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Resource recovery strategies and schemes: A regional case study on sewage sludge hub centres in Italy

Elisa Blumenthal [a], Alessia Foglia [a,*], Alberto Piasentin [c], Corinne Andreola [a], Anna Laura Eusebi [a], Nicola Frison [b], Francesco Fatone [a]

[a] Department of Science and Engineering of Matter, Environment and Urban Planning, Università Politecnica delle Marche, via Brecce Bianche, 60131 Ancona, Italy
[b] Department of Biotechnology, University of Verona, 37134 Verona, Italy
[c] Alto Trevigiano Servizi Srl, Public-Owned Water Utility, Via Schiavonesca Priula, 86, 31044 Montebelluna, Treviso, Italy

A R T I C L E I N F O

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A B S T R A C T

The establishment of regional sludge treatment hubs has been proposed as a solution to achieve the necessary economic scale for sustainable resource recovery and safe reuse in non-metropolitan areas. However, existing literature provides limited insights into their sustainability and effectiveness, and inadequate legal frameworks often hinder efficient sludge treatment, leading to improper disposal and increased risks. This article addresses these gaps by evaluating the environmental impact of a centralised sludge treatment system through a real case study, highlighting the importance of selecting sewage sludge with minimal risks for centralised resource recovery and safe reuse, in line with the recent review of the Sewage Sludge Directive.

Building on the Horizon 2020 SMART-Plant innovation action, a regional sludge hub has been designed to treat and valorise the sewage sludge from 52 municipalities in Treviso, serving around 500,000 residents. In the first phase, the potential for resource recovery and safe reuse was assessed by evaluating the long-term chemical and physical characteristics of the sewage sludge, while considering the replicability of this model. In the second phase, a Life Cycle Assessment (LCA) was conducted to evaluate the environmental impact of various valorisation pathways, including composting, biogas production, phosphorus salts, and biopolymer (PHA) recovery. The final environmental impacts were normalised using the revised ReCiPe 2016 normalisation values. The results indicated that phosphorus and biopolymer recovery was the most sustainable scenario, reducing emissions by an amount equivalent to 176 individuals per day compared to the decentralised system.

1. Introduction

The sewage sludge production is predicted to increase further according to recent environmental legislations such as the European Directive 2024/3019 concerning urban wastewater treatment. Regional Sludge Hub Centres have been identified as a key solution to ensure that sludge is treated to the highest standards for safe reuse in agriculture. Optimising the number and structure of these centres is essential for enhancing the efficiency of national sludge management systems. For example, Uisce Éireann and Irish water are expanding their sludge hub networks to improve the effectiveness and efficiency of sludge treatment [1,2]. Adequate scale remains one of the primary challenges to the technical, economic, and environmental sustainability of resource recovery and safe reuse from municipal wastewater and sewage sludge [3].

The scientific literature consistently underscores the advantages of centralising wastewater treatment plants for resource recovery [4,5]. However, in Italy, many regional water utilities manage multiple municipalities in non-metropolitan areas, where the decision between centralised and decentralised wastewater or sludge management often focuses on cost efficiency, neglecting the potential for resource recovery and environmental sustainability.

The recent evaluation of the EU Sewage Sludge Directive (SSD) [6] highlighted improvements in the quality of sludge produced by European wastewater treatment plants (WWTPs) over recent decades, but also called for higher quality standards for recovered sludge. In this context, water utilities operating across large, geographically diverse areas may face challenges in maintaining sludge quality and ensuring safe reuse of recovered resources.

Before considering the establishment of regional sludge hubs, it is critical to identify key contaminants and resource flows in treated

* Corresponding author.
E-mail address: a.foglia@staff.univpm.it (A. Foglia).

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Glossary

AD	Anaerobic digestion
CAM	Calcium ammonium nitrate
COD	Chemical oxygen demand
EF	Emission factor
EOFP	Photochemical ozone formation: ecosystem quality
FEP	Freshwater eutrophication
FFP	Fossil resource scarcity
GWP	Climate change
HOFP	Photochemical ozone formation: human health
K	Potassium
LCA	Life Cycle Assessment
LCC	Life Cycle Costing
LCI	Life cycle inventory
MEP	Marine eutrophication
MLE	Modified Ludzack-Ettinger
N	Nitrogen
ODP	Ozone depletion
P	Phosphorus
PE	Person equivalent
PHA	Polyhydroxyalkanoate
SBR	Sequencing batch reactor
TAP	Terrestrial acidification
TN	Total nitrogen
TP	Total phosphorus
TSP	Triple superphosphate
TSS	Total suspended solids
VFA	Volatile fatty acids
WWTP	Wastewater treatment plant

wastewater. Understanding pollutants such as heavy metals, organic chemicals, and pharmaceuticals is critical for selecting the most appropriate treatment technologies and ensuring that the recovered materials meet regulatory and environmental standards.

Additionally, assessing the potential for resource recovery, such as biogas or nutrients, is necessary for optimally designing regional hubs.

At the European level, there are several Innovation Actions among the water-related projects [7] that aim at recovering and reusing materials from centralised and decentralised WWTPs. In particular, SMART-Plant, was an Innovation Action aimed at reducing the energy and environmental footprint while promoting the reuse of water and the recovery of valuable materials such as cellulose, biopolymers like PHA (polyhydroxyalkanoate) and nutrients. These recovered products were valued in the construction, chemical, and agricultural supply chains (CORDIS, European Commission). Tested in the province of Treviso, SMART-Plant has and inspired the regional water utility to pursue the design and development of a regional circular sludge hub centre.

In particular, SMART-Plant demonstrated the feasibility of recovering cellulose, PHA, biogas, volatile fatty acids, and phosphorus within existing WWTPs [8].

The recovery of phosphorus from sewage sludge is now a priority, particularly following the new urban wastewater treatment directive [9] which includes both (a) the decrease of effluent concentration standard up to 0.5 mg/L for total phosphorus (P) for WWTPs bigger than 150′000 person equivalent (PE) and (b) the promotion of sustainable phosphorus recovery and safe reuse. In addition, the driver to valorise secondary streams such as sludge is combined with the need to reduce the disposal of solids, ensuring the sustainability of both energy and environmental processes.

Recent reviews, such as Odey et al. [10], have provided valuable insights into faecal sludge management in developing urban centres, focusing on key aspects such as collection, treatment, and composting. They highlight the significant environmental and public health risks associated with improper faecal sludge disposal and advocate for sustainable practices, such as stabilisation and composting, to convert faecal sludge into valuable soil amendments. However, their study does not provide a detailed environmental assessment or quantify the impacts of different treatment options, limiting the understanding of their relative sustainability and effectiveness. Similarly, Topanou et al. [11] highlight the challenges posed by inadequate faecal sludge treatment facilities and the indiscriminate discharge of sludge into the environment, emphasising the need for improved regulations and infrastructure to manage faecal sludge sustainably. While their work stresses the importance of enhanced treatment and valorisation to mitigate environmental pollution and health risks, it does not explore in depth the role of centralised sludge treatment facilities, or sludge hubs, which are crucial for the efficient management of faecal sludge in urbanising areas.

This presents a significant research gap, as the role of sludge hubs in sludge management remains underexplored. Specifically, the efficiency, scalability, and environmental impact of such centralised facilities are not well understood, making it difficult to assess their potential to contribute to sustainable urban sanitation. This study aims to fill this gap by examining the role of sludge hubs in sewage sludge management and evaluating their environmental impact, thereby contributing to the development of more effective and sustainable treatment solutions.

In addition, a critical gap in the current research is the lack of strong and comprehensive legislation governing sludge management. Inadequate legal frameworks can impede the development of efficient sludge treatment systems, leading to improper disposal and increased environmental and public health risks. This article also addresses this gap by considering the recent evaluation of the Sewage Sludge Directive, which presents an opportunity to strengthen regulatory frameworks and enhance the sustainability of sludge management practices across Europe.

After evaluating and benchmarking the characteristics of sewage sludge produced in the studied territory to define acceptable flows for sustainable resource recovery and safe reuse, this paper presents an environmental assessment of a sludge hub that is guiding the regional water utility's decision-making process beyond mere cost-efficiency. To the best of the author's knowledge, no articles in the literature compare territorial scenarios that adopt innovative technologies for nutrient and resource recovery in this manner.

2. Materials and methods

The analysis focused on a region that includes: i) the Salvatronda WWTP, with an initial design treatment capacity of 73,300 PE, ii) 75,501 PE currently not collected and treated in Imhoff tanks, and iii) the transport and composting of 11,023 kgTS/d of sludge produced in 9 WWTPs operated by the same water utility in the nearby Salvatronda area. The planned upgrade of Salvatronda WWTP to 120,000 PE, designated as the sludge hub, will enable the collection and treatment of both the PE currently not collected and the 11,023 kgTS/d of sludge produced in the external WWTPs across the province. Consequently, a regional system boundary (Fig. 1) was defined for the analysis. The final sludge produced at the hub will be sent to a composting plant for further processing.

The water utility operates 10 medium-large WWTPs and more than 40 smaller plants, each with a capacity of less than 10,000 PE. Table 1 presents the list of the 10 WWTPs, along with their respective maximum treatment capacities, expressed in person equivalent (PE). The five largest plants (Salvatronda, Paese, Treviso, Borgo Padova, and Montebelluna) can also co-treat water-based non-hazardous liquid waste such as septic tanks sludge, industrial effluents from paper mills (Borgo

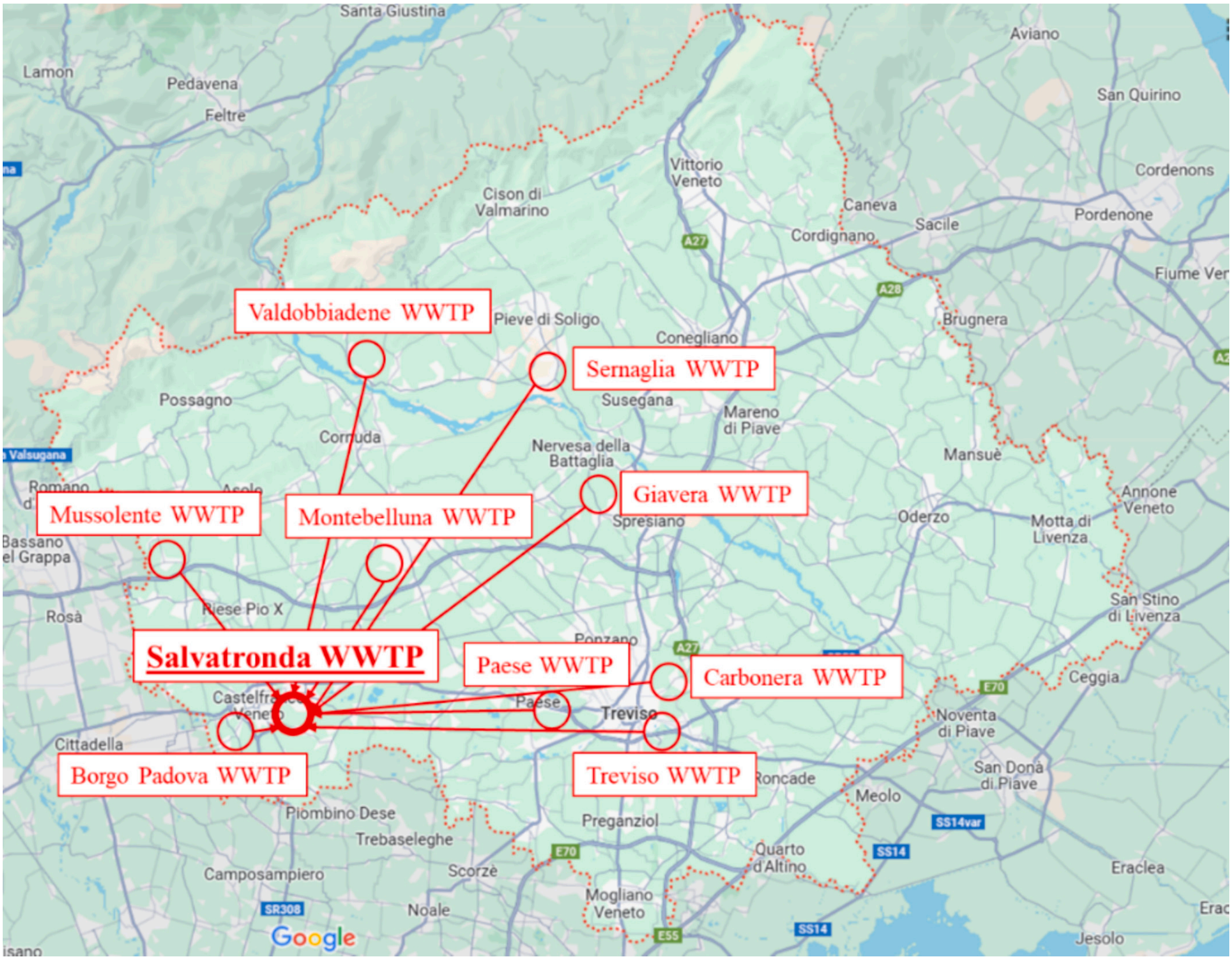

Fig. 1. Territorial map with flow paths of the sludge.

Table 1
Larger wastewater treatment plants of the region and nominal capacities.

WWTP	Maximum capacity (PE)
Salvatronda	73,300
Paese	45,000
Treviso	70,000
Borgo padova	40,000
Montebelluna	32,000
Carbonera	40,000
Giavera	18,000
Mussolente	22,000
Valdobbiadene	10,000
Sernaglia	9500

Table 2
Characteristics of the sludge originated from the main WWTPs of the province.

Parameter	UoM	Average ± SD
pH	-	7.04 ± 0.32
TS	%	19 ± 1.58
TVS/TS	%	74.6 ± 2.51
Cd	mg/kgTS	0.78 ± 0.3
Cu	mg/kgTS	234.6 ± 99.8
Ni	mg/kgTS	19.4 ± 6.47
Pb	mg/kgTS	42 ± 23.86
Zn	mg/kgTS	724.4 ± 185.71
Cr	mg/kgTS	24.2 ± 7.4
Hg	mg/kgTS	1.06 ± 0.64
As	mg/kgTS	3.29 ± 0.58
Se	mg/kgTS	0.77 ± 0.36
N	N%TS	5.24 ± 0.26
P	P%TS	1.87 ± 0.34
K	K%TS	0.38 ± 0.06

Padova), and municipal landfill leachate (Paese).

To preliminary select sewage sludge with minimum potential risk for centralised resource recovery and safe reuse, both the quality standard of the current SSD [12] and the typical characteristics of European urban sewage sludge [6,13] were considered and compared with the sludge produced by the main WWTPs in the province. In this context, Table 2 summarises the mean and standard deviation of key sludge characteristics from the WWTPs under consideration. The parameters assessed include pH, total solids percentage (TS%), the ratio of total volatile solids to total solids (TVS/TS), heavy metal concentrations, as well as macronutrient content, specifically nitrogen (N), phosphorus (P), and potassium (K), expressed in percentage. The comparison between the characteristics of the sludge under study and the typical European urban sewage sludge is reported in the Results and Discussion section (Table 4).

2.1. Description of the scenarios

Three scenarios for the Salvatronda regional sludge hub have been analysed, compared and discussed: 1) the "as it is" decentralised scenario, hereafter "SCENARIO 0"; 2) the centralised actually designed scenario (P-recovery & biogas production), hereafter "SCENARIO 1"; 3) the centralised potential scenario (P-recovery & PHA production), which can be realised by integrating the designed scenario, hereafter "SCENARIO 2".

2.1.1. SCENARIO 0

The Salvatronda WWTP has a design treatment capacity of 73,300 PE and an actual COD-based capacity of 40,567 PE receiving inflow from both combined and separate sewer systems. After primary sedimentation, the plant undergoes a conventional Modified Ludzack-Ettinger (MLE) process, with chemical precipitation for phosphorus removal. The plant produces 6110 t of sludge per year, which is composted off-site by third parties. In addition to urban wastewater, Salvatronda WWTP also treats water-based non-hazardous liquid waste, primarily from urban water cycle management (e.g. septic tanks sludge, etc.). For this scenario, 75,501 PE from the province, not currently collected in wastewater treatment plants (and treated in Imhoff tanks or septic systems) were considered, in line with the regional boundaries of this study. Furthermore, the transport and composting of 11,023 kgTS/d of sludge produced by 9 main WWTPs located near the Salvatronda sludge hub were considered. The mass flow diagram for SCENARIO 0 is depicted in Figure S1, while Figure S2 shows the flowchart of the sludge line for Salvatronda WWTP in the considered scenario.

2.1.2. SCENARIO 1

The Salvatronda WWTP is set to be upgraded from a designed treatment capacity of 73,300 PE to 120,000 PE, which will include the 75,501 PE not currently collected in SCENARIO 0. The sludge line will evolve into a regional hub capable of processing 11,023 kgTS/d of sludge produced by the 9 main WWTPs in the area managed by the same water utility. All the energy used in the plant will come from renewable sources.

For the water line, the planned scenario involves the operation of the secondary treatment through a Biological Nutrient Removal (BNR) configuration.

In the sludge line, several technological improvements are incorporated: i) fermentation of primary cellulosic sludge, ii) coupling of thermal hydrolysis with anaerobic digestion (AD), iii) drying of dewatered sludge, iv) P recovery, and v) treatment of the liquid flow from the sludge line via dissolved air flotation followed by the via nitrite biological treatment process SCENA [14]. This configuration aims to maximise energy recovery and minimise the volumes of treated sludge for safe recovery.

Specifically, primary sludge from the water line undergoes thickening, fermentation, and dewatering before being mixed with secondary and external sludge. Controlled acidogenic fermentation allows to recover the best quantity and quality of VFAs from the sludge, both in terms of yield and mixture, optimising the relative presence of propionic acid [15,16]. In particular, propionic acid promotes the growth of phosphorus-accumulating bacteria and enhances the biological removal of phosphorus in the BNR configuration. This approach also reduces the need for external carbon sources, thus mitigating associated environmental and economic impacts.

The secondary sludge is treated in a dynamic thickener and centrifuged for dewatering. Both primary and secondary sludges, along with the sludge produced in the external plants (11,023 kgTS/d), undergo thermal hydrolysis before AD and P-recovery processes.

Thermal hydrolysis, applied under controlled conditions (140–165 °C, 6–8 bar, 20–30 min) [17], breaks down the sludge into simpler, more biodegradable compounds, thereby enhancing biogas production and improving overall plant energy efficiency while reducing the volume of sludge to be disposed of [18].

The AD process generates 1,825,000 Nm^3/year of biogas, which is utilised to fuel the thermal hydrolysis boiler, heat the fermentation and AD units, or generate electricity through cogeneration. In SCENARIO 1, this biogas can produce 7261 kWh/d of electricity.

Dewatered sludge is then dried using a thermal dryer, reducing its volume to a dry concentration of up to 90 %. The exhaust air from the drying process is treated, with a total volume of 189,680,280 Nm^3/year. The dried sludge (4560 t/year) is transported to a recovery centre for composting, further reducing the environmental impact of fertiliser production.

The anaerobic supernatants, the evaporated streams, and all the streams with significant N and P loads undergo Dissolved Air Flotation (DAF) treatment to remove colloidal or inhibiting material, followed by a via nitrite biological treatment for nutrient removal before being sent to water line. Biological removal efficiencies of 85 % for both N and P are expected, with an estimated 10–15 % reduction in energy consumption compared to conventional treatments (SMART-Plant).

The possibility of recovering phosphorus is enhanced by the BNR process in the water line and the thermal hydrolysis process in the sludge line, which increases phosphorus solubilisation to over 100 mgP/L. As a result, 159 t $P\text{-}PO_4$/year of struvite can be recovered through the crystallisation process.

The mass flow for SCENARIO 1 is shown in Figure S3, and the sludge line flowchart for Salvatronda WWTP is presented in Figure S4.

2.1.3. SCENARIO 2

Starting from the design configuration of SCENARIO 1, SCENARIO 2 introduces the recovery of PHA as an alternative to biogas production. In this scenario, the plant retains the nominal capacity of 120,000 PE, including the 75,501 PE not collected in SCENARIO 0, and facilitates the recovery of both struvite and biopolymers. Similar to SCENARIO 1, all energy used by the plant is sourced from external renewable energy.

In SCENARIO 2, the selection of PHA-storing bacteria is integrated with a sidestream treatment process for nitrogen removal via nitrite from sewage sludge reject water [19]. This process enables the removal of nitrogen and phosphorus and supports the acclimation of biomass needed for PHA accumulation. The effluent from this treatment, characterised by low nitrogen and phosphorus concentrations, is sent to the water line along with reject water from pre-thickeners and biological sludge dewatering. The microorganisms from the nitritation and biomass selection tanks are directed to the polyhydroxyalkanoate accumulation tank, enhancing PHA storage. The daily production of PHA-rich biomass is equal to 837 kg of volatile suspended solids (VSS), resulting in the recovery of 293 kg of PHA per day. Final PHA-products are needed to be extracted from the PHA-rich sludge through chemical methods. The most established and widely used methods use halogenated solvents, such as chloroform. However, they are environmentally harmful and require excessive energy input, making the recovery process unsustainable and economically unfeasible [20]. For this reason, sodium hydroxide (NaOH) was adopted in this study, as a more sustainable alternative to chlorinated compounds for the extraction of PHA [20,21].

Similarly to SCENARIO 1, phosphorus is still recovered in the form of struvite through crystallisation (159 tons $P\text{-}PO_4$/year). In addition, the exhaust air from the dryer is treated (189,680,280 Nm^3/year) and the dried sludge (7034 t/year) is sent for composting. The mass flow for SCENARIO 2 is illustrated in Figure S5, and the flowchart of the sludge line for Salvatronda WWTP in this scenario is shown in Figure S6.

2.2. Life cycle assessment methodology

The environmental impacts of the scenarios described above were compared through a Life Cycle Assessment (LCA), carried out in four phases: goal and scope definition, inventory analysis, impact assessment, and interpretation. This methodology follows the framework and principles set out in ISO 14044 [22]. The analysis focused exclusively on the operational phase, excluding the construction, maintenance, and demolition of the plant from the overall assessment. The environmental impacts directly related to the treatment system (foreground system), as well as the background impacts of the supplementary supply chains providing energy, chemicals, or auxiliary materials (background system) were considered using the Ecoinvent v.3.9.1 database. This database is published and maintained by the Ecoinvent Centre in Switzerland and is the most renowned for Life Cycle Inventory (LCI) datasets.

The Life Cycle Impact Assessment (LCIA) phase was largely automated using the Umberto LCA+ v11 software, which supports graphical modelling of the product life cycle and allows for the analysis, evaluation, and visualisation of the environmental impacts across different categories. The LCIA was conducted using the ReCiPe 2016 Evaluation Method with the Hierarchist (H) variant for the midpoint impact (MID) categories. Eight environmental MID categories were considered in this analysis: climate change (GWP), fossil resource scarcity (FFP), freshwater eutrophication (FEP), marine eutrophication (MEP), ozone depletion (ODP), photochemical ozone formation: ecosystem quality (EOFP), photochemical ozone formation: human health (HOFP), and terrestrial acidification (TAP). These categories are particularly relevant for the environmental assessment of WWTPs [23].

The normalisation of impacts enables to identify, in a semi-quantitative way, which categories can be considered more relevant than the others, thus facilitating the interpretation of the results [24]. Many methods allow the results of impact category indicators to be compared against a reference (or normal) value. This involves dividing the impact category value by the reference value. A commonly used reference is the average annual environmental load in a country or continent, divided by its population. In this study, normalisation was carried out using the revised normalisation scores of ReCiPe 2016 [25].

2.2.1. Goal and scope definition, functional unit, and system boundaries

The functional unit selected for this study was the volume of effluent discharged daily from the WWTPs. To enable comparison of the impact assessment results across the scenarios, the impacts were normalised based on the nitrogen removed by each of the three schemes.

The physical system boundaries of the three scenarios considered were defined according to the goal and scope of the study and are illustrated in Figure S7 for SCENARIO 0 and in Figure S8 for SCENARIO 1 and SCENARIO 2. In particular, the system boundaries have been carefully defined to enable a fair comparison among the three scenarios and to focus on the entire process of sewage sludge treatment and valorisation. The system boundaries encompass all process stages, from wastewater treatment to final product recovery, including energy and reagent consumption, transport of reagents, direct emissions from processes, production, transport, and treatment of grit and screening waste, transport of external sludge and uncollected PE, as well as the recovery of compost, struvite, PHA, and biogas.

2.2.2. Data collection and inventory

The Life Cycle Inventory (LCI) for the three scenarios is presented in Table 3. This table provides the inventory data for the three scenarios analysed through LCA. It includes key parameters such as the influent characteristics (COD, TN, and TP), chemicals used in the treatment processes, and the associated transport distances from suppliers. The table also details the waste generated, its final destination, and the transport logistics involved. Additionally, the table presents data on the direct emissions of N_2O, CO_2, and CH_4 from biological, via-nitrite, biogas production (fugitive) and air treatment processes, as well as from the PE not collected and liquid waste treatments. Indirect emissions, including those dissolved in the effluent (N_2O, CO_2, CH_4, TN, and TP), are also included. Furthermore, the table provides information on biogas production, energy consumption and production, and the recovery of P and PHA.

The data are derived from the mass and energy balances outlined in

Table 3
LCI of the three scenarios analysed.

Water line	UoM	SCENARIO 0	SCENARIO 1	SCENARIO 2
Influent				
COD	kg/d	4868	13,500	13,500
TN	kg/d	519	1605	1605
TP	kg/d	77	8	8
Chemicals				
PAC	kg/d	575.4	0	0
Methanol	kg/d	0	3204	3204
Transport				
Sand and grit	t*km/d	28	83	83
PAC	t*km/d	15.8	0	0
Methanol	t*km/d	0	42	42
Emissions				
Biological process (direct emissions)				
N_2O	kg/d	5.8	8	7.4
CO_2	kg/d	924.8	1315	1315
CH_4	kg/d	1.3	5.4	5.4
Effluent of WWTP (indirect emissions)				
N_2O	kg/d	4.2	8.2	8.2
CO_2	kg/d	192.8	797.6	797.6
CH_4	kg/d	1.6	13.9	13.9
COD	kg/d	244	347	347
TN	kg/d	91	205	205
TP	kg/d	6	13	13
PE not collected (direct emissions)				
CO_2	kg/d	30	0	0
Effluent of PE not collected (indirect emissions)				
N	kg/d	899	0	0
P	kg/d	89.9	0	0
Liquid waste treatment (direct emissions)				
N_2O	kg/d	0.00003	0	0
CO_2	kg/d	4	0	0
CH_4	kg/d	0.034	0	0
Energy				
Electrical energy (Italian MIX)	kWh/d	4,012	0	0
Electrical energy (renewable)	kWh/d	0	10,322	10,322

Sludge line	UoM	SCENARIO 0	SCENARIO 1	SCENARIO 2
Produced sludge	kg/d	16,739	12,493	19,272
Chemicals				
Methanol	kg/d	0	267	0–
$MgCl_2$	kg/d	0	1,354	1,918
Poly	kg/d	18.3	246	349
NaOH	kg/d	0	0	167.4
Emissions				
Biogas production (direct emissions)				
N_2O	kg/d	0	0.009	0
CO_2	kg/d	0	5,221	0
CH_4	kg/d	0	104.9	0
Via nitrite process				
N_2O	kg/d	0	56.5	9.8
CO_2	kg/d	0	85.3	45.3
CH_4	kg/d	0	0.5	0.3
Air treatment				
N_2O	kg/d	0	1.9	1.9
CO_2	kg/d	0	322	322
CH_4	kg/d	0	25.4	25.4
Transport				
Sludge	t*km/d	2929	381	588

(continued on next page)

Table 3 (*continued*)

Sludge line	UoM	SCENARIO 0	SCENARIO 1	SCENARIO 2
Sludge (external)	t*km/d	16,820	0	0
Methanol	t*km/d	0	3.5	0
Poly	t*km/d	0.01	0.2	0.3
$MgCl_2$	t*km/d	0	0.2	0.3
Energy				
Electrical energy (Italian MIX)	kWh/d	732	0	0
Electrical energy (renewable)	kWh/d	0	19,119	18,455
Thermal energy	kWh/d	0	26,233	37,899
Produced electric energy	kWh/d	0	7261	0
Produced thermal energy	kWh/d	0	13,964	0
Valuable products				
Biogas production	Nm3/d	0	5,000	0
P recovered	kgP-PO_4/d	0	436	436
PHA recovered	kgPHA/d	0	0	293

Figs. S1, S2, and S3. For SCENARIO 0, the quantities of chemicals listed in Table 3 (polyelectrolyte and PAC) and the mass and energy balances are based on actual data from the Salvatronda WWTP. Regarding SCENARIO 1 (methanol, magnesium chloride, and polyelectrolyte) and SCENARIO 2 (magnesium chloride, polyelectrolyte, and sodium hydroxide), the quantities are estimated based on projected operational conditions.

For the sludge line, the mass and energy balances for SCENARIO 1 and SCENARIO 2 were performed also based on data obtained from the SMART-Plant project [26].

The transportation distances for the various waste streams to be treated in the three scenarios correspond to the current and projected future waste management destinations, meaning that the distances used are based on real-world values for both the present and anticipated future logistics.

Regarding greenhouse gas (GHG) emissions, the Emission Factors (EFs) used to calculate both direct emissions and dissolved emissions in the effluent for SCENARIO 0 were derived from field measurements. For SCENARIO 1 and SCENARIO 2, which represent potential or projected scenarios, the GHG emissions for various processes (e.g., biological processes, dissolved GHG in the effluent, biogas leakage, biogas combustion, via-nitrite, and air treatment) were calculated using theoretical EFs derived from similar processes, as reported by Marinelli et al. [27].

2.2.3. Processes implementation for LCA

2.2.3.1. Energy process. In SCENARIO 0 the electricity consumed by the plant is sourced from the national electricity grid, which has been modelled using the corresponding Italian market activity in the Ecoinvent v.3.9.1 database. This activity starts with 1 kWh of electricity fed into the medium voltage transmission network and ends with the transport of 1 kWh of medium voltage electricity in the transmission network over aerial lines and cables. The dataset includes: i) the electricity inputs generated in the country and from imports and transformed to medium voltage, ii) the transmission network, iii) the direct emissions to air, and iv) electricity losses during transmission.

In contrast, SCENARIO 1 and SCENARIO 2 are powered by renewable energy, with the energy mix consisting of 40 % hydroelectric, 22 % photovoltaic, 17 % biogas, 16 % wind energy, and 5 % geothermal.

2.2.3.2. Composting process. The sludge produced in the three scenarios is sent to composting facilities. To evaluate the potential environmental impact savings associated with the production of fertilisers and/or soil amendments from the compost, the impact savings of the production of the calcium ammonium nitrate (CAM, N-based fertiliser) and the triple superphosphate (TSP, P-based fertiliser) were considered [28]. Both fertilisers production processes are available in the Ecoinvent v.3.9.1 database, characterised by N and P_2O_5 contents of 27 % and 46 % respectively. The amount of compost produced was assumed to be the 49 % of the sludge [29,30]. The presence of macronutrients in the compost is 1.53 % of N (dry weight) and 1.34 % of P_2O_5 (dry weight) [31]. The moisture content of compost is generally 50 % [32–34]. Given these parameters, it was assumed that N and P_2O_5 produced with the compost could replace the production of these nutrients in the CAM and the TSP fertilisers.

2.2.3.3. P-recovery process. P-recovery in both SCENARIO 1 and SCENARIO 2 was considered by evaluating the environmental impact savings associated with the production of TSP [28].

2.2.3.4. PHA recovery process. The PHA extraction from biomass in SCENARIO 2 was carried out using sodium hydroxide (NaOH). a method that has been demonstrated to be a cost-effective and environmentally sustainable alternative to halogenated solvents [20,21].

In this study, the potential for producing plastic bags from PHA was evaluated, and the associated environmental benefits were compared with the production of fossil-based plastic bags The comparison was based on the methodology presented by Khoo, Tan, and Chng [35], who conducted a comprehensive life cycle assessment of plastic bags produced from fossil-based polypropylene (PP) versus those produced from bio-based PHA.

3. Results and discussion

3.1. Sewage sludge characteristics and comparison with EU data

The chemical composition of sewage sludges from urban wastewater treatment systems varies considerably depending on the source of the sewage, the degree of urbanisation in the region, and the treatment adopted. Directive 86/278/EEC (Sewage Sludge Directive) was adopted to promote the agricultural use of sewage sludge while preventing adverse effects on soil, vegetation, animals, and human health. It ensures that sludge use does not degrade soil or crop quality. The directive sets limit values only for 6 heavy metals (cadmium, copper, nickel, lead, zinc, and mercury) in soil as well as in sludge itself. Limits were also to be set for chromium, but the Directive has never been updated with a specific value for that metal. SSD does not consider persistent organic pollutants (POPs) nor potentially pathogenic organisms. Almost forty years after its adoption, the Directive is considered outdated, leading Member States to implement significantly stricter limit values for heavy metals and introduce regulations for contaminants not addressed in the Directive, in response to new scientific insights into the effects of sludge on land. The Final Report for the 'Support to the Evaluation of the Sewage Sludge Directive' study [36] presents the analysis of the effectiveness, efficiency, relevance, coherence, and EU-added value of the Sewage Sludge Directive (SSD). According to this study, 25 Member States have reported limits within the upper limit set out in the SSD for the concentration of heavy metals in sludge. Overall, there have been improvements in the quality of sewage sludge used in agriculture since the implementation of the SSD, largely attributed to the actions of the Directive. Recently, a Staff Working Document (SWD) was published by the European Commission in 2023 for evaluating and assessing sludge characteristics across the EU for agricultural uses. The concentration of heavy metals in the sewage sludge samples considered in this study is consistently below the limit values. This suggests that, based on these parameters, the sewage sludge is suitable for reuse in agriculture without posing a risk to soil quality, vegetation, or human health.

In addition, JRC organised the centralised collection of 61 sewage sludge samples in 15 European countries with the FATE-SEES campaign, including both industrial and municipal facilities [13]. Table 4 presents the average concentrations of various heavy metals in the sludge produced by the WWTPs considered in this study, comparing them with the typical content of heavy metals in European sewage sludges [13] and the quality standard for reuse in agriculture [12]. In general, the concentrations of heavy metals resulting from the historical analysis of the WWTPs (average data ± standard deviation (SD)) considered for this study are in line or lower than the statistical analysis performed by JRC about European sludge characterisation, reported in terms of average data ± standard deviation (SD) and of 90th percentile. The heavy metal concentrations are consistently below the limits set by the Council Directive 86/278/EEC for sludge reuse in agriculture. Moreover, the concentration of nutrients was also analysed, resulting in line with the average characteristics of sludge of the same origin [13,37].

It is important to note the low phosphorous content, which is typical in Italian WWTPs as compared to other European countries. In this context, phosphorus recovery must be carefully evaluated for its technical, economic and environmental sustainability, as also emerged in the discussion about the recent European Directive 2024/3019.

3.2. Results of LCA

The LCA results, presented in Table 5, show absolute values normalised by kg of N removed, along with the percentage variations (both positive and negative) relative to the baseline scenario (SCENARIO 0). In general, SCENARIO 0 exhibited higher environmental impacts than both SCENARIO 1 and SCENARIO 2 across most of the environmental categories considered, including GWP, FFP, FEP, MEP, ODP, EOFP, HOFP, and TAP. However, SCENARIO 1 displayed worse impacts in terms of climate change (−6 %) and ozone depletion (−98 %) compared to SCENARIO 0. Despite these two negative impacts, all other categories in SCENARIO 1 showed significant improvements of up to 116 % compared to SCENARIO 0. In SCENARIO 2, improvements were observed in all environmental categories analysed, with reductions in environmental impacts of up to 111 % compared to SCENARIO 0. This demonstrates the overall sustainability benefits of SCENARIO 2.

Regarding the GWP, Figure S9 presents a comparison of the impacts for the three scenarios (SCENARIO 0, SCENARIO 1, and SCENARIO 2) considered in this study, along with the contributions of the different processes involved. In SCENARIO 1 the primary contributor to its increase compared to SCENARIO 0 was the via nitrite process, primarily due to the air emissions. These emissions were equal to 16,950 kg CO_2 eq./d (12 kg CO_2 eq./kg N removed). According to Rodriguez-Garcia et al. [38], the direct emissions of this process, especially the N_2O emissions, are the main cause of the rise of GWP, while the direct CO_2 emissions do not significantly increase it. In this study, the via nitrite process in SCENARIO 1 was responsible for the emission of an amount of carbon dioxide, dinitrogen monoxide, and methane corresponding to 85 kg CO_2 eq./d, 16,847 kg CO_2 eq./d and 17 kg CO_2 eq./d, respectively. N_2O emissions from the via nitrite process accounted for 40 % of the total impact of the plant, while those from biological processes in the water line accounted for only 6 %. This is due to the high nitrogen load (707 kgN/d) in the stream fed to the via nitrite reactor, as the sludge is thermally hydrolysed and anaerobically digested in this scenario. In fact, the dissolved organic nitrogen increases with the thermal hydrolysis reaction [18]. In addition, the anaerobic digestion process causes a drastic increase in ammonia nitrogen. The degradation of proteins and other nitrogenous organic compounds is the main source of nitrogen release [39]. According to Rodriguez-Garcia et al. [38], the production of acetic acid as a carbon source is also among the main causes of the increase in GWP. However, in this study, the carbon source used in both SCENARIO 1 and SCENARIO 2 was methanol, and its production contributed respectively to 6 % and 11 % of the total impact (1.9 kg CO_2 eq./kg N removed and 1.8 kg CO_2 eq./kg N removed). Sludge composting was another significant contributor to GWP emissions in SCENARIO 0, representing 29 % of the total impacts (8.3 kg CO_2 eq./kg N removed). The high emissions associated with this process in SCENARIO 0 can be attributed to the inefficient management of sludge, primarily due to the absence of a dedicated sludge hub. This inefficiency led to the generation of a larger volume of sludge, which required composting, thus resulting in increased associated emissions. In contrast, SCENARIO 1 and SCENARIO 2, both involving the implementation of dedicated sludge hubs, resulted in substantial reductions in GWP emissions. Specifically, emissions were reduced by 93.8 % (0.51 kg CO_2 eq./kg N removed) in SCENARIO 1 and by 90.4 % (0.79 kg CO_2 eq./kg N removed) in SCENARIO 2. These improvements are due to more efficient sludge management practices facilitated by the sludge hubs, which optimised sludge treatment, reduced its volume, and ultimately lowered the emissions associated with composting.

The damages of the FFP category (Figure S10) were mainly related to transport and energy consumption in SCENARIO 0 and to methanol production in SCENARIO 1 and SCENARIO 2. In SCENARIO 0, energy consumption accounted for 35 % of the total impact in the category (1.39 kg oil eq./kg N removed), while external sludge transport contributed 34 % (1.36 kg oil eq./kg N removed). In SCENARIO 0, the energy consumption contributed by 35 % (1.39 kg oil eq./kg N removed, while the external sludge transport) by 34 % of the total impact in the category (1.36 kg oil eq./kg N removed). The emissions related to methanol production were the primary contributors to the impact on FFP in both SCENARIO 1 and SCENARIO 2. Specifically, this impact accounted for 81 % (1.8 kg oil eq./kg N removed) of the total FFP impact in SCENARIO 1, and 70 % (1.7 kg oil eq./kg N removed) in SCENARIO 2. However, despite the significant contribution of methanol production, the overall impact remains comparatively lower than that associated with transport and energy consumption in SCENARIO 1. The improvement in FFP for both SCENARIO 1 and SCENARIO 2 was mainly related to the use of renewable energy instead of the Italian energy mix used in SCENARIO 0. The impact associated with the use of renewable energy in SCENARIO 1 and SCENARIO 2 was 0.16 kg oil eq./kg N removed and 0.22 kg oil eq./kg N removed, respectively. Furthermore, the recovery of

Table 4
Average data from the historical analysis and comparison with the average concentration of heavy metals in sewage sludge in Europe [13] and the quality standard for reuse in agriculture [12].

Parameter	UoM	This study	European sludge campaign (JRC)		SSD (86/278/EEC)
		Average ± SD	Average ± SD	90th percentile	Limits
pH		7.04 ± 0.32	–	–	–
TS	%	19 ± 1.58	–	–	–
TVS/TS	%	74.6 ± 2.51	–	–	–
Cd	mg/kgTS	0.78 ± 0.3	0.9 ± 0.7	1.3	20 to 40
Cu	mg/kgTS	234.6 ± 99.8	257 ± 118	418	1000 to 1750
Ni	mg/kgTS	19.4 ± 6.47	29.0 ± 40.2	34.9	300 to 400
Pb	mg/kgTS	42 ± 23.86	47.6 ± 59.3	81.2	750 to 1200
Zn	mg/kgTS	724.4 ± 185.71	–	–	2500 to 4000
Cr	mg/kgTS	24.2 ± 7.4	79.8 ± 215	80.5	–
Hg	mg/kgTS	1.06 ± 0.64	0.4 ± 0.2	0.7	16 to 25
As	mg/kgTS	3.29 ± 0.58	–	–	–
Se	mg/kgTS	0.77 ± 0.36	–	–	–
N	N%TS	5.24 ± 0.26	–	–	–
P	P%TS	1.87 ± 0.34	3.1 ± 1.1	4.8	–
K	K%TS	0.38 ± 0.06	0.43 ± 0.39	0.74	–

Table 5
Environmental impacts in the selected environmental categories per kg of N removed for the three scenarios and variation percentages.

Impact categories	Acronym	UoM	SCENARIO 0	SCENARIO 1	SCENARIO 2	Variations between SCENARIO 0 and SCENARIO 1 (%)	Variations between SCENARIO 0 and SCENARIO 2 (%)
Climate change	GWP	kg CO_2 eq./kg N removed	2.85E+01	3.01E+01	1.56E+01	−6 %	45 %
Fossil resource scarcity	FFP	kg oil eq./kg N removed	3.95E+00	2.25E+00	2.41E+00	43 %	39 %
Freshwater eutrophication	FEP	kg P eq./kg N removed	2.26E-01	8.73E-03	9.16E-03	96 %	96 %
Marine eutrophication	MEP	kg N eq./kg N removed	6.89E-01	4.48E-02	4.50E-02	94 %	93 %
Ozone depletion	ODP	kg CFC_{11} eq./kg N removed	2.98E-04	5.91E-04	2.21E-04	−98 %	26 %
Photochemical ozone formation: ecosystems quality	EOFP	kg NO_x eq./kg N removed	4.29E-02	8.34E-03	1.14E-02	81 %	73 %
Photochemical ozone formation: human health	HOFP	kg NO_x eq./kg N removed	4.06E-02	7.51E-03	1.06E-02	81 %	74 %
Terrestrial acidification	TAP	kg SO_2 eq./kg N removed	2.10E-01	−3.41E-02	−2.32E-02	116 %	111 %

phosphorus and biopolymers, along with the resulting avoidance of mineral fertiliser and industrial plastic production, played a significant role, reducing the total impacts of SCENARIO 1 and SCENARIO 2 by 0.4 kg oil eq./kg N removed and 0.6 kg oil eq./kg N removed respectively.

Freshwater and marine eutrophication (Figure S11 and Figure S12) showed a strong improvement in both SCENARIO 1 and SCENARIO 2. The main cause of the damage in these two categories in SCENARIO 0 was the impact from the PE not collected, which was responsible for the emissions of 0.6 kg N/kg N removed in MEP and 0.2 kg P/kg N removed in FEP. In fact, in SCENARIO 1 and SCENARIO 2, the PE not collected was treated in the WWTP. This strategy of collection and centralisation serves as an effective mitigation measure for the environmental impacts, leading to a 96 % reduction in total impacts for FEP in both SCENARIO 1 and SCENARIO 2, and a 94 % and a 93 % reduction in MEP for SCENARIO 1 and SCENARIO 2 respectively, compared to SCENARIO 0.

Ozone depletion across the three scenarios was predominantly driven by N_2O emissions (Figure S13). Specifically, these emissions accounted for 86.3 % (0.0003 kg CFC-11 eq./kg N removed), 99.2 % (0.0006 kg CFC-11 eq./kg N removed), and 96.8 % (0.0002 kg CFC-11 eq./kg N removed) of the total impact in SCENARIO 0, SCENARIO 1, and SCENARIO 2, respectively. Notably, the ODP was significantly worse in SCENARIO 1 compared to SCENARIO 0 (−98 %), primarily due to N_2O emissions from the via nitrite process (0.62 kg CFC-11 eq/d). As explained for the GWP, this is attributed to the high nitrogen load (707 kg N/d) in the stream entering the via nitrite reactor. In contrast, SCENARIO 2 showed an improvement in ODP relative to SCENARIO 0, owing to a slight reduction in N2O emissions and a significant decrease in the volume of sludge sent for composting. The impact of sludge composting in SCENARIO 0 was 0.00004 kg CFC-11 eq./kg N removed, compared to just 0.000004 kg CFC-11 eq./kg N removed in SCENARIO 2.

Photochemical ozone formation (ecosystem quality and human health) was mainly influenced by transport (Figure S14 and Figure S15). In fact, the reduction in transport emissions per kilogram of nitrogen removed was the primary factor driving the improvements in both environmental categories for SCENARIO 1 and SCENARIO 2. This led to a decrease in the EOFP and HOFP of 81 % and 73 % in SCENARIO 1, and 81 % and 74 % in SCENARIO 2, respectively, compared to SCENARIO 0.

The impact on terrestrial acidification was primarily driven by the composting process (Figure S16). In SCENARIO 0, composting accounted for 97 % of the total impact (0.2 kg SO_2 eq./kg N removed). As explained for the GWP, this was largely due to the inefficient management of sludge in SCENARIO 0, in contrast to the more effective sludge management in SCENARIO 1 and SCENARIO 2, both of which involved the implementation of dedicated sludge hubs. SCENARIO 1 and SCENARIO 2 demonstrated reductions in this category, primarily due to the recovery of struvite, which prevented the need for producing the corresponding inorganic phosphorus fertiliser. As a result, the overall impact in this category was −0.11 kg SO_2 eq./kg N removed in SCENARIO 1 and − 0.08 kg SO_2 eq./kg N removed in SCENARIO 2.

The impact results in the eight environmental categories considered were also normalised to the impact potential of an average European over one year (Fig. 2). SCENARIO 0, SCENARIO 1, and SCENARIO 2 had a total impact equivalent to the one generated by 222 people per day, 55 people per day, and 46 people per day respectively. Considering the individual environmental categories, the GWP in SCENARIO 0, SCENARIO 1, and SCENARIO 2 had an impact equivalent to 1.5 people, 5.3 people, and 2.7 people respectively. The results obtained are comparable to those achieved by Yoshida et al. [40], where the GWP was 5.4 people eq. The impact of fossil resource scarcity was 1.7 people eq, 3.2 people eq, and 3.4 people eq in the SCENARIO 0, SCENARIO 1, and SCENARIO 2 respectively. Freshwater eutrophication was the environmental category that showed a better improvement in SCENARIO 1 and SCENARIO 2 compared to SCENARIO 0. In fact, the FEP in SCENARIO 0 had an emission equivalent to 148.7 people against an emission

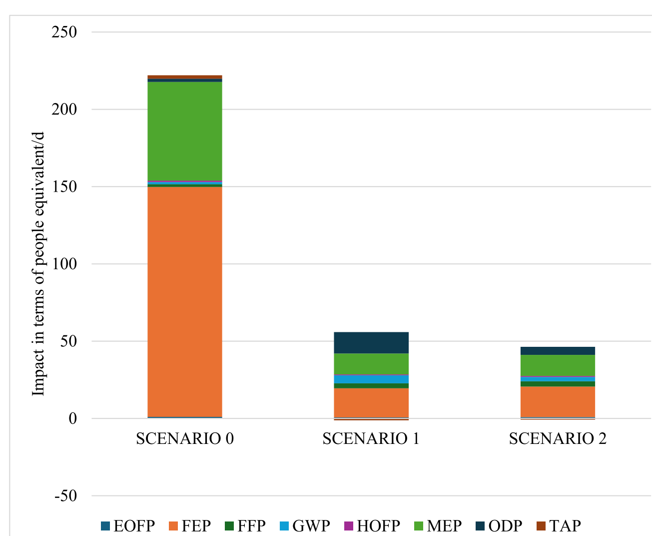

Fig. 2. Environmental impacts in the categories of the three scenarios normalised in terms of emissions per citizen.

equivalent to 18.8 people and 19.7 people in SCENARIO 1 and SCENARIO 2 respectively. Marine eutrophication also showed a large improvement in SCENARIO 1 and SCENARIO 2. MEP had an impact equivalent to 63.8 people in SCENARIO 0, while it decreased to 13.6 people eq in both SCENARIO 1 and SCENARIO 2. Ozone depletion had an impact equivalent to 2 people in SCENARIO 0 and increased slightly to 13.8 people eq and 5.2 people eq in SCENARIO 1 and SCENARIO 2.

In general, the main cause of the environmental impacts in all three scenarios was the emission of nutrients to water (phosphorus and nitrogen). In SCENARIO 0, nutrient emissions to water represented 95 % of the total impact (66 % was due to phosphorus emissions and the remaining 23 % was due to nitrogen emissions). In SCENARIO 1, nutrient emissions represented 61 % of the total impact (37 % phosphorus and 24 % nitrogen), followed by the emissions to air which represented 33 % of the total impact (dinitrogen monoxide represented 30 % of the total impact). SCENARIO 2 was characterised by an emission of nutrients to water accounting for 73 % of the total impact (44 % of phosphorus and 29 % of nitrogen), followed by the emissions to air, which represented 14 % of the total impact.

To conclude, the normalised results confirm that SCENARIO 1 shows better improvements than SCENARIO 0 in almost all the eight categories, and SCENARIO 2 is the best option from a sustainability point of view.

4. Conclusions

The recent discussions that led to the new EU wastewater treatment directive 2024/3019 and is driving the decisions for the revision of the EU Sewage Sludge Directive highlight the urgent need to align circular water and waste management policies with the EU's zero pollution ambition.

This study presented the environmental assessment of current, designed and potential best circular scenario for a regional sludge hub centre designed on the basis of innovative solutions demonstrated at pilot scale in EU-funded innovation actions.

After analysing and benchmarking the sewage sludge characteristics in the region, which highlighted relatively low phosphorus content, three different territorial scenarios for sewage sludge management were analysed, compared and discussed. The “as it is” scenario is decentralised and with only sludge composting as resource recovery process. In the centralised scenarios, phosphorus is recovered by crystallisation (struvite), and biogas or biopolymers, respectively for SCENARIO 1 and 2, are produced.

An LCA was carried out to evaluate the environmental impacts of the three scenarios and showed that the scenario where phosphorus and biopolymers are recovered is the more sustainable option, reducing the total emissions of the actual decentralised scenario by the equivalent of 176 people per day. In particular, the innovative technologies, validated in SMART-Plant and applied at full scale in this study for the first time, allowed a massive improvement compared to the actual decentralised scenario. In particular, the emissions in MEP and FEP categories improved by 93 % and 96 % respectively in the centralised scenario where phosphorus and biopolymers are recovered compared to the decentralised ones.

This article lays the foundation for a preliminary assessment methodology for the development of a sludge hub, starting with the evaluation of sludge quality and selecting only those that do not pose associated risks. Furthermore, it is essential to conduct an environmental assessment that justifies the choice of centralising the treatment process as the most sustainable option from an environmental perspective. Additionally, the recent evaluation of the Sewage Sludge Directive highlights the importance of strengthening regulatory frameworks to support such initiatives and ensure long-term sustainability in sludge management practices.

From a future-oriented perspective, the data obtained in this study can serve as a foundation for developing advanced strategies for centralised sludge treatment, with a focus on territorial resource recovery and environmental impact mitigation. Future research should investigate the scalability and replicability of the sludge hub model in other regions with similar characteristics. The province of Treviso is representative, collecting and treating 406,500 PE, with its diverse municipalities and strong agricultural focus, provides a relevant case for studying the sustainability of resource recovery from centralised sludge treatment, offering insights that could be applied to other medium-sized regions across Italy and Europe. However, a risk-based approach is needed for balancing the goal of circular economy and zero pollution ambition. In fact, a risk-based framework ensures that both objectives are achieved sustainably, maximizing benefits while minimizing adverse impacts on environment and human health.

CRediT authorship contribution statement

Elisa Blumenthal: Writing – original draft, Software, Methodology, Formal analysis, Conceptualization. **Alessia Foglia:** Writing – original draft, Visualization, Supervision, Methodology, Investigation. **Alberto Piasentin:** Validation, Data curation. **Corinne Andreola:** Software, Methodology, Formal analysis, Data curation, Conceptualization. **Anna Laura Eusebi:** Writing – review & editing, Visualization, Validation, Supervision, Methodology, Conceptualization. **Nicola Frison:** Writing – review & editing, Supervision. **Francesco Fatone:** Writing – review & editing, Validation, Supervision, Project administration, Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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Data availability

Data will be made available on request.

References

[1] NEBRA - sludge hubs [WWW Document], n.d. URL https://www.nebiosolids.org/sludge-hubs (accessed 2.15.25).
[2] Regional Sludge Hub Centres | Projects | Uisce Éireann (formerly Irish Water) [WWW Document], n.d. URL https://www.water.ie/projects/national-projects/regional-sludge-hub-centr (accessed 2.15.25).
[3] A. Taron, S. Singh, C. Ravishankar, A. Ulrich, RESOURCE RECOVERY & REUSE SERIES 23 Sewage Sludge: A Review of Business Models for Resource Recovery and Reuse, 2023, https://doi.org/10.5337/2023.211.
[4] P.K. Cornejo, Q. Zhang, J.R. Mihelcic, How does scale of implementation impact the environmental sustainability of wastewater treatment integrated with resource recovery? Environ. Sci. Technol. 50 (2016) 6680–6689, https://doi.org/10.1021/acs.est.5b05055.

[5] G.T. Daigger, Evolving urban water and residuals management paradigms: water reclamation and reuse, decentralization, and resource recovery, Water Environ. Res. 81 (2009) 809–823, https://doi.org/10.2175/106143009x425898.
[6] SWD, Commission Staff Working Document Evaluation Council Directive 86/278/EEC of 12 June 1986 on the Protection of the Environment, and in Particular of the Soil, When Sewage Sludge Is Used in Agriculture, 2023.
[7] European Commission, European Commission - Research & Innovation Projects Relevant to Water Research - HORIZON 2020 Calls 2014–2020, 2022.
[8] V. Conca, C. da Ros, F. Valentino, A.L. Eusebi, N. Frison, F. Fatone, Long-term validation of polyhydroxyalkanoates production potential from the sidestream of municipal wastewater treatment plant at pilot scale, Chem. Eng. J. 390 (2020), https://doi.org/10.1016/j.cej.2020.124627.
[9] European Parliament and Council Directive 2024/3019, DIRECTIVE (EU) 2024/3019 OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 27 November 2024 Concerning Urban Wastewater Treatment (Recast) (Text With EEA Relevance), 2024.
[10] E.A. Odey, Z. Li, X. Zhou, L. Kalakodio, Fecal sludge management in developing urban centers: a review on the collection, treatment, and composting, Environ. Sci. Pollut. Res. (2017), https://doi.org/10.1007/s11356-017-0151-7.
[11] N. Topanou, B. Agbatchi, G.J. Gbaguidi, J. Fatombi, F.P. Tchobo, Uncovering the challenges of faecal sludge management in Benin's urban hubs: the cases of Abomey-Calavi and Natitingou, J Environ Prot (Irvine, Calif) 15 (2024) 767–780, https://doi.org/10.4236/jep.2024.157044.
[12] Council Directive 86/278/EEC, Council Directive 86/278/EEC of 12 June 1986 on the Protection of the Environment, and in Particular of the Soil, When Sewage Sludge Is Used in Agriculture, 1986.
[13] JRC, Occurrence and Levels of Selected Compounds in European Sewage Sludge Samples, 2012, https://doi.org/10.2788/67153.
[14] S. Longo, N. Frison, D. Renzi, F. Fatone, A. Hospido, Is SCENA a good approach for side-stream integrated treatment from an environmental and economic point of view? Water Res. 125 (2017) 478–489, https://doi.org/10.1016/j.watres.2017.09.006.
[15] D. Crutchik, N. Frison, A.L. Eusebi, F. Fatone, Biorefinery of cellulosic primary sludge towards targeted short chain fatty acids, phosphorus and methane recovery, Water Res. 136 (2018) 112–119, https://doi.org/10.1016/j.watres.2018.02.047.
[16] S. Longo, E. Katsou, S. Malamis, N. Frison, D. Renzi, F. Fatone, Recovery of volatile fatty acids from fermentation of sewage sludge in municipal wastewater treatment plants, Bioresour. Technol. 175 (2015) 436–444, https://doi.org/10.1016/j.biortech.2014.09.107.
[17] J.L. Urrea, S. Collado, A. Laca, M. Díaz, Rheological behaviour of activated sludge treated by thermal hydrolysis, Journal of Water Process Engineering 5 (2015) 153–159, https://doi.org/10.1016/j.jwpe.2014.06.009.
[18] W.P.F. Barber, Thermal hydrolysis for sewage treatment: a critical review, Water Res. (2016), https://doi.org/10.1016/j.watres.2016.07.069.
[19] N. Frison, E. Katsou, S. Malamis, A. Oehmen, F. Fatone, Development of a novel process integrating the treatment of sludge reject water and the production of Polyhydroxyalkanoates (PHAs), Environ. Sci. Technol. 49 (2015) 10877–10885, https://doi.org/10.1021/acs.est.5b01776.
[20] A.M. Rodrigues, R.D.G. Franca, M. Dionísio, C. Sevrin, C. Grandfils, M.A.M. Reis, N. D. Lourenço, Polyhydroxyalkanoates from a mixed microbial culture: extraction optimization and polymer characterization, Polymers (Basel) 14 (2022), https://doi.org/10.3390/polym14112155.
[21] M. Mohammadi, M.A. Hassan, L.Y. Phang, Y. Shirai, H. Che Man, H. Ariffin, A. A. Amirul, S.N. Syairah, Efficient polyhydroxyalkanoate recovery from recombinant cupriavidus necator by using low concentration of NaOH, Environ. Eng. Sci. 29 (2012) 783–789, https://doi.org/10.1089/ees.2011.0255.
[22] ISO, ISO 14044 Environmental Management - Life Cycle Assessment - Requirements and Guidelines, 2006.
[23] M. Tsangas, I. Papamichael, D. Banti, P. Samaras, A.A. Zorpas, LCA of municipal wastewater treatment, Chemosphere 341 (2023), https://doi.org/10.1016/j.chemosphere.2023.139952.
[24] A.W. Sleeswijk, L.F.C.M. van Oers, J.B. Guinée, J. Struijs, M.A.J. Huijbregts, Normalisation in product life cycle assessment: an LCA of the global and European economic systems in the year 2000, Sci. Total Environ. 390 (2008) 227–240, https://doi.org/10.1016/j.scitotenv.2007.09.040.
[25] Normalization scores ReCiPe 2016 | RIVM [WWW Document], URL, https://www.rivm.nl/en/documenten/normalization-scores-recipe-2016, 2024. (Accessed 21 June 2024).
[26] SMART-Plant [WWW Document], URL, https://www.smart-plant.eu/, 2024. (Accessed 28 June 2024).
[27] E. Marinelli, S. Radini, A. Foglia, N. Lancioni, A. Piasentin, A.L. Eusebi, F. Fatone, Validation of an evidence-based methodology to support regional carbon footprint assessment and decarbonisation of wastewater treatment service in Italy, Water Res. 207 (2021), https://doi.org/10.1016/j.watres.2021.117831.
[28] A. Amann, O. Zoboli, J. Krampe, H. Rechberger, M. Zessner, L. Egle, Environmental impacts of phosphorus recovery from municipal wastewater, Resour. Conserv. Recycl. 130 (2018) 127–139, https://doi.org/10.1016/j.resconrec.2017.11.002.
[29] ECN, ECN Data Report 2022 Compost and Digestate for a Circular Bioeconomy, 2022.
[30] R. Gottschall, M. Thelen-Jüngling, M. Kranert, B. Kehres, Suitability of biowaste and green waste composts for organic farming in Germany and the resulting utilization potentials, Agriculture (Switzerland) 13 (2023), https://doi.org/10.3390/agriculture13030740.
[31] Organic Materials Management - CalRecycle Home Page [WWW Document], URL, https://calrecycle.ca.gov/organics/, 2024. (Accessed 21 June 2024).
[32] L. Almulla, B.M. Thomas, M.F.A. Jallow, A. Al-Roumi, Y. Devi, J. Jacob, Rotary drum composting of organic school wastes and compost valorization, Sustainability (Switzerland) 16 (2024), https://doi.org/10.3390/su16062428.
[33] E. Kim, D.H. Lee, S. Won, H. Ahn, Evaluation of optimum moisture content for composting of beef manure and bedding material mixtures using oxygen uptake measurement, Asian Australas. J. Anim. Sci. 29 (2016) 753–758, https://doi.org/10.5713/ajas.15.0875.
[34] The University of Maine, Compost Report Interpretation Guide % Solids, % Moisture, 2016.
[35] H.H. Khoo, R.B.H. Tan, K.W.L. Chng, Environmental impacts of conventional plastic and bio-based carrier bags, Int. J. Life Cycle Assess. 15 (2010) 284–293, https://doi.org/10.1007/s11367-010-0162-9.
[36] European Commission, Support to the Evaluation of the Sewage Sludge Directive Final Study Report, 2022.
[37] Metcalf & Eddy, Wastewater Engineering Treatment and Resource Recovery - Fifth Edition, 2014.
[38] G. Rodriguez-Garcia, N. Frison, J.R. Vázquez-Padín, A. Hospido, J.M. Garrido, F. Fatone, D. Bolzonella, M.T. Moreira, G. Feijoo, Life cycle assessment of nutrient removal technologies for the treatment of anaerobic digestion supernatant and its integration in a wastewater treatment plant, Sci. Total Environ. 490 (2014) 871–879, https://doi.org/10.1016/j.scitotenv.2014.05.077.
[39] J. Suschka, K. Grübel, Nitrogen in the process of waste activated sludge anaerobic digestion, Archives of Environmental Protection 40 (2014) 123–136, https://doi.org/10.2478/aep-2014-0021.
[40] H. Yoshida, J. Clavreul, C. Scheutz, T.H. Christensen, Influence of data collection schemes on the life cycle assessment of a municipal wastewater treatment plant, Water Res. 56 (2014) 292–303, https://doi.org/10.1016/j.watres.2014.03.014.