$$

$$\begin{aligned}
&\text{Step 3: Sewage sludge volume 2050 (Mt; with AD)}\\
&\qquad = \text{Sewage sludge volume 2050 (Mt; without AD)}\\
&\qquad - \text{sewage sludge volume reduction (Mt) with AD for WWTP} > \text{50k p.e.})
\end{aligned}$$

The expected nutrient load (Mt) in sewage sludge in 2050 is calculated as followed:

$$\begin{aligned}
&\text{Step 4: Nutrient load in sewage sludge 2050 (Mt; without AD)}\\
&\qquad = (\text{nutrient load sewage sludge 2019 (Mt)} * \text{population development (\%)}\\
&\qquad * \text{distance to target connection rate (\%)}) + (\text{distance to target}\\
&\qquad - \text{biological treatment (million PE)} * \text{annual pollutant load (kg C/N/P per inh yr} - 1)\\
&\qquad * (\text{removal efficiency secondary treatment (\%)}\\
&\qquad - \text{removal efficiency primary treatment (\%)}) + (\text{distance to target}\\
&\qquad - \text{biological treatment} + \text{NP removal (million PE)}\\
&\qquad * \text{annual pollutant load (kg C/N/P per inh yr} - 1)\\
&\qquad * (\text{removal efficiency tertiary treatment (\%)}\\
&\qquad - \text{removal efficiency secondary treatment (\%)}))
\end{aligned}$$

$$\begin{aligned}
&\text{Step 5: Nutrient load in sewage sludge 2050 (Mt, with AD)}\\
&\qquad = \text{Nutrient load in sewage sludge 2050 without AD (Mt)} * \text{loss of C, N, P due to AD (\%)}
\end{aligned}$$

Table 4. Assumptions to calculate the future sewage sludge production and nutrient within the sludge (baseline).

country	population development	UWWTD requirements distance to target (mio. p.e.)			anaerobic digestion		landfill
		connection rate	biological treatment	biological treatment + N/P removal	waste water to WWTP >50k p.e. (%)	WWTP >50k p.e. without AD (%)	SS to landfill 2019 (%)
BE	4%	0.01	0.03	0.06	58%	40%	0%
BG	-19%	0.38	1.20	1.19	62%	81%	7%

CZ	-1%	0.00	0.01	1.55	67%	0%	9%
DK	5%	0.00	0.02	0.07	72%	27%	1%
DE	0%	0.00	0.12	0.11	67%	0%	0%
EE	-5%	0.00	0.00	0.01	74%	43%	6%
IE	27%	0.00	2.51	2.77	69%	63%	0%
GR	-11%	0.00	0.40	0.04	76%	64%	36%
ES	5%	0.27	6.03	3.92	81%	77%	8%
FR	4%	0.00	4.62	2.61	65%	88%	1%
HR	-17%	0.53	2.63	2.12	67%	57%	94%
IT	-4%	0.57	9.12	2.24	71%	78%	48%
CY	19%	0.15	0.15	0.04	93%	100%	0%
LV	-27%	0.00	0.00	0.01	71%	67%	0%
LT	-23%	0.00	0.00	0.00	75%	0%	9%
LU	25%	0.00	0.00	0.00	67%	0%	0%
HU	-5%	0.43	1.43	1.11	68%	50%	0%
MA	35%	0.00	0.91	0.13	94%	67%	100%
NL	5%	0.00	0.00	0.00	79%	34%	0%
AT	5%	0.00	0.00	0.00	68%	0%	0%
PL	-10%	0.12	0.45	1.31	71%	49%	2%
PT	-9%	0.01	0.73	0.24	72%	42%	4%
RO	-20%	7.16	12.87	7.72	62%	81%	56%
SI	-2%	0.01	0.35	0.31	59%	46%	1%
SK	-6%	0.02	0.04	0.06	66%	48%	18%
FI	-4%	0.00	0.05	0.11	76%	42%	5%
SE	20%	0.00	0.10	0.34	69%	0%	1%
Source	(Eurostat, 2016).	(WISE, 2022)			(EEA, 2022)	(EBA, 2021)	(Eurostat, 2022)

For MS with no available data on number of installed biogas plants (BG, CY, HU, MA, PT, RO, SI, SK), it is assumed that only WWTP greater 100k p.e. have AD installed.

Table 5. Assumptions for sewage sludge production and removal efficiency of nutrients as a consequence of implemented waste water treatment steps.

Treatment steps	**Sewage sludge production** (kg DM person^{-1} yr^{-1}) (Imhoff et al., 2018)		**Nutrient removal efficiency** (%) (latest version of the revised UWWTD)		
	not stabilised	**stabilised**	**C**	**N**	**P**
No treatment	0	0	0%	0%	0%
Primary treatment	16.4	9.1	50%	25%	30%
Secondary treatment	12.8	7.1	94%	55%	60%
Tertiary treatment	3.7	2.0	96%	80%	90%

Due to AD, about 50% of organic carbon is transferred into biogas. Nitrogen is partly dissolved into ammonium, which re-enters the WWTP after dewatering and is then removed by nitrification/denitrification. The amount of N in sewage sludge reduces by 50%. In comparison, P is quite stable in the AD process, and if P dissolves during AD, it will be transferred again into the sewage sludge during the waste water treatment (Table 6).

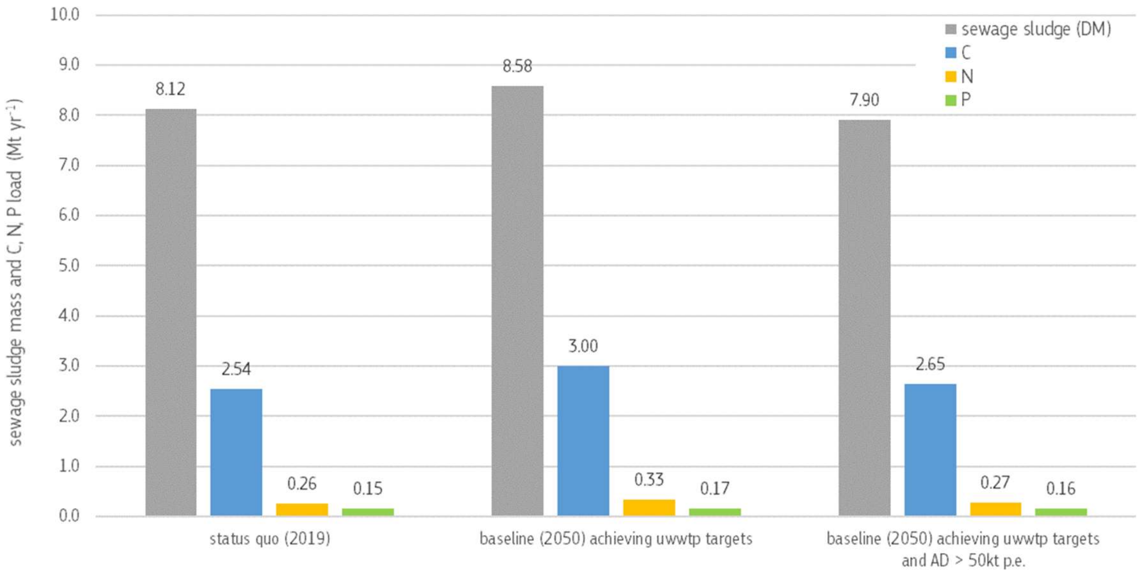
Table 6. Assumptions for transfer of carbon and nutrients into sewage sludge with or without anaerobic digestion.

Treatment steps	**Carbon and nutrient transfer into sewage sludge (%)**		
	P	**N**	**K**
Without anaerobic digestion	0.70	0.40	0.90
With anaerobic digestion	0.33	0.20	0.90
Assumed loss	50%	50%	0%

Table 7. Agronomic efficiencies of organic and mineral fertiliser material.

Organic fertiliser material	**P**	**N**	**K**
Sewage sludge (AD, wet, dewatered)	0.55	0.65	0.50
Compost	0.55	0.65	0.50
Mineral fertiliser material	**P**	**N**	**K**
Recovered secondary P fertiliser	1.00	-	1.00

Figure 38. Sewage sludge mass and C, N, P load in the status quo (2019) and baseline (2050) (Mt yr^{-1}) considering the baseline without and with the implementation of AD on WWTP > 50k p.e.



11.1.2.2 *Distribution of sewage sludge management options*

National legal framework on mandatory technical P recovery

DE and AT will have legal nutrient recovery frameworks in place which target 70% and 85% of waste water burden P (**Table 53**). These countries have in common, that the recovery of P will be mandatory for municipal WWTP exceeding a defined treatment capacity, that a technical P recovery is possible from sewage sludge or mono-incineration residues and that a percentual recovery rate is defined for the recovery. In countries as e.g. DK and SE but also through trans regional programs (HELCOM), strategies and measures for nutrient recovery have been discussed, developed, and presented, but without any legal commitment. At the present, it is unclear to what extent this will result in binding recycling targets. For the baseline it is therefore assumed, that only AT and DE have mandatory technical nutrient recovery in force. In countries were a certain percentage of sewage sludge undergoes mono-incinerator (BE, DK, ES, FR, FI, NL, PO) it is assumed, that P is recovered from the SSA by 2050 due to the increased cost-effectiveness of phosphorus recovery from mono-incineration ashes (efficiency of P-recovery technologies: 90%; fertilising efficiency of recovered P fertilisers: 100% mineral P substitution).

EU goals and strategies influencing sewage sludge management

The landfilling of biodegradable waste is increasingly being phased out in EU legislation. Based on further enforcement of waste hierarchy and the ambitions of the EU Methane Strategy (European Commission, 2020), it is assumed that landfilling of sewage sludge will no longer be carried out in countries with already low landfill ratio (<5%; DK, IE, FR, PO, PT, SI, FI, SE). In countries with a landfilling ration >5% (BG, CZ, EE, ES, GR, HR, IT, LT, MA, RO, SK), landfilling will be reduced by 50% in 2050 compared (Table 4). In countries with existing incineration sector, sewage sludge will be thermally treated (DK, GR, PO). In other countries, sewage sludge will be used on agricultural land (50%) or will be composted (50%). Countries with no sewage sludge to landfill remain unchanged.

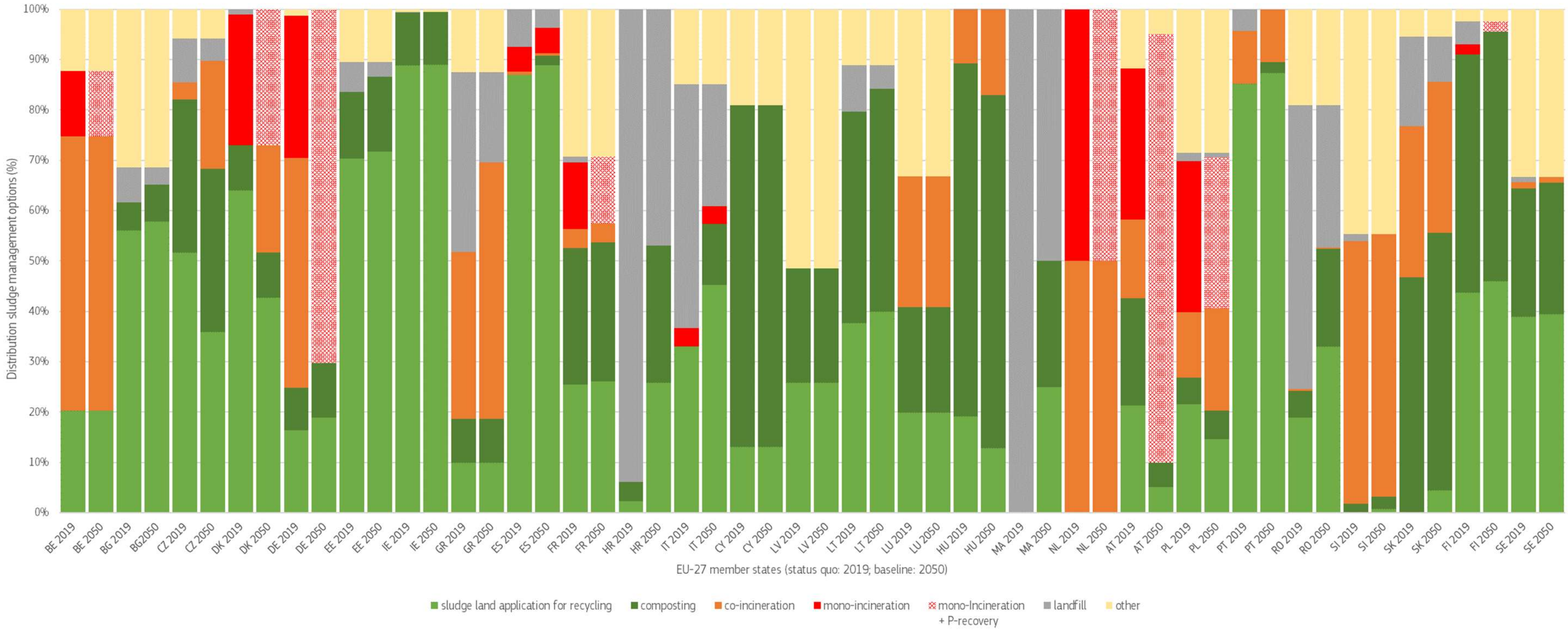
Reducing nutrient losses by 50% has been set out in the EU Biodiversity and Farm-to-Fork strategy. The latest implementation report of the Nitrates Directive indicates that MS with a high percentage of surface waters stations in eutrophic status include BE, CZ, DE, DK, EI, FI, HU, NL, and PL. About half of these countries have effectively restricted sewage sludge application in the environment, amongst others with a view to further reduce nutrient inputs to agricultural land. However, CZ, DK, HU, PL and FI apply more than 65% of their sewage sludge in the environment, thereby contributing to nutrient losses as a share of the nutrients contained in sewage sludge will be lost to surface and groundwater. Therefore, it is assumed that the land application of sewage sludge in these MS will be reduced by one third (33%) relative to the present-day situation, but that instead, the sewage sludge will be disposed through incineration without P-recovery.

Sewage sludge management option ‘other’

For sewage sludge within the definition ‘other’ it is often unclear how sewage sludge is recycled, treated or disposed. It is assumed that sewage sludge classified as ‘other’ is neither used in a nutrient efficient way nor thermally treated. As no corresponding data are available, it is assumed that the share of sewage sludge classified as ‘other’ remains unchanged. In case of implementation of national legal frameworks to recover P, the share of sewage sludge to ‘other’ is reduced proportionally.

In addition to the assumptions considered above, the sewage sludge volume, the nutrient content and also the costs can be influenced by further factors as e.g. economic development, changes of industrial discharge, changes in storm water overflow and urban runoff management, and changes in sewage sludge dewatering methods resulting in lower sewage sludge volumes (e.g. change from liming and dewatering with presses to the use of polymers and dewatering with centrifuges). However, insufficient information on these factors is available on EU-27 and national level to enable a fact-based forecast. **Table 39** summarises the distribution for the different sewage sludge management options for 2019 and 2050.

Figure 39. Distribution of sewage sludge management option in EU-27 MS for status quo 2019 and the baseline 2050.



11.1.2.3 Methane emissions from landfills

In 2019, 10.1% of total sewage sludge production or 0.83 Mt of sewage sludge dry matter was landfilled. Under anaerobic conditions, the methane formation potential of untreated raw sewage sludge is about 200-250 m^3 CH_4 t DM^{-1} ((Nielfa et al., 2015; Grosser, 2018; Alberici et al., 2022), meaning that the landfilled sewage sludge has a theoretical methane formation potential of 0.078-0.097 Mt CH_4 yr^{-1}.

11.1.2.4 Annual cost for sewage sludge management

Annual cost are calculated by multiplying the total amount of sewage sludge associated to the different sewage sludge management options with the cost presented in Table 44. Possible savings for e.g. nutrients applied on land or energy recovery from sewage sludge incineration are not taken into account. Cost for the current sewage sludge management in the EU-27 are estimated at **1883 M€ yr^{-1}**.

Considering the predicted sewage sludge production of 8 124 Mt in 2050, the average cost to recycle, treat or dispose 1 t of sewage sludge dry matter is **238 €.**

11.1.3 Policy options

To assess the different policy options, the WWTP were classified in six different size categories (SC): SC 1 (<2k p.e.); SC 2 (2–20k p.e.); SC 3 (20–50k p.e.); SC 4 (50–100k p.e); SC 5 (100–500k p.e.); SC 6 (>500k p.e.). For all policy options four different scenario were applied considering the implementation of P recovery goals for WWTP exceeding one of the defined size categories:

- SC 6 (>500k p.e.)
- SC 5–6 (>100k p.e.)
- SC 4–6 (>50k p.e.)
- SC 3–6 (>20k p.e.)

Distribution of sewage sludge management options to WWTP size category

Unfortunately, no data is publicly available on the sewage sludge management routes for each single WWTP in the EU. Therefore, it is unknown, how WWTP of different sizes categories recycle, treat or dispose their sewage sludge. However, this level of granularity is crucial to assess the different possible policy options addressing different WWTP size categories. Studies from DE and AT indicate, that incineration is the major disposal route for larger WWTP in countries with an existing incineration infrastructure (Wagner et al., 2020; A. Amann et al., 2021). Furthermore, **Table 51** indicates, that mono-incineration plants are operating in capital and bigger EU cities were the waste water is treated in large centralised WWTP. For the assessment it is therefore assumed, that the sewage sludge from WWTP with the largest size category is primarily mono-incineration, followed by co-incineration.

The methodology of sewage sludge management route distribution to the size categories of the WWTP is explained exemplarily for the cases of Ireland with no sewage sludge incineration and most of sewage sludge is directly applied in agriculture and for Poland were already a high percentage of sewage sludge is incinerated, including also mono-incinerated. Poland is also one of those countries, for which it was assumed, that the currently mono-incinerated sewage sludge will undergo P-recovery by 2050 in the baseline.

- Ireland has the following basic sewage sludge management distribution: agriculture: 82.9 kt DM yr^{-1}; composting: 4.6 kt DM yr^{-1}; co-incineration: 0.0 kt DM yr^{-1}; mono-incineration: 0.0 kt DM yr^{-1}; other: 0.5 kt DM yr^{-1}; landfill: 3.9 kt DM yr^{-1}: total: 88.0 kt DM yr^{-1}.
- Poland has the following basic sewage sludge management distribution: agriculture: 84.1 kt DM yr^{-1}; composting: 13.7 kt DM yr^{-1}; co-incineration: 97.9 kt DM yr^{-1}; mono-incineration: 144.6 kt DM yr^{-1}; landfill: 3.9 kt DM yr^{-1}; other: 137.9 kt DM yr^{-1}: total: 482.1 kt DM yr^{-1}.

In a first step, it is assumed, that the sewage sludge from the largest WWTP (SC 6) is 100% mono-incinerated (**Table 8**). The remaining mono-incinerated sewage sludge (41.9 kt DM yr^{-1}) is associated to the next WWTP size category (SC 5). The difference between incinerated sewage sludge (co- and mono-incineration) in SC5 and the total sewage sludge in SC5 (165.5 kt DM yr^{-1}) is distributed between agriculture, landfill, and other in relation to their share on the total amount of these three options (total amount: 326.6 kt DM yr^{-1}, share of each management option: agriculture: 35%, landfill: 2%, other: 63%).

Table 8. Methodology for final sewage sludge management route distribution in the baseline depending on the size categories of the WWTP for the case Ireland and Poland.

p.e.	Ireland			Poland		
	kt DM yr^{-1}	**management route**	**kt DM yr^{-1}**	**kt DM yr^{-1}**	**management route**	**kt DM yr^{-1}**
SC 6 >500k	25.4	mono-inc	-	102.7	mono-inc	102.7
		co-inc	-		co-inc	-
		agriculture	23.9		agriculture	-
		landfill	-		landfill	-
		other	1.5		other	-
SC 5–6 >100k	25.5	mono-inc	-	165.5	mono-inc	41.9
		co-inc	-		co-inc	97.9
		agriculture	24.0		agriculture	9.0
		landfill	-		landfill	0.4
		other	1.5		other	16.2
SC 4–6 >50k	9.4	mono-inc	-	74.6	mono-inc	-
		co-inc	-		co-inc	-
		agriculture	8.9		agriculture	26.8
		landfill	-		landfill	1.2
		other	0.5		other	48.8
SC 3–6 >20k	13.2	mono-inc	-	63.5	mono-inc	-
		co-inc	-		co-inc	-
		agriculture	12.4		agriculture	22.3
		landfill	-		landfill	1.0
		other	0.8		other	40.2
SC 2–3 <20k	14.4	mono-inc	-	74.0	mono-inc	-
		co-inc	-		co-inc	-
		agriculture	13.6		agriculture	26.0
		landfill	-		landfill	1.2
		other	0.8		other	46.8
Sum	**88.0**	**sum**	**88.0**	**482.1**	**Sum**	**482.1**

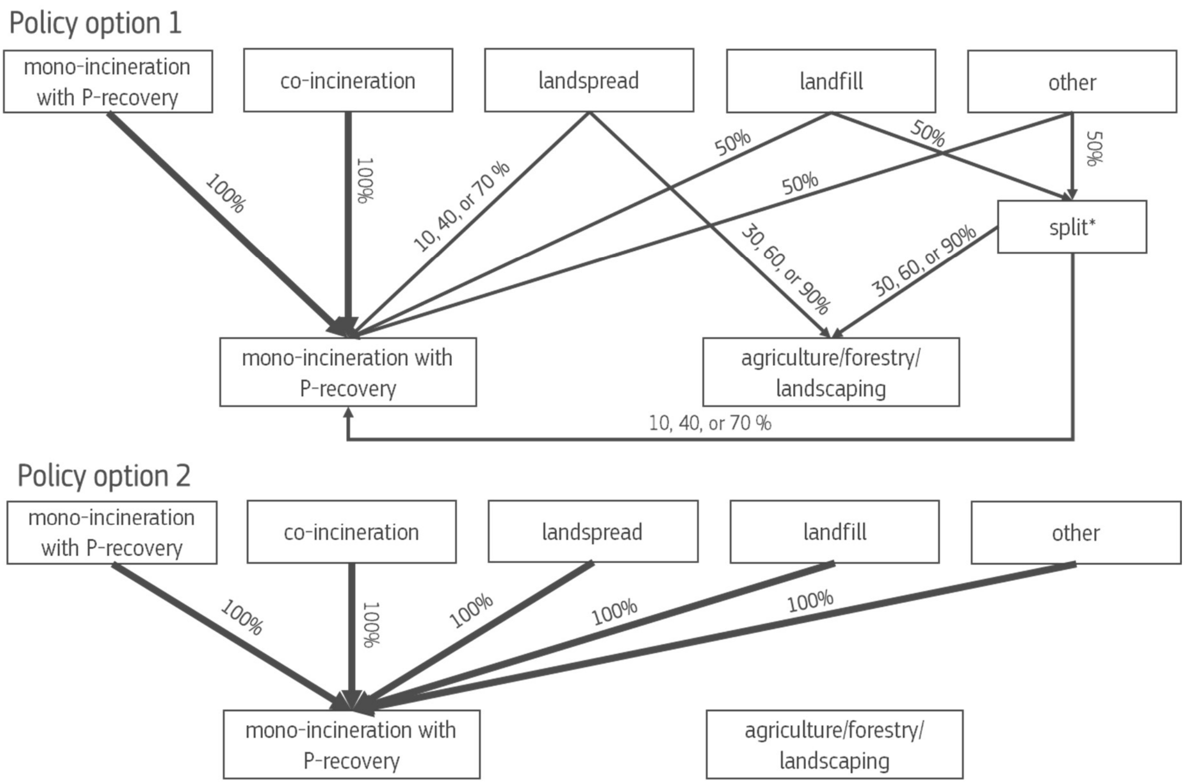
Re-routing of sewage sludge

For PO1, the following sub-options were defined for sewage sludge which is currently directly applied to agriculture and has to be transferred either to agriculture or mono-incineration with technical P-recovery in case of mandatory nutrient recycling:

- Sub-option 1: 90% agriculture, 10% mono-incineration with P-recovery (90/10)
- Sub-option 2: 60% agriculture, 40% mono-incineration with P-recovery (60/40)
- Sub-option 3: 30% agriculture, 70% mono-incineration with P-recovery (30/70)

Sewage sludge categorised as co-incineration is transferred 100% to mono-incineration with technical P-recovery. Sewage sludge categorised as landfill and 'other' (in this case compost and other use are combined) are transferred to agriculture (50%) and mono-incineration with technical P-recovery (50%). For the sewage sludge transferred to agriculture, the three sub-option will be applied in case of mandatory nutrient recycling. In comparison, for PO2, if mandatory technical P-recovery goals are set, 100% of the sewage sludge from all sewage sludge management options will be mono-incinerated with subsequent technical P-recovery (0/100) (Figure 40).

Figure 40. Assumed flows for the re-routing of sewage sludge under the different policy options for sewage sludge originating from waste water treatment plants exceeding a certain size.



The results of the re-distribution of sewage sludge are given in the following Table 9,

Table 10, Table 11 and Table 12 and serve as basis to perform the various assessment of the two policy options.

Table 9. Results of the sewage sludge distribution for the different (sub-) options for PO1 in the case of Ireland.

Options (agri/mono-inc + P-recovery)	**agri**	**comp**	**co-inc**	**mono-inc**	**mono-inc + P-rec.**	**other**	**landfill**	**control**
	kt DM yr^{-1}							
baseline	82.9	4.6	0.0	0.0	0.0	0.5	0.0	88.0
SC 6_90/10	81.1	3.3	0.0	0.0	3.2	0.3	0.0	88.0
SC 6_60/40	73.7	3.3	0.0	0.0	10.6	0.3	0.0	88.0
SC 6_30/70	66.3	3.3	0.0	0.0	18.0	0.3	0.0	88.0
SC 5–6_90/10	79.4	1.9	0.0	0.0	6.4	0.2	0.0	88.0
SC 5–6_60/40	64.5	1.9	0.0	0.0	21.3	0.2	0.0	88.0
SC 5–6_30/70	49.7	1.9	0.0	0.0	36.1	0.2	0.0	88.0
SC 4–6_90/10	78.8	1.4	0.0	0.0	7.6	0.1	0.0	88.0
SC 4–6_60/40	61.2	1.4	0.0	0.0	25.2	0.1	0.0	88.0
SC 4–6_30/70	43.6	1.4	0.0	0.0	42.8	0.1	0.0	88.0
SC 3–6_90/10	77.9	0.8	0.0	0.0	9.3	0.1	0.0	88.0
SC 3–6_60/40	56.4	0.8	0.0	0.0	30.7	0.1	0.0	88.0
SC 3–6_30/70	35.0	0.8	0.0	0.0	52.1	0.1	0.0	88.0

Table 10. Results of the sewage sludge distribution for the different (sub-) options for PO2 in the case of Ireland.

Options (agri/mono-inc + P-recovery)	**agri**	**comp**	**co-inc**	**mono-inc**	**mono-inc + P-rec.**	**other**	**landfill**	**control**
	kt DM yr^{-1}							
baseline	82.9	4.6	0.0	0.0	0.0	0.5	0.0	88.0
SC 6_0/100	58.9	3.3	0.0	0.0	25.4	0.3	0.0	88.0
SC 5–6_0/100	34.8	1.9	0.0	0.0	51.0	0.2	0.0	88.0
SC 4–6_0/100	26.0	1.4	0.0	0.0	60.4	0.1	0.0	88.0
SC 3–6_0/100	13.6	0.8	0.0	0.0	73.5	0.1	0.0	88.0

Table 11. Results of the sewage sludge distribution for the different (sub-) options for PO1 in the case of Poland.

Options (agri/mono-inc + P-recovery)	**agri**	**comp**	**co-inc**	**mono-inc**	**mono-inc + P-rec.**	**other**	**landfill**	**control**
	kt DM yr^{-1}							
baseline	144.6	13.7	97,9	0.0	144.6	137.9	3.9	482.1
SC 6_90/10	84.1	13.7	97.9	0.0	144.6	137.9	3.9	482.1
SC 6_60/40	84.1	13.7	97.9	0.0	144.6	137.9	3.9	482.1
SC 6_30/70	84.1	13.7	97.9	0.0	144.6	137.9	3.9	482.1
SC 5–6_90/10	90.7	12.3	0.0	0.0	252.5	123.1	3.4	482.1
SC 5–6_60/40	85.5	12.3	0.0	0.0	257.7	123.1	3.4	482.1
SC 5–6_30/70	80.3	12.3	0.0	0.0	262.9	123.1	3.4	482.1
SC 4–6_90/10	110.3	7.9	0.0	0.0	282.5	79.1	2.2	482.1
SC 4–6_60/40	89.7	7.9	0.0	0.0	303.2	79.1	2.2	482.1
SC 4–6_30/70	69.0	7.9	0.0	0.0	323.9	79.1	2.2	482.1
SC 3–6_90/10	126.7	4.2	0.0	0.0	307.4	42.6	1.2	482.1
SC 3–6_60/40	93.1	4.2	0.0	0.0	341.0	42.6	1.2	482.1
SC 3–6_30/70	59.5	4.2	0.0	0.0	374.5	42.6	1.2	482.1

Table 12. Results of the sewage sludge distribution for the different (sub-) options for PO2 in the case of Poland.

Options (agri/mono-inc + P-recovery)	**agri**	**comp**	**co-inc**	**mono-inc**	**mono-inc + P-rec.**	**other**	**landfill**	**control**
	kt DM yr^{-1}							
baseline	144.6	13.7	97,9	0.0	144.6	137.9	3.9	482.1
SC 6_0/100	84.1	13.7	97.9	0.0	144.6	137.9	3.9	482.1
SC 5–6_0/100	75.1	12.3	0.0	0.0	268.1	123.1	3.4	482.1
SC 4–6_0/100	48.3	7.9	0.0	0.0	344.6	79.1	2.2	482.1
SC 3–6_0/100	26.0	4.2	0.0	0.0	408.1	42.6	1,2	482.1

11.1.3.1 Investment cost and annual cost

Mandatory technical P-recovery will shift sewage sludge into mono-incineration resulting in necessary investments into the so far underdeveloped mono-incineration- but also recovery infrastructure. Based on the sewage sludge volume re-directed to mono-incineration, considering an ash content of 40% in sewage sludge, and the assumptions on plant capacity and related investment cost given in

Table 13, the total needed investment cost can be calculated. With the knowledge on the cost distribution of these two processes (section 12.2.5 and 12.2.6), it is possible to annualise the cost and distinguish between capital- and operational cost.

Table 13. Assumptions on annual treatment capacity and investment cost for a mono-incineration- and P-recovery plant.

Infrastructure	Capacity	Unit	Investment cost	Unit
Mono-incineration plant	30	kt DM yr^{-1}	1.700	€ t DM^{-1}
P-recovery plant	30	kt SSA yr^{-1}	20	M€

Annual cost are calculated by multiplying the total amount of sewage sludge newly associated to the sewage sludge management options with the cost presented in Table 44.

Table 14. Sewage sludge associated to the different sewage sludge management options for EU-27 MS (kt DM yr^{-1}).

sludge land application for recycling	BE	BG	CZ	DK	DE	EE	IE	GR	ES	FR	HR	IT	CY	LV	LT	LU	HU	MA	NL	AT	PL	PT	RO	SL	SK	FI	SE
SC 6_90/10	31.8	32.5	116.8	62.6	424.6	17.4	81.1	12.0	997.5	452.6	18.9	478.9	4.1	7.6	19.7	3.4	105.9	6.4	0.0	18.5	84.1	94.0	149.8	0.7	14.6	93.6	142.1
SC 6_60/40	31.8	28.2	113.3	62.6	424.6	17.4	73.7	12.0	893.9	435.1	15.5	428.9	4.1	6.1	18.1	3.4	102.1	6.4	0.0	18.5	84.1	93.0	134.9	0.7	14.6	88.1	129.9
SC 6_30/70	31.8	23.9	109.7	62.6	424.6	17.4	66.3	12.0	790.3	417.6	12.1	378.9	4.1	4.6	16.5	3.4	98.3	6.4	0.0	18.5	84.1	92.0	120.1	0.7	14.6	82.7	117.8
SC 5–6_90/10	31.8	34.2	121.4	62.5	424.6	17.6	79.4	15.4	969.7	508.1	19.5	516.2	5.3	8.3	21.2	3.4	115.1	9.8	0.0	18.5	90.7	89.4	170.2	0.7	16.6	96.4	152.6
SC 5–6_60/40	31.8	26.3	104.4	62.2	424.6	13.4	64.5	13.1	754.2	424.4	15.5	411.9	3.9	6.2	16.2	3.4	94.9	6.9	0.0	18.5	85.5	73.7	135.0	0.7	14.6	77.2	125.2
SC 5–6_30/70	31.8	18.4	87.3	61.9	424.6	9.2	49.7	10.8	538.7	340.7	11.6	307.6	2.4	4.2	11.2	3.4	74.7	4.0	0.0	18.5	80.3	58.0	99.7	0.7	12.6	57.9	97.7
SC 4–6_90/10	31.8	35.3	124.2	60.9	424.6	17.6	78.8	18.1	960.8	536.8	21.1	536.6	5.6	8.6	21.5	4.4	118.4	10.3	0.0	18.5	110.3	88.1	176.8	1.7	18.3	97.6	160.9
SC 4–6_60/40	31.8	25.0	99.0	51.9	424.6	13.2	61.2	14.0	709.4	418.8	15.7	402.6	3.8	6.3	15.9	3.4	92.4	7.0	0.0	18.5	89.7	68.5	135.0	1.3	14.6	72.5	121.4
SC 4–6_30/70	31.8	14.7	73.8	42.9	424.6	8.9	43.6	9.9	458.0	300.9	10.3	268.5	2.1	4.0	10.3	2.5	66.4	3.7	0.0	18.5	69.0	49.0	93.1	1.0	10.9	47.4	81.8
SC 3–6_90/10	33.8	35.7	126.8	60.0	425.5	17.7	77.9	20.8	953.9	564.6	22.9	556.2	5.7	9.1	22.0	4.9	121.8	10.5	0.0	19.1	126.7	86.8	182.1	4.3	20.3	98.5	166.8
SC 3–6_60/40	29.2	24.5	94.0	46.0	353.0	12.5	56.4	14.9	674.6	413.4	15.9	393.6	3.8	6.4	15.3	3.5	89.7	7.0	0.0	18.3	93.1	62.8	135.0	3.0	14.6	69.2	118.7
SC 3–6_30/70	24.7	13.4	61.1	31.9	280.4	7.4	35.0	9.0	395.4	262.3	8.9	231.1	2.0	3.6	8.6	2.0	57.5	3.5	0.0	17.5	59.5	38.9	87.8	1.7	8.9	39.9	70.6
SC 6_0/100	31.8	19.6	106.1	62.6	424.6	17.4	58.9	12.0	686.7	400.1	8.7	329.0	4.1	3.1	15.0	3.4	94.5	6.4	0.0	18.5	84.1	91.0	105.3	0.7	14.6	77.3	105.7
SC 5–6_0/100	31.8	10.5	70.3	61.7	424.6	5.1	34.8	8.5	323.1	257.0	7.7	203.4	1.0	2.1	6.2	3.4	54.5	1.1	0.0	18.5	75.1	42.3	64.4	0.7	10.6	38.6	70.3
SC 4–6_0/100	31.8	4.4	48.5	33.9	424.6	4.5	26.0	5.7	206.6	182.9	4.9	134.4	0.3	1.7	4.7	1.5	40.4	0.4	0.0	18.5	48.3	29.4	51.3	0.6	7.1	22.3	42.3
SC 3–6_0/100	20.1	2.2	28.2	17.9	207.8	2.2	13.6	3.1	116.2	111.2	1.9	68.6	0.1	0.8	2.0	0.6	25.4	0.0	0.0	16.7	26.0	14.9	40.7	0.3	3.2	10.7	22.5
compost	**BE**	**BG**	**CZ**	**DK**	**DE**	**EE**	**IE**	**GR**	**ES**	**FR**	**HR**	**IT**	**CY**	**LV**	**LT**	**LU**	**HU**	**MA**	**NL**	**AT**	**PL**	**PT**	**RO**	**SL**	**SK**	**FI**	**SE**
SC 6_90/10	0.0	1.2	33.0	6.0	95.0	1.6	3.3	3.7	7.2	138.7	3.0	38.9	2.9	0.9	5.3	1.2	69.2	2.1	0.0	6.2	13.7	1.1	23.9	0.4	12.4	27.1	26.2
SC 6_60/40	0.0	1.2	33.0	6.0	95.0	1.6	3.3	3.7	7.2	138.7	3.0	38.9	2.9	0.9	5.3	1.2	69.2	2.1	0.0	6.2	13.7	1.1	23.9	0.4	12.4	27.1	26.2
SC 6_30/70	0.0	1.2	33.0	6.0	95.0	1.6	3.3	3.7	7.2	138.7	3.0	38.9	2.9	0.9	5.3	1.2	69.2	2.1	0.0	6.2	13.7	1.1	23.9	0.4	12.4	27.1	26.2
SC 5–6_90/10	0.0	0.6	21.9	5.9	95.0	0.5	1.9	2.6	3.4	89.1	2.7	24.0	0.7	0.6	2.2	1.2	39.9	0.4	0.0	6.2	12.3	0.5	14.6	0.4	9.0	13.5	17.5
SC 5–6_60/40	0.0	0.6	21.9	5.9	95.0	0.5	1.9	2.6	3.4	89.1	2.7	24.0	0.7	0.6	2.2	1.2	39.9	0.4	0.0	6.2	12.3	0.5	14.6	0.4	9.0	13.5	17.5
SC 5–6_30/70	0.0	0.6	21.9	5.9	95.0	0.5	1.9	2.6	3.4	89.1	2.7	24.0	0.7	0.6	2.2	1.2	39.9	0.4	0.0	6.2	12.3	0.5	14.6	0.4	9.0	13.5	17.5
SC 4–6_90/10	0.0	0.3	15.1	3.2	95.0	0.4	1.4	1.8	2.2	63.4	1.7	15.9	0.2	0.5	1.7	0.5	29.6	0.1	0.0	6.2	7.9	0.4	11.6	0.4	6.1	7.8	10.5
SC 4–6_60/40	0.0	0.3	15.1	3.2	95.0	0.4	1.4	1.8	2.2	63.4	1.7	15.9	0.2	0.5	1.7	0.5	29.6	0.1	0.0	6.2	7.9	0.4	11.6	0.4	6.1	7.8	10.5
SC 4–6_30/70	0.0	0.3	15.1	3.2	95.0	0.4	1.4	1.8	2.2	63.4	1.7	15.9	0.2	0.5	1.7	0.5	29.6	0.1	0.0	6.2	7.9	0.4	11.6	0.4	6.1	7.8	10.5
SC 3–6_90/10	0.0	0.1	8.8	1.7	46.5	0.2	0.8	0.9	1.2	38.5	0.7	8.1	0.1	0.3	0.7	0.2	18.6	0.0	0.0	5.6	4.2	0.2	9.2	0.2	2.7	3.7	5.6
SC 3–6_60/40	0.0	0.1	8.8	1.7	46.5	0.2	0.8	0.9	1.2	38.5	0.7	8.1	0.1	0.3	0.7	0.2	18.6	0.0	0.0	5.6	4.2	0.2	9.2	0.2	2.7	3.7	5.6
SC 3–6_30/70	0.0	0.1	8.8	1.7	46.5	0.2	0.8	0.9	1.2	38.5	0.7	8.1	0.1	0.3	0.7	0.2	18.6	0.0	0.0	5.6	4.2	0.2	9.2	0.2	2.7	3.7	5.6
SC 6_0/100	0.0	1.2	33.0	6.0	95.0	1.6	3.3	3.7	7.2	138.7	3.0	38.9	2.9	0.9	5.3	1.2	69.2	2.1	0.0	6.2	13.7	1.1	23.9	0.4	12.4	27.1	26.2
SC 5–6_0/100	0.0	0.6	21.9	5.9	95.0	0.5	1.9	2.6	3.4	89.1	2.7	24.0	0.7	0.6	2.2	1.2	39.9	0.4	0.0	6.2	12.3	0.5	14.6	0.4	9.0	13.5	17.5
SC 4–6_0/100	0.0	0.3	15.1	3.2	95.0	0.4	1.4	1.8	2.2	63.4	1.7	15.9	0.2	0.5	1.7	0.5	29.6	0.1	0.0	6.2	7.9	0.4	11.6	0.4	6.1	7.8	10.5
SC 3–6_0/100	0.0	0.1	8.8	1.7	46.5	0.2	0.8	0.9	1.2	38.5	0.7	8.1	0.1	0.3	0.7	0.2	18.6	0.0	0.0	5.6	4.2	0.2	9.2	0.2	2.7	3.7	5.6
co-incineration	**BE**	**BG**	**CZ**	**DK**	**DE**	**EE**	**IE**	**GR**	**ES**	**FR**	**HR**	**IT**	**CY**	**LV**	**LT**	**LU**	**HU**	**MA**	**NL**	**AT**	**PL**	**PT**	**RO**	**SL**	**SK**	**FI**	**SE**
SC 6_90/10	85.1	0.0	0.0	28.3	0.0	0.0	0.0	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.9	0.0	0.0	168.0	0.0	97.9	0.0	0.0	18.4	5.0	0.0	0.0
SC 6_60/40	85.1	0.0	0.0	28.3	0.0	0.0	0.0	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.9	0.0	0.0	168.0	0.0	97.9	0.0	0.0	18.4	5.0	0.0	0.0
SC 6_30/70	85.1	0.0	0.0	28.3	0.0	0.0	0.0	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.9	0.0	0.0	168.0	0.0	97.9	0.0	0.0	18.4	5.0	0.0	0.0
SC 5–6_90/10	41.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	142.6	0.0	0.0	0.0	0.0	5.1	0.0	0.0	0.0
SC 5–6_60/40	41.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	142.6	0.0	0.0	0.0	0.0	5.1	0.0	0.0	0.0
SC 5–6_30/70	41.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	142.6	0.0	0.0	0.0	0.0	5.1	0.0	0.0	0.0
SC 4–6_90/10	15.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	70.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SC 4–6_60/40	15.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	70.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SC 4–6_30/70	15.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	70.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SC 3–6_90/10	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	16.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

SC 3–6_60/40	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	16.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SC 3–6_30/70	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	16.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SC 6_0/100	85.1	0.0	0.0	28.3	0.0	0.0	0.0	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.9	0.0	0.0	168.0	0.0	97.9	0.0	0.0	18.4	5.0	0.0	0.0
SC 5–6_0/100	41.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	142.6	0.0	0.0	0.0	0.0	5.1	0.0	0.0	0.0
SC 4–6_0/100	15.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	70.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SC 3–6_0/100	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	16.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
mono-inc + P-recovery	**BE**	**BG**	**CZ**	**DK**	**DE**	**EE**	**IE**	**GR**	**ES**	**FR**	**HR**	**IT**	**CY**	**LV**	**LT**	**LU**	**HU**	**MA**	**NL**	**AT**	**PL**	**PT**	**RO**	**SL**	**SK**	**FI**	**SE**
SC 6_90/10	20.3	4.9	51.2	35.8	1220	0.0	3.2	41.2	106.0	213.2	6.0	98.8	0.0	2.8	1.7	0.0	41.8	0.0	168.0	209.4	144.6	11.5	25.7	0.0	9.6	7.4	19.5
SC 6_60/40	20.3	9.2	54.8	35.8	1220	0.0	10.6	41.2	209.6	230.7	9.4	148.8	0.0	4.3	3.3	0.0	45.6	0.0	168.0	209.4	144.6	12.5	40.5	0.0	9.6	12.8	31.6
SC 6_30/70	20.3	13.5	58.3	35.8	1220	0.0	18.0	41.2	313.2	248.2	12.8	198.7	0.0	5.8	4.9	0.0	49.4	0.0	168.0	209.4	144.6	13.5	55.4	0.0	9.6	18.2	43.8
SC 5–6_90/10	63.8	8.9	64.8	64.2	1220	3.0	6.4	47.8	152.8	312.8	7.0	172.2	2.2	3.9	5.6	2.5	61.9	5.4	193.4	209.4	252.5	16.7	59.9	13.3	17.9	19.4	40.2
SC 5–6_60/40	63.8	16.8	81.8	64.5	1220	7.2	21.3	50.1	368.3	396.4	10.9	276.5	3.7	5.9	10.6	2.5	82.1	8.3	193.4	209.4	257.7	32.4	95.2	13.3	19.9	38.7	67.6
SC 5–6_30/70	63.8	24.7	98.8	64.8	1220	11.4	36.1	52.4	583.8	480.1	14.8	380.8	5.1	8.0	15.6	2.5	102.3	11.2	193.4	209.4	262.9	48.1	130.4	13.3	21.9	57.9	95.0
SC 4–6_90/10	90.1	11.6	73.0	68.5	1220	3.2	7.6	51.8	167.8	364.3	9.6	212.6	2.7	4.3	6.2	4.6	69.0	6.1	265.8	209.4	282.5	18.1	70.8	19.7	20.8	24.5	56.6
SC 4–6_60/40	90.1	21.9	98.2	77.5	1220	7.5	25.2	55.9	419.2	482.3	15.0	346.6	4.5	6.6	11.8	5.5	95.0	9.4	265.8	209.4	303.2	37.7	112.6	20.1	24.5	49.6	96.1
SC 4–6_30/70	90.1	32.2	123.5	86.5	1220	11.9	42.8	60.0	670.6	600.3	20.4	480.7	6.3	8.9	17.4	6.5	121.0	12.7	265.8	209.4	323.9	57.3	154.4	20.5	28.3	74.7	135.7
SC 3–6_90/10	110.5	12.6	80.7	70.9	1270	3.7	9.3	55.6	179.4	414.2	12.4	251.1	2.8	5.2	7.4	5.4	76.5	6.5	319.1	210.6	307.4	19.7	79.6	22.9	24.1	28.1	68.2
SC 3–6_60/40	115.1	23.8	113.6	85.0	1342	8.8	30.7	61.5	458.6	565.4	19.4	413.6	4.7	7.9	14.1	6.9	108.7	10.0	319.1	211.4	341.0	43.6	126.7	24.3	29.8	57.4	116.3
SC 3–6_30/70	119.6	34.9	146.4	99.0	1415	14.0	52.1	67.4	737.9	716.5	26.4	576.1	6.5	10.7	20.7	8.3	140.8	13.5	319.1	212.2	374.5	67.6	173.9	25.6	35.5	86.7	164.4
SC 6_0/100	20.3	17.9	61.9	35.8	1221	0.0	25.4	41.2	416.8	265.7	16.2	248.7	0.0	7.3	6.5	0.0	53.2	0.0	168.0	209.4	144.6	14.5	70.2	0.0	9.6	23.6	55.9
SC 5–6_0/100	63.8	32.6	115.9	65.1	1221	15.6	51.0	54.7	799.3	563.8	18.8	485.0	6.6	10.1	20.6	2.5	122.5	14.1	193.4	209.4	268.1	63.8	165.7	13.3	23.9	77.2	122.4
SC 4–6_0/100	90.1	42.5	148.7	95.5	1221	16.3	60.4	64.2	922.0	718.2	25.8	614.8	8.0	11.1	23.0	7.5	147.0	15.9	265.8	209.4	344.6	76.9	196.3	20.8	32.0	99.8	175.2
SC 3–6_0/100	124.1	46.1	179.3	113.1	1488	19.1	73.5	73.3	1017	867.6	33.4	738.6	8.3	13.5	27.4	9.8	173.0	17.0	319.1	213.0	408.1	91.5	221.0	26.9	41.2	116.0	212.5
other	**BE**	**BG**	**CZ**	**DK**	**DE**	**EE**	**IE**	**GR**	**ES**	**FR**	**HR**	**IT**	**CY**	**LV**	**LT**	**LU**	**HU**	**MA**	**NL**	**AT**	**PL**	**PT**	**RO**	**SL**	**SK**	**FI**	**SE**
SC 6_90/10	19.2	10.0	12.0	0.0	3.5	2.3	0.3	10.5	0.0	294.6	0.0	95.6	1.6	4.3	2.7	3.7	0.0	0.0	0.0	12.3	137.9	0.0	47.1	15.7	2.7	2.7	67.0
SC 6_60/40	19.2	10.0	12.0	0.0	3.5	2.3	0.3	10.5	0.0	294.6	0.0	95.6	1.6	4.3	2.7	3.7	0.0	0.0	0.0	12.3	137.9	0.0	47.1	15.7	2.7	2.7	67.0
SC 6_30/70	19.2	10.0	12.0	0.0	3.5	2.3	0.3	10.5	0.0	294.6	0.0	95.6	1.6	4.3	2.7	3.7	0.0	0.0	0.0	12.3	137.9	0.0	47.1	15.7	2.7	2.7	67.0
SC 5–6_90/10	19.2	5.3	8.0	0.0	3.5	0.7	0.2	7.5	0.0	189.3	0.0	59.1	0.4	2.9	1.1	3.7	0.0	0.0	0.0	12.3	123.1	0.0	28.8	15.7	2.0	1.4	44.6
SC 5–6_60/40	19.2	5.3	8.0	0.0	3.5	0.7	0.2	7.5	0.0	189.3	0.0	59.1	0.4	2.9	1.1	3.7	0.0	0.0	0.0	12.3	123.1	0.0	28.8	15.7	2.0	1.4	44.6
SC 5–6_30/70	19.2	5.3	8.0	0.0	3.5	0.7	0.2	7.5	0.0	189.3	0.0	59.1	0.4	2.9	1.1	3.7	0.0	0.0	0.0	12.3	123.1	0.0	28.8	15.7	2.0	1.4	44.6
SC 4–6_90/10	19.2	2.3	5.5	0.0	3.5	0.6	0.1	5.1	0.0	134.7	0.0	39.1	0.1	2.3	0.8	1.6	0.0	0.0	0.0	12.3	79.1	0.0	22.9	13.4	1.3	0.8	26.8
SC 4–6_60/40	19.2	2.3	5.5	0.0	3.5	0.6	0.1	5.1	0.0	134.7	0.0	39.1	0.1	2.3	0.8	1.6	0.0	0.0	0.0	12.3	79.1	0.0	22.9	13.4	1.3	0.8	26.8
SC 4–6_30/70	19.2	2.3	5.5	0.0	3.5	0.6	0.1	5.1	0.0	134.7	0.0	39.1	0.1	2.3	0.8	1.6	0.0	0.0	0.0	12.3	79.1	0.0	22.9	13.4	1.3	0.8	26.8
SC 3–6_90/10	12.2	1.1	3.2	0.0	1.7	0.3	0.1	2.7	0.0	81.9	0.0	19.9	0.1	1.1	0.4	0.6	0.0	0.0	0.0	11.1	42.6	0.0	18.2	7.7	0.6	0.4	14.3
SC 3–6_60/40	12.2	1.1	3.2	0.0	1.7	0.3	0.1	2.7	0.0	81.9	0.0	19.9	0.1	1.1	0.4	0.6	0.0	0.0	0.0	11.1	42.6	0.0	18.2	7.7	0.6	0.4	14.3
SC 3–6_30/70	12.2	1.1	3.2	0.0	1.7	0.3	0.1	2.7	0.0	81.9	0.0	19.9	0.1	1.1	0.4	0.6	0.0	0.0	0.0	11.1	42.6	0.0	18.2	7.7	0.6	0.4	14.3
SC 6_0/100	19.2	10.0	12.0	0.0	3.5	2.3	0.3	10.5	0.0	294.6	0.0	95.6	1.6	4.3	2.7	3.7	0.0	0.0	0.0	12.3	137.9	0.0	47.1	15.7	2.7	2.7	67.0
SC 5–6_0/100	19.2	5.3	8.0	0.0	3.5	0.7	0.2	7.5	0.0	189.3	0.0	59.1	0.4	2.9	1.1	3.7	0.0	0.0	0.0	12.3	123.1	0.0	28.8	15.7	2.0	1.4	44.6
SC 4–6_0/100	19.2	2.3	5.5	0.0	3.5	0.6	0.1	5.1	0.0	134.7	0.0	39.1	0.1	2.3	0.8	1.6	0.0	0.0	0.0	12.3	79.1	0.0	22.9	13.4	1.3	0.8	26.8
SC 3–6_0/100	19.2	2.3	5.5	0.0	3.5	0.6	0.1	5.1	0.0	134.7	0.0	39.1	0.1	2.3	0.8	1.6	0.0	0.0	0.0	12.3	79.1	0.0	22.9	13.4	1.3	0.8	26.8
landfill	**BE**	**BG**	**CZ**	**DK**	**DE**	**EE**	**IE**	**GR**	**ES**	**FR**	**HR**	**IT**	**CY**	**LV**	**LT**	**LU**	**HU**	**MA**	**NL**	**AT**	**PL**	**PT**	**RO**	**SL**	**SK**	**FI**	**SE**
SC 6_90/10	0.0	1.1	8.8	0.0	0.0	0.6	0.0	15.0	28.7	0.0	10.4	155.5	0.0	0.0	1.1	0.0	0.0	8.5	0.0	0.0	3.9	0.0	69.5	0.0	4.3	0.0	0.0
SC 6_60/40	0.0	1.1	8.8	0.0	0.0	0.6	0.0	15.0	28.7	0.0	10.4	155.5	0.0	0.0	1.1	0.0	0.0	8.5	0.0	0.0	3.9	0.0	69.5	0.0	4.3	0.0	0.0
SC 6_30/70	0.0	1.1	8.8	0.0	0.0	0.6	0.0	15.0	28.7	0.0	10.4	155.5	0.0	0.0	1.1	0.0	0.0	8.5	0.0	0.0	3.9	0.0	69.5	0.0	4.3	0.0	0.0
SC 5–6_90/10	0.0	0.6	5.8	0.0	0.0	0.2	0.0	10.7	13.5	0.0	9.2	96.1	0.0	0.0	0.5	0.0	0.0	1.4	0.0	0.0	3.4	0.0	42.5	0.0	3.1	0.0	0.0
SC 5–6_60/40	0.0	0.6	5.8	0.0	0.0	0.2	0.0	10.7	13.5	0.0	9.2	96.1	0.0	0.0	0.5	0.0	0.0	1.4	0.0	0.0	3.4	0.0	42.5	0.0	3.1	0.0	0.0
SC 5–6_30/70	0.0	0.6	5.8	0.0	0.0	0.2	0.0	10.7	13.5	0.0	9.2	96.1	0.0	0.0	0.5	0.0	0.0	1.4	0.0	0.0	3.4	0.0	42.5	0.0	3.1	0.0	0.0
SC 4–6_90/10	0.0	0.3	4.0	0.0	0.0	0.2	0.0	7.2	8.6	0.0	5.9	63.5	0.0	0.0	0.3	0.0	0.0	0.5	0.0	0.0	2.2	0.0	33.9	0.0	2.1	0.0	0.0
SC 4–6_60/40	0.0	0.3	4.0	0.0	0.0	0.2	0.0	7.2	8.6	0.0	5.9	63.5	0.0	0.0	0.3	0.0	0.0	0.5	0.0	0.0	2.2	0.0	33.9	0.0	2.1	0.0	0.0
SC 4–6_30/70	0.0	0.3	4.0	0.0	0.0	0.2	0.0	7.2	8.6	0.0	5.9	63.5	0.0	0.0	0.3	0.0	0.0	0.5	0.0	0.0	2.2	0.0	33.9	0.0	2.1	0.0	0.0
SC 3–6_90/10	0.0	0.1	2.3	0.0	0.0	0.1	0.0	3.9	4.9	0.0	2.3	32.4	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	1.2	0.0	26.9	0.0	0.9	0.0	0.0
SC 3–6_60/40	0.0	0.1	2.3	0.0	0.0	0.1	0.0	3.9	4.9	0.0	2.3	32.4	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	1.2	0.0	26.9	0.0	0.9	0.0	0.0

SC 3–6_30/70	0.0	0.1	2.3	0.0	0.0	0.1	0.0	3.9	4.9	0.0	2.3	32.4	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	1.2	0.0	26.9	0.0	0.9	0.0	0.0
SC 6_0/100	0.0	1.1	8.8	0.0	0.0	0.6	0.0	15.0	28.7	0.0	10.4	155.5	0.0	0.0	1.1	0.0	0.0	8.5	0.0	0.0	3.9	0.0	69.5	0.0	4.3	0.0	0.0
SC 5–6_0/100	0.0	0.6	5.8	0.0	0.0	0.2	0.0	10.7	13.5	0.0	9.2	96.1	0.0	0.0	0.5	0.0	0.0	1.4	0.0	0.0	3.4	0.0	42.5	0.0	3.1	0.0	0.0
SC 4–6_0/100	0.0	0.3	4.0	0.0	0.0	0.2	0.0	7.2	8.6	0.0	5.9	63.5	0.0	0.0	0.3	0.0	0.0	0.5	0.0	0.0	2.2	0.0	33.9	0.0	2.1	0.0	0.0
SC 3–6_0/100	0.0	0.1	2.3	0.0	0.0	0.1	0.0	3.9	4.9	0.0	2.3	32.4	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	1.2	0.0	26.9	0.0	0.9	0.0	0.0

Table 15. P to land within sewage sludge or as mineral component (kg P yr^{-1}).

P within sewage sludge	BE	BG	CZ	DK	DE	EE	IE	GR	ES	FR	HR	IT	CY	LV	LT	LU	HU	MA	NL	AT	PL	PT	RO	SL	SK	FI	SE
SC 6_90/10	0.6	0.7	2.3	1.2	8.0	0.4	1.7	0.2	20.3	9.0	0.4	9.7	0.1	0.2	0.4	0.1	2.1	0.1	0.0	0.3	1.6	1.9	3.0	0.0	0.3	1.9	2.9
SC 6_60/40	0.6	0.6	2.2	1.2	8.0	0.4	1.5	0.2	18.0	8.6	0.3	8.6	0.1	0.1	0.4	0.1	2.0	0.1	0.0	0.3	1.6	1.9	2.7	0.0	0.3	1.8	2.6
SC 6_30/70	0.6	0.5	2.2	1.2	8.0	0.4	1.3	0.2	15.8	8.2	0.2	7.5	0.1	0.1	0.3	0.1	1.9	0.1	0.0	0.3	1.6	1.9	2.4	0.0	0.3	1.7	2.4
SC 5–6_90/10	0.6	0.7	2.4	1.2	8.0	0.4	1.6	0.3	19.7	10.0	0.4	10.3	0.1	0.2	0.4	0.1	2.3	0.2	0.0	0.3	1.7	1.8	3.4	0.0	0.3	1.9	3.1
SC 5–6_60/40	0.6	0.5	2.0	1.2	8.0	0.3	1.3	0.2	15.0	8.3	0.3	8.1	0.1	0.1	0.3	0.1	1.9	0.1	0.0	0.3	1.6	1.5	2.6	0.0	0.3	1.5	2.5
SC 5–6_30/70	0.6	0.4	1.7	1.2	8.0	0.2	1.0	0.2	10.4	6.5	0.2	5.9	0.0	0.1	0.2	0.1	1.4	0.1	0.0	0.3	1.5	1.1	1.9	0.0	0.2	1.1	1.9
SC 4–6_90/10	0.6	0.7	2.4	1.2	8.0	0.4	1.6	0.3	19.5	10.5	0.4	10.7	0.1	0.2	0.4	0.1	2.3	0.2	0.0	0.3	2.1	1.8	3.5	0.0	0.4	2.0	3.2
SC 4–6_60/40	0.6	0.5	1.9	1.0	8.0	0.3	1.2	0.3	14.0	8.1	0.3	7.8	0.1	0.1	0.3	0.1	1.8	0.1	0.0	0.3	1.7	1.4	2.6	0.0	0.3	1.4	2.4
SC 4–6_30/70	0.6	0.3	1.4	0.8	8.0	0.2	0.8	0.2	8.8	5.7	0.2	5.1	0.0	0.1	0.2	0.0	1.3	0.1	0.0	0.3	1.3	0.9	1.8	0.0	0.2	0.9	1.6
SC 3–6_90/10	0.6	0.7	2.5	1.2	8.0	0.4	1.6	0.4	19.3	11.1	0.5	11.0	0.1	0.2	0.4	0.1	2.4	0.2	0.0	0.4	2.4	1.8	3.6	0.1	0.4	2.0	3.3
SC 3–6_60/40	0.5	0.5	1.8	0.9	6.6	0.2	1.1	0.3	13.2	7.9	0.3	7.6	0.1	0.1	0.3	0.1	1.7	0.1	0.0	0.3	1.7	1.2	2.6	0.1	0.3	1.4	2.3
SC 3–6_30/70	0.5	0.3	1.2	0.6	5.2	0.1	0.7	0.2	7.5	4.9	0.2	4.4	0.0	0.1	0.2	0.0	1.1	0.1	0.0	0.3	1.1	0.7	1.7	0.0	0.2	0.8	1.3
SC 6_0/100	0.6	0.4	2.1	1.2	8.0	0.4	1.2	0.2	13.5	7.9	0.2	6.4	0.1	0.1	0.3	0.1	1.9	0.1	0.0	0.3	1.6	1.8	2.1	0.0	0.3	1.5	2.1
SC 5–6_0/100	0.6	0.2	1.3	1.2	8.0	0.1	0.7	0.2	6.1	4.9	0.1	3.8	0.0	0.0	0.1	0.1	1.0	0.0	0.0	0.3	1.4	0.8	1.2	0.0	0.2	0.7	1.3
SC 4–6_0/100	0.6	0.1	0.9	0.6	8.0	0.1	0.5	0.1	3.9	3.4	0.1	2.5	0.0	0.0	0.1	0.0	0.8	0.0	0.0	0.3	0.9	0.6	1.0	0.0	0.1	0.4	0.8
SC 3–6_0/100	0.4	0.0	0.5	0.3	3.8	0.0	0.3	0.1	2.1	2.1	0.0	1.3	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.3	0.5	0.3	0.8	0.0	0.1	0.2	0.4
P as mineral component	**BE**	**BG**	**CZ**	**DK**	**DE**	**EE**	**IE**	**GR**	**ES**	**FR**	**HR**	**IT**	**CY**	**LV**	**LT**	**LU**	**HU**	**MA**	**NL**	**AT**	**PL**	**PT**	**RO**	**SL**	**SK**	**FI**	**SE**
SC 6_90/10	0.4	0.1	1.0	0.7	23.0	0.0	0.1	0.8	2.2	4.2	0.1	2.0	0.0	0.1	0.0	0.0	0.8	0.0	3.1	3.9	2.8	0.2	0.5	0.0	0.2	0.2	0.4
SC 6_60/40	0.4	0.2	1.1	0.7	23.0	0.0	0.2	0.8	4.2	4.6	0.2	3.0	0.0	0.1	0.1	0.0	0.9	0.0	3.1	3.9	2.8	0.3	0.8	0.0	0.2	0.3	0.6
SC 6_30/70	0.4	0.3	1.2	0.7	23.0	0.0	0.4	0.8	6.2	4.9	0.2	3.9	0.0	0.1	0.1	0.0	1.0	0.0	3.1	3.9	2.8	0.3	1.1	0.0	0.2	0.4	0.9
SC 5–6_90/10	1.2	0.2	1.3	1.2	23.0	0.1	0.1	0.9	3.1	6.2	0.1	3.4	0.0	0.1	0.1	0.0	1.2	0.1	3.5	3.9	4.8	0.3	1.2	0.2	0.3	0.4	0.8
SC 5–6_60/40	1.2	0.3	1.6	1.3	23.0	0.1	0.4	0.9	7.3	7.7	0.2	5.4	0.1	0.1	0.2	0.0	1.6	0.2	3.5	3.9	4.8	0.6	1.9	0.2	0.4	0.8	1.3
SC 5–6_30/70	1.2	0.5	1.9	1.3	23.0	0.2	0.7	1.0	11.3	9.2	0.3	7.3	0.1	0.2	0.3	0.0	2.0	0.2	3.5	3.9	4.9	0.9	2.5	0.2	0.4	1.1	1.8
SC 4–6_90/10	1.7	0.2	1.4	1.3	23.0	0.1	0.2	1.0	3.4	7.2	0.2	4.2	0.1	0.1	0.1	0.1	1.4	0.1	4.8	3.9	5.3	0.4	1.4	0.4	0.4	0.5	1.1
SC 4–6_60/40	1.7	0.4	1.9	1.5	23.0	0.1	0.5	1.0	8.3	9.3	0.3	6.8	0.1	0.1	0.2	0.1	1.8	0.2	4.8	3.9	5.7	0.7	2.2	0.4	0.5	1.0	1.9
SC 4–6_30/70	1.7	0.6	2.4	1.6	23.0	0.2	0.8	1.1	12.9	11.4	0.4	9.2	0.1	0.2	0.3	0.1	2.3	0.2	4.8	3.9	6.0	1.1	3.0	0.4	0.5	1.4	2.6
SC 3–6_90/10	2.1	0.3	1.6	1.4	23.9	0.1	0.2	1.0	3.6	8.1	0.2	5.0	0.1	0.1	0.1	0.1	1.5	0.1	5.8	3.9	5.8	0.4	1.6	0.4	0.5	0.6	1.4
SC 3–6_60/40	2.1	0.5	2.2	1.6	25.1	0.2	0.6	1.1	9.0	10.9	0.4	8.0	0.1	0.2	0.3	0.1	2.1	0.2	5.8	3.9	6.4	0.9	2.5	0.4	0.6	1.1	2.3
SC 3–6_30/70	2.2	0.7	2.8	1.9	26.3	0.3	1.0	1.2	14.0	13.5	0.5	10.9	0.1	0.2	0.4	0.2	2.7	0.3	5.8	3.9	6.9	1.3	3.3	0.5	0.7	1.6	3.1
SC 6_0/100	0.4	0.3	1.2	0.7	23.0	0.0	0.5	0.8	8.2	5.2	0.3	4.9	0.0	0.1	0.1	0.0	1.1	0.0	3.1	3.9	2.8	0.3	1.4	0.0	0.2	0.5	1.1
SC 5–6_0/100	1.2	0.6	2.2	1.3	23.0	0.3	1.0	1.0	15.1	10.7	0.4	9.2	0.1	0.2	0.4	0.0	2.3	0.3	3.5	3.9	5.0	1.2	3.1	0.2	0.5	1.5	2.3
SC 4–6_0/100	1.7	0.8	2.8	1.8	23.0	0.3	1.1	1.2	17.2	13.4	0.5	11.5	0.1	0.2	0.4	0.1	2.8	0.3	4.8	3.9	6.4	1.5	3.7	0.4	0.6	1.9	3.3
SC 3–6_0/100	2.3	0.8	3.3	2.1	27.5	0.4	1.4	1.3	18.8	16.0	0.6	13.6	0.2	0.2	0.5	0.2	3.2	0.3	5.8	3.9	7.5	1.7	4.1	0.5	0.8	2.1	3.9

Table 16. N+C to land as sewage sludge (kg yr^{-1}).

P within sewage sludge	BE	BG	CZ	DK	DE	EE	IE	GR	ES	FR	HR	IT	CY	LV	LT	LU	HU	MA	NL	AT	PL	PT	RO	SL	SK	FI	SE
SC 6_90/10	0.6	0.7	2.3	1.2	8.0	0.4	1.7	0.2	20.3	9.0	0.4	9.7	0.1	0.2	0.4	0.1	2.1	0.1	0.0	0.3	1.6	1.9	3.0	0.0	0.3	1.9	2.9
SC 6_60/40	0.6	0.6	2.2	1.2	8.0	0.4	1.5	0.2	18.0	8.6	0.3	8.6	0.1	0.1	0.4	0.1	2.0	0.1	0.0	0.3	1.6	1.9	2.7	0.0	0.3	1.8	2.6
SC 6_30/70	0.6	0.5	2.2	1.2	8.0	0.4	1.3	0.2	15.8	8.2	0.2	7.5	0.1	0.1	0.3	0.1	1.9	0.1	0.0	0.3	1.6	1.9	2.4	0.0	0.3	1.7	2.4
SC 5–6_90/10	0.6	0.7	2.4	1.2	8.0	0.4	1.6	0.3	19.7	10.0	0.4	10.3	0.1	0.2	0.4	0.1	2.3	0.2	0.0	0.3	1.7	1.8	3.4	0.0	0.3	1.9	3.1
SC 5–6_60/40	0.6	0.5	2.0	1.2	8.0	0.3	1.3	0.2	15.0	8.3	0.3	8.1	0.1	0.1	0.3	0.1	1.9	0.1	0.0	0.3	1.6	1.5	2.6	0.0	0.3	1.5	2.5
SC 5–6_30/70	0.6	0.4	1.7	1.2	8.0	0.2	1.0	0.2	10.4	6.5	0.2	5.9	0.0	0.1	0.2	0.1	1.4	0.1	0.0	0.3	1.5	1.1	1.9	0.0	0.2	1.1	1.9
SC 4–6_90/10	0.6	0.7	2.4	1.2	8.0	0.4	1.6	0.3	19.5	10.5	0.4	10.7	0.1	0.2	0.4	0.1	2.3	0.2	0.0	0.3	2.1	1.8	3.5	0.0	0.4	2.0	3.2
SC 4–6_60/40	0.6	0.5	1.9	1.0	8.0	0.3	1.2	0.3	14.0	8.1	0.3	7.8	0.1	0.1	0.3	0.1	1.8	0.1	0.0	0.3	1.7	1.4	2.6	0.0	0.3	1.4	2.4
SC 4–6_30/70	0.6	0.3	1.4	0.8	8.0	0.2	0.8	0.2	8.8	5.7	0.2	5.1	0.0	0.1	0.2	0.0	1.3	0.1	0.0	0.3	1.3	0.9	1.8	0.0	0.2	0.9	1.6
SC 3–6_90/10	0.6	0.7	2.5	1.2	8.0	0.4	1.6	0.4	19.3	11.1	0.5	11.0	0.1	0.2	0.4	0.1	2.4	0.2	0.0	0.4	2.4	1.8	3.6	0.1	0.4	2.0	3.3
SC 3–6_60/40	0.5	0.5	1.8	0.9	6.6	0.2	1.1	0.3	13.2	7.9	0.3	7.6	0.1	0.1	0.3	0.1	1.7	0.1	0.0	0.3	1.7	1.2	2.6	0.1	0.3	1.4	2.3
SC 3–6_30/70	0.5	0.3	1.2	0.6	5.2	0.1	0.7	0.2	7.5	4.9	0.2	4.4	0.0	0.1	0.2	0.0	1.1	0.1	0.0	0.3	1.1	0.7	1.7	0.0	0.2	0.8	1.3
SC 6_0/100	0.6	0.4	2.1	1.2	8.0	0.4	1.2	0.2	13.5	7.9	0.2	6.4	0.1	0.1	0.3	0.1	1.9	0.1	0.0	0.3	1.6	1.8	2.1	0.0	0.3	1.5	2.1
SC 5–6_0/100	0.6	0.2	1.3	1.2	8.0	0.1	0.7	0.2	6.1	4.9	0.1	3.8	0.0	0.0	0.1	0.1	1.0	0.0	0.0	0.3	1.4	0.8	1.2	0.0	0.2	0.7	1.3
SC 4–6_0/100	0.6	0.1	0.9	0.6	8.0	0.1	0.5	0.1	3.9	3.4	0.1	2.5	0.0	0.0	0.1	0.0	0.8	0.0	0.0	0.3	0.9	0.6	1.0	0.0	0.1	0.4	0.8
SC 3–6_0/100	0.4	0.0	0.5	0.3	3.8	0.0	0.3	0.1	2.1	2.1	0.0	1.3	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.3	0.5	0.3	0.8	0.0	0.1	0.2	0.4
P as mineral component	**BE**	**BG**	**CZ**	**DK**	**DE**	**EE**	**IE**	**GR**	**ES**	**FR**	**HR**	**IT**	**CY**	**LV**	**LT**	**LU**	**HU**	**MA**	**NL**	**AT**	**PL**	**PT**	**RO**	**SL**	**SK**	**FI**	**SE**
SC 6_90/10	0.4	0.1	1.0	0.7	23.0	0.0	0.1	0.8	2.2	4.2	0.1	2.0	0.0	0.1	0.0	0.0	0.8	0.0	3.1	3.9	2.8	0.2	0.5	0.0	0.2	0.2	0.4
SC 6_60/40	0.4	0.2	1.1	0.7	23.0	0.0	0.2	0.8	4.2	4.6	0.2	3.0	0.0	0.1	0.1	0.0	0.9	0.0	3.1	3.9	2.8	0.3	0.8	0.0	0.2	0.3	0.6
SC 6_30/70	0.4	0.3	1.2	0.7	23.0	0.0	0.4	0.8	6.2	4.9	0.2	3.9	0.0	0.1	0.1	0.0	1.0	0.0	3.1	3.9	2.8	0.3	1.1	0.0	0.2	0.4	0.9
SC 5–6_90/10	1.2	0.2	1.3	1.2	23.0	0.1	0.1	0.9	3.1	6.2	0.1	3.4	0.0	0.1	0.1	0.0	1.2	0.1	3.5	3.9	4.8	0.3	1.2	0.2	0.3	0.4	0.8
SC 5–6_60/40	1.2	0.3	1.6	1.3	23.0	0.1	0.4	0.9	7.3	7.7	0.2	5.4	0.1	0.1	0.2	0.0	1.6	0.2	3.5	3.9	4.8	0.6	1.9	0.2	0.4	0.8	1.3
SC 5–6_30/70	1.2	0.5	1.9	1.3	23.0	0.2	0.7	1.0	11.3	9.2	0.3	7.3	0.1	0.2	0.3	0.0	2.0	0.2	3.5	3.9	4.9	0.9	2.5	0.2	0.4	1.1	1.8
SC 4–6_90/10	1.7	0.2	1.4	1.3	23.0	0.1	0.2	1.0	3.4	7.2	0.2	4.2	0.1	0.1	0.1	0.1	1.4	0.1	4.8	3.9	5.3	0.4	1.4	0.4	0.4	0.5	1.1
SC 4–6_60/40	1.7	0.4	1.9	1.5	23.0	0.1	0.5	1.0	8.3	9.3	0.3	6.8	0.1	0.1	0.2	0.1	1.8	0.2	4.8	3.9	5.7	0.7	2.2	0.4	0.5	1.0	1.9
SC 4–6_30/70	1.7	0.6	2.4	1.6	23.0	0.2	0.8	1.1	12.9	11.4	0.4	9.2	0.1	0.2	0.3	0.1	2.3	0.2	4.8	3.9	6.0	1.1	3.0	0.4	0.5	1.4	2.6
SC 3–6_90/10	2.1	0.3	1.6	1.4	23.9	0.1	0.2	1.0	3.6	8.1	0.2	5.0	0.1	0.1	0.1	0.1	1.5	0.1	5.8	3.9	5.8	0.4	1.6	0.4	0.5	0.6	1.4
SC 3–6_60/40	2.1	0.5	2.2	1.6	25.1	0.2	0.6	1.1	9.0	10.9	0.4	8.0	0.1	0.2	0.3	0.1	2.1	0.2	5.8	3.9	6.4	0.9	2.5	0.4	0.6	1.1	2.3
SC 3–6_30/70	2.2	0.7	2.8	1.9	26.3	0.3	1.0	1.2	14.0	13.5	0.5	10.9	0.1	0.2	0.4	0.2	2.7	0.3	5.8	3.9	6.9	1.3	3.3	0.5	0.7	1.6	3.1
SC 6_0/100	0.4	0.3	1.2	0.7	23.0	0.0	0.5	0.8	8.2	5.2	0.3	4.9	0.0	0.1	0.1	0.0	1.1	0.0	3.1	3.9	2.8	0.3	1.4	0.0	0.2	0.5	1.1
SC 5–6_0/100	1.2	0.6	2.2	1.3	23.0	0.3	1.0	1.0	15.1	10.7	0.4	9.2	0.1	0.2	0.4	0.0	2.3	0.3	3.5	3.9	5.0	1.2	3.1	0.2	0.5	1.5	2.3
SC 4–6_0/100	1.7	0.8	2.8	1.8	23.0	0.3	1.1	1.2	17.2	13.4	0.5	11.5	0.1	0.2	0.4	0.1	2.8	0.3	4.8	3.9	6.4	1.5	3.7	0.4	0.6	1.9	3.3
SC 3–6_0/100	2.3	0.8	3.3	2.1	27.5	0.4	1.4	1.3	18.8	16.0	0.6	13.6	0.2	0.2	0.5	0.2	3.2	0.3	5.8	3.9	7.5	1.7	4.1	0.5	0.8	2.1	3.9

Table 17. Number of mono-incinerators and P-recovery plants.

P within sewage sludge	BE	BG	CZ	DK	DE	EE	IE	GR	ES	FR	HR	IT	CY	LV	LT	LU	HU	MA	NL	AT	PL	PT	RO	SL	SK	FI	SE
SC 6_90/10	0.6	0.7	2.3	1.2	8.0	0.4	1.7	0.2	20.3	9.0	0.4	9.7	0.1	0.2	0.4	0.1	2.1	0.1	0.0	0.3	1.6	1.9	3.0	0.0	0.3	1.9	2.9
SC 6_60/40	0.6	0.6	2.2	1.2	8.0	0.4	1.5	0.2	18.0	8.6	0.3	8.6	0.1	0.1	0.4	0.1	2.0	0.1	0.0	0.3	1.6	1.9	2.7	0.0	0.3	1.8	2.6
SC 6_30/70	0.6	0.5	2.2	1.2	8.0	0.4	1.3	0.2	15.8	8.2	0.2	7.5	0.1	0.1	0.3	0.1	1.9	0.1	0.0	0.3	1.6	1.9	2.4	0.0	0.3	1.7	2.4
SC 5–6_90/10	0.6	0.7	2.4	1.2	8.0	0.4	1.6	0.3	19.7	10.0	0.4	10.3	0.1	0.2	0.4	0.1	2.3	0.2	0.0	0.3	1.7	1.8	3.4	0.0	0.3	1.9	3.1
SC 5–6_60/40	0.6	0.5	2.0	1.2	8.0	0.3	1.3	0.2	15.0	8.3	0.3	8.1	0.1	0.1	0.3	0.1	1.9	0.1	0.0	0.3	1.6	1.5	2.6	0.0	0.3	1.5	2.5
SC 5–6_30/70	0.6	0.4	1.7	1.2	8.0	0.2	1.0	0.2	10.4	6.5	0.2	5.9	0.0	0.1	0.2	0.1	1.4	0.1	0.0	0.3	1.5	1.1	1.9	0.0	0.2	1.1	1.9
SC 4–6_90/10	0.6	0.7	2.4	1.2	8.0	0.4	1.6	0.3	19.5	10.5	0.4	10.7	0.1	0.2	0.4	0.1	2.3	0.2	0.0	0.3	2.1	1.8	3.5	0.0	0.4	2.0	3.2
SC 4–6_60/40	0.6	0.5	1.9	1.0	8.0	0.3	1.2	0.3	14.0	8.1	0.3	7.8	0.1	0.1	0.3	0.1	1.8	0.1	0.0	0.3	1.7	1.4	2.6	0.0	0.3	1.4	2.4
SC 4–6_30/70	0.6	0.3	1.4	0.8	8.0	0.2	0.8	0.2	8.8	5.7	0.2	5.1	0.0	0.1	0.2	0.0	1.3	0.1	0.0	0.3	1.3	0.9	1.8	0.0	0.2	0.9	1.6

SC 3–6_90/10	0.6	0.7	2.5	1.2	8.0	0.4	1.6	0.4	19.3	11.1	0.5	11.0	0.1	0.2	0.4	0.1	2.4	0.2	0.0	0.4	2.4	1.8	3.6	0.1	0.4	2.0	3.3
SC 3–6_60/40	0.5	0.5	1.8	0.9	6.6	0.2	1.1	0.3	13.2	7.9	0.3	7.6	0.1	0.1	0.3	0.1	1.7	0.1	0.0	0.3	1.7	1.2	2.6	0.1	0.3	1.4	2.3
SC 3–6_30/70	0.5	0.3	1.2	0.6	5.2	0.1	0.7	0.2	7.5	4.9	0.2	4.4	0.0	0.1	0.2	0.0	1.1	0.1	0.0	0.3	1.1	0.7	1.7	0.0	0.2	0.8	1.3
SC 6_0/100	0.6	0.4	2.1	1.2	8.0	0.4	1.2	0.2	13.5	7.9	0.2	6.4	0.1	0.1	0.3	0.1	1.9	0.1	0.0	0.3	1.6	1.8	2.1	0.0	0.3	1.5	2.1
SC 5–6_0/100	0.6	0.2	1.3	1.2	8.0	0.1	0.7	0.2	6.1	4.9	0.1	3.8	0.0	0.0	0.1	0.1	1.0	0.0	0.0	0.3	1.4	0.8	1.2	0.0	0.2	0.7	1.3
SC 4–6_0/100	0.6	0.1	0.9	0.6	8.0	0.1	0.5	0.1	3.9	3.4	0.1	2.5	0.0	0.0	0.1	0.0	0.8	0.0	0.0	0.3	0.9	0.6	1.0	0.0	0.1	0.4	0.8
SC 3–6_0/100	0.4	0.0	0.5	0.3	3.8	0.0	0.3	0.1	2.1	2.1	0.0	1.3	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.3	0.5	0.3	0.8	0.0	0.1	0.2	0.4
P as mineral component	**BE**	**BG**	**CZ**	**DK**	**DE**	**EE**	**IE**	**GR**	**ES**	**FR**	**HR**	**IT**	**CY**	**LV**	**LT**	**LU**	**HU**	**MA**	**NL**	**AT**	**PL**	**PT**	**RO**	**SL**	**SK**	**FI**	**SE**
SC 6_90/10	0.4	0.1	1.0	0.7	23.0	0.0	0.1	0.8	2.2	4.2	0.1	2.0	0.0	0.1	0.0	0.0	0.8	0.0	3.1	3.9	2.8	0.2	0.5	0.0	0.2	0.2	0.4
SC 6_60/40	0.4	0.2	1.1	0.7	23.0	0.0	0.2	0.8	4.2	4.6	0.2	3.0	0.0	0.1	0.1	0.0	0.9	0.0	3.1	3.9	2.8	0.3	0.8	0.0	0.2	0.3	0.6
SC 6_30/70	0.4	0.3	1.2	0.7	23.0	0.0	0.4	0.8	6.2	4.9	0.2	3.9	0.0	0.1	0.1	0.0	1.0	0.0	3.1	3.9	2.8	0.3	1.1	0.0	0.2	0.4	0.9
SC 5–6_90/10	1.2	0.2	1.3	1.2	23.0	0.1	0.1	0.9	3.1	6.2	0.1	3.4	0.0	0.1	0.1	0.0	1.2	0.1	3.5	3.9	4.8	0.3	1.2	0.2	0.3	0.4	0.8
SC 5–6_60/40	1.2	0.3	1.6	1.3	23.0	0.1	0.4	0.9	7.3	7.7	0.2	5.4	0.1	0.1	0.2	0.0	1.6	0.2	3.5	3.9	4.8	0.6	1.9	0.2	0.4	0.8	1.3
SC 5–6_30/70	1.2	0.5	1.9	1.3	23.0	0.2	0.7	1.0	11.3	9.2	0.3	7.3	0.1	0.2	0.3	0.0	2.0	0.2	3.5	3.9	4.9	0.9	2.5	0.2	0.4	1.1	1.8
SC 4–6_90/10	1.7	0.2	1.4	1.3	23.0	0.1	0.2	1.0	3.4	7.2	0.2	4.2	0.1	0.1	0.1	0.1	1.4	0.1	4.8	3.9	5.3	0.4	1.4	0.4	0.4	0.5	1.1
SC 4–6_60/40	1.7	0.4	1.9	1.5	23.0	0.1	0.5	1.0	8.3	9.3	0.3	6.8	0.1	0.1	0.2	0.1	1.8	0.2	4.8	3.9	5.7	0.7	2.2	0.4	0.5	1.0	1.9
SC 4–6_30/70	1.7	0.6	2.4	1.6	23.0	0.2	0.8	1.1	12.9	11.4	0.4	9.2	0.1	0.2	0.3	0.1	2.3	0.2	4.8	3.9	6.0	1.1	3.0	0.4	0.5	1.4	2.6
SC 3–6_90/10	2.1	0.3	1.6	1.4	23.9	0.1	0.2	1.0	3.6	8.1	0.2	5.0	0.1	0.1	0.1	0.1	1.5	0.1	5.8	3.9	5.8	0.4	1.6	0.4	0.5	0.6	1.4
SC 3–6_60/40	2.1	0.5	2.2	1.6	25.1	0.2	0.6	1.1	9.0	10.9	0.4	8.0	0.1	0.2	0.3	0.1	2.1	0.2	5.8	3.9	6.4	0.9	2.5	0.4	0.6	1.1	2.3
SC 3–6_30/70	2.2	0.7	2.8	1.9	26.3	0.3	1.0	1.2	14.0	13.5	0.5	10.9	0.1	0.2	0.4	0.2	2.7	0.3	5.8	3.9	6.9	1.3	3.3	0.5	0.7	1.6	3.1
SC 6_0/100	0.4	0.3	1.2	0.7	23.0	0.0	0.5	0.8	8.2	5.2	0.3	4.9	0.0	0.1	0.1	0.0	1.1	0.0	3.1	3.9	2.8	0.3	1.4	0.0	0.2	0.5	1.1
SC 5–6_0/100	1.2	0.6	2.2	1.3	23.0	0.3	1.0	1.0	15.1	10.7	0.4	9.2	0.1	0.2	0.4	0.0	2.3	0.3	3.5	3.9	5.0	1.2	3.1	0.2	0.5	1.5	2.3
SC 4–6_0/100	1.7	0.8	2.8	1.8	23.0	0.3	1.1	1.2	17.2	13.4	0.5	11.5	0.1	0.2	0.4	0.1	2.8	0.3	4.8	3.9	6.4	1.5	3.7	0.4	0.6	1.9	3.3
SC 3–6_0/100	2.3	0.8	3.3	2.1	27.5	0.4	1.4	1.3	18.8	16.0	0.6	13.6	0.2	0.2	0.5	0.2	3.2	0.3	5.8	3.9	7.5	1.7	4.1	0.5	0.8	2.1	3.9

12 Supplementary information – cost assessment

Besides the data from literature, own cost calculation were performed to on the one hand cross-check the values from literature but on the other hand get a deeper inside in the cost structure for the different sewage sludge management options. Different treatment capacities were considered to see the effect of size scaling. With this own cost calculation we are able elaborate the cost drivers for a certain sewage sludge management option and highlight the effect of scaling which in turn allows for the consideration of uncertainties.

12.1 Methodology private cost

Cost were calculated for the different sewage sludge management options, considering operational cost and an annualisation of capital cost. This cost were then used to calculate the investment and annual cost for the PO. The functional unit is the management of 1 ton of sewage sludge dry matter. The system boundary is the gate of a management option including all the relevant processes that are necessary to achieve the final goal of the process (e.g. composted sewage sludge for composting).

12.1.1 Annual costs

Annual costs consist of capital- and operating costs. Investment cost are divided into four cost components and if information on the allocation of investment costs are not available, the following distribution is assumed: Site acquisition (SA, 0.5%) process equipment (PE, 67.5%), building and civil work (B&CW, 29.0 %), and project development (PD, 3.0 %).

To annualise the investment cost, the capital recovery factor is calculated. Then investment costs are multiplied with the capital recovery factor. The calculation of the capital recovery factor includes the rate of interest (3%; ECB, 2022)) and the expected typical depreciation times of the plant components. The expected useful life is, unless otherwise known, 30 years for land, building and civil works, 20 years for process equipment and 7 years for fleet vehicles.

Capital recovery factor $=\frac{i * (1+i)^{n}}{(1+i)^{n}-1}$ Equation 1

i = rate of interest, n = expected useful life

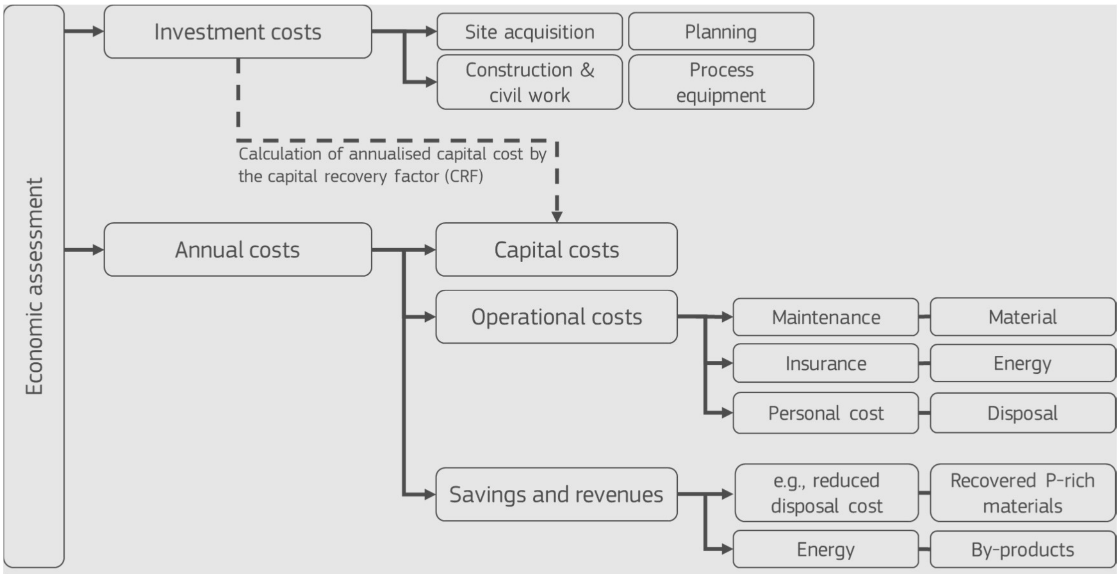
Annualised capital cost [€ yr^{-1}] = Investment cost [€] * capital recovery factor [-] Equation 2

The operational costs are divided into fixed and variable cost. Fixed costs include maintenance (2% of investment costs), insurance (0.5% of investment costs), and personal costs (29 €/man*h per man-year, Eurostat, 2020). In the EU, personal cost vary from 7 to 46 € $man^{-1}h^{-1}$. This aspect is relevant for labour intensive processes and is considered in the uncertainty analysis.

To calculate the variable costs, all relevant resource- and energy input- and output flows were quantified (life cycle inventory) and multiplied with their market prices (**Table 47**). The annual costs are calculated by summing annualised capital cost and operating cost.

Annual cost [€ yr^{-1}] = Annualised capital cost [€ yr^{-1}] + operating cost [€ yr^{-1}] Equation 3

Figure 41. Overview on the cost groups considered in the cost analysis.



12.1.2 Economies of scale

Costs are known to be subject to economies of scale. The cost of infrastructure and many types of equipment (e.g. vessels, engines) does not increase in the same ratio as size. In addition, the number of workers and thus labours cost does not increase in direct proportion to the capacity of a plant. To show this effect, cost calculations are carried out taking into account different treatment capacities for the considered sewage sludge treatment processes. Based on the knowledge of typical capacities for the different processes, costs are calculated for small, medium and large plants (**Table 18**).

Table 18. Treatment capacity for cost calculation of the selected sewage sludge treatment processes.

Sewage sludge treatment process	Treatment capacity in t DM yr^{-1}		
	small	**medium**	**large**
Composting (open, closed)	15k	40k	75k
Drying (solar, thermal)	10k	25k	50k
Co-incineration (Waste-to-energy)	100k	200k	300k
Mono-incineration	100k	200k	300k

12.1.3 Savings and revenues

The application of certain processes can result in savings within the whole process chain. The considered savings include, for example, reduced cost for mineral fertiliser due to sewage sludge application or reduced disposal costs for a mono-incinerator due to the conversion of the total sewage sludge ash into a fertiliser by a P recovery process.

Revenues can results from the production of electricity and heat through combustion of biogas from anaerobic digesters and landfills, incineration of sewage sludge and from the production of tradeable goods through certain processes (e.g. composting, P-recovery technologies). For electricity and heat, revenues are calculate by multiplying the produced energy (kWh yr^{-1}) with market prices.

Compared to energy, the nutrient content of organic and mineral materials cannot be simply multiplied with the associated market value. This is justified by the heterogeneity of the materials and the different agronomic efficiency of the contained nutrients. Therefore, revenues are calculated multiplying the nutrient content [t] of the material with the specific market value [€ t^{-1}] (**Table 19**), World Bank, 2022), involving the agronomic efficiency [-][23] of the nutrients N, P and K within the materials (Table 20; Huygens & Saveyn, 2018; Herzel et al., 2016; Herzel et al., 2022; Smol et al., 2020; Stemann et al., 2015; Tonini et al., 2019; You et al., 2021).

[23] Agronomic efficiency: Plant response (e.g. dry matter yield, P use efficiency) derived from the nutrients in organic and inorganic fertilising materials compared to plant response from nutrients from mined and synthetic P-fertilisers.

Table 19. Market value for P, N and K based on mineral fertilisers.

	P	N	K
Market value (€ kg^{-1})	low: 0.8 [±0.3] [24]; high: 2.2 [±0.9] [25]	1 [±0.6] [26]	0.55 [±0.15] [27]

Table 20. Agronomic efficiencies of organic and mineral fertiliser material.

Organic fertiliser material	**P**	**N**	**K**
Sewage sludge (AD, wet, dewatered)	0.55 [±0.05]	0.65 [±0.05]	0.50 [±0.10]
Compost	0.55 [±0.05]	0.65 [±0.05]	0.50 [±0.10]
Mineral fertiliser material	**P**	**N**	**K**
Untreated sewage sludge ash	0.25 [±0.15]	-	0.25 [±0.15]
Phosphoric acid	1.00	-	-
TSP based on SSA	1.00	-	1.00
Precipitated Calcium-Phosphate	0.85 [±0.15]	-	-
Rhenania-Phosphate based on SSA	0.80 [±0.10]	-	0.50 [±0 .10]
Fertiliser industry [SSP with SSA content]	1.00	-	1.00

12.2 Cost for sludge management options

12.2.1 Direct agricultural use

Literature data

Sewage sludge can be brought to land in wet condition with silo trailer (dry matter content of 3–5%) or after dewatering (dry matter content: 15–25%). Cost associated to direct agricultural land application are transport, storage and spreading on land. In some countries, farmers are paid by WWTP operators to accept the application of sewage sludge on their land, e.g. 100 € t DM^{-1} in Lithuania and 100–560 € t DM^{-1} in Germany (see Evaluation report). Typical cost for WWTP operators for the use of sewage sludge in agriculture are given in **Table 21**.

Table 21. Typical cost for direct agricultural sewage sludge application.

Country	**€ t DM^{-1}**		**Source**
	min	max	
Wet sewage sludge	100	175	(A. Amann et al., 2021)
	200	320	(Roskosch et al., 2018)
Dewatered sewage sludge	80	150	(A. Amann et al., 2021)
	125	175	(Roskosch et al., 2018)

12.2.2 Composting

Literature data

Composting is a process that is either done directly at the WWTP or at third party composting plants. Composting plants for sewage sludge reach from simple technologies with open windrow to more complex technologies with closed windrows and advanced exhaust gas treatment. Therefore, cost for sewage sludge composting can spread widely (**Table 22**).

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24 based on not immediately plant available P from raw phosphate rock (70–170 € t^{-1}, P_2O_5 content: 32%)
25 based on the immediately plant available P from triple superphosphate (270–630 € t^{-1}, P_2O_5 content: 46%)
26 based on KCl (200–360 € t^{-1}, K_2O content: 60%)
27 based on urea (210–810 € t^{-1}, N content: 47%)

Table 22. Literate data for sewage sludge composting.

Type of composting	€ t DM^{-1}		Source
	min	max	
Composting (all technologies)	100	500	Stakeholder consultation (see evaluation report)
	125	280	Eunomia 2002
	150	300	(A. Amann et al., 2021)

Data for cost calculation

To cover the broad spectrum of costs not only for plants with different capacities, but also different technological approaches, cost are calculated for an open and closed windrow composting plant. The cost are heavily dependent on the choice of technology but also the legal and quality constraints applied to the output. Therefore composting cost can vary strongly within the EU. Steinfelt et al 2002 published detailed cost analysis for open and closed composting, both approaches with or without post rotting (**Table 23**).

Table 23. Basic data for cost assessment of open and closed composting considering different plant sizes.

Type of composting	**Open**	**Closed**
CapEx (M€; small, medium, large)	4.5, 10, 17	9, 14, 22
Personal (PAX)	5, 7, 10	4
Resource demand		
- water (L t input^{-1})	250	100
- fuel (L t input^{-1})	1.7	0.3
- electricity (kwh t input^{-1})	0.5	50
Final compost (t t input^{-1})	0.4	0.4
Solid and liquid waste		
- solid residues (rejects) to landfill/incineration (t t input^{-1})	0.05	0.05
- waste water (L t input^{-1})	25	200

Revenues

Even though, high revenues can be achieved for specific compost products (e.g. small bags for hobby gardeners or wholesalers), the revenues for compost for the large area use in agriculture or landscaping can be in the range from 1.3 to 10 € t^{-1} (Steinmann and Noell, 2000; Hogg et al., 2002; Saveyn and Eder, 2014). For the economic assessment, a revenue of 5.5 € t^{-1} compost (=8.5 € m^{-3}) is assumed.

Result for open composting

Table 24. Cost distribution and potential revenues for open windrow composting considering different plant sizes.

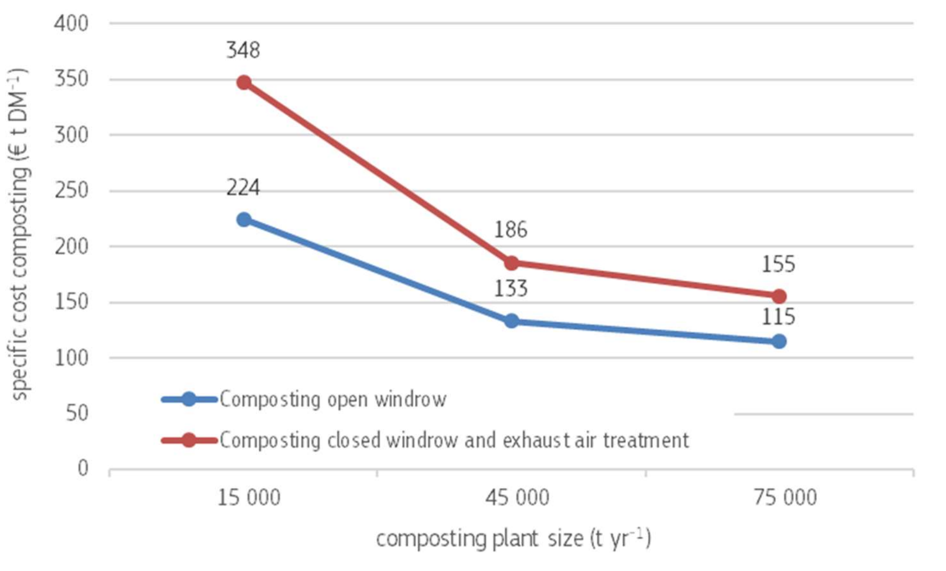
Cost position open composting	**Plant size: 15k t yr^{-1}**		**Plant size: 45k t yr^{-1}**		**Plant size: 75k t yr^{-1}**	
	€ t DM^{-1}	%	€ t DM^{-1}	%	€ t DM^{-1}	%
Capital cost	83.5	37%	69.6	52%	15.7	55%
Labour cost	80.2	36%	42.1	32%	8.0	28%
Maintenance & insurance	30.0	13%	20.0	15%	4.5	16%
Operational cost (energy, resources)	9.4	4%	0.3	0%	0.1	0%
Operational cost (disposal cost)	21.2	9%	1.2	1%	0.3	1%
Total	**224.5**	**100%**	**133.3**	**100%**	**28.7**	**100%**
Revenues	**8**		**8**		**8**	

Result for closed composting

Table 25. Cost distribution and potential revenues for closed windrow composting considering different plant sizes.

Cost position closed composting	Plant size: 15k t yr^{-1}		Plant size: 45k t yr^{-1}		Plant size: 75k t yr^{-1}	
	€ t DM^{-1}	%	€ t DM^{-1}	%	€ t DM^{-1}	%
Capital cost	167.0	48%	97.5	52%	81.7	53%
Labour cost	80.2	23%	42.1	23%	32.1	21%
Maintenance & insurance	60.0	17%	28.0	15%	23.5	15%
Operational cost (energy, resources)	10.7	3%	8.6	5%	8.6	6%
Operational cost (disposal cost)	29.60	9%	9.6	5%	9.6	6%
Total	**347.7**	**100%**	**185.8**	**100%**	**155.4**	**100%**
Revenues	**8**		**8**		**8**	

Figure 42. Specific cost for sewage sludge composting depending on composting technology and plant size (€ t DM^{-1}).



12.2.3 Solar and thermal drying

Literature data

Compared to thermal drying, solar drying is a simple technology but requires a lot of space (0.6-2.0 m^2 t dewatered sewage sludge^{-1}. Greater space is needed for regions with lower solar radiation. Solar drying can be combined with an external heat source, reducing the drying time and space demand (0.15-0.35 m^2 t dewatered sewage sludge^{-1}; e.g. Murcia (ES) and Bottrop (DE); (Thermo-Systems, 2022)) and consequently (investment) cost. For thermal drying, a variety of technologies exist (e.g. drum dryer, disc dryer, belt dryer). Perhaps the main explanation for the wide range of cost for thermal drying is the source of energy used for drying. For example, waste heat is much cheaper than primary energy sources as gas or oil (

Table **26**).

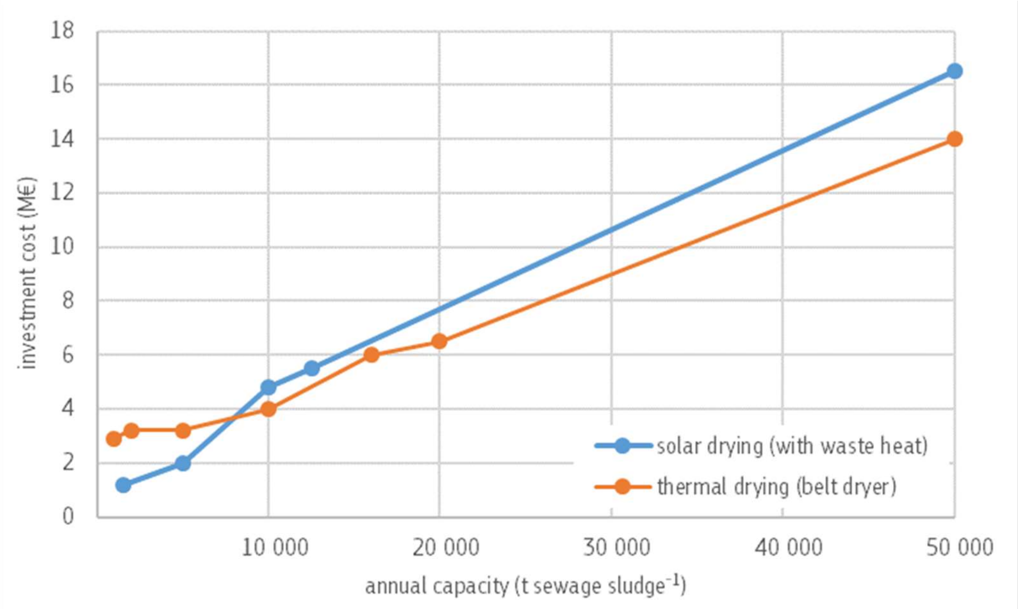
Table 26. Literature data for sewage sludge for solar- and thermal drying.

Type of drying	€ t DM^{-1}		Source
	min	max	
Solar drying	270	360	(Roskosch et al., 2018)
	140	230	(Kurt et al., 2015)
Thermal drying	255	715	(Bratina et al., 2016)
	150	500	Stakeholder consultation (see evaluation report)
	220	360	(Roskosch et al., 2018)

Data for cost calculation

Based on available literature data, the following investment costs are used for the calculation of the annual cost: solar drying: 5, 9.5 and 16 M€; thermal drying: 4, 8, 14 M€ (**Figure 43**). The personnel requirement for drying is minor and the facilities are managed by the existing operating staff. For the cost calculation, therefore, a maximum of one person was assumed to be in charge of the facilities (**Table 27**).

Figure 43. Investment cost for solar- and thermal drying in relation to the plant capacity.



To cover the broad spectrum of costs not only for drying plants with different capacities (10k, 25k, 50k t yr^{-1}), but also different technological approaches, cost are calculated for solar- and thermal drying plant (belt dryer). The cost are heavily dependent on the choice of technology and used energy source, especially for thermal drying. For this cost calculation, gas was assumed as energy source **Table 27**. If sewage sludge is dried thermally, an ammonium (~200–3 000 mg NH_4-N L^{-1}) and carbon, expressed as chemical oxygen demand (COD) (1 000–10 000 mg COD L^{-1}) rich condensate and exhaust air is produced (e.g. belt dryer: up to 80 000 m^3 h^{-1}; contact dryer: >100 000 m^3 h^{-1}). For solar drying, higher cost for the site acquisition was considered (Bux and Baumann, 2003; Geyer, 2013; Kurt, 2014; Kurt et al., 2015; Wolf, 2019; RePhoNOH, 2021).

Table 27. Basic data for cost assessment of solar- and thermal drying and different plant sizes.

Type of drying	Solar	Thermal
CapEx (M€; small, medium, large)	5; 9.5; 16	4, 8, 14
Personal (PAX)	1; 0.75; 0.5	1; 0.75; 0.5
Space demand (m^2 t^{-1})	0.8	-
Water evaporation		
Water content input	25% DM	25% DM
Water content output	65% DM	90% DM
Water to condensate (t H_2O t dewatered sewage sludge^{-1})	0.4	0.7
Evaporation rate (t H_2O m^{-2} yr^{-1})	0.8	-

Resource demand		
- electricity (kwh t input^{-1})	2.2	52
- gas (MJ t input^{-1})	-	553
Solid and liquid waste		
- solid residues (t t input^{-1})	-	-
- waste water (% of water in input sewage sludge)	10%	100%
- waste water (mg NH_4-N)	-	1 100
- waste water (mg COD)	-	3 500

Result for solar drying

Table 28. Cost distribution and potential revenues for solar drying considering different plant sizes.

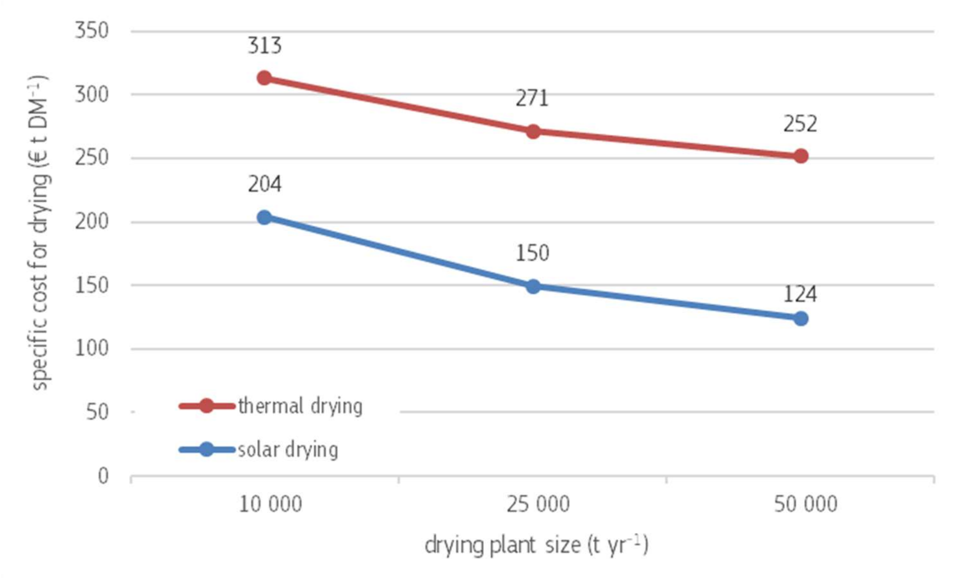
Cost position solar drying	Plant size: 10k t yr^{-1}		Plant size: 25k t yr^{-1}		Plant size: 50k t yr^{-1}	
	€ t DM^{-1}	%	€ t DM^{-1}	%	€ t DM^{-1}	%
Capital cost	139.2	68%	105.8	71%	89.1	72%
Labour cost	24.1	12%	18.1	12%	12.0	10%
Maintenance & insurance	30.0	15%	15.2	10%	12.8	10%
Operational cost (energy, resources)	8.6	4%	8.6	6%	8.6	7%
Operational cost (disposal cost)	1.9	1%	1.9	1%	1.9	2%
Total	**203.8**	**100%**	**149.6**	**100%**	**124.5**	**100%**

Result for thermal drying

Table 29. Cost distribution and potential revenues for thermal drying considering different plant sizes.

Cost position thermal drying	Plant size: 10k t yr^{-1}		Plant size: 25k t yr^{-1}		Plant size: 50k t yr^{-1}	
	€ t DM^{-1}	%	€ t DM^{-1}	%	€ t DM^{-1}	%
Capital cost	122.3	39%	97.8	36%	85.6	34%
Labour cost	24.1	8%	18.1	7%	12.0	5%
Maintenance & insurance	24.0	8%	12.8	5%	11.2	4%
Operational cost (energy, resources)	111.5	36%	111.5	41%	111.5	44%
Operational cost (disposal cost)	31.2	10%	31.2	11%	31.2	12%
Total	**313.1**	**100%**	**271.4**	**100%**	**251.6**	**100%**

Figure 44. Specific cost for sewage sludge drying depending on the technological approach and plant size (€ t DM^{-1}).



12.2.4 Co-incineration

Literature data

In Europe, sewage sludge is incinerated in coal-fired power plants, cement kilns and waste-to-energy (WtE) plants in dewatered and dried form. In comparison to the other sewage sludge management options, no separate cost calculation were performed for the co-incineration in coal- and cement plants, as sewage sludge represents only a minor part of the fuel input and an allocation of cost to the sewage sludge is difficult. As shown in **Table 30**, the treatment cost differ strongly dependent on the water content of the sewage sludge and the type of co-incineration.

Table 30. Typical cost for thermal sewage sludge treatment (co-incineration).

Sewage sludge co-incineration		Dewatered sludge (€ t DM^{-1}) (25-45% DM)			Dried sludge (€ t DM^{-1}) (>85% DM)			Source
		min	max	average	min	max	average	
Coal power plants	brown coal	50	75	208	-	-	-	(Wiechmann et al., 2013)
	stone coal	75	100	292	80	130	124	(Wiechmann et al., 2013)
	coal	-	60	200	-	-		(RePhoNOH, 2021)
Cement industry	cement kilns I	35	75	183	25	100	74	(Montag et al., 2014)
	cement kilns II	-	-	-	90	100	112	(Wiechmann et al., 2013)
	cement kilns III	55	75	217	-	-	-	(RePhoNOH, 2021)
WtE	WtE I	80	100	300	-	-	-	(Wiechmann et al., 2013)
	WtE II	80	140	367	-	-	-	(Montag et al., 2014)

Data for cost calculation

Annual costs are calculated for co-incineration plant with different annual capacities of 100k, 200k and 300k t waste input. Based on available literature data presented in Figure 45, the following investment costs are used for the calculation of the annual cost: 50, 75 and 110 M€. The personnel requirement is assumed to be 20, 22 and 25 full time equivalents. As sewage sludge is usually not the main waste input of WtE plants, no sewage sludge specific cost are indicated for WtE.

Figure 45. Investment cost for co-incineration WtE in relation to the plant capacity.

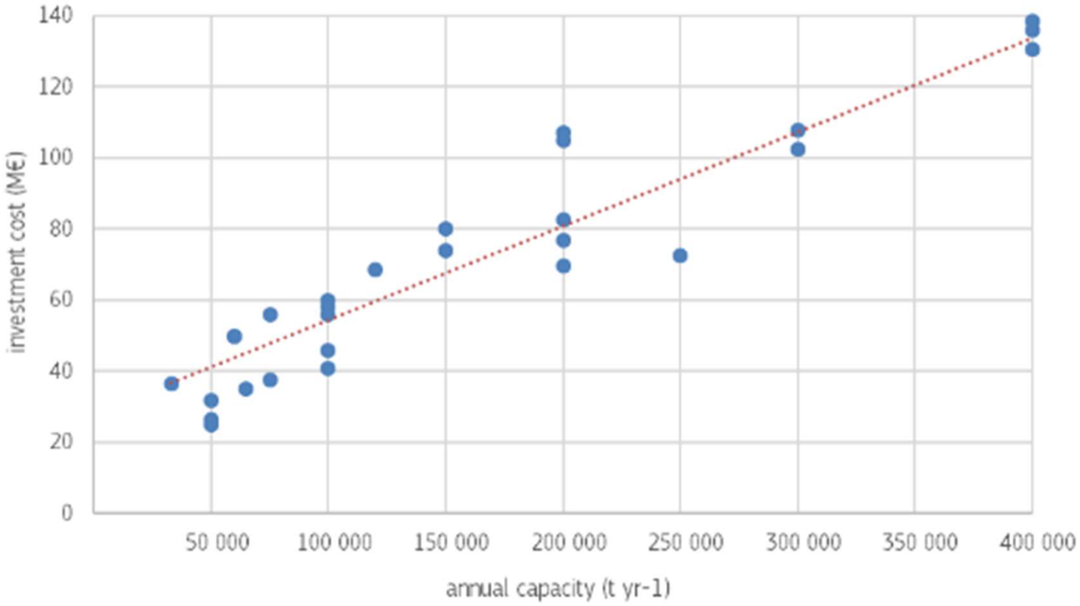


Table 31. Resource demand, waste occurrence and energy recovery for co-incineration in WtE.

Resource demand	**€**
- electricity (kWh t input^{-1})	63.3
- natural gas (MJ t input^{-1})	118
- water (kg t input^{-1})	42.5
- NH_3 (25%) (kg t input^{-1})	6
- $CaCO_3$ (kg t input^{-1})	7.1
- activated carbon (kg t input^{-1})	1
Solid and liquid waste	
- fly ash (kg t input^{-1})	124
- bottom ash (kg t input^{-1})	14
- flue gas treatment residues (kg t input^{-1})	1.5
- waste water (L t input^{-1})	42.5
Energy recovery	
- electricity (efficiency) (%)	30
- electricity (kWh t input^{-1})	59
- heat recovery (efficiency) (%)	55
- net heat recovery (kWh t input^{-1})	200

Result

Table 32. Cost distribution and potential revenues for co-incineration considering different plant sizes.

Cost position co-incineration	Plant size: 100k t yr^{-1}		Plant size: 200k t yr^{-1}		Plant size: 300k t yr^{-1}	
	€ t DM^{-1}	%	€ t DM^{-1}	%	€ t DM^{-1}	%
Capital cost	150.6	48	113.0	46	111.1	47
Labour cost	48.2	15	26.5	11	20.1	9
Maintenance & insurance	50.0	16	37.5	15	36.7	16
Operational cost (energy, resources)	33.8	11	33.8	14	37.4	16
Operational cost (disposal cost)	32.3	10	32.3	13	29.1	12
Total	**314.9**	**100**	**243.1**	**100**	**234.4**	**100**
Revenues from energy recovery (based on municipal solid waste input)	**78.8**		**78.8**		**78.8**	

Compared to sewage sludge mono-incineration plants (see Table 35), the revenues from energy recovery are significantly higher, as the main input of WtE incinerator is municipal solid waste with a high energy content (~11 MJ kg^{-1}). Sewage sludge, due to its characteristics (high water and low energy content, high fine ash content), is not a typical and often unwanted input for WtE plants and the share of sewage sludge in the input is low (<5%).

12.2.5 Mono-incineration

Literature data

Sewage sludge mono-incineration in dewatered, semi-dried and dried form is already state of the art and is fluidised bed reactor is the technology of choice for sewage sludge incineration. In comparison co-incineration plants, sewage sludge is the only or main waste input in mono-incineration plants. As shown in **Table 33**, the treatment cost differ strongly and the main cost driver is the capacity of the mono-incinerator. (A. Amann et al., 2021 reports mono-incineration cost of 510 € t DM^{-1} for an incinerator with a capacity of 2kt DM yr^{-1} and 160 € t DM^{-1} for treatment capacity of 35kt DM yr^{-1}. No specific data is available for cost for the incineration of dried sewage sludge. Reason could be, that the drying is performed at the incineration plant itself with the produced excess heat.

Table 33. Typical cost for thermal sewage sludge treatment (mono-incineration).

Country	€ t DM^{-1}		Source
Austria	160	510	(A. Amann et al., 2021)
Germany	180	400	(Wiechmann et al., 2013)
Germany	280	480	(Roskosch et al., 2018)
Italy	360		(Castorini, 2021)

Data for cost calculation

Annual costs are calculated for mono-incineration plant with different annual capacities of 100kt, 200kt and 300kt dewatered sewage sludge input (25% DM). Based on available literature data presented in **Figure 46**, the following investment costs are used for the calculation of the annual cost: 55, 80 and 100 M€. The personnel requirement is assumed to be 20, 22 and 25 full time equivalents. The median investment cost for 1 t DM sewage sludge is ~1 700 € (25% percentile: 1 550 € t DM^{-1}: 75% percentile: 2 280 € t DM^{-1}). Typically, the incineration capacity of an incineration line for fluidized bed reactors is 20–30kt DM per year.

Figure 46. Investment cost for mono-incinerators in relation to the plant capacity.

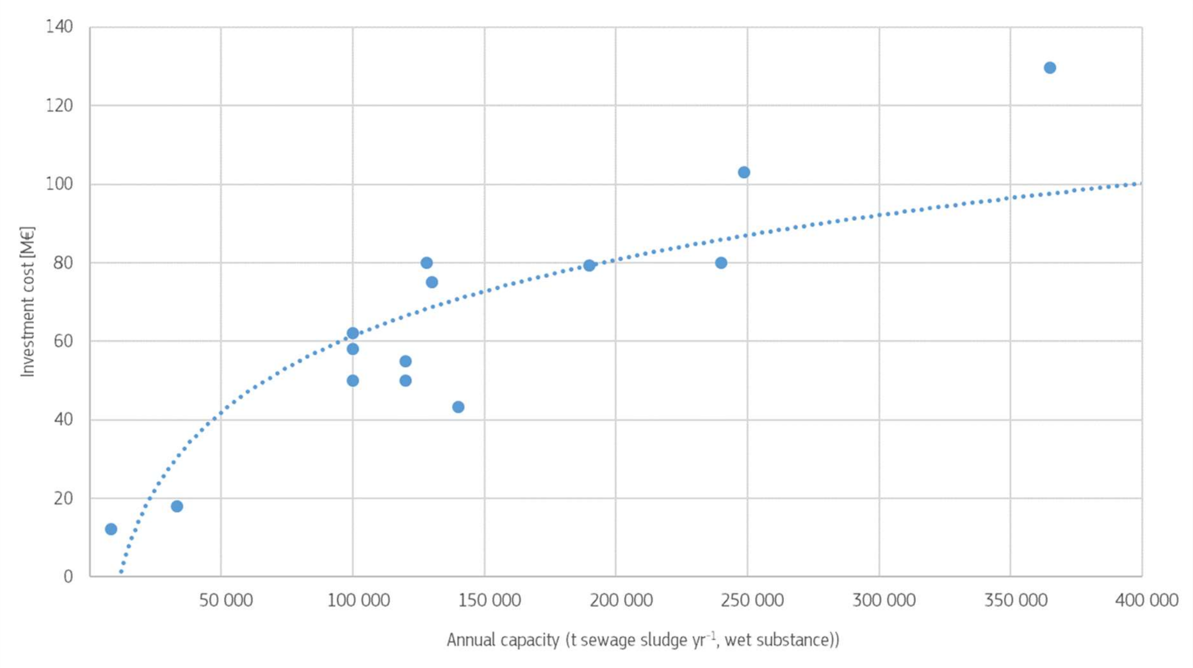


Table 34. Resource demand, waste occurrence and energy recovery for mono-incineration.

Resource demand	**€**
- electricity (kWh t input^{-1})	63.3
- natural gas (MJ t input^{-1})	118
- water (kg t input^{-1})	42.5
- NH_3 (25%) (kg t input^{-1})	6
- $CaCO_3$ (kg t input^{-1})	7.1
- activated carbon (kg t input^{-1})	1
Solid and liquid waste	
- fly ash (kg t input^{-1})	138
- bottom ash (kg t input^{-1})	25
- flue gas treatment residues (kg t input^{-1})	1.5
- waste water (L t input^{-1})	42.5
Energy recovery	
- electricity (efficiency) (%)	30
- electricity (kWh t input^{-1})	59
- heat recovery (efficiency) (%)	55
- net heat recovery (kWh t input^{-1})	200

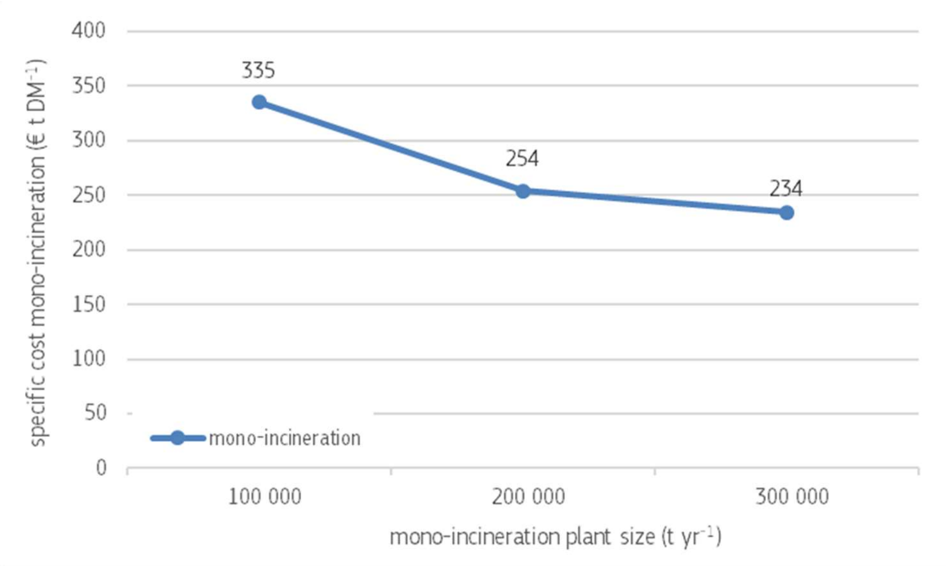
Result for mono-incineration

Table 35. Cost distribution and potential revenues for mono-incineration considering different plant sizes.

Cost position mono-incineration	**Plant size: 100kt yr^{-1}**		**Plant size: 200kt yr^{-1}**		**Plant size: 300kt yr^{-1}**	
	€ t DM^{-1}	%	€ t DM^{-1}	%	€ t DM^{-1}	%
Capital cost	165.7	49%	121.2	48%	111.1	47%
Labour cost	48.2	14%	26.5	10%	20.1	9%
Maintenance & insurance	55.0	16%	40.0	16%	36.7	16%
Operational cost (energy, resources)	37.4	11%	37.4	15%	37.4	16%
Operational cost (disposal cost)	29.1	9%	29.1	11%	29.1	12%
Total	335.4	100%	254.2	100%	234.4	100%
revenues from energy recovery	**76.2**		**76.2**		**76.2**	

In addition to the revenues from the energy recovery (electricity and heat), also savings could be possible in the future in case that sewage sludge ash no longer needs to be deposited and could be used in the fertiliser and/or recycling industry instead.

Figure 47. Specific cost for sewage sludge mono-incineration depending on plant size (€ t DM^{-1})



12.2.6 P-recovery from sewage sludge ash

To cover the various approaches the following P recovery technologies are considered:

— Acid wet chemical leaching and production of phosphoric acid or solid calcium phosphate

Aim is the transformation of P in different uniformly usable and marketable forms (e.g. phosphoric acid, calcium phosphates). P is leached with mineral acids (e.g. hypochloric, sulphuric or phosphoric acid) and as such separated from the ash. Depending on the technological approach, metals are removed by e.g. ion-exchange, liquid-liquid separation or precipitation. Certain technologies also aim for the recovery of iron- and aluminium as iron- and aluminium salts, which can be used as by-products (e.g. as coagulants at WWTP). Due to the specific removal processes, the P-rich output materials contain less contaminants.

— Acid wet chemical extraction and production of a SSA based single- and triple-super-phosphate (TSP):

Transformation of the P from SSA into a plant available form by mixing the ash with mineral acids as e.g. sulphuric or phosphoric acid. All the other compounds of the ash are fully incorporated into the fertiliser, so no removal of contaminants takes places. The fertiliser industry follows this approach to produce single-or triple-superphosphate from raw phosphate rock and could use a limited percentage of SSA to substitute raw phosphate rock.

— Thermo-chemical treatment and production of SSA with improved bio-availability (similar to Rhenania-phosphate):

Aim of this approach is the partial removal of metals and the transformation of the P into a better plant available form. This can be achieved by adding chlorine and a treatment temperature of 750–1 000 ºC (below ash melting temperatures). Latest developments of this technology focus on the further improvement of the plant availability of SSA by adding sodium compounds instead of chloride, with the trade-off of significantly lower metal removal.

Data for cost calculation

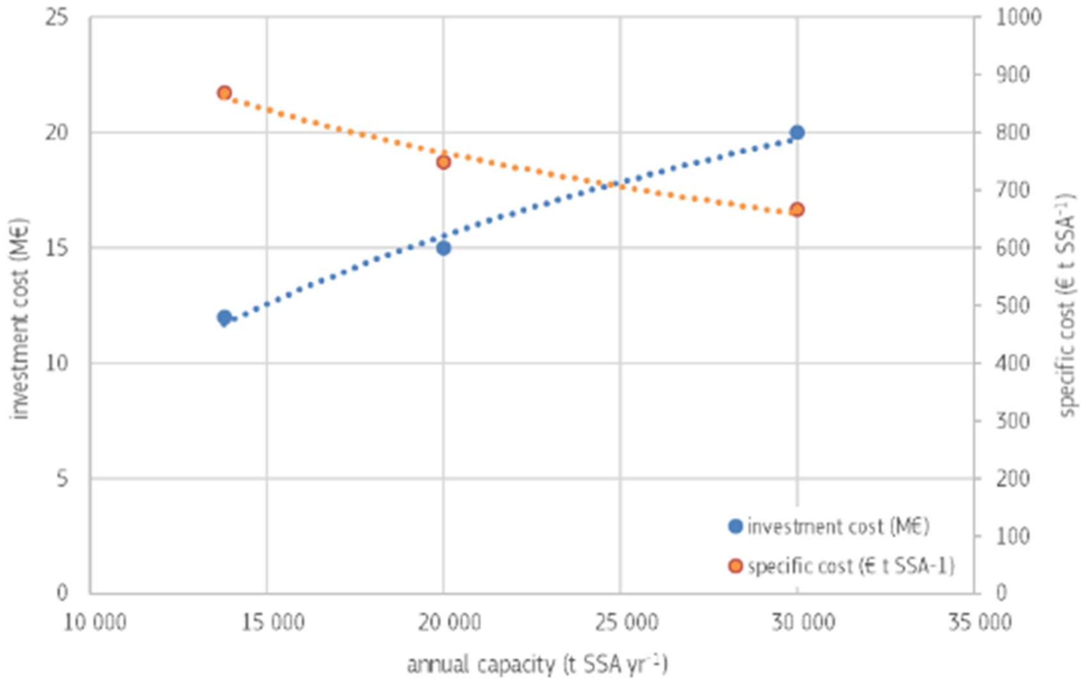
Currently, only a handful of P recovery technologies are full scale implemented or will be implemented at an industrial scale in the upcoming years. The two in the EU full scale implemented recovery plants operate with an annual capacity of 20kt and 30kt of SSA, respectively. Feasibility studies for upcoming technologies include annual capacities of around 30kt SSA, therefore, due to a lack of data, cost calculation is only performed for this capacity. The investment costs of the recovery processes vary according to the technology used. Particularly for those technologies that have not yet been implemented on a large scale, it is necessary to take into account the corresponding uncertainties in the investment costs. The same applies to personnel requirements. Based on the available literate data, the following assumptions were made (Table 36):

Table 36. Investment cost for P-recovery technologies and personal demand in relation to the plant capacity.

Technology	**Capacity [t yr^{-1}]**	**Investment cost [M€]**	**Employees [full time equivalents]**	**Space demand [m^2]**	**TRL**
Fertiliser industry	Depending on the process and capacity of the plant	<1 (only for storage silos and plant components to ash to the process)	<1	<100	9
Acid wet chemical leaching with P-acid production	20kt	17.5	5	12kt	9
Acid wet chemical leaching with precipitated phosphate production	30kt	30.0	5	12kt	7-8
Acid wet chemical extraction	30kt	22.5	20	15kt	9
Thermo-chemical treatment	30kt	20.0	10	12kt	7

As **Figure 48** indicates, the size of the recovery plant can have significant influence on the cost.

Figure 48. Investment cost for thermal-chemical P recovery in relation to the plant capacity.



12.2.6.1 *Resource demand, material output and waste occurrence for the selected P recovery technologies*

Technology data have been collect from available literature that describe a set of different technologies. It is noted, that the numbers provided to not correspond to real technologies.

Table 37. Resource demand, material output and waste occurrence for P-recovery technologies.

Technological approach		Acid wet chemical extraction	Acid wet chemical leaching	Acid wet chemical leaching	Acid wet chemical extraction	Thermo-chemical
Recovered P rich material		*Single super-phosphate (SSP)*[28] or Triple-super-phosphate (TSP)	Phosphoric acid	Precipitated calcium phosphate (PCP)	Triple-super-phosphate (TSP)	SSA with improved bioavailability
Resource demand						
- electricity	kWh t SSA^{-1}	0.07	190	633.3	273	70
- oil	t t SSA^{-1}	0.17	-	-	-	-
- natural gas	kWh t SSA^{-1}	-	-	-	-	500
- steam	kWh t SSA^{-1}	-	720	266.7	-	-
- process water	m^3 t SSA^{-1}	0.15	5.0	3.17	0.28	0.60
- HCl (36%)	t t SSA^{-1}	-	0.33	0.97	-	-
- H_2SO_4 (96%)	t t SSA^{-1}	0.55	0.21	-	-	-
- H_3PO_4 (85%)	t t SSA^{-1}	-	-	-	0.70	-
- CaO (dry)	t t SSA^{-1}	-	-	0.32	-	-
- $Ca(OH)_2$	t t SSA^{-1}	-	-	-	-	0.008
- NaOH (50%)	t t SSA^{-1}	-	-	0.18	-	0.008
- $Al(OH)_3$ (dry)	t t SSA^{-1}	-	-	0.02	-	-
- resin	kg t SSA^{-1}	-	0.011	-	-	-
- Na_2CO_3, $NaHCO_3$, $NaSO_4$[29]	t t SSA^{-1}	-	-	-	-	0.24, 0.36, 0.40
Material output						
Recovered P rich material	t t SSA^{-1}	1.35 Single-super-phosphate (SSP, 8.7%)	0.35 Phosphoric acid (75%, 24% P)	0.53 Precipitated calcium phosphate (16.5% P, 95% DM)	1.71 Triple-super-phosphate (TSP, 16.6 %)	1.05 Rhenania-phosphate (7-9% P)[30]
$FeCl_3$ (??% DS)	t t SSA^{-1}	-	0.19	-	-	-
$FeCl_3$ (15% DS)	t t SSA^{-1}	-	-	0.30	-	-
$AlCl_3$ (15% DS)	t t SSA^{-1}	-	0.12	-	-	-
$NaAlO_2$ (38% DS)	t t SSA^{-1}	-	-	0.60	-	-
Gypsum	t t SSA^{-1}	-	0.54	-	-	-
Solid and liquid waste						
- silica sand	t t SSA^{-1}	-	0.75 (65% DM)	0.87 (55% DM)	-	-
- heavy metal cake (wet)	t t SSA^{-1}	-	0.02 (?% DM)	0.10 (45% DM)	-	-

[28] Data presented for the production of single-super-phosphate
[29] Different sodium sources possible for the thermo-chemical processes
[30] P content in the final material depends on the P content of the input SSA

- heavy metal cake concentrate (dry)	t t SSA^{-1}	-	-	-	-	0.04
- waste water	m^3 t SSA^{-1}	0.45	3.00	3.50	-	-
Literature source		(Egle et al., 2016; Amann et al., 2018)	(Everding and Montag, n.d.; Hanssen et al., 2016; Lebek et al., 2018; Rak, 2018)	(Cohen, 2018; DPP, 2019; Easymining, 2022)	(Amann et al., 2018; ICL-Fertilizers, 2019; Kirchhof and Brumme, 2020; Seraplant, 2021)	(Adam et al., 2008; Egle et al., 2016; Herzel et al., 2016, 2021; Everding and Montag, 2017; Smol et al., 2020)

Result for P-recovery technologies

Table 38. Cost distribution and potential revenues for P-recovery.

Cost position P-recycling	Acid wet chemical leaching		Acid wet chemical leaching		Acid wet chemical extraction		Thermo-chemical	
	€ t SSA^{-1}	%	€ t SSA^{-1}	%	€ t SSA^{-1}	%	€ t SSA^{-1}	%
Capital cost	65.8	23	75.2	24	47.3	9	50.2	28
Labour cost	15.0	5	10.0	3	34.4	7	20.1	11
Maintenance & insurance	21.9	8	25.0	8	15.7	3	16.7	9
Operational cost (energy, resources)	144.2	50	4136.7	43	408.8	81	55.8	32
Operational cost (disposal cost)	42.5	15	67.5	21	0.0	0	8.8	5
Total	**289.4**	**100**	**314.4**	**100**	**506.2**	**100**	**151.4**	
Revenue low P market value [31,32]	**245.0**		**131.2**		**425**		**89.3**	
Revenue high P market value	**332.5**		**179.2**		**568**		**178.5**	
€ kg P^{-1} (without revenues)	**3.8**		**3.5**		**5.6**		**2.0**	
€ kg P^{-1} (incl. highest revenue)	**-0.6**		**1.5**		**-0.7**		**-0.1**	

12.2.7 Landfill

Literature data

There are two possibilities in terms of sewage sludge landfilling:

- mono-landfill, where only sewage sludge is disposed of, and
- mixed-landfill (most commonly observed), when the landfill is also used for municipal solid wastes

In mixed deposit with municipal solid wastes, sewage sludge is not the principal ingredient. Its proportion reaches usually 10 to 25% of the total deposit (EPA USA, 2003; O'kelly, 2005). Therefore, the cost for municipal solid waste landfilling was applied for sewage sludge. The cost are calculated for 1 t sewage sludge DM, assuming that sewage sludge has a water content of 25%. The following costs were extracted from the EEA report ((EEA, 2013a, 2013b):

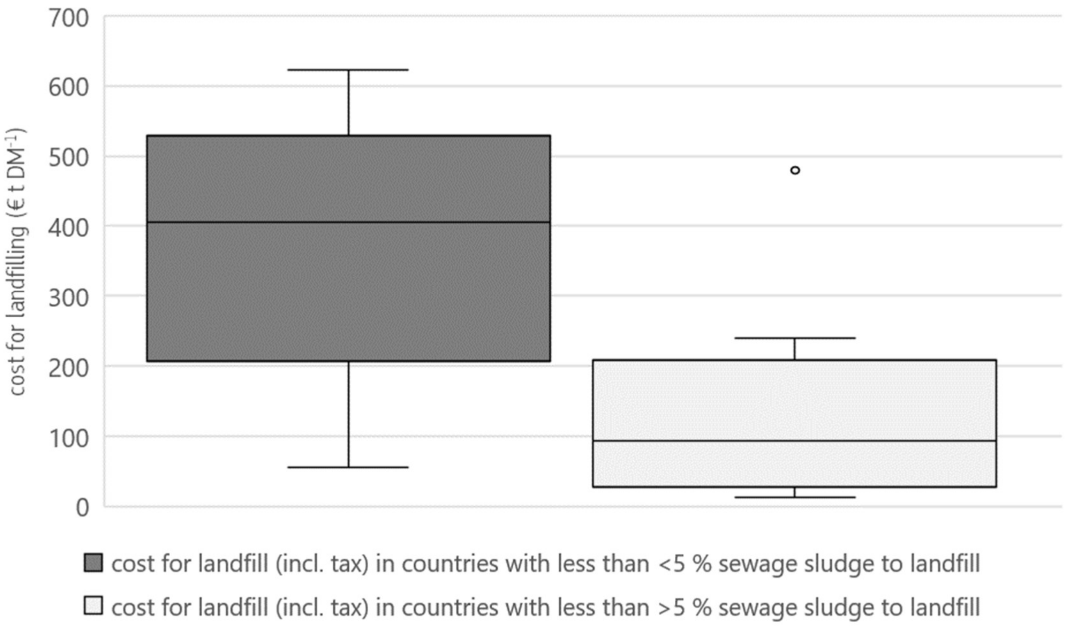
Table 39. Cost for landfilling of sewage sludge in EU-27 member states.

Landfill cost in MS with <5% sewage sludge to landfill					**Landfill cost in MS with >5% sewage sludge to landfill**				
Country	Gate fee without tax	Landfill tax	Total	Sludge to landfill	Country	Gate fee without tax	Landfill tax	Total	Sludge to landfill
AT	280	104	384	N	BG	no data	12	12	Y (25%)
BE-Fl	200	328	528	N	ES	120	120	240	Y (10%)
BE-Wal	200	260	460	N	ES-Cat	131	50	181	Y (10%)
CY	224	no data	224	N	HR	62		62	Y (>90%)
DK	176	252	428	Y (<5%)	CZ	64	80	144	Y (<10%)
FI	238	120	358	Y (<5%)	EE	160	48	208	Y (10%)
FR	242	80	322	N	GR	94	no data	94	Y (30%)
DE	560	no data	560	Y (<5%)	IT	360	120	480	Y (50%)
HU	140	no data	140	N	MA	80	no data	80	Y (100%)
IE	280	200	480	N	RO	15	no data	15	Y (70%)
LT	65	no data	65	Y (<5%)	SK	27	no data	27	Y (10%)
LV	120	32	152	N					
LU	598	no data	598	N					
NL	100	429.96	530	Y (<5%)					
PL	278	106	384	Y (<5%)					

[31] Market values for P are given in Table 19
[32] Including revenues from other recovered materials (e.g. iron and aluminium compounds)

PT	42	14	56	N					
SE	426	196	622	Y (<5%)					
SL	422	44	466	N					

Figure 49. Overview cost for landfilling of sewage sludge in EU-27 member states.



Data for cost calculation

To calculate the cost for landfilling of sewage sludge, two different sources of data were used (Martinez-Sanchez et al., 2015) published data considering investment cost of 26.7 M€. The manpower to operate a landfill is 13 200 man*h yr^{-1} and the specification and resource demand given is in **Table 40**. Furthermore, detailed cost data from (Damgaard et al., 2011) were applied to perform the cost assessment (**Table 41**).

Table 40. Landfill characteristics and resource demand for landfilling.

Landfill characteristics	**€**
- land demand (m^2)	300 000
- depth (m)	15
- in-place density (t m^{-3})	0.9
- usage rate (t yr^{-1})	100 000
- expected filling time (yr)	40
Resource demand	**€**
- daily cover soil (m^3 m^{-2})	0.2
- vegetation (plant m^{-2})	1.0
- topsoil / compost (m^3 m^{-2})	0.2
- soil (m^3 m^{-2})	0.8
- gravel for drainaging layer (m^3 m^{-2})	0.2
- leachate and hauling treatment (m^3 m^{-2})	13.5
- diesel (L m^{-2})	0.33
- electricity (kWh yr^{-1})	100 000

Table 41. Cost position for landfilling.

Landfill characteristics	**€**
- depth (m)	12.5
- usage rate (t yr^{-1})	50 000

Cost position	€ t input DM^{-1} (uncertainty)
- baseline cost	160
- top cover	12 (±4)
- bottom liner	16 (±4)
- leachate collection	10 (±0.5)
- leachate treatment	45 (±8)
- gas collection	4
- bio filter	0.4
- flare	0.6
- electricity plant	8 (±0.5)
- heat plant	8

Result

Based on the two data sets used, the cost for landfilling of sewage sludge are 150 to 260 (±20) € t DM^{-1}. This calculated cost are in the cost range of EU-27 MS that still landfill more than 5% of the sewage sludge (Table 39). For countries with a landfill rate of less than 5%, landfill cost are significantly higher.

12.2.8 Transport

12.2.8.1 Transport distances

In a recent study which serves a basis for the development of a nationwide P-recycling strategy in Austria, WWTP operators were questioned about the one-way transport distance in relation to the sewage sludge utilization, treatment and disposal (Arabel Amann et al., 2021). The survey reveals, that wet sewage sludge (3-5% DM) is at maximum transported over 10 km (median: 4 km). Agricultural disposal of dewatered sewage sludge (15-35% DM) and composting at the WWTP site rarely exceeded 20 km (median: 15 and 0.25 km, respectively). For external composting transport distances above 100 km are rare (median: 50 km) and for incineration the distances can be as twice as high (median: 120 km; ranging from 0-375 km). With regard to possible P-recovery strategies from sewage sludge ash, the transport distances can increase, as centralised mono-incinerators are installed to produce SSA suitable for P recycling. However, for WWTP that produce amounts of sewage sludge for operating an on-site mono-incinerator, the transport distances can come to 0 km.

12.2.8.2 Transport cost

Literature data

When looking at the costs for sewage sludge management options quoted in literature, it is often not clear whether transport costs are included or not.

Table 42. Typical cost for sewage sludge transport.

Type of transport	€ km^{-1}	Source
Truck (unknown capacity)	0.3	Diaz, Gracia and Canziani, 2015
Truck (unknown capacity)	1.3-1.6	(Bratina et al., 2016)
Truck (30 t) - Eastern Europe - Western Europe	 1.2 1.6	(BME, 2022; TI et al., 2022)

Data for cost calculation

Average fuel demand for a tractor with 10-20 t sewage trailer is around 0.03 L km t^{-1}. Fuel accounts for around 40 % of the overall cost for the operation of a tractor. For the cost calculation it was assumed, that a tractor is equipped with a trailer (silo or spreader) to transport 14 t of wet or dewatered sewage sludge to the agricultural field. The transport distance to agriculture was assumed with 7.5 km for one direction (15 km two ways). If dewatered sewage sludge is transported over longer distances, trucks with a capacity of 30 t are used, assuming cost of 1.5 € per km.

Result

As an example, the costs for the transport of sewage sludge to agriculture were calculated. Especially in the case of transporting wet sewage sludge by tractor, the transport costs per ton of dewatered sewage sludge are several factors higher than for dewatered sewage sludge (Table 43). In the case of wet sewage sludge application, transport costs can account for about 50% of the total cost of agricultural use (Table 21). The cheapest option by far is the transport of sewage sludge with trucks (approx. 3-4€ DM^{-1}). For this variant, however, additional costs for reloading sewage sludge onto agricultural spreaders must be taken into account. Exemplarily, if sewage sludge is transported over 120 km (one way) to an incineration plant, in addition to the incineration costs, about 24€ DM^{-1} transport cost must be included. This corresponds, for example, to about 7-10% of mono-incineration costs (Table 35).

Table 43. Cost for transporting sewage sludge to agriculture.

Type of transport	€ t DM
Truck (15 km, 30 t)	3-4
Truck (240 km, 30 t)	24
Tractor with trailer (15 km, 14 t) - wet sludge (3% DM) - dewatered sludge (25% DM)	 60-70 8-10

Table 44. Assumed cost for the different sludge management options to perform the cost for policy options (€ t DM^{-1} including CapEx and OpEx).

Sewage sludge management	€ t DM^{-1}
Agriculture	150
Composting	200
Co-incineration	250
Mono-incineration	350
P-recovery	500
Landfill	country specific
Other	200

Table 45. Cost for landfilling of sludge (€ t DM^{-1} including country specific taxes).

Country	€ t DM^{-1}	Country	€ t DM^{-1}
BG	12	IT	480
CZ	144	LT	65
EE	208	MA	80
GR	94	RO	15
ES	480	SK	27
HR	62		

12.2.9 Assumed cost for sludge management options for baseline and policy options

Table 46. Cost for sludge management options (€ t DM^{-1} incl. CapEx and OpEx).

Sludge management	€ t DM^{-1}
Agriculture	150
Composting	200
Co-incineration	250
Mono-incineration	350
P-recovery	500
Landfill	country specific
Other	200

12.3 Cost for energy, resources and labour

Table 47. Summary table for the cost of energy, resources and chemical, waste disposal, labour and revenues for sewage sludge management option outputs.

Energy	**€**	**Unit**	**Liquid and solid waste disposal**	**€**	**Unit**
Electricity	0.086	€ kWh^{-1}	Waste water	0.012	€ m^{-3}
Gas (CH_4)	0.011	€ MJ^{-1}	Incineration	100	€ t^{-1}
Diesel	1.16	€ L^{-1}	Non-hazardous waste landfill (e.g. bottom ash)	50	€ t^{-1}
Heating oil	0.7	€ L^{-1}	Non-hazardous waste landfill (e.g. fly ash)	135	€ t^{-1}
Steam (high pressure)	0.07	€ kWh^{-1}	Hazardous waste landfill (e.g. heavy metal cake)	250	€ t^{-1}
Resources and chemicals	**€**	**Unit**	**Labour cost**	**€**	**Unit**
Water	1.25	€ m^{-3}	Labour	29	€ h^{-1}
$Al(OH)_3$ (dry)	0.50	€ kg^{-1}	**Process output**	**€**	**Unit**
$Ca(OH)_2$ (dry)	0.11	€ kg^{-1}	Compost	0.005	€ kg^{-1}
CaO (dry)	0.10	€ kg^{-1}	PCP (95%)	0.14	€ kg^{-1}
$CaCO_3$ (dry)	0.094	€ kg^{-1}	TSP	0.3	€ kg^{-1}
$Al(OH)_3$ (dry)	0.50	€ kg^{-1}	H_3PO_4 (85%)	0.55	€ kg^{-1}
HCl (36%)	0.10	€ kg^{-1}	$FeCl_3$ (15%)	0.10	€ kg^{-1}
H_2SO_4 (96%)	0.13	€ kg^{-1}	$NaAlO_2$ (38%)	0.55	€ kg^{-1}
H_3PO_4 (85%)	0.55	€ kg^{-1}	$AlCl_3$ (% unknown)	0.037	€ kg^{-1}
Na_2CO_3 (dry)	0.23	€ kg^{-1}	Rhenania phosphate	0.13	€ kg^{-1}
$NaHCO_3$ (dry)	0.15	€ kg^{-1}	**Landfill specific cost**	**€**	**unit**
NH_3 (25%)	0.30	€ kg^{-1}	Soil		
NaOH (50%)	0.45	€ kg^{-1}	Daily cover or topsoil/composted waste	0.2	€ m^{-3}
$NaSO_4$ (dry)	0.08	€ kg^{-1}	Vegetation	1.0	€ $plant^{-1}$
Activated carbon	1.0	€ kg^{-1}	Soil	0.8	€ m^{-3}
Resin	1.0	€ kg^{-1}	Gravel (drainage layer)	0.2	€ m^{-3}

12.4 Cost for compliance (additional analysis and reporting)

Table 48. Additional cost for sewage sludge analysis and reporting to meet compliance for the different sub-options for PO1.

Scenario	agri/mono-inc.	WWTP to agri	WWTP to mono-inc.	Cost for additional analysis (€ yr^{-1})	Cost for reporting (€ yr^{-1})
>500k p.e.	90/10	149	17	241 725	12 177
	60/40	99	66	174 900	12 177
	30/70	50	116	108 075	12 177
>100k p.e.	90/10	1 114	124	1 813 670	91 364
	60/40	743	495	1 312 280	91 364
	30/70	371	867	810 890	91 364
>50k p.e.	90/10	2 369	263	3 855 880	194 242
	60/40	1 579	1 053	2 789 920	194 242
	30/70	790	1 842	1 723 960	194 242
>20k p.e.	90/10	5 114	568	8 324 130	419 332
	60/40	3 409	2 273	6 022 920	419 332
	30/70	1 705	3 977	3 721 710	419 332

Table 49. Additional cost for sewage sludge analysis and reporting to meet compliance for the different sub-options for PO2.

Sub-option	agri/mono-inc.	WWTP to agri	WWTP to mono-inc.	Cost for additional analysis (€ yr^{-1})	Cost for reporting (€ yr^{-1})
>500k p.e.	0/100	0	165	41 250	12 177
>100k p.e.	0/100	0	1 238	309 500	91 364
>50k p.e.	0/100	0	2 632	658 000	194 242
>20k p.e.	0/100	0	5 682	1 420 500	419 332

12.5 External costs

12.5.1 Background

Costs borne by third parties, thus actors that are not directly involved in sewage sludge management, are externalities or external costs. External costs can be calculated based on the emission data and "shadow prices" that express the social cost of environmental emissions based on the willingness-to-pay for preventing pollution and other unwanted (Afman et al., 2017; de Bruyn et al., 2018). Hence, they can occur because, for instance, citizens experience adverse effects from pollution. Here, external costs are calculated based on assumed losses of carbon, nitrogen, phosphorus and metals. Other contaminants, including PFAS, PAH and PCDD/F, that are emitted to air and soil during incineration and sewage sludge land application have not been taken into consideration in this assessment as no shadow process for these compounds are available for emissions to soils. Therefore, the external cost associated to sewage sludge land spreading may be underestimated in this assessment.

12.5.2 Methodology and assumptions

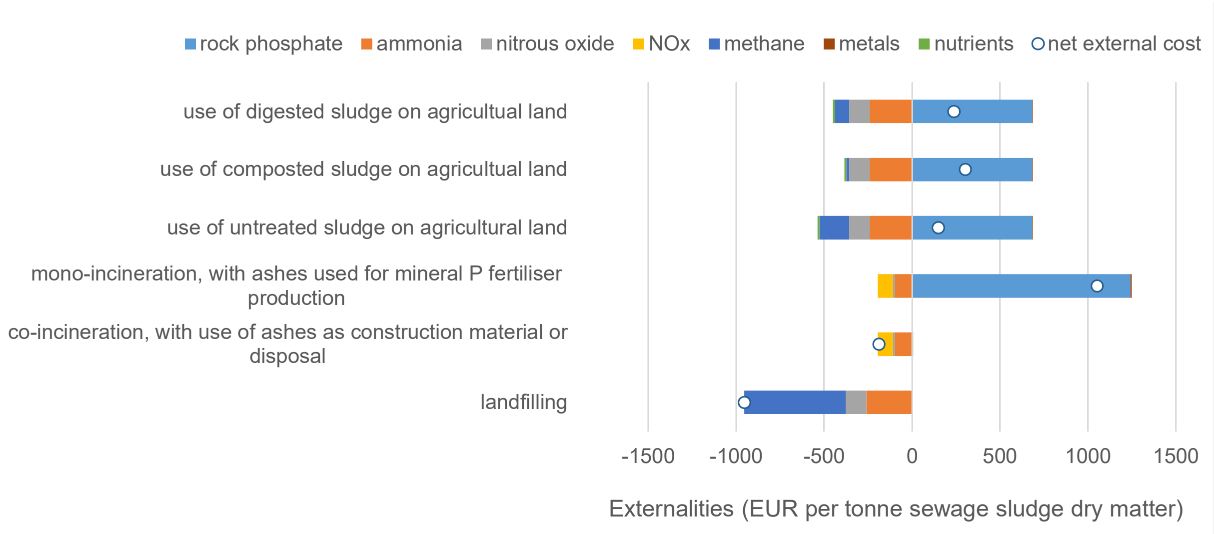
The assessment was based on following data and assumptions.

- Sewage sludge is assumed to have N content of 3.15%, a P content of 1.8%, and an organic carbon content of 30%. It is assumed that sewage sludge was temporarily stored for >30 days prior to land application, and that 20% of the nitrogen and phosphorus applied on agricultural land lost through leaching.
- The phosphorus fertiliser efficiency of sewage sludge was assumed to be 55% of the P in a mineral fertiliser (Oenema et al., 2012; Tonini et al., 2019). Mineral P fertilisers manufactured from sewage sludge have the same fertiliser efficiency as their counterparts derived from primary raw materials.
- A 27% NH_3 loss from sewage sludge landfilling is assumed (Sutton et al., 2000).
- Ammonia emissions during sewage sludge storage drying stage prior to incineration were assumed 10% of the total sewage sludge N content (conservative estimate; associated to a high variation and thus possible higher external cost for incineration). Ammonia emissions from sewage sludge storage, composting and the use on land were assumed 25%.
- Emission factors for nitrous oxide emissions from sewage sludge storage followed by use on land and landfilling were assumed 1.5% of the N in sewage sludge, in line with IPCC guidelines for emissions from land. Nitrous oxide emissions from incineration were budgeted based on emission data reported in the BREF on waste incineration (Neuwahl et al., 2019).
- NO_x emissions for incineration were estimated based on emission data reported in the EMEP/EEA air pollutant emission inventory guidebook 2019 (EEA, 2019b). The same source was used to preliminary assess the impacts from PAH, PCDD/F and metal emissions from incineration, for which the calculations indicated minor external costs.
- Methane emissions for all sewage sludge management routes were based on emission data presented by the IPCC/OECD/IEA expert group (Hobson, 1999).
- Metals emissions to soil were based on observed metal concentrations observed in sewage sludge as documented in Tavazzi et al. (2012). Mercury, cadmium and lead emissions during incineration were considered, but external costs largely (70%) originate from Hg emissions as the remaining metals do not volatilise to a significant extent.
- Shadow prices for the emissions were used based on work from CE Delft (Afman et al., 2017; de Bruyn et al., 2018): rock phosphate avoidance – 69 € kg P^{-1}; ammonia 30.5 € kg NH_3^{-1}; nitrous oxide: 25 € kg N_2O^{-1}; NO_x: 34.7 € kg NO_x^{-1} (as NO_2); CH_4: 3 € kg CH_4^{-1}; NO_3^- loss from agricultural soils: 0.71 kg NO_3-N^{-1}; PO_4^{3-}: 1.9 kg PO_4-P^{-1}.

12.5.3 Results

A comparison of external costs across the different sewage sludge management routes in given in **Figure 50**:

Figure 50. Externalities for sewage sludge management routes (positive values are benefits to society, whereas societal adverse effects are evident when the net values are negative).



Significant negative externalities are observed from methane emissions (particularly from landfilling, and to a smaller extent from methane emissions during sewage sludge digestion) and ammonia emissions (all routes, but more important for landspreaded and landfilled sewage sludge; Figure 50). Negative externalities from the summed emissions of nitrous oxide and NOx are similar for all pathways. External costs from nutrient losses and metal emissions are low across all pathways. Note that emissions from persistent organic pollutants such as PFAS, PAH, and PCDD/F have not been considered in this assessment as no shadow prices for emissions of these contaminants to soils are available. Therefore, adverse impacts from consumers that consume food products from sewage sludge amended soils are underestimated for sewage sludge management routes that involve land spreading.

Net externalities are positive for all pathways that return resources, more particularly phosphorus to agricultural land, but negative for co-incineration and landfilling as disposal pathways (Figure 50). The processing pathway that uses mono-incineration ashes to produce a mineral phosphorus fertiliser is associated the highest positive externality due to the high plant-availability of the material. Values for untreated and biologically treated sewage sludge are lower because of the plant availability of the sewage sludge-contained phosphorus is lower. The positive externalities for these pathways are significant and often exceed the internal or private costs for operators involved in sewage sludge management. The disposal pathways co-incineration and landfilling are associated to negative externalities (Figure 50).

13 Supplementary information – additional data

13.1 Size distribution UWWTD in the EU-27

Based on the reported data from (EEA, 2022), the number of UWWTD with treatment capacities >2k p.e. and size distribution of UWWTD capacities could be determined for each MS. For the EU 27 member states the following picture emerges:

Table 50. Size distribution of UWWTD in the EU-27.

Size category	**Number**	**p.e. capacity (in Mio)**
2-20k	15 558	110.2
20-50k	3 050	99.7
50-100k	1 394	102.4
100-500k	1 073	221.1
>500k	165	178.6
sum	**21 240**	**712.0**

13.2 Sewage sludge (mono)-incineration plants in the EU-27

Based on the preliminary findings from (Sichler et al., 2022), the list of sewage sludge incineration plants was extended through an extensive research in literature, screening of relevant webpages of plant engineering companies and operators (public and private) in the EU-27. **Table 51** highlights, that at least 60 sewage sludge incineration plants operate in the EU with a total capacity of about 1.4 Mt DM^{-1}, corresponding to around 17% of sewage sludge produced in 2019. At this point it is noted that it is not always clear whether only sewage sludge is incinerated, or also other types are incinerated with the sewage sludge.

Table 51. Overview sewage sludge incineration capacities in the EU-27.

Country	**City**	**Capacity (t DM yr^{-1})**	**Country**	**City**	**Capacity (t DM yr^{-1})**
AT	Vienna	68 000	FR	Colombes	64 000
AT	Bad Voeslau	2 500	FR	Dammarie les Lys	7 000
BE	Brueges	19 800	FR	Le Havre	10 000
DE	Altenstadt	55 000	FR	Lyon Saint-Fons	28 800
DE	Balingen	2 400	FR	Paris	no data
DE	Berlin-Ruhleben	84 100	FR	Rosny-sur-Seine	9 600
DE	Bitterfeld-Wolfen	15 200	FR	Rouen Petit-Quevilly	24 000
DE	Bonn	8 000	FR	Seine-Aval	no data
DE	Bottrop	44 000	FR	Valenton	12 000
DE	Dinkelsbuehl	5 326	FR	Vitré	no data
DE	Dueren	14 000	NL	Moerdijk	60 000
DE	Elverlingsen-Werdohl	61 320	NL	Moerdijk	55 000
DE	Frankfurt	52 560	IT	Bologna	6 250
DE	Gendorf	10 000	IT	Milan	14 950
DE	Hamburg	78 840	IT	Prato	7 360
DE	Herne	22 200	IT	Pustertal	5 681
DE	Karlsruhe	20 000	PL	Bydgoszcz	7 800
DE	Luenen	95 000	PL	Gdansk	14 000
DE	Muenchen	22 000	PL	Gdyania	9 000
DE	Stuttgart	32 000	PL	Zielona Gora	6 400
DE	Neu-Ulm	16 000	PL	Kielce	6 200
DE	Wuppertal	32 000	PL	Cracow	23 000
DE	Sande/Wilhelmshaven	2 250	PL	Lodz	21 000
DE	Straubing	40 000	PL	Lomza	1 500
DE	Mannheim	no data	PL	Szczcin	6 000
DK	Avedore	7 418	PL	Warsaw	62 200
DK	Kopenhagen	18 560	RO	Bucharest	18 150
DK	Lynette	14 493	RO	Pitesti	no data
DK	Lundtofte	3 200	ES	Bilbao	23 250

FI	Rovaniemi	2 500		Saragossa	36 800

13.3 Uptake and state-of-the-art of emerging phosphorus recovery technologies from sewage sludge and sewage sludge ash

In the EU, the first use of SSA in the fertilizer industry and the first fully operational P recovery plants from SSA were commissioned around 2020. The fertiliser industry uses SSA to substitute raw phosphate rock to produce marketable fertilisers. ICL Fertilisers reports the production of super phosphate fertiliser and NPK fertilisers that are 100% based on SSA. For other fertiliser producer as e.g. Borealis L.A.T the use of SSA is limited due to the technical fertiliser production approach and the physical-chemical properties of the SSA (maximum substitution potential for phosphate rock within the fertiliser production process: 3-10%).

Besides the fertiliser industry, two P recovery facilities operate in the EU by now (SERAPLANT (DE), TetraPhos® (DE) with a total annual SSA capacity of 55 000 t yr^{-1}. Combined with the fertiliser industry presently about 5kt P yr^{-1} is recovered from around 65kt SSA yr^{-1} (Table 52).

The already operating technologies, but also those technologies that are most advanced and could be implemented on a large scale in the foreseeable future use proven processes from the fertilizer industry in combination with proven processes for the purification of concentrates and apply them to SSA.

These technologies are primarily so-called wet chemical processes, which in a first step dissolve P out of the ash matrix with mineral acids into a liquid phase. This enables the subsequent separation of undesirable elements that unavoidably also go into solution. After the purifying steps, either a concentrated phosphoric acid is produced, or the phosphoric acid can be seen as an intermediate product to produce different solid marketable fertilisers (e.g. Mono-Ammonium-Phosphate) (MAP), Di-Ammonium-Phosphate (DAP), (Nitrogen-Phosphorus-Potassium (NPK), different P-salts). Compared to conventional fertilizer production processes, these newly developed processes are somewhat more complex, since the chemical composition and the contaminant content of SSA differs significantly from that of rock phosphates.

Beside the wet chemical processes, thermo-chemical (treatment temperature below the ash melting temperature) and thermo-electrical (treatment temperature above the ash melting temperature) approaches are currently under investigation.

For sewage sludge no full-scale P-recovery plant is operation in the EU at the present time. Three technologies with TRL 8-9 are planning to set up full scale plants in the EU in the upcoming years (EuPhore, Pyreg, and TherraNova). The last two mentioned, transform the sewage sludge into biochar and on the one hand use the biochar directly as a fertiliser (Pyreg) or the biochar is subjected to further chemical processing steps to obtain mineral phosphate salts at the end (TerraNova).

With the legal certainty that now exists with regard to mandatory P recycling in certain countries (e.g. AT, CH, DE), numerous new plants, both for SS and SSA are expected to come online in the upcoming 10 years.

Table 52 gives an overview on technologies to recover P addressing SS and SSA at different steps of development (full scale plants or near full scale implementation (TRL 8-9), tested in relevant environment (TRL 6+), and at R&D scale with promising outcome). Recovery technologies that apply on other stages at a WWTP (e.g. liquid phase after dewatering) are not considered, as these technologies address only a specific nutrient stream within the WWTP .This table builds up on extensive and regularly updated P recovery catalogue from ESPP (ESPP et al., 2022) and was complemented by additional research.

Table 52. Overview on P recovery technologies from sewage sludge and incineration resides from sewage sludge (sewage sludge ash) (alphabetical order).

Process (source)	Objectives	Output	By-products/ waste	Rec. rate (%)	Country (installed/ *planned*)
Full scale plants in operation or under permitting/construction (TRL 8-9)					
Ash2Phos/ EasyMining (SSA)	Acidic wet chemical leaching: P is leached from SSA with HCl and H_3PO_4 and as such separated from the ash. Contaminants that go into solution together with P are removed by ion-exchange, liquid-liquid separation or precipitation (Easymining, 2021). Two investments in prospect:	PCP, $FeCl_3$, $NaAlO_2$	Sand, HM concentrate	80-95%	SE (*2025*) DE (*2027*)

	- 30kt yr^{-1} ash, Helsingborg (SE) (permit application submitted) - 30kt yr^{-1} ash planned, Schkopau (DE) (permit application ongoing)				
EuPhore (dewatered and dried SS)	The EuPhoRe process uses a specifically-designed rotary kiln incinerator. Depending on the input material quality, alkali- and/or earth-alkali chlorides may be added for removal of part of the HM into the gas phase (e.g. Cd, Hg, Cr(VI) (90-95%); Cu (20-40%). Two industrial scale plant are planned in Germany: Offenbach (100kt SS yr^{-1}) and Mannheim (135kt yr^{-1}).	SSA with improved plant availability.	Flue gas residues.	95%	DE (unknown)
Fertiliser industry (SSA)	ICL Fertilisers: Mixing SSA with with mineral acids (e.g. H_3SO_4 or H_3PO_4) to transformation of the P into a plant available form. All compounds of the SSA are fully incorporated into the fertiliser, so no removal of contaminants takes places (ICL-Fertilizers, 2019). Borealis L.A.T**:** Mixing SSA with raw PR and leaching of P, Ca and other elements with nitric acid into a liquid phosphoric-nitric acid. After removal of the sand, the leached acid is used to produce a multicomponent NPK fertiliser and a calcium nitrogen fertiliser. Some contaminants are removed with the sand, but most of them are incorporated into the fertilisers. Technical limitations, thereof SSA can only substitute 3-5% of raw PR (Huber and Amann, 2022).	SSP, TSP, NPK NPK, NAC	No Sand	100% 95-100%	NL (2022) AT (*2022*)
Kubota (dried SS or SSA)	Thermal treatment at 1 300 °C and addition of iron oxide to retain P in the solid slag whereas part of the heavy metals, copper and zinc are volatilised and removed. To improve the P plant availability calcium hydroxide is added. Around 20 full-scale furnaces operate in Japan with sewage sludge or SSA as input material.	P-containing slag with plant availability compared to comercial P fertilisers	Ironslag, Flue gas residues	90%	Japan (unknown)
Metawater (SSA)	Caustic wet chemical leaching: P is leached from SSA with NaOH and then separated from the ash. To avoid the dissolution of HM into the process, the pH is only increased slightly, resulting also in a low P recovery rate. Two full-scale plants operate in Japan: Gifu and Tottori	CaP	Sand	30%	Japan (2010)
PYREG (SS)	Producing biochar operating at temperature from 500-800ºC. Biochar produced from SS is registered as a fertiliser in Sweden (PYREGphos) and 4 plants using sewage sludge as input are operating at full scale in DE, SE, USA (capacity of around 1 200 t DM yr^{-1} each).	biochar	unknown	100%	DE, SE, USA (2015)
Seraplant/ PHOS4Green (SSA)	Mixing of SSA with H_3PO_4 and producing a fertiliser with spray granulation in a fluidised bed. Other nutrient elements (e.g. N, K, S) can be added to the SSA/acid suspension. All compounds of the SSA are fully incorporated into the fertiliser, so no removal of contaminants takes places (SERAPLANT, 2021). SSA input: 35kt yr^{-1}: Product output: 60kt yr^{-1}.	TSP or NPK fertilisers	No	100%	DE (2021)
TerraNova (HTC) (SS)	Hydrothermal hydrolysis carbonization process (175°C, 20-25 bars) and acid treatment of the sludge to dissolve P. Precipitation and/or crystallisation on calcium-silicat-hydrat granulate (CSH) of the P from the liquid phase after mechanical separation. Full-scale operation in China with 2 t h^{-1}. Developments also in DE with a demonstration plant (250 kg h^{-1}).	Mg/Ca-P salt	80%	80%	China (2016)
TetraPhos/ Remondis (SSA)	Acidic wet chemical leaching: P is leached from SSA with HCl and H_3PO_4 and as such separated from the ash. Contaminants that go into solution together with P are removed by ion-exchange, liquid-liquid separation or precipitation (Rak, 2018). Operational full scale plant in Hamburg (DE) with capacity of 20k t SSA.	P-acid (technical grade), lime, $FeCl_3$, AlCl	Sand, HM concentrate	80-95%	DE (2020)

PAKU (Endev)	A full scale PAKU facility incinerates the sewage sludge from the Rovaniemi city (63 000 inhabitants). The SSA from the incinerator is directly used as fertiliser in the forest industry.	SSA	-	100%	FI (2021)
Technologies TRL 6+					
AshDec/ Metso Outotec) (SSA)	Thermo-chemical approach: Aim is the partial removal of metals and the transformation of the P into plant available form. This can be achieved by adding Cl and a treatment temperature of 750 ºC to 1 000 ºC (below ash melting temperatures: (Adam et al., 2008)). Latest developments of this technology focus on the further improvement of the plant availability of SSA by adding Na instead of Cl, with the trade-off of significantly lower metal removal (Herzel et al., 2021). A full scale plant (30kt SSA yr-1) is planned in Bavaria (DE).	SSA with improved plant availability	Flue gas residues	97%	DE (unknown)
Geocycle/ LafargeHolcim (SS)	A pilot plant is operating at the Holcim cement plant in Untervaz (CH) to first gasify the sewage sludge at 1 000 °C. The energy rich syngas is used in the cement kilns. The occurring ash undergoes acidic wet chemical leaching. Details about the further treatment steps are not known.	P_4	unclear	80-95%	CH (*2026*)
Phos4Life, Técnicas Reunidas SA (SSA)	Acidic wet chemical leaching: P is leached from SSA with HCL and as such separated from the ash. Contaminants that go into solution together with P are removed by liquid-liquid separation. A recovery plant with a capacity of 40kt SSA yr^{-1} is planned for 2026 ((Morf, 2022)).	P-acid (fertiliser grade)	Sand, HM concentrate	80-95%	CH (*2026*)
Prayon/former Ecophos (SSA)	Acidic wet chemical leaching: P is leached from SSA with H_3PO_4 as such separated from the ash. Contaminants that go into solution together with P are removed by a cascade of ion-exchangers. A semi-industrial pilot plant is in operation with a capacity of 200 kg SSA h^{-1} in Varna (BG). The two incinerators SNB and HVC treating half of the sewage sludge in NL and producing 57kt SSA yr^{-1} planned to recover P with this technology (Sonveaux, 2022).	DCP or P-acid (technical grade), $FeCl_3$, AlCl, Ca/MgCl	Sand	>95%	BG (2020) NL (unknown)
ViViMag® (WETSUS - Kemira) (SS)	Aim is the precipitation of iron phosphate in the form of vivianite, by reducing iron(III) to iron(II) in anaerobic conditions (digester). The vivianite is then recovered by magnetic separation. Vivianite can be used as an Fe fertilizer or optional P can be extracted from the vivianite to produce liquid PK fertiliser and recycle iron a coagulant to WWTP. Currently pilot phase (Korving and Hansen, 2022).	Vivianite: iron(II)-phosphate	no	80%	DE, NL, DK (Pilot phase 2022)
Technologies at R&D scale					
CarboREM (SS)	First step is the hydrothermal carbonisation (HTC) of SS. After solid liquid separation, the hydrochar is treated with citric- or hypochloric acid. From the liquid phase, P is precipitated by pH adjustment. High removal rates of HM can be observed (80-90%). Industrial-scale continuous HTC plant installed 2019 (WWTP, Mezzocorona, Italy). Capacity: 1.4 t h^{-1} of wet digested sewage sludge.	Precipitated P salts	hydrochar	90%	IT (unknown)
Flashphos (Univ. Stuttgart, Italmatch) (SS)	Sewage sludge is dried and then flash gasified at high temperatures with CaO (lime) as slagging agent to produce elemental phosphorus (P_4). A 2 t day^{-1} dry matter input pilot plant is funded by a Horizon 2020 project (Leverenz et al., 2021).	P_4, cement binder, iron alloy	HM concentrate	85-90%	IT, DE (unknown)
Parforce (SSA)	Acidic wet chemical leaching: P is leached from SSA with HCL or HNO_3. Unwanted substances can be removed with ion exchange, solvent extraction, and membrane electrodialysis.	P-acid, struvite, $FeCl_3$, AlCl, Ca/MgCl	Sand	80%	DE (unknown)

RSR (Green Sentinel) (SS)	Modular approach to recover a mixed aqueous solution of minerals (P, Ca, K). The SS low in P can be used as energy source.	Nutrient rich aqueous solution	Sewage sludge, HM concentrate	75%	AT (*2022*)
Susphos (SSA)	Acidic wet chemical leaching: P is leached from SSA with H_3SO_4. The produced P-acid is purified by organic solvent extraction process and MAP and DAP is precipitated. Full scale plant of 50kt yr^{-1} is planned in NL.	MAP, DAP, $FeCl_3$, AlCl	Sand, gypsum	80-95%	NL (*2023-2024*)

13.4 National legal frameworks for nutrient recovery from sewage sludge in the EU-27

For two EU-27 countries, namely AT and DE, mandatory legal frameworks for P-recovery will be or are already in place. With CH, a third country in Europe has implemented already mandatory P-recovery.

Table 53. Overview on legal frameworks and ongoing discussions on nutrient recovery from municipal sewage sludge for specific countries.

Country	Objectives	Implementation timeframe	Status/Policy
AT	**Mandatory P recovery for municipal WWTP ≥20k p.e.** (compromises ~85% of waste water in Austria) Three options: 1) Mono-incineration and recovery of **P from SSA efficiency of at least 80%** 2) All the incineration ash itself is used for production of a fertiliser respecting national fertiliser regulation specifications (Düngemittelgesetz 2021, BGBl. I 103/2021) 3) Recovery of P by thermal, chemical or physico-chemical" processes at or nearby the WWTP from **sewage sludge with recovery efficiency of 60%** with regard to **WWTP influent**. WWTP <20k p.e. are not affected and can choose their way of sewage sludge treatment.	Until 2030	**Notified to the EU**/ Amendment of the existing 'Waste Incineration Ordinance' 2002 Expected 2022, (BMU, 2022)
DE	Implementation in two phases: Phase 1: **Mandatory P recovery for municipal WWTP ≥100k p.e.** (compromises ~ 50% of waste water P in Germany Phase 2: **Mandatory P recovery for municipal WWTP ≥50k p.e.** (compromises ~ 65% of waste water P in Germany Two options: 1) Mono-incineration and **recovery of P from SSA efficiency of at least 80%** 2) Recovery of P from sewage sludge to **reduce P content in sewage sludge below 20 g kg DM^{-1}**. Sewage sludge low in P than can be (co-)incinerated. WWTP <50k p.e. are not affected and can choose their way of sewage sludge treatment.	Until 2029 Until 2032	**Implemented**/ Sewage Sludge Ordinance (BMJ, 2017)
DK	The Danish Resource Strategy 'Denmark without waste' contains the chapter 4 'Better exploitation of important nutrients such as phosphorus' (Government, 2013). Until 2018, 80% of P from sewage sludge is to be recycled through: 1) Recovery by utilization of sewage sludge on agricultural soil 2) Recovery of phosphorus from the sewage sludge incineration ash as fertilizer However, Figure 36 reveals that this goal was not reached by 2019. There are ongoing discussion on possible mandatory P recovery.	-	Ongoing discussion/ No policy in place
SE	In 2012 the Swedish EPA (SEPA, 2012) proposed a target that **40% of P from waste** (incl. sewage sludge) should be recycled on arable land by 2018. Further targets were set to continuously lower the content of undesired substances in the sewage sludge by upstream measures. The Commission of Inquiry on Non-toxic and Circular Recycling of Phosphorus from sewage sludge has submitted its report on sustainable sewage sludge management to the government in 2020 (SOU, 2020). Exemplarily, one suggestion is the **60% P recovery** from sewage sludge for WWTP from about 20k p.e.	-	Ongoing discussion/ No policy in place

CH	Total ban on agricultural sewage sludge application in 2006. Currently 100% (co-) incineration. Mandatory P recovery from P rich waste: Ordinance on the Prevention and Disposal of Waste, Art. 15 Waste rich in phosphorus – P must be recovered from municipal waste water, from sewage sludge from central waste water treatment plants or from the ash from the thermal treatment of such sewage sludge and recycled. – P contained in meat-and-bone meal shall be recycled, unless the meat-and-bone meal is used as animal feed. – When P is recovered from waste in accordance with paragraph 1 or 2, the pollutants contained in this waste must be removed in accordance with the state of the art. If the recovered phosphorus is used for the production of a fertilizer, the requirements of Annex 2.6 Number 2.2.4 ChemRRV16 must also be met. In the corresponding implementation guide to the waste ordinance, the minimum target of a 50 % recovery rate by 2026 has been set. By 2036, as much as **75% of the phosphorus from sewage sludge**, sewage sludge ash and other P-rich wastes are to be returned to the cycle.	Until 2026	**Implemented**/ Waste Ordinance (BAFU, 2021)
HELCOM	**Baltic Marine Environment Protection Commission** (Helsinki Commission, **HELCOM**). Members: EU, DK, DE, EE, FI, LT, LV, PO, SE 2021 HELCOM published the Draft for the Baltic Sea Regional Nutrient Recycling Strategy. One objective is, that the Baltic Sea region shall be a model area for nutrient recycling. Mentioned, as a possible measure is the **'Promotion of the development and application of new technologies for removal and recovery of nutrients from WWTP** (HELCOM, 2021). However, no further specific targets are indicated.	-	Ongoing discussion/ No policy in place

13.5 Voluntary standardisation schemes for sewage sludge

The REVAQ certification system is operated by the Swedish Water & Wastewater Association, the Federation of Swedish Farmers (LRF), The Swedish Food Federation, and the Swedish food retailer's federation, in close cooperation with the Swedish Environmental Protection Agency (IEA Bioenergy Task 37, 2015). The goals of the REVAQ work are to (i) avoid unacceptable accumulation of metals or undesired organic substances on agricultural land in the long term, (ii) have no accumulation of cadmium taking place from 2025, and (iii) reduce accumulation of non-essential substances to a maximum of 0.2% per year from 2025. More than 50% of the Swedish population is connected to a REVAQ certified waste water treatment plant. For each batch of sewage sludge produced at a certified waste water treatment plants, specific parameters must be checked. The sewage sludge is required to be analysed (yearly composite sample) at least every three years for 60 trace elements, including metals. Each waste water treatment plant will as a result of this analysis identify a number of prioritised trace elements. This quality control system emphasizes upstream actions to reduce the quantity of undesirable substances entering wastewater streams and is based on trust and confidence between stakeholders (Persson et al., 2015; (Persson et al., n.d.; I'Ons et al., 2015)).

The German Association for Water, Wastewater and Waste offers a voluntary quality assurance system for the agricultural utilization of secondary raw material fertilizers within the framework of the "Quality Assurance of Agricultural Waste Utilization" (QLA, 2017). The goal is to promote the sustainable agricultural utilization of residual and waste materials (incl. sewage sludge) according to the current state of science and technology. The quality assessment involves controls on feedstocks, sewage sludge and application methods for land spreading. Depending on the size of the waste water treatment plant, an audit occurs in 2 or 3 years interval. With regard to the quality of the sewage sludge, measurement requirements and limit values are set for nutrients, metals, certain organic pollutants (PFOS, PFOA, AOX, PCB, PAH, DEHP), and microbiological parameters. However, certain organic priority contaminants, such as other PFAS, chlorinated paraffins and others, are not regulated in this scheme, and the proposed limit values (e.g. for PFAS) are possibly still causing health and environmental risk as outlined in Huygens et al. (Huygens et al., 2022). Other voluntary standaridsation schemes in Germany for compost derived from sewage sludge are also available. (Quality assurance compost from sewage sludge, Quality assurance RAL-258 Refinement products from sewage sludge, https://www.kompost.de/guetesicherung/guetesicherung-kompost-aus-abwasserschlamm; BGK - Quality Assurance Recycling of Sewage Sludge, Quality assurance RAL-GZ 247, https://www.kompost.de/guetesicherung/guetesicherung-verwertung-von-abwasserschlamm).

The UK had a similar certification system while being part of the EU (Biosolids Assurance Scheme). The purpose of the Biosolids Assurance Scheme is to provide food chain and consumer reassurance that BAS certified biosolids can be safely and sustainably used on agricultural land. It combines legislative requirements and best practice, and has been developed following a joint initiative of stakeholders and the UK Water to enhance the overall performance of sewage sludge treatment and biosolids recycling to agricultural land. Member organisations are audited by an independent third-party certification body to ensure that they conform to the scheme standard (BAS, 2022). The voluntary standardisation scheme is focused on limiting microbiological hazards and metals in sewage sludge applied on agricultural land, but limit values for organic contaminants are not proposed in this standardisation scheme.

14 Annex: Supplementary information - modelling of metal inputs from sewage sludge to agricultural land

14.1 Methodology

Impact assessment of sewage sludge application was estimated by using three sewage sludge formulations (sewage sludge lower limit: SS_LL, sewage sludge upper limit: SS_UL and sewage sludge quality according to the data from Tavazzi et al. (2012): SS_Tavazzi in Figure 4) and both upper and lower limit values of concentrations of heavy metals in soils (EU_LL and EU_UL) from SSD (Annex IA). Concentrations of heavy metals from LUCAS 2009 topsoil database (ESDAC, 2013) were used as experimental data for impact assessment. In total, 14 726 agricultural soil samples from LUCAS 2009 topsoil database were used to assess the actual scenario and their capability for receiving sewage sludge when heavy metal burden is analyzed. Soil factors such as soil-water adsorption coefficient (Ksw), partition coefficients (KD) and leaching factor (Kleach) were used for each of the heavy metals (Cu, Cd, Ni, Hg, Pb, and Zn) to determine the fraction of the substance that remains in the topsoil layer over time. Details of modelling and maps for all tested scenarios are shown in Annex I for all agricultural point and aggregated at NUTS2 level.

The accumulative effect of heavy metals in soils over 10 consecutive years applying three different sewage sludge formulations at 5 Mg ha^{-1} rate was assessed. Contamination rates (percentage) before and after sewage sludge applications were determined in order to assess how much sewage sludge application is affecting on soil contamination rate.

This objective is very convenience in order to meet with a specific statement of the SSD (Directive 86/278/EEC. Article 5) as Member States shall regulate the use of sludge in such a way that the accumulation of heavy metals in the soil does not lead to the limit values being exceeded.

Composition of sewage sludge to be tested

Impact assessment of sewage sludge application on agricultural soils is going to be carried out as difference between final and initial contamination rate after application of three different sewage sludge formulations at 5 tonnes per hectare rate over 10 years. Sewage sludge formulations such as SS_LL and SS_UL (Table 4) were proposed from regulation limit values for heavy metals concentration in sludge for their use in agriculture (Directive 86/278/EEC) and already used in previous section. Third one (SS_Tavazzi in **Table 54**) was arranged as reported by Tavazzi et al. (2012) by using the average concentration values from 61 sewage sludge samples along Europe. Initial contamination rates in agricultural soils were determined by using those limit values (mg kg DM^{-1}) for HMs in the soils (EU_LL and EU_UL Annex IA SSD) and they will be compared after application of those three sewage sludge formulations (SS_LL, SS_UL and SS_Tavazzi in **Table 54**). Hence six different scenarios will be arranged (i.e. LLUL, LLLL, LLTav, ULLL, ULLL and ULTav where two first letters are indicating the limit values for HM in soils and the second ones the sewage sludge formulations) to figure out the impact of the application of sewage sludge on agricultural soils.

Table 54. Values for heavy metals concentrations (mg HM kg of sewage sludge^{-1}) for each sewage sludge formulation tested. SS_LL and SS_UL formulations from SSD 86/278/EEC and SS_Tavazzi (Tavazzi et al. 2012).

Heavy metal	SS_LL	SS_UL	SS_Tavazzi
Cd	20	40	0.90
Cu	1 000	1 750	257
Ni	300	400	29.0
Pb	750	1 200	47.6
Zn	500	4 000	700
Hg	16	25	0.40

Impact assessment of sewage sludge application on agricultural soils in Europe

For sewage sludge spreading to agricultural soils an application rate of 5 Mg ha^{-1} dry weight per year is assumed. Accumulation of the heavy metal occur when sewage sludge is applied over consecutive years. In this report is assumed that each sewage sludge formulation will be applied for 10 consecutive years. Accumulated content is estimated by quantifying the losses of heavy metals in soils via leaching. Thus the fraction for each of heavy metals that remains in the soil after each sewage sludge application may be determined. Calculations were done in accordance with the European Chemicals Agency (ECHA, 2016)

A pseudo first-order rate constant for leaching has been calculated from the amount of rain flushing the liquid-phase of the soil compartment (eq. 1).

$$K_{lea} \quad \frac{Finf_{soil} \cdot RAIN_{rate}}{K_{soil-water} \cdot DEPTH_{soil}} \qquad \text{Equation 4}$$

$$K_{soil-water} = \frac{K_D \cdot 2500}{1000} \qquad \text{Equation 5}$$

Where:

K_{leach} is the pseudo first-order rate constant for leaching from soil layer [day^{-1}]

$Finf_{soil}$ is the fraction of rain water that infiltrates into soil (0.25)

$RAIN_{rate}$ is the rate of wet precipitation [m/day]. Rain rate will be determined for each LUCAS 2009 point as precipitation data are available (mm $year^{-1}$)

$K_{soil-water}$ is the partitioning coefficient (m^{-3} m^{-3}). It was determined by using solid/liquid partition coefficient K_D (L kg^{-1}) and soil density (2 500 kg m^{-3}) as stated in eq. 12

$DEPTH_{soil}$ is a mixing depth of soil (m). For agronomic soils usually 20 cm is used as rhizosphere

Solid/liquid partition coefficients (K_D) used in eq. 12 for each of heavy metals were taken from the literature. Initially KD values from Sheppard et al (2011) were taking into consideration as soil texture and land use (wetlands and agricultural soils) were used as discriminant factors but large variability was observed and no other relevant soil properties such as soil pH was used. Finally, KD values from Janssen et al. (1999) were used for As, Cd, Cr, Cu, Ni, Pb and Zn as it may be deduced that pH already explains a high percentage in the variation of KD for all metals, except arsenic (**Table 55**). Other soil parameters such as Feox and Alox (oxalate extraction) would enhance the KD accuracy but they are not available in LUCAS 2009 database.

Table 55. Correlation between KD and soil characteristics according to $LogK_D = a \cdot pH_{CaCl_2} + c$ (Janseen et al. 2009).

Metal	R^2 adjusted	Proton coefficient (a)	Constant (c)
As	$0.36^{n.s.}$ (20)[a]	0.27	1.81
Cd	0.74*** (18)	0.48	0.28
Cr	0.54*** (19)	0.21	2.64
Cu	0.54*** (20)	0.33	0.68
Ni	0.61*** (19)	0.29	1.30
Pb	0.65*** (19)	0.35	2.06
Zn	0.85*** (20)	0.61	−0.65

No consistency information was got for mercury as the total mercury content of soils is generally very low (Lindsay W 1979). Solid/liquid partition coefficient (KD) value for Hg was taken from Allison et al. (2005). Regression equation $LogK_{d,waste} = 0.7 \cdot logK_{d,soil} + 0.3$ was used to calculate the mercury KD value (2.96 L kg^{-1}) from sewage sludge. It is assumed that the availability of mercury when applied from sewage sludge is higher than that of soil native mercury (3.8 L kg^{-1}).

Fraction of the substance that remains in the topsoil layer at the end of a year is given by eq. 3

$$F_{acc} = e^{-365 \cdot K_{le}} \qquad \text{Equation 6}$$

Where:

Kleach is the first order rate constant for removal from topsoil (day^{-1}). Other removal processes may be important (e.g. uptake by plants) but for this work only pseudo-first order rate constant for leaching from topsoil (eq. 11) has been taken into consideration in order to estimate the accumulative effect of heavy metals in soils when sewage sludge is applied as the worst scenario.

F_{acc} is the fraction accumulation in one year

The concentration for each heavy metal after the first application of each sewage sludge formulation is given by eq. 4

$$Csludge_{soil\_1}(0) = \frac{C_{sludge} \cdot APPL_{sludge}}{DEPTH_{soil} \cdot RHO_{soil}} \qquad \text{Equation 7}$$

Where:

$Csludge_{soil_1}(0)$ is the concentration in soil (mg/kg) due to sludge in first year at t=0

C_{sludge} is the concentration in dry sewage sludge (mg kg^{-1}). Concentration of each heavy metal for each tested sewage sludge formulation (Table 4) were entered. Three sewage sludge formulations were managed ($C_{SS\_UL}, C_{SS\_LL}, and\ C_{SS\_Tavazzi}$)

$APPL_{sludge}$ is the dry sewage sludge application rate. For this work a sewage sludge rate of 0.5 kg m^{-2} y^{-1} has been used (corresponding to 5 tonnes per hectare)

$DEPTH_{soil}$ is the mixing depth of soil (20 cm)

RHO_{soil} is the bulk density of soil. 1700 kg m^{-3} has been taken as average value as no bulk density parameter was measured in LUCAS 2009 topsoil survey

The accumulated concentration after 10 applications of sewage sludge was calculated by using eq. 15. It was determined for each heavy metal of each sewage sludge tested (Table 4)

$$Csludge_{soil_{10}}(0) = Csludge_{soil\_1}(0) \cdot [1 + \sum_{n=1}^{9} F_{acc}^{n}] \qquad \text{Equation 8}$$

Total concentration of heavy metals in soils ($C_{soil}\ (0)$) was determined by taking into consideration the background concentrations as measured in LUCAS 2009 topsoil survey ($Cdep_{soil_{10}}(0)$) (eq. 6)

$$C_{soil}\ (0) = Cdep_{soil_{10}}(0) + Csludge_{soil\_10}(0) \qquad \text{Equation 9}$$

It was assumed that background concentration in soil ($Cdep_{soil_{10}}(0)$) for each heavy metal after 10 year applying sewage sludge was the same as that measured in LUCAS 2009. No further inputs and/or removals were considered. Three different $C_{soil}\ (0)$ will be got as three sewage sludge formulations were tested; $C_{soil_{SS\_UL}}\ (0), C_{soil_{SS\_LL}}\ (0), and\ C_{soil_{SS\_Tavazzi}}\ (0)$

Fresh contamination rates (%) after application of those three sewage sludge formulations for 10 years at 5 Mg ha^{-1} rate were determined. Those limit values of concentrations of heavy metals at European level (eq. 4 and eq. 5) in accordance with Sewage Sludge Directive were used for each of the three sewage sludge formulations (eq. 7-12)

$$Contamination\ Rate\ HM_{ULUL}\ (\%) = \frac{C_{soil_{SS\_UL}}}{\text{[HM]EU\_UL}} \cdot 100 \qquad \text{Equation 10}$$

$$Contamination\ Rate\ HM_{ULLL}(\%) = \frac{C_{soil_{SS\_LL}}}{\text{[HM]EU\_UL}} \cdot 100 \qquad \text{Equation 11}$$

$$Contamination\ Rate\ HM_{ULTav}(\%) = \frac{C_{soil_{SS\_Tavazzi}}}{\text{[HM]EU\_UL}} \cdot 100 \qquad \text{Equation 12}$$

$$Contamination\ Rate\ HM_{LLUL}(\%) = \frac{C_{soil_{SS\_UL}}}{\text{[HM]EU\_LL}} \cdot 100 \qquad \text{Equation 13}$$

$$Contamination\ Rate\ HM_{LLLL}(\%) = \frac{C_{soil_{SS\_LL}}}{\text{[HM]EU\_LL}} \cdot 100 \qquad \text{Equation 14}$$

$$Contamination\ Rate\ HM_{LLTav}(\%)\ = \frac{C_{soil_{SS\_Tavazzi}}}{\text{[HM]EU\_LL}} \cdot 100 \qquad \text{Equation 15}$$

Contamination rate will be used to identify and quantify those new sites from LUCAS 2009 topsoil database that reach those limit values and therefore they will be labelled as contaminated sites. Additionally the impact of the application of different sewage sludge formulations will be assessed as the difference between those contamination rate before and after the sewage sludge applications in agricultural soils. The use of two limit values of concentrations of heavy metals in soils and the three potential sewage sludge formulations will make possible to identify the actual reliability of the threshold values lay down in the current SSD

Impact assessment of application of sewage sludge on agricultural soils will be quantified as the difference between the contamination rate after application of three different sewage sludge formulations at 5 Mg ha^{-1} for 10 years before application (equation 13-18) and the contamination rate from the background concentration in soils (eq. 4,5) as follows:

$$Diff_{ULUL} = Contamination\ Rate\ HM_{ULUL}(\%) -\ Contamination\ Rate\ HM_{EU\_UL} \qquad \text{Equation 16}$$

$$Diff_{ULLL} = Contamination\ Rate\ HM_{ULLL}(\%) -\ Contamination\ Rate\ HM_{EU\_UL} \qquad \text{Equation 17}$$

$$Diff_{ULTav} = Contamination\ Rate\ HM_{ULTav}(\%) -\ Contamination\ Rate\ HM_{EU\_UL} \qquad \text{Equation 18}$$

$$Diff_{LLUL} = Contamination\ Rate\ HM_{LLUL}(\%) -\ Contamination\ Rate\ HM_{EU\_LL} \qquad \text{Equation 19}$$

$$Diff_{LLLL} = Contamination\ Rate\ HM_{LLLL}(\%) -\ Contamination\ Rate\ HM_{EU\_LL} \qquad \text{Equation 20}$$

$$Diff_{LLTav} = Contamination\ Rate\ HM_{LLTav}(\%) -\ Contamination\ Rate\ HM_{EU\_LL} \qquad \text{Equation 21}$$

14.2 Results

14.2.1 Accumulation of heavy metals in soils

Descriptive statistics for partitioning coefficients (K_s), solid/liquid partition coefficients (K_D), pseudo first-order rate constant for leaching (K_{Leach}) and the fraction accumulation in one year (F_{acc}) are shown in **Table 56** to **Table 59**, respectively. Significant uncertainties are assumed when partition coefficients for heavy metals in soils are stated. Is well-know that soil properties such as pH, organic matter, clay content, metal oxy(hydr)oxides and dissolved Organic matter are significantly affecting in different way on partition coefficients for each of heavy metals. Partition coefficients for Cd, Cu, Ni, Pb, and Zn were calculated for each of LUCAS 2009 points by taking into consideration a linear mathematical model by using the potential soil acidity as factor. As already done in previous section, no arsenic and chromium were included into analysis as no reference values were available in current Sewage Sludge Legislation. Additionally, the partition coefficients for heavy metals when applied from sewage sludge are lower than those observed in soil naturally polluted as already introduced for the Mercury partition coefficient. The resulting patterns of decreasing KD are in line with those reported by Allison (2005). Highest KD value was found for Lead and lowest KD value was recorded for copper regardless of solid phase to be analysed (suspended matter, sediment or soil). A single partition coefficient was used for mercury as no soil properties such as dissolved organic carbon and content in fulvic- and humic acids are available in LUCAS 2009 topsoil database. Factors such as pseudo first-order rate constant for leaching and fraction accumulation are used to explain the mobility of heavy metals in soil over time in complementary way. More than 99.7% of the concentration for each of heavy metals remains in the soil (in average). In cumulative terms it means that more than 97% of the initial concentration for each of heavy metals will remain in the soil after 10 years

Table 56. Descriptive statistics for the partitioning coefficients (Ks in m^{-3} m^{-3}) for each of heavy metals from LUCAS 2009 database. They were determined by using solid/liquid partition coefficient KD (L kg^{-1}) and soil density (2 500 kg m^{-3}).

	n	**mean**	**median**	**StandDev**	**kurtosis**	**skewness**	**min**	**max**
Ks_Cd	21682	6204	2772	6852	5.7	1.31	4.8	131202
Ks_Cu	21682	1433	952	1225	-0.4	0.72	12.0	13503

Ks_Ni	21682	3211	2335	2492	-0.8	0.62	49.9	24013
Ks_Pb	21682	47006	29781	41921	-0.1	0.78	287.0	495952
Ks_Zn	21682	6084	1826	7862	44.5	2.73	0.6	245720
Ks_Hg	21682	2280	2280	0	-2.0	1.00	2280.0	2280

Table 57. Descriptive statistics for Solid/liquid partition coefficients (KD) for each of heavy metals from LUCAS 2009 database.

	n	**mean**	**median**	**StandDev**	**kurtosis**	**skewness**	**min**	**max**
KD_Cd	21682	2482	1109	2741	5.7	1.31	2	52481
KD_Cu	21682	573	381	490	-0.4	0.72	5	5401
KD_Ni	21682	1284	934	997	-0.8	0.62	20	9605
KD_Pb	21682	18802	11912	16768	-0.1	0.78	115	198381
KD_Zn	21682	2433	730	3145	44.5	2.73	0	98288
KD_Hg	21682	912	912	0	-2.0	1.00	912	912

Table 58. Descriptive statistics for the pseudo first-order rate constant for leaching (KLeach) from soil layer [day^{-1}] for each of heavy metals from LUCAS 2009 database.

	n	**mean**	**median**	**StandDev**	**kurtosis**	**skewness**	**min**	**max**
KLeach_Cd	21682	2.86E-06	8.48E-07	6.04E-06	2784	38.1	1.42E-08	4.82E-04
KLeach_Cu	21682	4.68E-06	2.46E-06	5.45E-06	100	4.58	1.38E-07	1.92E-04
KLeach_Ni	21682	1.68E-06	1.01E-06	1.72E-06	33.7	2.79	7.76E-08	4.60E-05
KLeach_Pb	21682	1.60E-07	7.88E-08	1.99E-07	171.5	6.10	3.76E-09	8.00E-06
KLeach_Zn	21682	7.93E-06	1.29E-06	3.91E-05	8456	85.8	7.59E-09	4.10E-03
KLeach_Hg	21682	1.02E-06	9.79E-07	2.74E-07	3.3	1.20	3.02E-07	3.11E-06

Table 59. Descriptive statistics for the fraction accumulation (Facc) in one year for each of heavy metals from LUCAS 2009 database.

	n	**mean**	**median**	**StandDev**	**kurtosis**	**skewness**	**min**	**max**
Facc__Cd	21682	0.99896	0.99969	0.00212	2355	-33.83	0.83873	0.99999
Facc__Cu	21682	0.99830	0.99910	0.00198	90.2	-4.34	0.93238	0.99995
Facc__Ni	21682	0.99939	0.99963	0.00063	32.8	-2.77	0.98334	0.99997
Facc__Pb	21682	0.99994	0.99997	0.00007	170.7	-6.08	0.99709	1.00000

Facc__Zn	21682	0.99718	0.99953	0.00877	4877	-57.7	0.22382	1.00000
Facc__Hg	21682	0.99963	0.99964	0.00010	3.3	-1.20	0.99887	0.99989

Impact assessment of cumulative sewage sludge application on agricultural soils in Europe

Impact assessment of cumulative sewage sludge application has been performed by comparing contamination rate for each of heavy metals and for overall contamination before and after application of **three** proposed sewage sludge formulations (SS_LL, SS_UL and SS_Tavazzi in Table 54) over 10 year at 5 Mg ha^{-1} rate. Concentrations of heavy metals from 14 726 agricultural soils of LUCAS 2009 topsoil database were used as experimental data for impact assessment. Contamination rate was determined taking into consideration **both** Upper and lower limit values of concentrations of heavy metals in soils from Sewage Sludge Directive (EU_LL and EU_UL; Figure 51). Therefore **six** different scenarios were managed to estimate the impact assessment. Distribution of contamination rate before and after application of each of sewage sludge formulations as well as the contamination rate difference at NUTS2 regions are shown in Figure 51 to Figure 62.

Impact of application of three different sewage sludge formulations by using Upper limit values of concentrations for each heavy metals in soils (EU_**UL**) are represented from Figures 20 to 25. Contamination rates for each of heavy metals are shown in Figure 20. New contamination rates for Cd are around 20-40% for the most of the NUTS2 regions except for Ireland where IE2 NUTS2 region is increased by 20% to reach a contamination rate around 70%. Some regions of Italy and Greece are increasing the copper contamination rate up to 60%. New contamination rates for mercury and lead are around 20-40% and 10-20% respectively. Largest contamination rates were recorded for Nickel as already pointed out that could be one of the key metals controlling the amount of sewage sludge to be applied in agricultural soils. In terms of overall contamination rates, some regions from Greece, Cyprus and north of Italy reach contamination rates above 80% over 10 years applying 5 Mg SS_**UL** per hectare. In general terms, the overall contamination rate is increased around 20% for the most of the NUTS2 regions when SS_**UL** formulation is applied (Figure 21). Lower effects on contamination rates are observed for SS_**LL** and SS_**Tavazzi** formulations as they have the lowest concentrations of heavy metals. Thus maximum differences of 10% were observed when application of SS_**LL** formulation was analyzed (Figures 22 and 23) and differences lower than 3.5% when the effect of application of SS_**Tavazzi** formulation was quantified.

Impact of application of three different sewage sludge formulations by using the Lower limit values of concentrations for each heavy metals in soils (EU_**LL**) are represented from Figures 26 to 31. The strictest scenario is shown in Figures 26 and 27. Sewage sludge formulation with high content in heavy metals (SS_**UL**) is applied on agricultural soils controlled by those lower limit values of concentrations in soils (EE_**LL**). Contamination rates for Copper and Nickel would be above 70% for the most of the NUST2 regions after SS application (Figure 26). All European soils show contamination rates higher than 70% as contamination rates were increased around 20-40% when SS_**UL** were applied (Figure 27). Contamination rates above 90% would be found for some countries such as Ireland, Italy, Greece, Romania, Bulgaria, Austria, Eslovenia, Cyprus and Belgium. They cannot be labelled as contaminated sites and Sewage Sludge formulations could be carefully applied on agricultural lands. Same trends are observed when application of SS_**LL** formulation is analyzed (Figures 28 and 29). Copper and Nickel seems those heavy metals to be carefully followed up (Figure 28) as overall contamination rates reach values above 90% for Italy, Greece, Estonia, Ireland and Estonia (Figure 29). Lately, application of SS_**Tavazzi** formulation in agricultural soils is significantly affecting on some specific regions in Italy, Greece, Bulgaria, Austria, Ireland, Cyprus and Romania where large baseline contamination rates for Copper and Nickel would be producing new contamination rates above 90% (Figure 30). Application of SS_**Tavazzi** would be increasing the overall contamination rates below 7% (Figure 31) for all NUTS2 regions.

Limit values of concentrations of heavy metals in soils which they lay down in the Sewage Sludge Directive should be compared with other soil screening values in Europe in order to find out the actual suitability of these limit values. Carlon (2007) reported soil screening values as generic quality standards to match soil pollution and human health. They were classified on the basis of the risk levels (negligible, intermediate and potentially unacceptable). Negligible risk values could correspond to background level and they are no related to the land use. The other ones, intermediate and potentially unacceptable risks, use to be related to the anthropogenic practices. Lower and upper limit values of concentrations of heavy metals in soils as lay down in Sewage Sludge Directive (Table 1) can be compared to those median values (mg/Kg) for negligible, intermediate and potentially unacceptable levels as reported by Carlon (2007) that are 29-30-50 (As), 0.7-3.0-6.0 (Cd), 115-250-275 (Cr), 36-110-345 (Cu), 0.3-2.5-10 (Hg), 35-140-175 (Ni), 83-195-450 (Pb), and 140-500-700 (Zn) respectively.

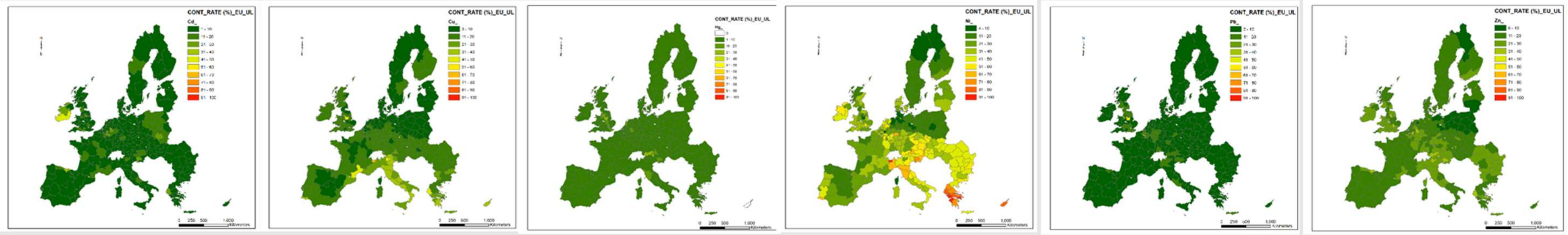
Maximum limit values of concentrations for Cadmium (1-3), Mercury (1.0-1.5), Nickel (30-75) and Zn (150-300) fall between negligible and intermediate risk levels. Lower limit value for Pb (50) is below negligible level. Upper limit values for Pb (300) and Cu (140) are above intermediate risk level. Limit values above intermediate risk level should be revised as significant and harmful effects on human health could be reported as they are exceeded. Limit value for Arsenic (25.5) and Chromium (100) were used as average values from those national legislations with available data as they are not reported in the current sewage sludge European Legislation. Both of them are below intermediate risk levels and, therefore, they could be proposed to be used as limit values. Other more stringent threshold values for Arsenic (5mg/kg) have been reported elsewhere (Ministry of the Environment of Finland, 2007) and further analysis should be conducted in order to include limit values for Arsenic in sewage sludge legislation as it is a naturally occurring ubiquitous metalloid and a Class I human Carcinogen (Chen et al. 2019). The use of contamination rate is shown to be a very useful and powerful tool not only to discriminate between contaminated and no contaminated sites but also to quantify the soil pollution load when different limit values of concentrations of heavy metals in soils are used and to identify potential contaminated sites. Additionally, this factor has been also used to quantify the impact of the application of three sewage sludge formulations with different composition of heavy metals for 10 years at 5 Mg ha^{-1} rate. Thus, the application of sewage sludge over 10 successive years at 5 t/ha rate will increase the overall contamination rate by 50% whether sewage sludge formulations with content of heavy metals close to the upper limits are applied on agricultural soils in Europe. However overall contamination rate in agricultural soils in Europe will increase by 3% whether sewage sludge formulations with content of heavy metals by using the average concentration values from 61 sewage sludge samples analysed in Europe (Tavazzi et al. 2012) are applied on agricultural soils in Europe. These findings related to the content of heavy metals in sewage sludge are very relevant as combined with those limit values of concentrations of heavy metal in soils as lay down in the Sewage Sludge Directive. Hence, sewage sludge formulations with content (mg kg^{-1}) of heavy metals of Cd (40), Cu (1 750), Ni (400), Pb (1 200), Zn (4 000) and Hg (22) should be carefully applied for 10 year at 5 Mg ha^{-1} rate on agricultural soils in Europe whether lower limits of concentrations of heavy metals in soils of Cd (1), Cu (50), Ni (30), Pb (1 200), Zn (4 000) and Hg (22) are implemented as overall contamination rate is higher than 70% and above 80% in the most of the NUTS2 level.

Overall contamination rate in agricultural soils in Europe will increase by 3% whether sewage sludge formulations with content of heavy metals by using the average concentration values from 61 sewage sludge samples (Tavazzi et al. 2012) are applied on agricultural soils in Europe. However, application of sewage sludge over 10 consecutive years at 5 Mg ha^{-1} rate will increase the overall contamination rate up to 40% whether sewage sludge formulations with content of heavy metals close to the upper limits are applied. Hence, sewage sludge with content (mg kg^{-1}) of heavy metals of Cd (40), Cu (1750), Ni (400), Pb (1 200), Zn (4 000) and Hg (22) should be carefully applied over 10 year at 5 Mg ha^{-1} rate on agricultural soils in Europe when lower limits of concentrations of heavy metals in soils of Cd (1), Cu (50), Ni (30), Pb (1 200), Zn (4 000) and Hg (22) are used as overall contamination rate is higher than 70% in the most of the NUTS2 regions.

The application of SS_UL; SS_LL and SS_Tav sewage sludge formulations increases the percentage of contaminated sites by 23%, 12% and 1% respectively when applied on agricultural lands with EU_UL limit values. Thus, the number of contaminated sites goes from 977 before application to 1 198, 1092 and 987 respectively

The application of SS_UL; SS_LL and SS_Tav sewage sludge formulations increases the percentage of contaminated sites by 46%, 25% and 2% respectively when applied on agricultural lands with EU_LL limit values. Thus, the number of contaminated sites goes from 5 046 before application to 7 357, 6 307 and 5 165 respectively

Figure 51. Effect of the application of sewage sludge formulation (SS_UL) at 5 Mg ha^{-1} for 10 years on soil contamination rate at NUTS2 level for each of the heavy metals (Cd, Cu, Hg, Ni, Pb, and Zn) by using the upper limit values of concentrations of heavy metals (mg kg^{-1}) in soils as stated in the SSD (EU_UL).



Heavy Metal	EU_LL	EU_UL
Cd	1.00	3.00
Cu	50.0	140
Ni	30.0	75.0
Pb	50.0	300
Zn	150	300
Hg	1.00	1.50

Heavy metal	SS_LL	SS_UL	SS_Tavazzi
Cd	20	40	0.900
Cu	1000	1750	257
Ni	300	400	29.0
Pb	750	1200	47.6
Zn	500	4000	0.070
Hg	16	22	0.400

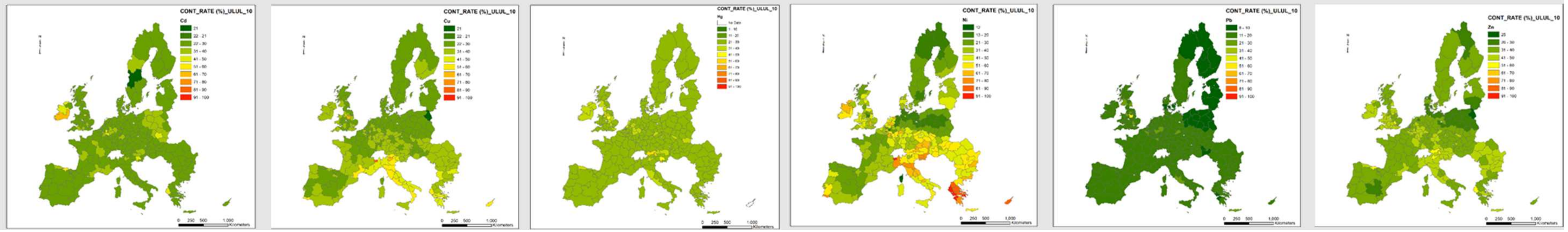


Figure 52. Effect of the application of sewage sludge formulation (SS_UL) at 5 Mg ha^{-1} for 10 years on the overall soil contamination rate at NUTS2 level by using the upper limit values of concentrations of heavy metals (mg kg^{-1}) in soils as stated in the SSD (EU_UL). Overall contamination rates before and after sewage sludge application are represented by orange and white bars respectively. Contamination rates for each of the heavy metals after sewage sludge application are represented by lines.

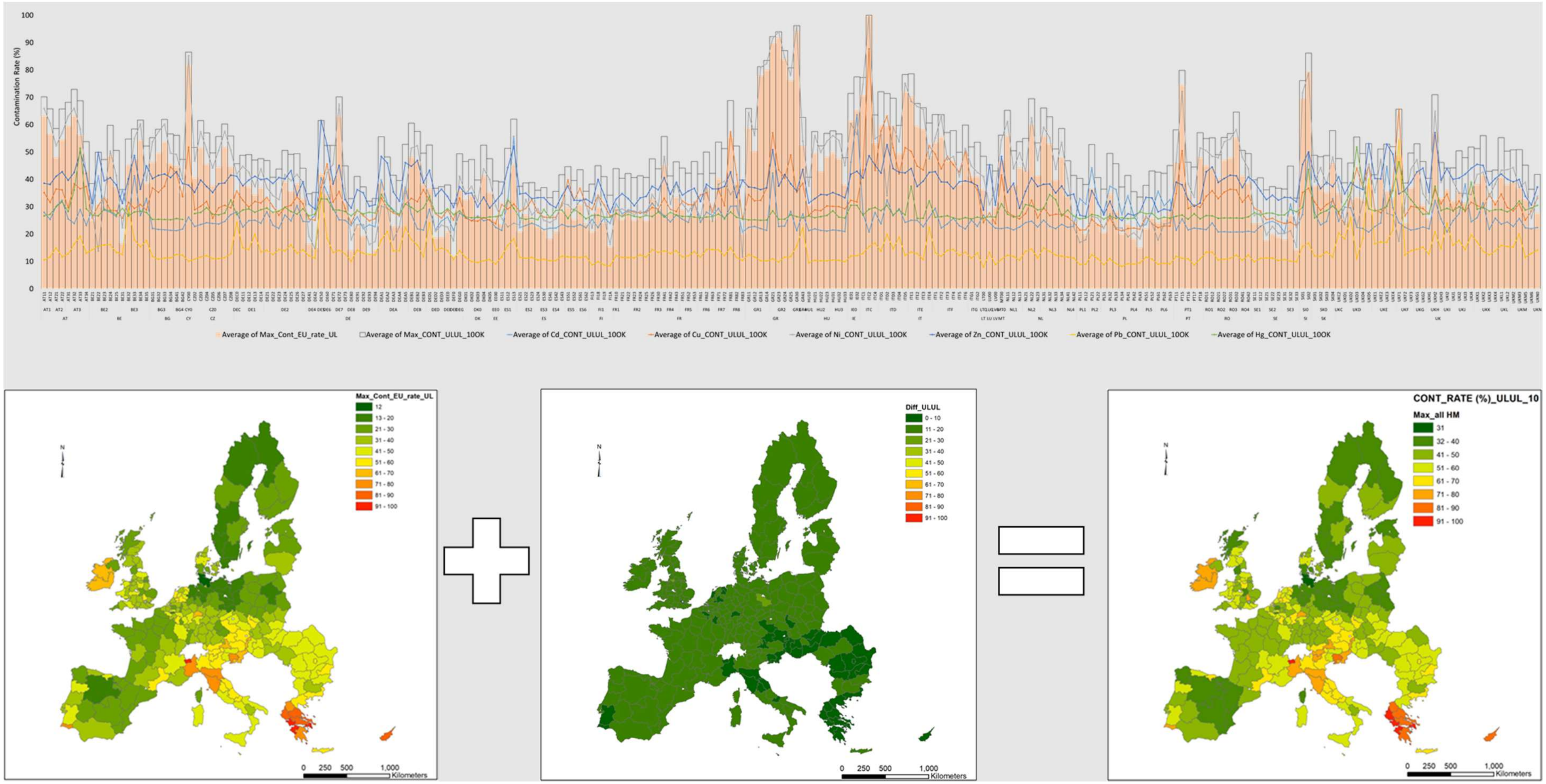
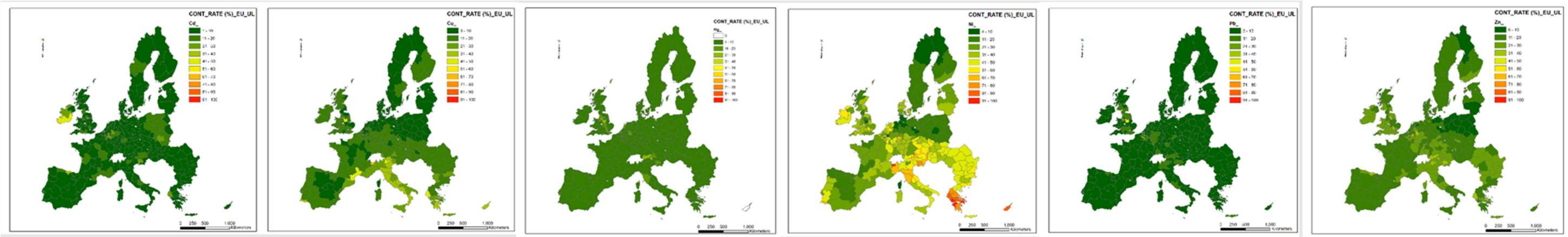


Figure 53. Effect of the application of sewage sludge formulation (SS_LL) at 5 Mg ha^{-1} for 10 years on soil contamination rate at NUTS2 level for each of the heavy metals (Cd, Cu, Hg, Ni, Pb, and Zn) by using the upper limit values of concentrations of heavy metals (mg kg^{-1}) in soils as stated in the SSD (EU_UL).



Heavy Metal	EU_LL	EU_UL
Cd	1.00	3.00
Cu	50.0	140
Ni	30.0	75.0
Pb	50.0	300
Zn	150	300
Hg	1.00	1.50

Heavy metal	SS_LL	SS_UL	SS_Tavazzi
Cd	20	40	0.900
Cu	1000	1750	257
Ni	300	400	29.0
Pb	750	1200	47.6
Zn	500	4000	0.070
Hg	16	22	0.400

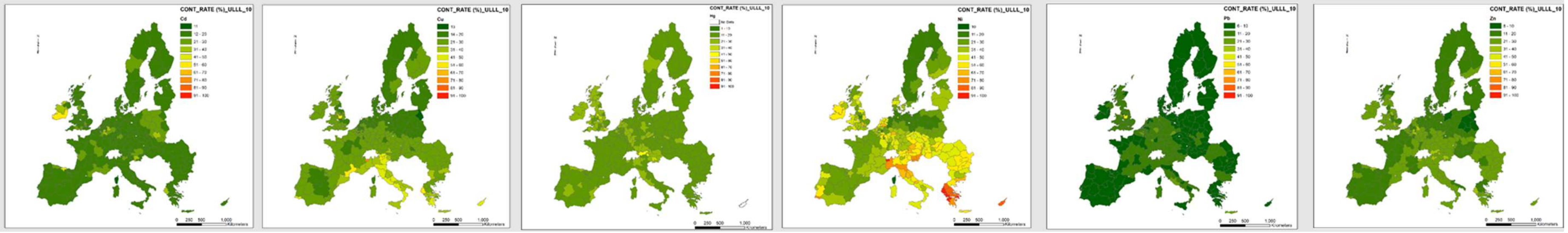


Figure 54. Effect of the application of sewage sludge formulation (SS_LL) at 5 Mg ha^{-1} for 10 years on the overall soil contamination rate at NUTS2 level by using the Upper Limit values of concentrations of heavy metals (mg kg^{-1}) in soils as stated in the SSD (EU_UL). Overall contamination rates before and after sewage sludge application are represented by orange and white bars respectively. Contamination rates for each of the heavy metals after sewage sludge application are represented by lines.

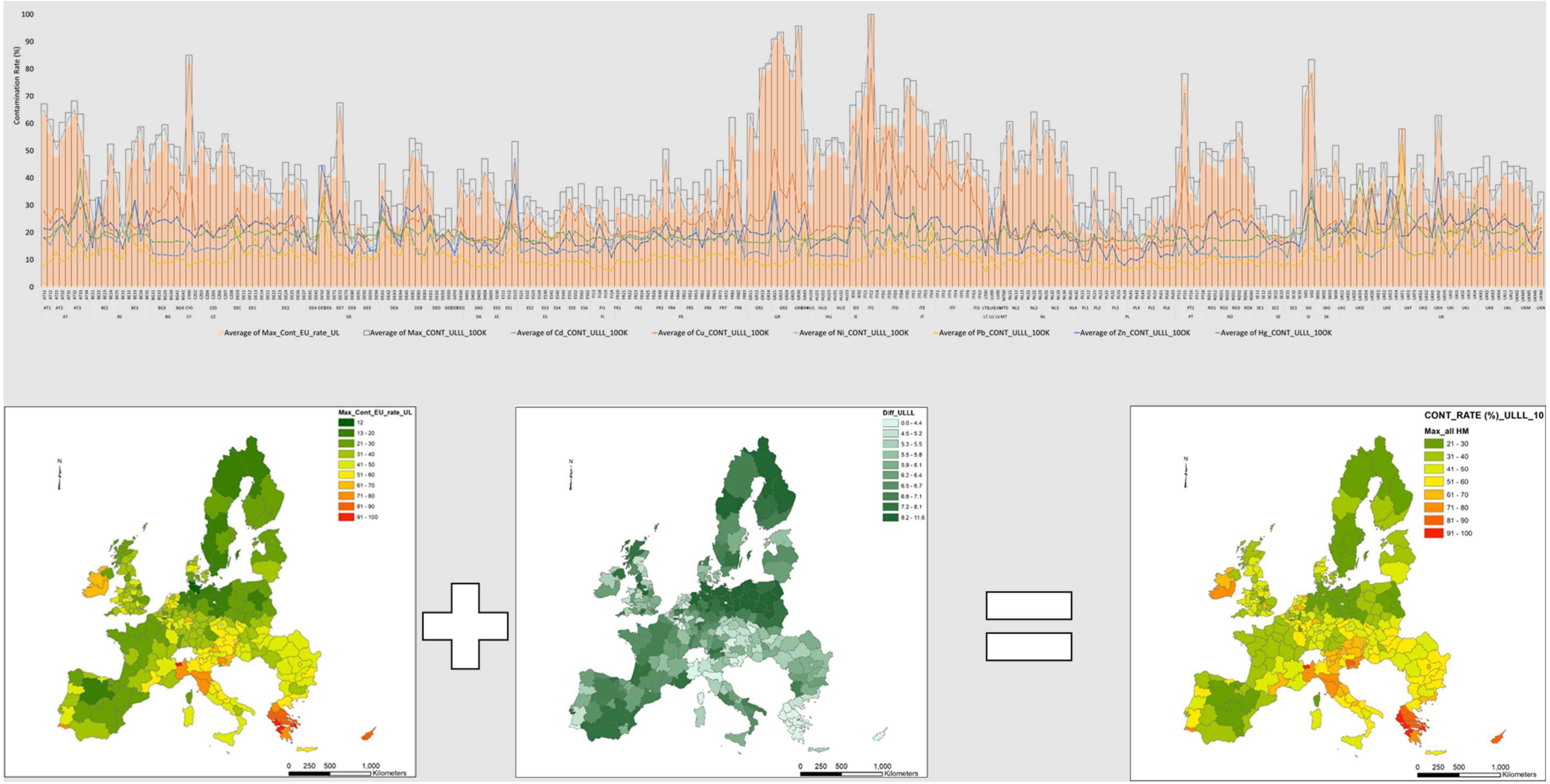
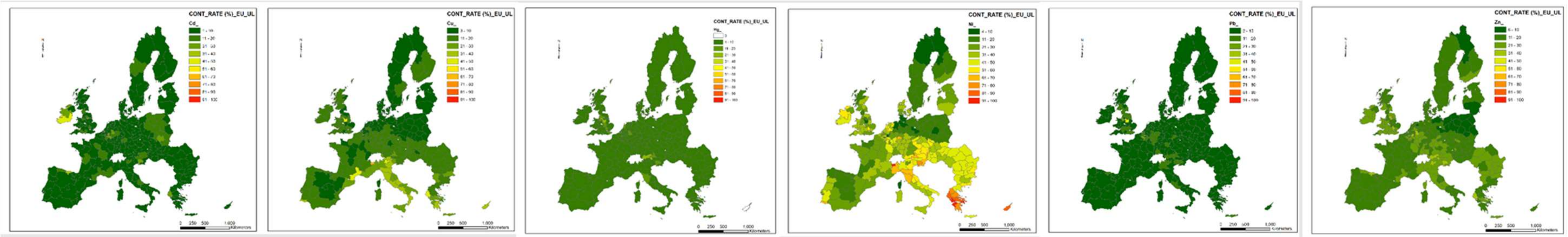


Figure 55. Effect of the application of sewage sludge formulation (SS_Tavazzi) at 5 Mg ha^{-1} for 10 years on soil contamination rate at NUTS2 level for each of the heavy metals (Cd, Cu, Hg, Ni, Pb, and Zn) by using the upper limit values of concentrations of heavy metals (mg kg^{-1}) in soils as stated in the SSD (EU_UL).



Heavy Metal	EU_LL	EU_UL
Cd	1.00	3.00
Cu	50.0	140
Ni	30.0	75.0
Pb	50.0	300
Zn	150	300
Hg	1.00	1.50

Heavy metal	SS_LL	SS_UL	SS_Tavazzi
Cd	20	40	0.900
Cu	1000	1750	257
Ni	300	400	29.0
Pb	750	1200	47.6
Zn	500	4000	0.070
Hg	16	22	0.400

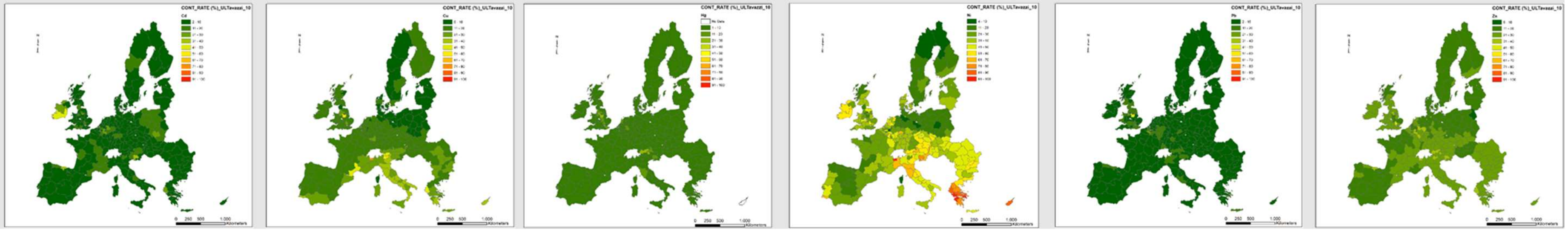


Figure 56. Effect of the application of sewage sludge formulation (SS_Tavazzi) at 5 Mg ha^{-1} for 10 years on the overall soil contamination rate at NUTS2 level by using the upper limit values of concentrations of heavy metals (mg kg^{-1}) in soils as stated in the SSD (EU_UL). Overall contamination rates before and after sewage sludge application are represented by orange and white bars respectively. Contamination rates for each of the heavy metals after sewage sludge application are represented by lines.

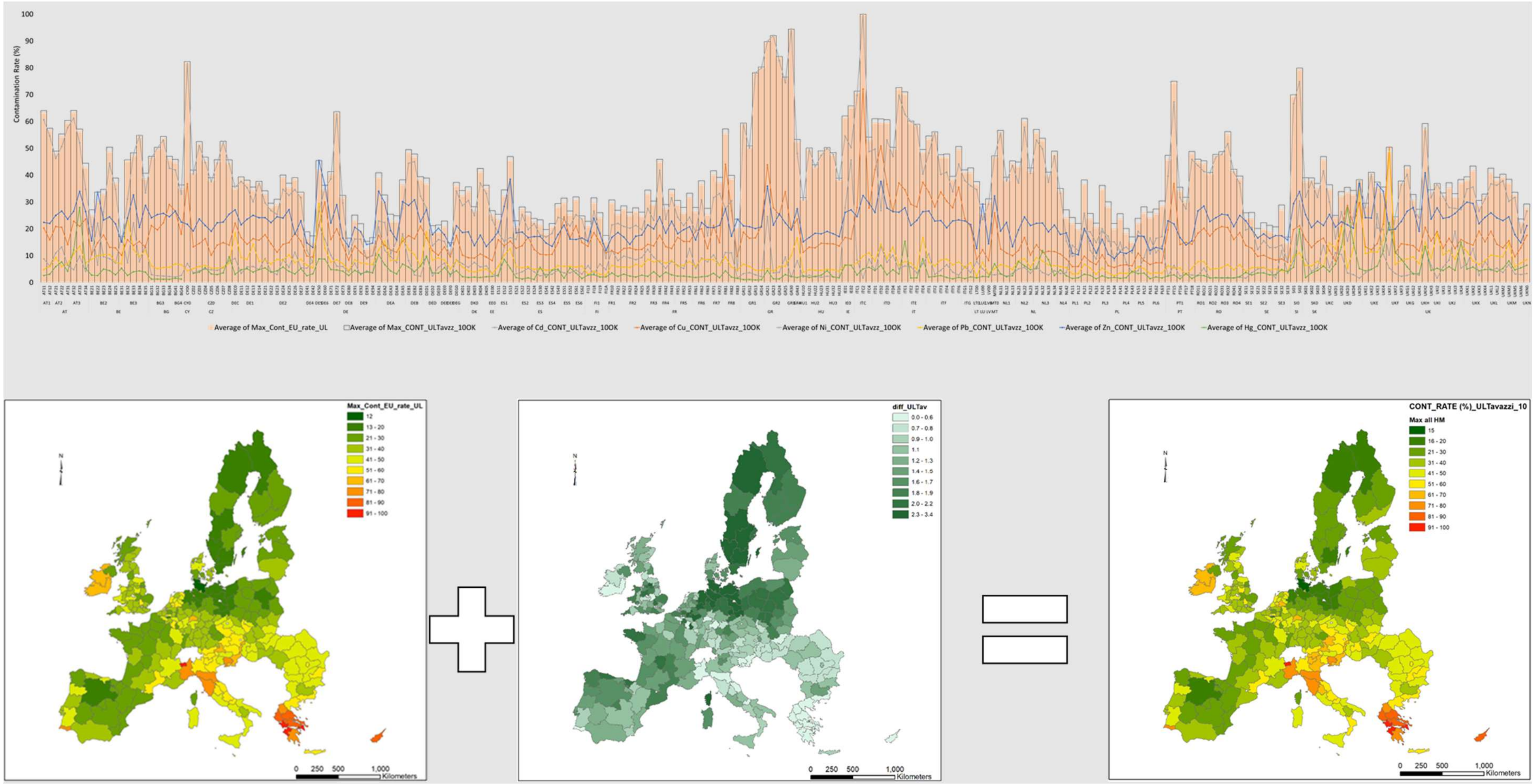
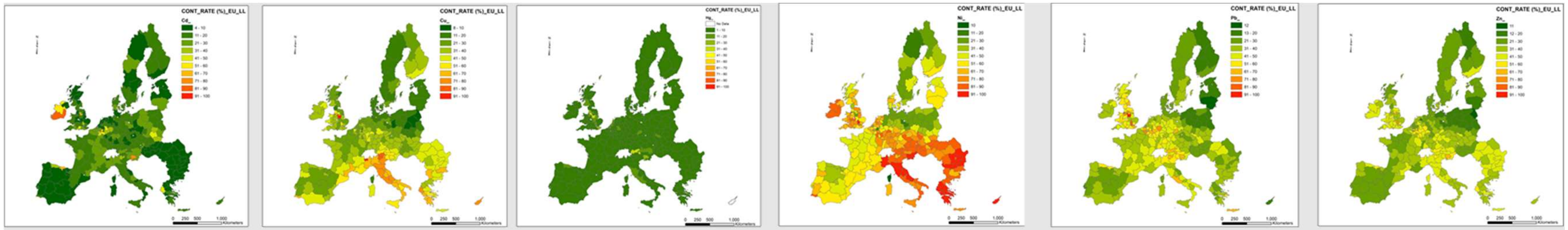


Figure 57. Effect of the application of sewage sludge formulation (SS_UL) at 5 Mg ha^{-1} for 10 years on soil contamination rate at NUTS2 level for each of the heavy metals (Cd, Cu, Hg, Ni, Pb, and Zn) by using the lower limit values of concentrations of heavy metals (mg kg^{-1}) in soils as stated in the SSD (EU_LL).



Heavy Metal	EU_LL	EU_UL
Cd	1.00	3.00
Cu	50.0	140
Ni	30.0	75.0
Pb	50.0	300
Zn	150	300
Hg	1.00	1.50

Heavy metal	SS_LL	SS_UL	SS_Tavazzi
Cd	20	40	0.900
Cu	1000	1750	257
Ni	300	400	29.0
Pb	750	1200	47.6
Zn	500	4000	0.070
Hg	16	22	0.400

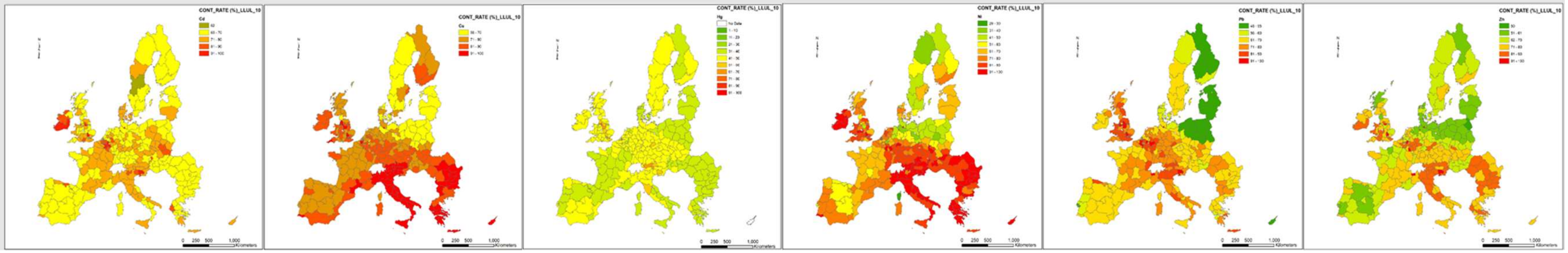
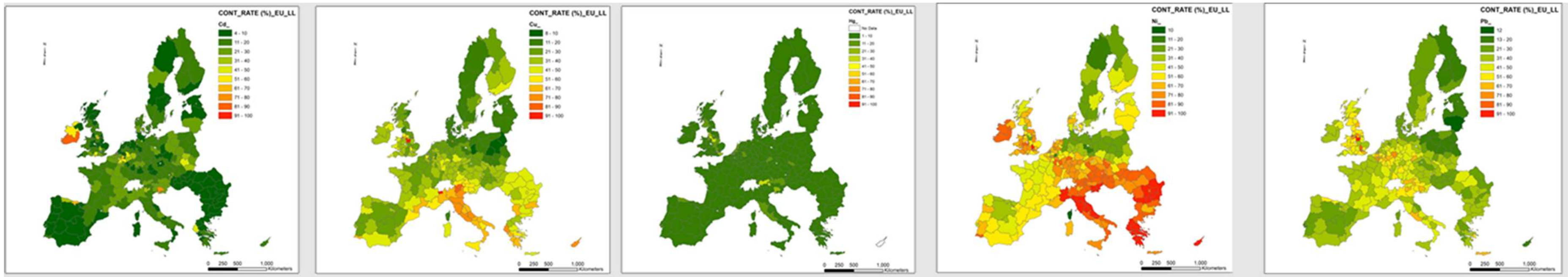


Figure 58. Effect of the application of sewage sludge formulation (SS_LL) at 5 Mg ha^{-1} for 10 years on soil contamination rate at NUTS2 level for each of the heavy metals (Cd, Cu, Hg, Ni, Pb, and Zn) by using the lower limit values of concentrations of heavy metals (mg kg^{-1}) in soils as stated in the SSD (EU_LL).



Heavy Metal	EU_LL	EU_UL
Cd	1.00	3.00
Cu	50.0	140
Ni	30.0	75.0
Pb	50.0	300
Zn	150	300
Hg	1.00	1.50

Heavy metal	SS_LL	SS_UL	SS_Tavazzi
Cd	20	40	0.900
Cu	1000	1750	257
Ni	300	400	29.0
Pb	750	1200	47.6
Zn	500	4000	0.070
Hg	16	22	0.400

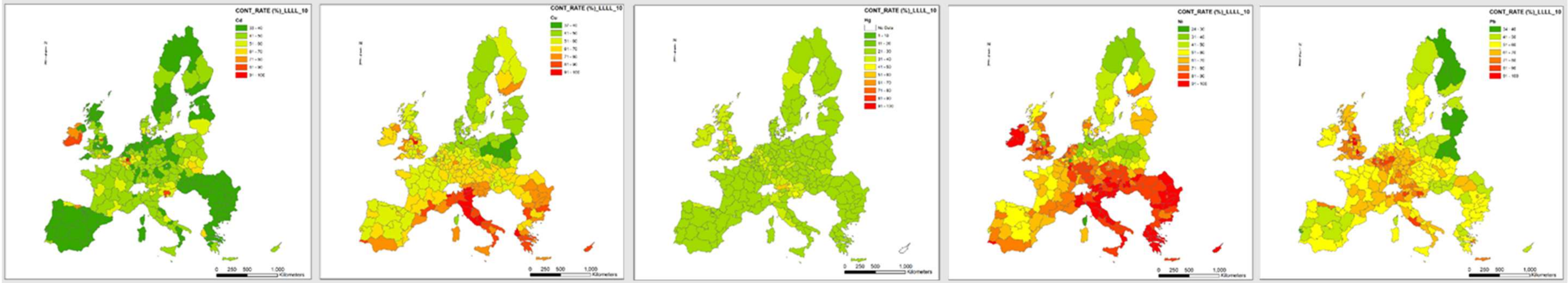


Figure 59. Effect of the application of sewage sludge formulation (SS_LL) at 5 Mg ha^{-1} for 10 years on the overall soil contamination rate at NUTS2 level by using the lower limit values of concentrations of heavy metals (mg kg^{-1}) in soils as stated in the SSD (EU_LL). Overall contamination rates before and after sewage sludge application are represented by green and white bars respectively. Contamination rates for each of the heavy metals after sewage sludge application are represented by lines.

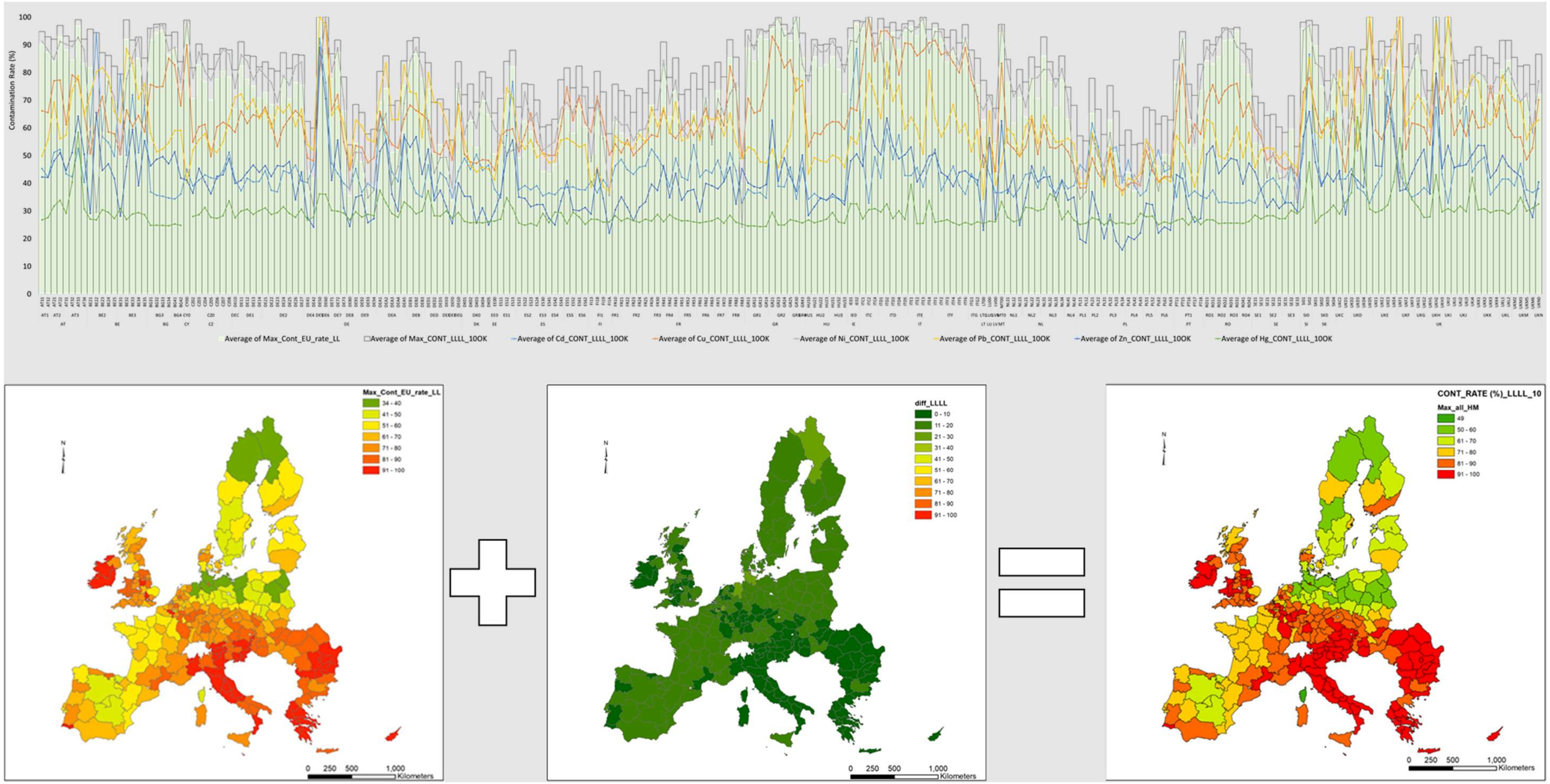
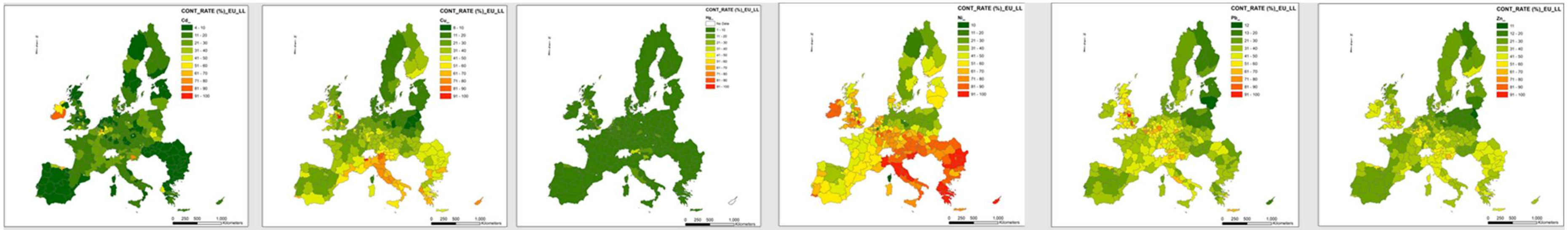


Figure 60. Effect of the application of sewage sludge formulation (SS_Tavazzi) at 5 Mg ha^{-1} for 10 years on soil contamination rate at NUTS2 level for each of the heavy metals (Cd, Cu, Hg, Ni, Pb, and Zn) by using the lower limit values of concentrations of heavy metals (mg kg^{-1}) in soils as stated in the SSD (EU_LL).



Heavy Metal	EU_LL	EU_UL
Cd	1.00	3.00
Cu	50.0	140
Ni	30.0	75.0
Pb	50.0	300
Zn	150	300
Hg	1.00	1.50

Heavy metal	SS_LL	SS_UL	SS_Tavazzi
Cd	20	40	0.900
Cu	1000	1750	257
Ni	300	400	29.0
Pb	750	1200	47.6
Zn	500	4000	0.070
Hg	16	22	0.400

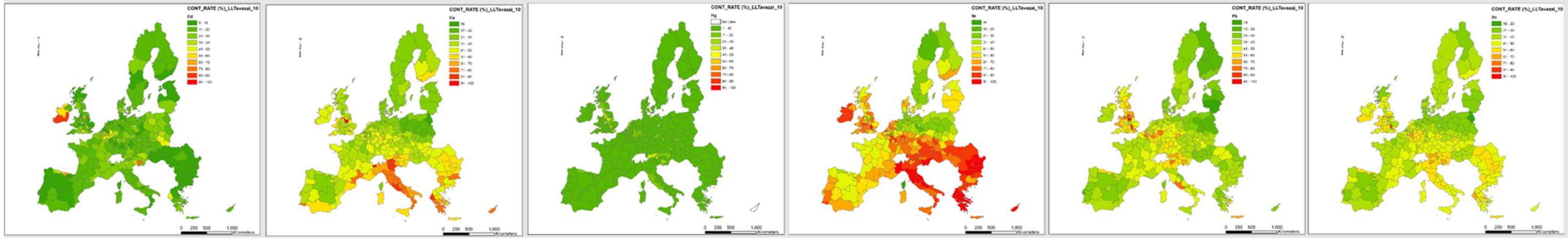


Figure 61. Effect of the application of sewage sludge formulation (SS_Tavazzi) at 5 Mg ha^{-1} for 10 years on the overall soil contamination rate at NUTS2 level by using the lower limit values of concentrations of heavy metals (mg kg^{-1}) in soils as stated in SSD (EU_LL). Overall contamination rates before and after sewage sludge application are represented by green and white bars respectively. Contamination rates for each of the heavy metals after sewage sludge application are represented by lines.

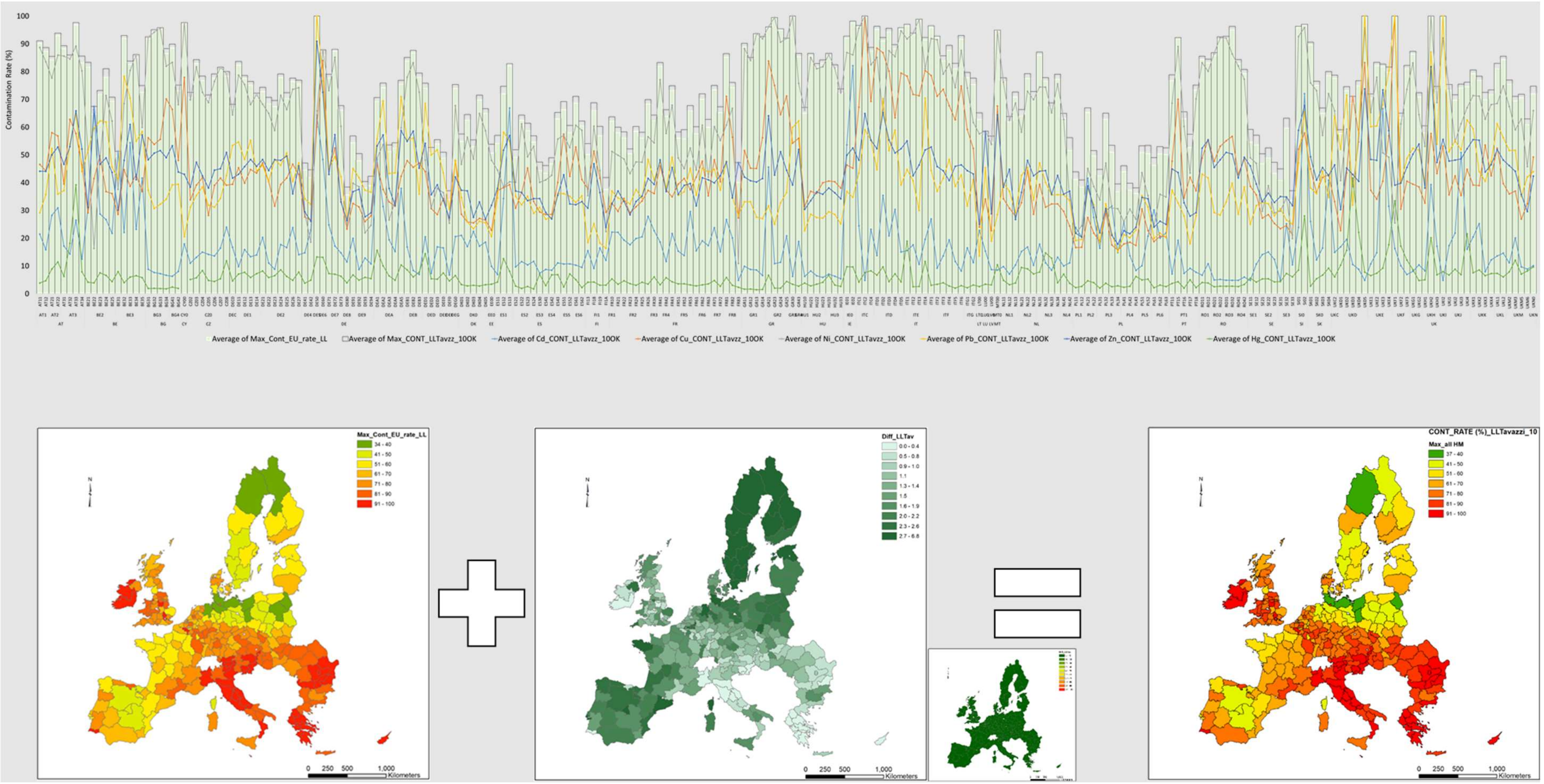
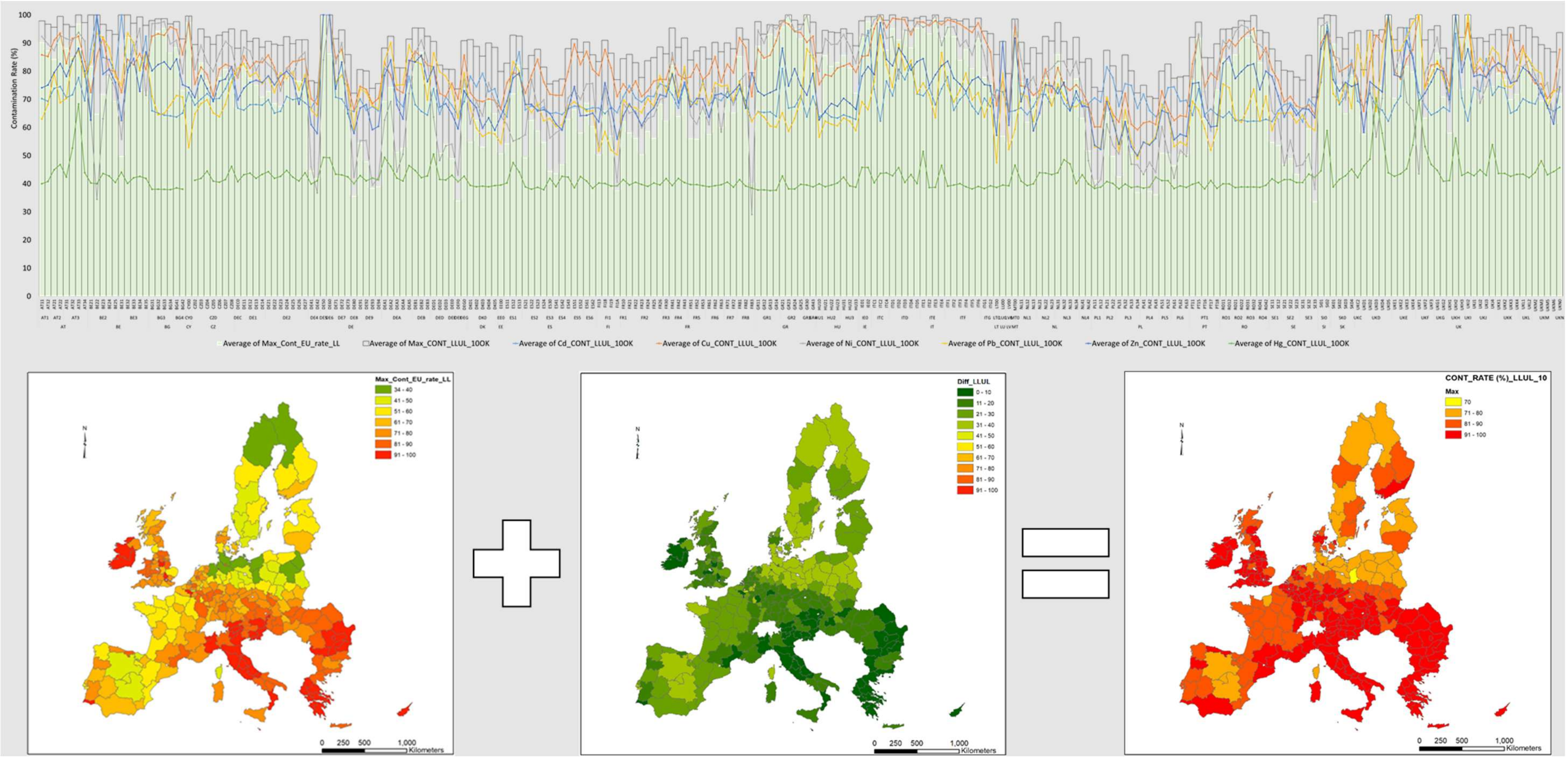


Figure 62. Effect of the application of sewage sludge formulation (SS_UL) over 10 consecutive years at 5 Mg ha^{-1} rate on the overall soil contamination rate at NUTS2 level by using the Lower Limit values of concentrations of heavy metals (mg kg^{-1}) in soils lay down in the Sewage Sludge Directive (EU_LL). Overall contamination rates before (left map) and after sewage sludge application (right map) are represented by green and white bars respectively as well. Contamination rates for each of the heavy metals after sewage sludge application are represented by lines.



14.2.2 Nitrogen input exceeding 145 kg ha^{-1} yr^{-1}

Table 60. Detailed overview on % of area exceeding the 145 kg ha^{-1} yr^{-1}on NUTS2 level.

NUTS2_ID	***TOT Area IN ha***	***NUTS2 Area km²***	***% area > 145 kg N ha***	***NUTS2_ID***	***TOT Area IN ha***	***NUTS2 Area km²***	***% area > 145 kg N ha***	***NUTS2_ID***	***TOT Area IN ha***	***NUTS2 Area km²***	***% area > 145 kg N ha***
AT11	*392165*	*3922*	*99*	*DE94*	*1472245*	*14722*	*83*	*HU33*	*1835431*	*18354*	*29*
AT12	*1917242*	*19172*	*99*	*DEA1*	*530454*	*5305*	*83*	*IE01*	*3368058*	*33681*	*28*
AT13	*43193*	*432*	*99*	*DEA2*	*738451*	*7385*	*83*	*IE02*	*3704825*	*37048*	*27*
AT21	*957151*	*9572*	*98*	*DEA3*	*696518*	*6965*	*83*	*ITC1*	*2545502*	*25455*	*25*
AT22	*1637083*	*16371*	*98*	*DEA4*	*652335*	*6523*	*81*	*ITC2*	*326604*	*3266*	*25*
AT31	*1202653*	*12027*	*98*	*DEA5*	*792135*	*7921*	*80*	*ITC3*	*533569*	*5336*	*23*
AT32	*721957*	*7220*	*97*	*DEB1*	*805598*	*8056*	*80*	*ITC4*	*2378803*	*23788*	*23*
AT33	*1261168*	*12612*	*97*	*DEB2*	*489117*	*4891*	*80*	*ITF1*	*1088405*	*10884*	*22*
AT34	*258518*	*2585*	*97*	*DEB3*	*687513*	*6875*	*80*	*ITF2*	*450102*	*4501*	*21*
BE10	*13299*	*133*	*97*	*DEC0*	*259589*	*2596*	*80*	*ITF3*	*1359354*	*13594*	*20*
BE21	*286393*	*2864*	*97*	*DED2*	*800714*	*8007*	*79*	*ITF4*	*1944054*	*19441*	*20*
BE22	*232242*	*2322*	*96*	*DED2*	*800714*	*8007*	*79*	*ITF5*	*1007526*	*10075*	*19*
BE23	*306215*	*3062*	*96*	*DED4*	*643850*	*6439*	*78*	*ITF6*	*1525278*	*15253*	*19*
BE24	*216289*	*2163*	*96*	*DED5*	*397839*	*3978*	*75*	*ITG1*	*1168297*	*11683*	*17*
BE25	*317299*	*3173*	*96*	*DEE0*	*2058427*	*20584*	*73*	*ITG2*	*2409907*	*24099*	*16*
BE31	*101361*	*1014*	*96*	*DEF0*	*1538602*	*15386*	*71*	*ITH1*	*740327*	*7403*	*15*
BE32	*383661*	*3837*	*95*	*DEG0*	*1609098*	*16091*	*67*	*ITH2*	*626430*	*6264*	*14*
BE33	*383117*	*3831*	*95*	*ES11*	*2953395*	*29534*	*67*	*ITH3*	*1778970*	*17790*	*13*
BE34	*447194*	*4472*	*95*	*ES12*	*1043516*	*10435*	*67*	*ITH4*	*764530*	*7645*	*12*
BE35	*367356*	*3674*	*94*	*ES13*	*527870*	*5279*	*66*	*ITH5*	*2267603*	*22676*	*9*
BG31	*1911102*	*19111*	*94*	*ES21*	*725107*	*7251*	*65*	*ITI1*	*2290044*	*22900*	*8*
BG32	*921505*	*9215*	*94*	*ES22*	*1026477*	*10265*	*64*	*ITI2*	*846920*	*8469*	*8*
BG33	*48554*	*486*	*94*	*ES23*	*514390*	*5144*	*64*	*ITI3*	*942332*	*9423*	*8*
BG34	*476557*	*4766*	*94*	*ES24*	*4791299*	*47913*	*64*	*ITI4*	*1728111*	*17281*	*8*
BG41	*2025319*	*20253*	*94*	*ES30*	*800812*	*8008*	*64*	*LI00*	*16045*	*160*	*7*
BG42	*1812076*	*18121*	*94*	*ES41*	*9404276*	*94043*	*63*	*LT00*	*6491541*	*64915*	*7*
CH01	*869855*	*8699*	*94*	*ES42*	*7940276*	*79403*	*63*	*LU00*	*263430*	*2634*	*7*
CH02	*1006152*	*10062*	*93*	*ES43*	*4154098*	*41541*	*62*	*LV00*	*6465881*	*64659*	*6*
CH03	*201051*	*2011*	*93*	*ES51*	*3220350*	*32204*	*62*	*ME00*	*1392173*	*13922*	*6*
CH04	*174584*	*1746*	*93*	*ES52*	*2327667*	*23277*	*62*	*MK00*	*2551099*	*25511*	*5*
CH05	*1145064*	*11451*	*93*	*ES53*	*500245*	*5002*	*59*	*NL11*	*243864*	*2439*	*5*
CH06	*446554*	*4466*	*93*	*ES61*	*8180446*	*81804*	*56*	*NL12*	*334374*	*3344*	*4*
CH07	*285646*	*2856*	*93*	*ES62*	*1130419*	*11304*	*55*	*NL13*	*275091*	*2751*	*4*
CZ01	*47660*	*477*	*93*	*FR10*	*1199548*	*11995*	*55*	*NL21*	*348772*	*3488*	*4*
CZ02	*1108924*	*11089*	*93*	*FR21*	*2560569*	*25606*	*55*	*NL22*	*512678*	*5127*	*4*
CZ03	*1758895*	*17589*	*92*	*FR22*	*1942907*	*19429*	*54*	*NL23*	*148072*	*1481*	*3*

CZ04	870757	8708	92	FR23	1239316	12393	53	NL31	144282	1443	3
CZ05	1252282	12523	92	FR24	3955289	39553	52	NL32	282546	2825	3
CZ06	1395478	13955	92	FR25	1786878	17869	51	NL33	324214	3242	3
CZ07	928669	9287	91	FR26	3171862	31719	50	NL34	186489	1865	3
CZ08	534291	5343	91	FR30	1254800	12548	48	NL41	512756	5128	3
DE11	1057304	10573	91	FR41	2373773	23738	48	NL42	230803	2308	3
DE12	698202	6982	91	FR42	832458	8325	45	PL11	1823693	18237	3
DE13	938031	9380	90	FR43	1628693	16287	44	PL12	3552044	35520	3
DE14	916261	9163	89	FR51	3219134	32191	43	PL21	1515095	15151	2
DE21	1749962	17500	89	FR52	2751605	27516	43	PL22	1232515	12325	2
DE22	1027086	10271	88	FR53	2615814	26158	42	PL31	2499538	24995	2
DE23	972598	9726	87	FR61	4203970	42040	42	PL32	1780812	17808	2
DE24	728604	7286	87	FR62	4559990	45600	41	PL33	1174571	11746	2
DE25	721232	7212	87	FR63	1709473	17095	41	PL34	2022879	20229	2
DE26	860458	8605	86	FR71	4499124	44991	41	PL41	2994607	29946	2
DE27	1005152	10052	86	FR72	2614097	26141	40	PL42	2242574	22426	2
DE30	88540	885	86	FR81	2783407	27834	39	PL43	1404197	14042	2
DE40	2946903	29469	86	FR82	3169343	31693	38	PL51	1991316	19913	2
DE50	42170	422	86	FR83	869148	8691	37	PL52	941086	9411	1
DE60	74653	747	86	HR04	3178268	31783	36	PL61	1801236	18012	1
DE71	730276	7303	85	HR03	2375334	23753	35	PL62	2423160	24232	1
DE72	546453	5465	85	HU10	686023	6860	33	PL63	1810807	18108	1
DE73	828910	8289	85	HU21	1089008	10890	32	PT11	2135056	21351	1
DE80	2356813	23568	84	HU22	1148643	11486	31	PT15	225055	2251	1
DE91	819189	8192	84	HU23	1427613	14276	31	PT16	2469204	24692	1
DE92	910241	9102	84	HU31	1343004	13430	31	PT18	2201701	22017	1
DE93	1555853	15559	84	HU32	1776929	17769	29	RO11	3413819	34138	1