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Research article

Multi-criteria analysis of strategies towards sustainable recycling of phosphorus from sewage sludge in Austria

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ABSTRACT

To promote optimal phosphorus (P) recovery from municipal wastewater and sewage sludge with viable legal instruments, it is imperative to understand the regional and national consequences of different legal requirements for recycling. In this study we develop a scenario-based analysis to assess the environmental and economic impact of different national P recovery strategies in the context of a detailed representation of the existing Austrian wastewater infrastructure. This assessment combines material flow analysis, life cycle assessment and life cycle costing and includes the indicators P recycling rate, P utilization degree, heavy metal removal rate, share of heavy metals' content in wastewater redirected to agricultural soils, global warming potential, cumulated energy demand, terrestrial acidification potential, volume of freight transport and annual costs. The following main conclusions can be drawn. P recovery from ash shows the highest potential regarding the utilization of P from wastewater. A high P utilization from wastewater should rely on recovery technologies that decontaminate products, otherwise pollutant loads to agricultural soils might increase. P recovery to the extent of 60–85 % of P in WWTPs influent can be achieved by savings/costs of −0.8 to +4.7 EUR inhabitant^{-1} yr^{-1} in addition to current cost of the wastewater treatment/sludge disposal system. Key factors to be considered for costs are the choice of recovery process, revenues from products, and the use of existing incineration infrastructure. P recovery can lead to the reduction of greenhouse gas emissions in Austria if nitrous oxide emissions from sludge incineration are limited and efficient heat utilization strategies are implemented. There is a trade-off in terms of environmental and economic costs in choosing a more centralized or decentralized mono-incineration strategy.

1. Introduction

Demand for sustainable phosphorus (P) management has gained traction in recent years due to the recognition of its critical role for contemporary society, most notably for food security in the form of fertilizer (Brownlie et al., 2022). With few internal reserves, the European Union (EU) and its member countries are highly dependent on imports of phosphate rock from Morocco (24%) and Russia (20%), as well as white P from Kazakhstan (71%) (European Comission, 2020). In 2022, the Russia-Ukraine conflict has further interrupted supply chains, leading to further scarcity and price increases for P resources (Mbah and Wasum, 2022; Brownlie et al., 2023).

The discovery of two secondary P resources with a high potential for primary raw material substitution, namely municipal wastewater and meat and bone meal has given EU members viable options to improve their national biogeochemical P cycles (Egle et al., 2014; Zoboli et al., 2016; Załuszniewska and Nogalska, 2022).

Two decades of intense research on P recovery from wastewater has led to a variety of suitable technologies to produce plant-available and low polluted fertilizers that are awaiting full-scale implementation (Remy and Jossa, 2015; Kraus and Seis, 2015; Egle et al., 2016c; Kraus et al., 2019a; Chrispim et al., 2019). Even with high substitution potentials and existing technologies for recovery from wastewater, for the time being, the challenge of slightly unfavorable economics for P recycling with high recovery rates remains (Schoumans et al., 2015; Egle et al., 2016). In conclusion, customized legislation or incentives enforcing the recycling of P from secondary resources are required to ensure EU nations recognize their responsibility for better P management.

To promote optimal P recycling with viable legal instruments, it is

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imperative to understand the regional and national consequences of different legal requirements for recycling, being dependent on the individual country and/or region (Schoumans et al., 2015; Hukari et al., 2016; Chrispim et al., 2019). It is not enough to set challenging recovery rates if the monetary and environmental impacts of different strategies and potential policies are unknown (Smol et al., 2020).

Scenario-based analysis can help to identify those who are best fit to avoid too soft or too rigorous standards for P recovery and/or recycling (Bauwens et al., 2020). Further, it can help eliminate those that contradict other goals for wastewater treatment and climate protection (e.g., energy neutrality, pollutant destruction; e.g. Jacobi et al., 2018). For well-developed scenarios analysis, in-depth knowledge of the *status quo* of a system is mandatory (Grafström and Aasma, 2021; Yang et al., 2021).

Austria is an example of an EU member state with a strong background in P management research and adequate data availability to perform such analyses. Amann et al. (2022) developed an in-depth operational and spatial model of the current Austrian wastewater treatment system. This sets a unique basis for a holistic evaluation of effects various P recovery strategies would exert on Austria's economy and ecology, which is provided in this paper.

Previous research on P recovery strategies has focused on evaluating single plants (Orner et al., 2022; Faragò et al., 2021) and/or a limited selection of evaluation criteria (Law and Pagilla, 2019). While this gives a good overview on pathways to consider, it is not enough to analyze circularity and its effects on a micro- or meso-level only (Jacobi et al., 2018). To improve overall circularity, as is the quest in this case for P, a detailed analysis of how materials and substances move within a larger system (i.e., on a national level) is required (Zoboli et al., 2016). A multi-criteria evaluation concept can help to guide decision making in the circular economy and mitigate potential negative impacts of new recycling strategies (Yazdani et al., 2019; dos Santos Gonçalves and Campos, 2022).

In this study, the research focus is on the development of a multi-criteria evaluation concept that allows a scenario-based analysis to assess the environmental and economic impact of different national P recovery strategies in the context of a detailed representation of the existing Austrian wastewater infrastructure. This assessment combines material flow analysis, life cycle assessment and life cycle costing and includes the indicators P recycling rate, P utilization degree, heavy metal removal rate, share of heavy metals content of sewage sludge potentially redirected to agricultural soils, global warming potential, cumulated energy demand, terrestrial acidification potential, volume of freight transport and annual costs. This work is based on extensive data collection and evaluation methods and is able to set its focus on the development and presentation of a one-of-a-kind approach to sustainably guide resource policy in establishing laws, and practitioners in setting-up and executing P recovery concepts.

As a result, the development of legal requirements adapted to local conditions can be promoted to close national and regional nutrient cycles. Scientifically justified guidelines and the public communication of these results are further seen as an opportunity to increase the acceptance of future legislative adjustments.

2. Materials & methods

To carry out this study, first, the considered processes and system boundaries as well as economic and ecological criteria (evaluation approaches) for a quantitative assessment of different strategies for P recovery were defined (chapter 2.1). Second, detailed data collection and analysis was carried out to be able to accurately represent the current situation of wastewater and sewage sludge treatment, utilization, and disposal in Austria. Main assumptions and results on the *Status Quo* can be found in detail in Amann et al. (2021, 2022). Third, different scenarios (chapter 2.2) were developed to illustrate multiple P reuse or recovery strategies, achieving a P utilization degree (chapter 2.3.1) of at least 50 % or more of the P load of all municipal wastewater treatment plants in Austria. Detailed life cycle inventories (Amann et al., 2022 and chapter 2.4.1) on all relevant processes (reference processes) provided a further basis for material flow analysis (MFA), cost estimation and life cycle assessments (LCA), which were used to implement the economic and ecological evaluation (chapter 2.3).

2.1. System boundaries & status quo

The spatial system boundary for this study contains all processes that are likely changed, impacted, or substituted by P recovery. The analyses conducted in this work necessitate the use of large datasets, which were collected over the period 2018–2021. The most recent year for which comprehensive data were available was 2016. Therefore, this reference year was defined as the temporal boundary for the *Status Quo*. Scenarios were originally developed to estimate changes as compared to the year 2016. Since the COVID-19 crisis and Russia's invasion of Ukraine have created large economic turbulences on the market – especially in the EU (Mbah and Wasum, 2022) – an additional sensitivity analysis was performed on the initial cost estimation and based on resource and utility prices in Q2/2023.

The processes considered as part of the overall system are *municipal wastewater treatment*, *sewage sludge recycling and disposal*, *P recovery*, and the *use of recovered products* (Fig. 1; see also Amann et al., 2022). *Municipal wastewater treatment* includes the treatment of wastewater and the resulting sewage sludge at all wastewater treatment plants (WWTPs) in Austria with a treatment capacity of more than 2000 population equivalents ($PE_{capacity}$) including dewatering and potential drying of sewage sludge as well as onsite recovery at the WWTP, in case it was assumed in the scenario definition. In the process of *sewage sludge recycling and disposal*, the transport of sewage sludge, as well as the reuse, thermal recycling or disposal of sewage sludge were considered. *P recovery* from mono-incinerated sewage sludge ash by specific recovery processes as well as the *use of the P products* in agriculture or industry were included as a further process.

Inputs to the system are the untreated municipal wastewater (containing P) and the resources required to maintain it. Outputs are all emissions from the system, the products as well as any potential energy surplus (e.g., from incineration).

2.2. Scenario development

2.2.1. Considered WWTPs

Different scenarios of our study consider different sizes of WWPTs to be included into a strategy for P recovery. We start with the biggest WWTPs $\geq$100,000 $PE_{capacity}$ (*>100k scenarios*, see Table 1) treating the highest loads, having better treatment efficiencies and better financial as well as operational opportunities for the adaptation of new recycling technologies (Haslinger et al., 2016). This is followed by other scenarios stepwise including smaller WWTPs into the strategy of P recovery. *Scenarios 50k, >20k, >2k* indicate scenarios that include all WWTPs with $\geq$50,000, $\geq$20,000 and $\geq$ 2000 $PE_{capacity}$, respectively (Table 1).

2.2.2. Centralized versus decentralized concepts

With the range of existing technologies, two different approaches to P recovery can be pursued, which differ greatly in their importance for the development of regional concepts and required infrastructure. These are (i) centralized concepts that go beyond individual WWTPs collecting sludge from surrounding regions to treat by mono-incineration (*MI*) and to provide sewage sludge ash (*SSA*) for further industrial processing (*MI + PR-SSA scenarios*) and (ii) decentralized concepts with P recovery (PR) directly at the WWTP (*PR-WWTP - scenarios*, see Table 1).

The maximum P recovery potential related to $P_{influent}$ ranges from 45 to 70% for decentralized concepts, while up to 90% can be achieved with centralized processing (Egle et al., 2014). Further, decentralized recovery at WWTPs is generally characterized by higher resource (Law

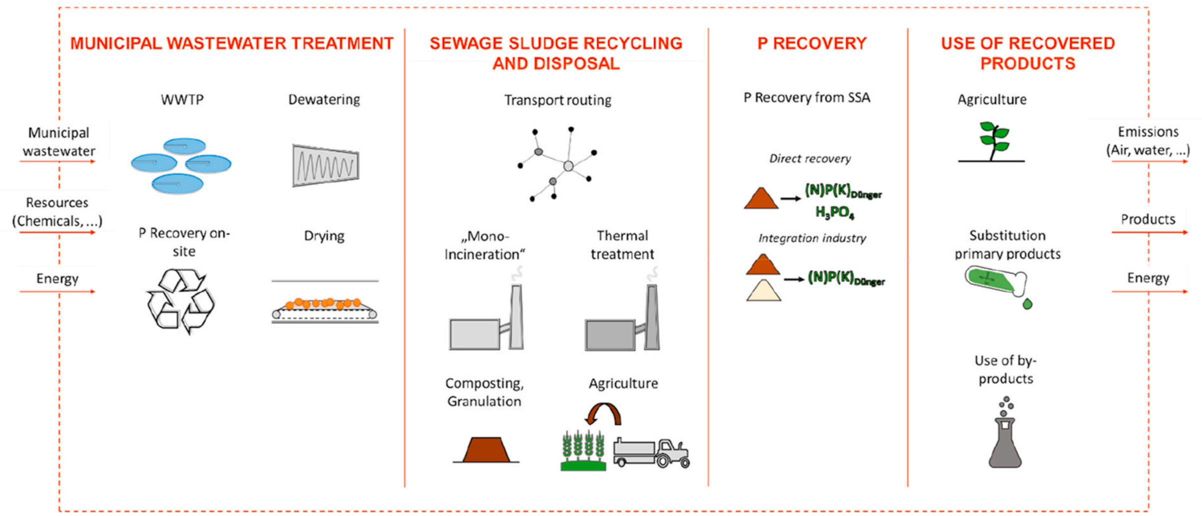

Fig. 1. Depiction of processes covered in the economic and environmental analysis. SSA: sewage sludge ash.

Table 1
Overview on main scenarios. Scenario names indicate (i) the size of WWTP included into recovery strategies (>20k …), (ii) centralized (MI) or decentralized (PR-WWTP) concepts, (iii) for centralized concepts if MI is expanded (expansion) or not (*status quo*) and for decentralized concepts which technology group has been considered (TG1 or TG2). (iv) P recovery form SSA (PR-SSA) is performed in all cases except in PR-WWTP scenarios for WWTPs ≥50,000 $PE_{capacity}$.

Scenario	Where	Type of recovery	Incineration through …
>20k MI-status-quo + PR-SSA	WWTPs ≥20,000 $PE_{capacity}$	from SSA with AshDec, EcoPhos, Phos4Life or by direct integration into the fertilizer industry	existing MI infrastructure +1 new plant
>20k AGR/MI-status-quo + PR-SSA	WWTPs ≥20,000 $PE_{capacity}$ that do not use sewage sludge or sewage sludge compost for agricultural purposes (AGR)	from SSA with AshDec, EcoPhos, Phos4Life or by direct integration into the fertilizer industry	existing MI infrastructure
>50k MI-status-quo + PR-SSA	WWTPs ≥50,000 $PE_{capacity}$	from SSA with AshDec, EcoPhos, Phos4Life or by direct integration into the fertilizer industry	existing MI infrastructure
>100k MI-status-quo + PR-SSA	WWTPs ≥100,000 $PE_{capacity}$	from SSA with AshDec, EcoPhos, Phos4Life or by direct integration into the fertilizer industry	existing MI infrastructure
>2k MI-expansion + PR-SSA	WWTPs ≥2000 $PE_{capacity}$	from SSA with AshDec, EcoPhos, Phos4Life or by direct integration into the fertilizer industry	existing MI infrastructure +3 additional plants
>20k MI-expansion + PR-SSA	WWTPs ≥20,000 $PE_{capacity}$	from SSA with AshDec, EcoPhos, Phos4Life or by direct integration into the fertilizer industry	existing MI infrastructure +3 additional plants
>50k MI-expansion + PR-SSA	WWTPs ≥50,000 $PE_{capacity}$	from SSA with AshDec, EcoPhos, Phos4Life or by direct integration into the fertilizer industry	existing MI infrastructure +3 additional plants
>20k PR-WWTP TG1 + PR-SSA	WWTPs ≥20,000 $PE_{capacity}$	technology group one (TG1): Airprex process at Bio-P WWTPs ≥100,000 $PE_{capacity}$ and Stuttgart process at WWTPs ≥50,000 $PE_{capacity}$	existing MI infrastructure
>20k PR-WWTP TG2 + PR-SSA	WWTPs ≥20,000 $PE_{capacity}$	technology group two (TG2): Wasstrip + Lysotherm process at Bio-P plants ≥100,000 $PE_{capacity}$ and TerraNova process at WWTPs ≥50,000 $PE_{capacity}$	existing MI infrastructure
>50k PR-WWTP TG1 + PR-SSA	WWTPs ≥50,000 $PE_{capacity}$	technology group one (TG1): Airprex process at Bio-P WWTPs ≥100,000 $PE_{capacity}$ and Stuttgart process at WWTPs ≥50,000 $PE_{capacity}$	existing MI infrastructure
>50k PR-WWTP TG2 + PR-SSA	WWTPs ≥50,000 $PE_{capacity}$	technology group two (TG2): Wasstrip + Lysotherm process at Bio-P WWTPs ≥100,000 $PE_{capacity}$ and TerraNova process at WWTPs ≥50,000 $PE_{capacity}$	existing MI infrastructure

and Pagilla, 2019) and lower transport requirements, as nutrient products can be sold regionally (Rufí-Salís et al., 2020). For centralized recovery, new MI capacities must be created and positioned at strategically favorable points, otherwise transport volumes could potentially increase.

2.2.3. Mono-incineration capacities and P recovery technologies from ash

To incinerate the sludge of WWTPs ≥100,000 $PE_{capacity}$ no new capacities are needed in Austria (Table S 1). Scenario >100k therefore only relies on existing MI facilities (*MI-status-quo scenarios*, see Table 1). Without accounting for plant down-times, comparing the minimum MI capacity (without new constructions) per year with sewage sludge production in Austria, we see that the sewage sludge of all wastewater treatment plants ≥50,000 $PE_{capacity}$ could be treated with the currently available MI capacity as well (*Scenario >50k MI-status-quo*). In contrast, scenarios >20k and >2k with MI and recovery from ash (*MI + PR-SSA*) must include (i) the assumption that treatment and recovery from sludges exceeding the available MI capacity in Austria are executed in neighboring countries (*MI-status-quo scenarios*) or (ii) the hypothetical placement of new MI capacity somewhere in Austria (*MI-expansion* *scenarios*). To pair WWTPs with an existing or hypothetical new MI plant, transport distances from each plant i_n, to each MI j_n, were queried from OpenStreetMap (OpenStreetMap contributors, 2022). Then, a transport model was applied, accounting for both volumes and distances, to minimize overall transport volumes within the boundaries of available MI capacity at each site (see exemplary scenario Fig. 2 and other scenarios Figure S 1 – S 3). Sludge pre-drying was assumed to happen on-site of the MI plants.

The goal of this study was to provide a range of environmental and economic effects that P recycling could have on the current wastewater treatment system in Austria. To set this focus and due to the data intensive analysis, a pre-selection of P recycling processes had to be made. A selection of technologies with very different recycling approaches (e.g., higher energy demand vs. resource intensive) ensured that the range of potential effects would show in the results.

For recovery from ash, four different reference processes were chosen (Table S 2): AshDec, EcoPhos, Phos4Life and direct integration into the fertilizer industry.

2.2.4. *Decentralized P recovery*

Below 50,000 $PE_{capacity}$, energy consumption and costs at Austrian WWTPs are considerably higher due to a lack of supervisory personnel (Haslinger et al., 2016). Accordingly, it was assumed that decentralized P recovery is only performed at WWTPs ≥50,000 $PE_{capacity}$. Further on, since most decentralized recovery technologies are especially efficient in WWTPs with anaerobic digestion and enhanced biological P removal (EBPR), the preferred decentralized technology to be applied had to be determined for each plant. Therefore, two technology groups have been considered. Technology group one 1 (*TG1*): Airprex process at WWTPs ≥100,000 $PE_{capacity}$ with EBPR and Stuttgart process at all other WWTPs ≥50,000 $PE_{capacity}$; Technology group two 2 (*TG2*): Wasstrip + Lysotherm process at plants ≥100,000 $PE_{capacity}$ with EBPR and TerraNova process at all other WWTPs ≥50,000 $PE_{capacity}$. For sludges from WWTPs where decentralized recovery was economically unfavorable (lack of anaerobic digestion or an established MI concept), again MI and recovery from ash was considered (Table S 2). All main scenarios are summarized in Table 1.

2.2.5. *Sub-scenarios for drying*

In addition to drying on-site of MI plants (central drying), as used in the primary model for the main scenarios, decentralized drying at WWTPs is a further possibility to increase the calorific value of sludge for incineration. The WWTP itself, by reducing sludge mass, lowers the costs for sludge transport and could use surplus energy if available. On-site of a MI plant, surplus heat from incineration is usually sufficient to dry the sludge. At WWTPs, it is often insufficient to meet the demands for drying. Haberkern et al. (2008) assumes that 30 % of the heat demand for drying can be covered by waste heat from the internal combined heat and power unit. As the heat supply is a decisive issue, three sub-scenarios were investigated: drying with 0 %, 30 % or 100 % of the drying heat being supplied from surplus, carbon-free energy, and the remaining heat by gas. Within the scope of this analysis, those Austrian WWTPs potentially able to install drying capacity due to their size were selected for the scenarios. For the scenarios *>20k MI-status-quo* and *MI-expansion + PR-SSA*, 7 and 20 plants were selected, respectively. Using these, the drying requirements of the associated MI plants were determined. If a MI plant is already equipped with a dryer or one is under construction, no demand for decentralized drying was considered.

2.3. *Scenario evaluation and evaluation criteria*

The following evaluation methods and criteria were identified as relevant for this study (for detailed description see Amann et al., 2022). While Amann et al. (2022) provide results for the Status Quo, in this study we focus on the changes between scenarios and Status Quo to indicate performance of the different scenarios.

2.3.1. *Material flow analysis (MFA)*

The methodology of material flow analysis (MFA) according to ÖNORM S 2024; Austrian Standards, 2005) was applied to survey all flows into, out of and within the system. The basic principles of mass and energy conservation are used. Both the flows of goods and the flows of substances contained in the goods, such as P or heavy metals, can be balanced (Brunner and Rechberger, 2016). This enables the fate of substances within the system and substance emissions from it to be determined.

The considered substances in those goods are the nutrients nitrogen (N) and P, as well as heavy metals arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), mercury (Hg), nickel (Ni), lead (Pb) and zinc (Zn). These substances were chosen for their relevance in related legislation for sludge and fertilizer use in Austria (Fertilizer Ordinance, 2004; Compost Ordinance, 2001; Regulation, 2019).

The *P recycling rate* (%) is used to determine how much of the P contained in the wastewater treatment plant influent is applied in

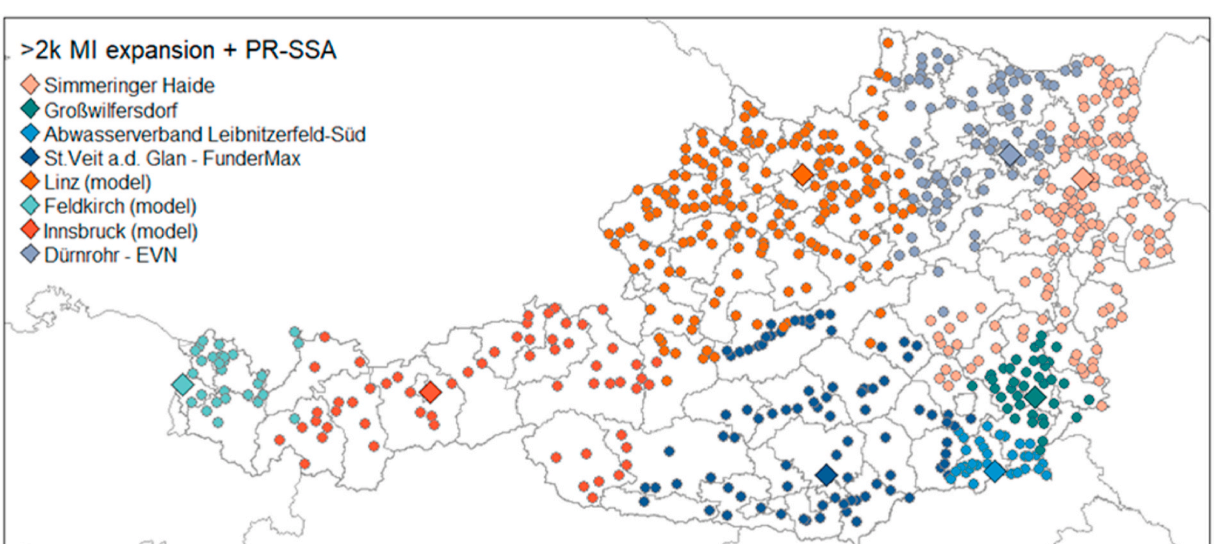

Fig. 2. MI and WWTPs transporting to a specific MI indicated by the same color for the exemplary scenario >2k MI-expansion + PR-SSA. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

agriculture or used. The *P utilization degree* (%) considers only the plant-available part of recovered P via MFA, reflecting the fraction of P available for fertilization.

To determine the fate of pollutants in the analyzed system, the following criteria were developed: the *heavy metal removal rate* (in %) represents the percentage of heavy metals originating from municipal wastewater and landfilled safely as inert material. The *rate of heavy metals directed to agricultural soils* (in %) instead considers the proportion of heavy metals that ends up in agriculture and could potentially harm humans and the environment.

Lack of sufficient data inhibits the modelling of organic trace substance behavior. Therefore, we gave their concentration in recovery products (in $\mu g\ kg^{-1}$ or $mg\ kg^{-1}$), determined using literature data, as a proxy for their potential for harm.

2.3.2. Life cycle assessment (LCA)

LCA is a method for evaluating systems and quantifying their environmental effects based on the emissions and consumption of resources in all relevant processes. The assessment considers both the direct environmental effects in the system itself (foreground system) and the indirect environmental effects in the upstream or downstream chain. The life cycle assessment was performed according to the specifications of ISO 14040 (2006) and ISO 14044 (2006). LCA is used to evaluate the mid-point indicators global warming potential (GWP), terrestrial acidification potential (TAP) and cumulative energy demand (CED). To ensure that P recycling strategies will not contribute to climate change and an increased energy use, it is required to compare GWP and CED for the different scenarios, as well as with current P mining and fertilizer production. As sulfuric acid is a main reagent in many P recycling technologies as well as for beneficiating primary P resources, TAP was additionally chosen to evaluated the impact of sulfuric emissions.

To determine all flows affecting GWP, TAP and CED, a life cycle inventory was set up (Chapter 2.4.1). The ecoinvent database 3.6 was then used to intersect the calculated flows with datasets on indirect environmental effects from background processes (Table S 3). Typical examples of these so-called background processes are electricity production, the manufacture of chemicals or subsequent waste disposal.

- *GWP* (in kg CO_2e; $CO_2 = 1$, $CH_4 = 29.7$, $N_2O = 264.8$; IPCC, 2013): emissions of CO_2 equivalents associated with the *Status Quo* and scenarios of wastewater treatment and sewage sludge utilization and disposal, are determined.
- *TAP* (in kg SO2e; $SO_2 = 1$, $NH_3 = 2.45$ and $NO_x = 0.56$; Goedkoop et al., 2009): emissions of SO_2 equivalents associated with the *Status Quo* and scenarios of wastewater treatment and sewage sludge utilization and disposal, including all upstream (e.g. chemical production) and downstream (e.g. disposal of residual materials) processes, are determined.
- *CED nuclear & fossil or renewable* (in MJe; VDI, 2015): the energy input associated with the *Status Quo* and the scenarios of wastewater treatment and sewage sludge utilization and disposal including all upstream (e.g. chemical production) and downstream (e.g. disposal of residual materials) processes is determined and assigned to the categories nuclear and fossil or renewable depending on the existing energy mix.

In addition to energy consumption and emissions from increased sewage sludge transport, actual freight volumes are of interest (noise pollution, particulate matter, space requirements, etc.). By means of transport routing, *volume of freight transport* (in tkm) is therefore determined for all scenarios.

2.3.3. Cost estimation

To predict changes in treatment costs if P recycling is enforced, a cost comparison calculation was performed to estimate the *additional costs for recovery* (in EUR). To give an indication of the expected cost change with or without valorization of products, the annual costs of the *Status Quo* of wastewater treatment and sewage sludge recycling and disposal are calculated considering and excluding revenues that can be obtained via (recovered) products, respectively.

2.3.4. Functionals units

Functional units are used as a standard, against which environmental impacts of different treatment alternatives can be assessed. The functional unit serves as a normalization factor, enabling a comparison of products or systems that may have different characteristics or lifespans. In this work two functional units are used to be able to establish a connection to inhabitants (inh), as well as to product-specific quantities (kg P recovered). The reference to actual Austrian inhabitants allows for a better comparison of total Austrian costs and environmental costs per scenario. The reference to kg P recovered enables a comparison of actual efficiencies of the different scenarios under analysis.

2.4. Data

2.4.1. Life cycle inventory

The individual P recovery processes considered in this study are not discussed in detail, as they are only *reference processes* for modeling costs and environmental impacts. A summary of these processes for the established life cycle inventory can be found in Table S 2; sources for the data were (Egle et al., 2014, 2016; Kraus et al., 2019a,b; Mehr and Hellweg, 2018; Morf, 2018; Schlumberger, 2019; Zhou et al., 2019 and Buttmann, personal communication, 2021). A detailed description of the processes can be found in reports by Egle et al. (2014, 2018) and Kraus et al. (2019a).

All required resources and energy, as well as products and wastes produced by the different technologies were considered in this analysis. The resources for the operation of decentralized processes can be found in Table S 4, while those of centralized processes are summarized in Table S 5. Infrastructure (basins, tanks, etc.) was also estimated following Kraus et al. (2019a,b). Transport distance for chemicals was assumed to be 100 km, transport of ash to the recovery plant at 200 km. Residues from the processes are shown in Table S 6. They are assumed to be treated and, if necessary, deposited.

Credits may apply for the substitution of primary P materials (P fertilizer, P acid), but only the plant-available P fraction of the recovery products was considered (Table S 7). It was defined as the relative net P uptake (net rPU in %) of plants compared to a water-soluble mineral fertilizer according to Kratz et al. (2019).

Besides P, other substances can also be recovered from SSA, mostly precipitating salts in the form of $FeCl_3$ or $AlCl_3$ and other chemical agents such as $CaCl_2$. Some processes also receive credits for electricity and heat, e.g., due to a reduced electricity consumption for sewage sludge dewatering (Table S 8).

2.4.2. Substance flows

To be able to map the fate of heavy metals originally contained in sewage sludge by means of MFA, transfer coefficients for the different cleaning, treatment and recycling processes had to be determined. The transfer coefficients finally chosen to represent an average situation are shown in Table S 9 and were sourced from Egle et al. (2014), Mehr and Hellweg (2018), Amann et al. (2018) and data of Buttmann (personal communication, 2021). For direct integration into the fertilizer industry, it was assumed that all heavy metals from SSA end up in the product.

2.4.3. Cost estimation

When possible, investment costs of P recovery were determined for two plant capacities. Then, following Kraus et al. (2019a,b), a ***r-factor*** was determined and applied to account for a non-linear price increase between capacities (Table S 10).

For integration into the fertilizer industry, the additional cost of an ash storage facility was applied to the investment costs. For the modeling of *PR + SSA* scenarios, it was assumed that a maximum of two P recovery plants could work cost-effectively in Austria. As further costs, according to Kraus et al. (2019a,b), the planning costs were set at 15% of the investment costs and additional peripheral costs for the AirPrex, Stuttgart process and Wasstrip + Lysotherm processes at 20% of the investments. In addition, maintenance cost was determined at 2 % of the investment costs per year. Personnel costs were calculated based on the manpower required and a cost of 60,000 EUR per person-year (Table S 10). For direct integration into the fertilizer industry, an additional requirement of 6 person-years was assumed, although with the sizes of existing plants in Austria, the requirement could possibly be covered by existing personnel.

Utility costs for resource demand and revenues from selling products were calculated using the life-cycle inventory of processes and defined unit prices per item based on Egle et al. (2014), Parravicini et al. (2020), Kreutzer, Fischer & Partner (2016), Kraus et al. (2019a,b), alibaba.com (2021), durchblicker.at (2021) and agrarheute.com (2021) (Table S 11).

Since the initial cost estimation, prices for utilities and products have increased substantially (Mbah and Wasum, 2022). Therefore, a cost sensitivity analysis was performed using updated prices from Q2/2023 (most recent ones at the time of preparing the manuscript) (businessanalytiq (2023), chemanalyst (2023), alibaba.com (2023), e-control (2023), WKO (2023), Statistik Austria, 2023a, Statistik Austria, 2023b, Stadt Wien (2023), agrarheute.com (2023); Table S 12).

3. Results

3.1. Material flow analysis (MFA)

Results for P recycling rates of each scenario can be seen in Table 2. The maximum achievable P recycling rate, including all WWTPs ≥2000 $PE_{capacity}$, is currently 87 % of the influent load of P to municipal WWTPs in Austria (mean for all *reference processes*). As expected, the recycling rate for the scenarios with mixed recovery from SSA or, if possible, on site of WWTPs is lower than for MI-only scenarios. With this strategy, only 42 to 61 (P recovery only) or 56–69 % of P (including direct agricultural reuse) could be recycled.

After subtracting the share of P in products or directly applied via sewage sludge that is not considered as plant-available, the P utilization degree is obtained (Table 2). The maximum P utilization degree achievable with the chosen *reference processes* and scenarios is 80 %. Considering the scenarios for WWTPs ≥20,000 $PE_{capacity}$, this is 73 % (MI scenarios) and 59 % (mixed scenarios) of the influent load, for those ≥50,000 $PE_{capacity}$ it is 63 % and 49 %, respectively. If direct agricultural application of sewage sludge were banned entirely, the P utilization degree would decrease by 0–12 percentage points, depending on the scenario.

For SSA processes, an acceptable P concentration in the ash is essential for economic viability. Sewage sludge ash competes with the primary feedstock (processed) rock phosphate, which usually has concentrations of at least 9–10 % P. Looking at the empirical distribution function (Fig. S 4) of calculated P concentrations in the theoretical SSA of each Austrian WWTP, it can be seen that at least 65% of the sludges, given separate MI, would reach a P ash concentration of 9%. However, lower concentrations down to 5 % are also possible. The average P concentration from theoretical mixing of all sludge from Austria and MI is 8.2–8.8 %, depending on the scenario (Table S 13).

Earlier studies on heavy metal contamination of sewage sludge have shown that despite the massive decrease in heavy metal concentrations in sewage sludge, the potential loads from sewage sludge application are not insignificant compared to other inputs to agriculture (Amann et al., 2021). This shows that a possible increase through the implementation of P recovery strategies must be considered.

For P products from precipitation at the WWTP (e.g., struvites, calcium-phosphates, …), no risk can be determined, since the corresponding products are very low in heavy metals, and only a small percentage of sludge metals is retained (Tables S 14 – S 15; Kraus and Seis, 2015; LfU, 2015; Krüger et al., 2016; Mehr and Hellweg, 2018; Buttmann, 2021). In contrast, a large proportion of the heavy metals from the influent of WWTPs ends up in sludge and thus predominantly also in conventional SSA. With the current implementation of sewage sludge utilization and disposal (*Status Quo*), a considerable proportion of heavy metals is already deposited together with the ash in landfills via incineration and ash deposition and thus discharged into a final sink. Heavy metals enter streams via wastewater treatment plant effluent, soils or groundwater via agricultural application of sewage sludge, the atmosphere via sewage sludge incineration, or material storage or landfill via sewage sludge ash.

Table 2
Results for P recycling rate and P utilization degree given as % of WWTP-P inlet loads. For each scenario, the mean of all reference processes is shown.

Scenario:	**P recycling rate in % of WWTP-P inlet through …**			**P utilization degree in % of WWTP-P inlet through …**		
	Sludge use in agriculture (direct, compost)	Recovery and application in agriculture or industry	Total	Sludge use in agriculture (direct, compost)	Recovery and application in agriculture or industry	Total
Status Quo (Amann et al., 2022)	21%	0%	21%	12%	0%	12%
>2k MI-expansion + PR-SSA	0%	87%	87%	0%	80%	80%
>20k MI-status-quo + PR-SSA	8%	75%	83%	4%	69%	73%
>20k AGR/MI-status-quo + PR-SSA	23%	61%	84%	12%	56%	69%
>20k MI-expansion + PR-SSA	8%	75%	83%	4%	69%	73%
>20k PR-WWTP TG1 + PR-SSA	9%	56%	65%	5%	53%	58%
>20k PR-WWTP TG2 + PR-SSA	8%	61%	69%	4%	55%	59%
>50k MI-status-quo + PR-SSA	14%	61%	74%	7%	56%	63%
>50k MI-expansion + PR-SSA	14%	61%	74%	7%	56%	63%
>50k PR-WWTP TG1 + PR-SSA	15%	42%	56%	8%	40%	48%
>50k PR-WWTP TG2 + PR-SSA	14%	47%	60%	7%	41%	49%
>100k MI-status-quo + PR-SSA	17%	53%	70%	9%	49%	58%

The average *heavy metal removal rate* for the *Status Quo* is 32%. The *rate of heavy metals directed to agricultural soils* in the *Status Quo* is currently about 20% (Amann et al., 2021). A large portion is not retained by WWTPs and enters water bodies via the effluent.

If no significant deterioration compared to the *Status Quo* is to be accepted, two things must be considered: Even if the *heavy metal removal rate* increases with the help of recovery processes, there may still be an increase in those directed to agriculture, since the amount of sewage sludge from which P recovery is realized increases in the scenarios. At the same time, however, it is possible that the heavy metals to agriculture will decrease while the removal rate remains the same because some processes produce phosphoric acid and other products for industrial purposes and thus those heavy metals no longer end up in agriculture. Whether heavy metals in these products can still cause significant health or environmental damages depends on their use. Iron chlorides for example, which are produced in the EcoPhos and Phos4Life processes, could be used again in the wastewater treatment plant, and the heavy metals they contained would also be recycled. It can therefore be stated that the future strategy should not lead to a significant deterioration of the *heavy metal removal rate* if higher loads of sewage sludge are recycled. As can be seen from Table 3 this only applies to the direct integration of sewage sludge ash into the fertilizer industry, since here no targeted separation of heavy metals takes place in the process. In the sense of limiting the heavy metal load, additional specifications for heavy metal removal would therefore have to be defined in the legal regulations.

A strong input of organic trace substances via P recovery products into agriculture is unlikely, although a general distinction must be made between products recovered from sludge/WWTPs and those recovered from SSA. Organic pollutants are marginally transferred from sludge/liquor to precipitated P products (Tables S 18 and S 19; LfU, 2015; Morf, 2018). Significant depletion from the WWTP influent to the product was detected for pharmaceutical residues (Stenzel et al., 2019). Previous risk assessments also show no increased negative effects on the soil, groundwater, and humans due to use of struvite (Kraus et al., 2019b). To keep the concentration of organic pollutants low (especially with precipitation from digested sludge), an optimal separation of struvite crystals or other P forms from the organic sludge mass should be achieved. This can be accomplished by multi-stage reactor systems and by washing the material.

Table 3
Results for the heavy metal removal rate and the share of heavy metal transferred to agricultural soils given as the share of each heavy metal (mean over all heavy metals) in relation to its WWTP influent load. Results for the four considered SSA reference processes AshDec (AD), Direct Integration (DI), EcoPhos (EP) and Phos4Life (PL) are shown. Individual values for each heavy metal are shown in Table S 16 and S 17.

Scenario	Heavy metal removal rate				Share of heavy metals redirected to agricultural soils			
	AD	DI	EP	PL	AD	DI	EP	PL
Status Quo (Amann et al., 2022)	32%				20%			
>2k MI-expansion + PR-SSA	45%	16%	34%	32%	27%	55%	0%	0%
>20k MI-status-quo + PR-SSA	39%	15%	30%	29%	29%	54%	6%	6%
>20k AGR/MI-status-quo + PR-SSA	31%	12%	24%	23%	38%	57%	20%	20%
>20k MI-expansion + PR-SSA	39%	15%	30%	29%	29%	54%	6%	6%
>20k PR-WWTP TG1 + PR-SSA	47%	34%	41%	42%	18%	31%	7%	7%
>20k PR-WWTP TG2 + PR-SSA	48%	35%	42%	42%	19%	32%	6%	6%
>50k MI-status-quo + PR-SSA	34%	15%	27%	26%	30%	49%	11%	11%
>50k MI-expansion + PR-SSA	34%	15%	27%	26%	30%	49%	11%	11%
>50k PR-WWTP TG1 + PR-SSA	41%	34%	38%	39%	19%	26%	12%	12%
>50k PR-WWTP TG2 + PR-SSA	42%	34%	39%	39%	19%	27%	11%	11%
>100k MI-status-quo + PR-SSA	32%	15%	25%	25%	31%	47%	14%	14%

If sewage sludge is thermally treated at temperatures above 500 °C, the resulting ash typically shows levels of organic pollutants below the current analytical detection limit (LfU, 2015; Stenzel et al., 2019). In some studies, sewage sludge has even been found to inhibit the formation of persistent organic pollutants in co-incineration with other PVC rich-waste or municipal waste (Conesa, 2021; Gandon-Ros et al., 2021). The co-incineration of plastics should be treated with caution, as this in turn can lead to increased formation of trace organic substances, such as polycyclic aromatic hydrocarbons, that also show in the ash composition (Conesa et al., 2021). At present, it is still controversial how per- and polyfluorinated alkyl compounds (PFAS) behave during incineration. Some studies showed that the best-known PFAS, PFOA and PFOS, are generally destroyed during incineration (Winchell et al., 2021). On the other hand, there is concern that PFAS transform into other, unknown, and potentially toxic substances (Stoiber et al., 2020).

3.2. Life cycle assessment (LCA)

GWP for the *Status Quo* of wastewater treatment and sludge reuse and disposal in Austria is 61 kg CO_2e inh^{-1} yr^{-1} (Amann et al., 2022). In comparison, the total *GWP* per inhabitant in Austria is about 9100 kg CO_2e yr^{-1} (Anderl et al., 2021a). Thus, the wastewater treatment/-sewage sludge disposal system accounts for about 0.7 % of total *GWP* in Austria.

Compared to the *Status Quo*, minor changes, or improvements in the range of −3.2 to +0.4 kg CO_2e PE^{-1} yr^{-1} can be observed for most scenarios (Fig. S 5). An increased electricity and heat demand is offset by credits from P recovery and product generation. It can also be observed that *GWP* tends to decrease as more and more WWTPs are included in the recovery strategy.

Fig. 3 compares the *P utilization degree* achieved in each scenario to the respective change in *GWP* per each additional kg P recovered. The scenarios show a variation in the range of −4.5 to +4.5 kg CO_2e kg^{-1} P. Scenarios with a low P utilization can increase and reduce CO_2e emissions through P recovery, depending on the applied technology. Those with expansion of MI capacity and the involvement of WWTPs ≥20,000 or ≥2000 $PE_{capacity}$ tend to offer both the highest P utilization degree and a reduction in *GWP* due to energy recovery during MI. However, the achievable P utilization degree is highly dependent on the recovery process. Those scenarios which require P recovery from the ash only for WWTPs ≥50,000 $PE_{capacity}$ are below a 70 % P utilization and contribute less to a reduction of *GWP*.

Nitrous oxide emissions from MI account for around 70 % of the GWP of direct emissions from MI (Amann et al., 2022). In the primary model, an emission value of 150 mg m^{-3} flue gas was applied. Assuming mean emissions in the range of 300 mg m^{-3} (based on German Environment Agency, 2018), mean GWP of the scenarios would increase by up to an additional 6 % relative to the Status Quo (Table S 20). As a result, the net decrease in GWP compared to the Status Quo found for all scenarios in the primary model would no longer be present, and shift to an increase.

TAP for the *Status Quo* is 0.22 kg SO_2e inh^{-1} yr^{-1} (Amann et al., 2022). In comparison, the total *TAP* per inhabitant in Austria is about 30 kg SO2e yr^{-1} (Anderl et al., 2021b). Thus, the wastewater treatment/-sewage sludge disposal system accounts for about 0.7 % of total *TAP* in Austria. For all considered scenarios, the *TAP* decreases compared to the

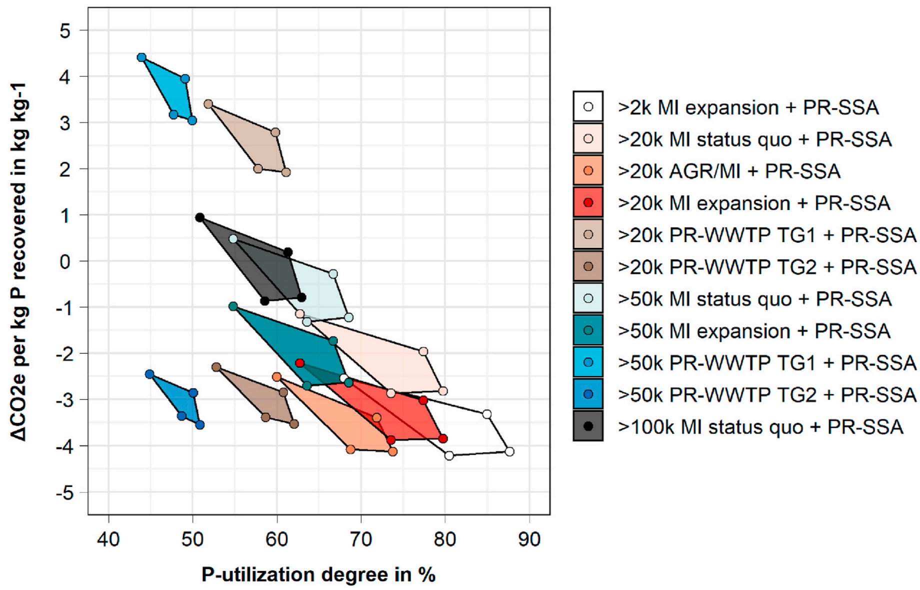

Fig. 3. Comparison of the *P utilization degree* and the additional *GWP* per kg of P recovered in relation to the *Status Quo*. The four dots per scenario mark the results for different ash recovery technologies (*AshDec, EcoPhos, Direct integration, Phos4Life*).

Status Quo. The net improvement is in the range of −0.15 to −0.02 kg SO2 SO_2e inh^{-1} yr^{-1} (Fig. S 6). This is mainly due to the decrease of direct agricultural utilization of sewage sludge and, therefore, ammonia emissions in the scenarios.

The additional *TAP* for scenarios with recovery at the WWTP and for the *MI-status-quo* with inclusion of WWTPs larger than 100,000 $PE_{capacity}$ is less favorable than for the other scenarios (Fig. S 7). Although it could theoretically achieve the highest P utilization rate and at the same time the greatest mean improvement in terms of acidification potential, the *MI-expansion* scenario is characterized by the greatest variability for WWTPs with 2000 $PE_{capacity}$ or more, depending on the technology.

For *CED*, the relatively low relevance of the wastewater treatment/sewage sludge disposal system in the overall context is clear. With a value of about 410 MJe PE^{-1} yr^{-1}, the wastewater treatment and sludge disposal system currently accounts for about 0.2% of the total *CED* (166,000 MJe PE^{-1} yr^{-1}, Eurostat, 2022). The expected changes in CED_{total} range for most scenarios from −77 to +43 MJe PE^{-1} yr^{-1} (Fig. S 8). Therefore, an improvement in *CED* can, but is not necessarily achieved. In addition to differences in the scenarios, *CED* also shows a significant dependence on the recovery process used.

Fig. 4 contrasts the P utilization rate with the change in total *CED* between *Status Quo* and the different scenarios per kg P. The scenarios with recovery at the WWTP (*PR-WWTP*) show a change in total *CED* that varies in the range −150 - +125 MJe kg^{-1} P, where the technology group two (TG2) with Wasstrip + Lysotherm and TerraNova show much better results as the technology group one (TG1, Airprex and Stuttgart process). The *MI-only* scenarios are in a range of −60 - +80 MJe kg^{-1} P again highly dependent on the assumed reference process.

Heat utilization concepts of MI plants represent a major factor in the ecological impact of P recovery from SSA. In the primary model, it was assumed that the overall energy efficiency of MI plants is 64 %. In principle, however, total efficiencies in the fluidized bed of up to 80 %

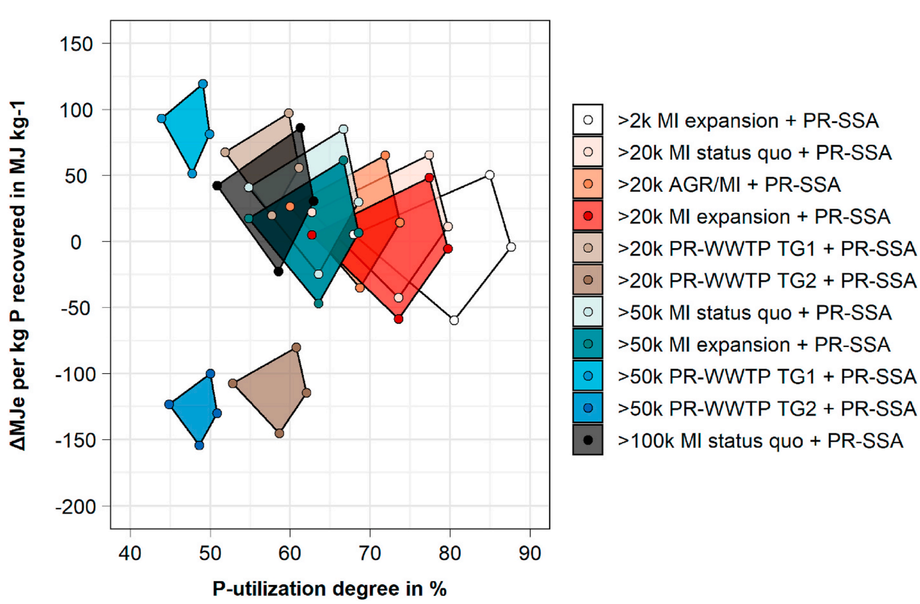

Fig. 4. Comparison of the P utilization degree and the additional CED per kg of P recovered in relation to the *Status Quo*. The four dots per scenario mark the results for different ash recovery technologies (AshDec, EcoPhos, Direct integration, Phos4Life).

are possible (Böhmer et al., 2007). Increasing the overall efficiency to 80 % results in an additional decrease of the GWP on top of the primary model up to 3 % and the CED up to 7 % relative to the *Status Quo* (Table S 21).

Table 4 shows the *volume of freight transport* per scenario. The *Status Quo* has a freight volume of about 60 million tkm per year. This corresponds to about 0.1 % of total Austrian freight transport (50–57 billion tkm yr^{-1} in 2017–2021, Statistik Austria, 2017; Statistik Austria, 2018). In the scenarios assuming *MI-status-quo* with consequent P recovery, volume of freight transport for wastewater treatment, sludge management and P recovery would increase by 40–60 % as compared to the current situation (*Status Quo*). Combining *MI-status-quo* with increased P recovery at the WWTP (*PR-WWTP* scenarios), this increase is reduced to 8–20 %. If MI capacities were to be expanded the associated *volume of freight transport* would remain at the current level.

By implementing sludge drying on-site of WWTPs, the *volume of freight transport* of the scenario *>20k MI-status-quo + PR-SSA* could be reduced by about 4 %, and those with *MI-expansion + PR-SSA* by as much as 12 % (Table S 22). Provided that no surplus heat is available, however, no advantage of increased decentralized drying over centralized drying can be identified in the (environmental) costs.

Replacing 30 % with surplus heat may result in slight advantages in the reduction of GWP and CED. From an economic point of view, no advantage can be seen in this case either. Decentralized drying appears to be advantageous only if the entire energy is obtained from surplus heat. This could reduce the GWP of the system by up to 6 % and the CED by up to 15 % compared to the *Status Quo*.

Another factor influencing the life cycle impact of P recovery from wastewater are byproduct production and use. In addition to fertilizers, some processes (e.g., *EcoPhos*, *Phos4Life*) also produce products such as precipitants. Since future legal obligations will most likely only demand recovery of P, it remains a question of economic viability whether these by-products will be produced and offered on the market. If they are not, this would result in an additional increase of *GWP* up to 5.5 %, *TAP* up to 13 %, and *CED* up to 12% compared to the *Status Quo* and depending on the technology (Table S 23). Thus, the impact of by-product-use is not negligible, as these products generate further credits by reducing energy consumption and emissions from the primary production of these products.

Table 4
Volume of freight transport in Mio. tkm yr^{-1} per scenario and for wastewater & sludge treatment as well as P recovery.

volume of freight transport in Mio. tkm yr^{-1}			Δtkm to Status Quo
Scenario	Wastewater & sludge treatment	P recovery	
Status Quo	60	0	–
>2k MI-expansion + PR-SSA	48	16	8%
>20k MI-status-quo + PR-SSA	81	14	60%
>20k AGR/MI-status-quo + PR-SSA	81	11	53%
>20k MI-expansion + PR-SSA	46	14	0%
>20k PR-WWTP TG1 + PR-SSA	66	7	23%
>20k PR-WWTP TG2 + PR-SSA	64	7	19%
>50k MI-status-quo + PR-SSA	85	11	62%
>50k MI-expansion + PR-SSA	46	11	−4%
>50k PR-WWTP + PR-SSA TG1	63	5	14%
>50k PR-WWTP + PR-SSA TG2	61	5	11%
>100k MI-status-quo + PR-SSA	75	10	42%

3.3. Cost estimation

The additional cost for P recovery (net value of expenses and savings or revenues) for the scenarios with MI and recovery from ash (PR-SSA) is in the range of −0.8 to +2.6 EUR inh^{-1} yr^{-1} (Fig. S 10) and −1.8 to +4.3 EUR kg^{-1} P recovered (Fig. 5) with the cost basis of 2020, respectively. For the scenarios with combination of recovery from ash and recovery at the WWTP (PR-WWTP), determined costs are higher, in the range of +1.8 to +3.6 EUR inh^{-1} yr^{-1} and +3.2 to +8.8 EUR kg^{-1} P recovered, respectively. Highest cost contributors are generally capital and utility costs, with transport costs being an additional factor in *MI-status-quo* scenarios.

Differences between the applied recovery processes are striking. However, it is important to point out that available investment cost estimations by technology providers might be based on (i) specific plant sizes and/or locations, (ii) a low level of technological readiness (no state-of-the-art), (iii) different assumptions on building material pricing and (iv) different system boundaries. Interestingly, processes with higher returns also need to compensate a higher demand for heat and/or utilities, following the assumption that production for market-ready specifications takes more technological effort, but will achieve higher returns.

MI-status-quo scenarios are shown as slightly more favorable than the expansion of MI capacities, with a reduction in mean cost of about 0.5 EUR inh^{-1} yr^{-1} or 1 EUR kg^{-1} P recovered – mainly due to a better utilization of existing plants. In the "worst case", i.e. if no revenues are generated from the sale of products, additional costs of +0.4 to +5.5 EUR inh^{-1} yr^{-1} are expected due to the recovery. With processes where a high effort is made to recover other by-products, the impact of revenues is particularly striking. In this case, products account for savings of around 3 EUR inh^{-1} yr^{-1}.

Looking at the progression of costs in the three scenarios *MI-expansion* + PR-SSA ≥ 2k, ≥20,000 and ≥ 50,000 $PE_{capacity}$, both mean costs per inhabitant and per kg of P recovered decrease with the consideration of more WWTPs in the recovery strategy (Fig. S 11). Mean costs for each size class also tend to decrease with *MI-expansion* scenarios the more WWTPs are included. The economics of MI plants but also of SSA recovery plants play a role here, improving with increasing plant size (Kraus et al., 2019a,b). The largest mean cost increase, at 4.2 EUR inh^{-1} yr^{-1} would result with the scenario ≥50k *MI-expansion* + PR-SSA. This is largely because there is no additional MI capacity required to incinerate the sewage sludge of WWTPs ≥50,000 $PE_{capacity}$ (see Table S 1). Additional plants would reduce transport distances but still lead to an excessive increase in cost.

Marginal costs (costs incurred for an additional kg of P produced) were evaluated for the *MI status quo* and *MI expansion* scenario groups. Fig. S 12 shows the marginal costs in EUR kg^{-1} P and for each of the four ash processes considered. For the marginal costs of the *MI status quo* scenarios the following can be said: The additional costs to be paid by including WWTPs ≥50,000 to 100,000 $PE_{capacity}$ are on average 1.5 to 2 EUR kg^{-1} P cheaper. To additionally include the wastewater treatment plants between 20,000 and 50,000 $PE_{capacity}$, a higher price per kg of P must be paid. If these WWTPs are included, a further mono-incineration plant would have to be built and higher capital costs are to be expected as a result. The marginal cost analysis of the *MI expansion* scenarios shows a different picture. While the initial costs for the >50k scenario are still comparatively high, they decrease significantly by 1–3 EUR kg^{-1} P for the additional expansion of the WWTPs between 20,000 and 50,000 $PE_{capacity}$. The additional costs to move from the >20k scenario to the >2k scenario differ only slightly from the step from >50k to >20k. Depending on the process, the marginal costs for the plants between 2000 and 20,000 PE expansion can increase or decrease again.

Finally, the increase in the cost of the system is also primarily dependent on the current costs, and the current disposal method of the sewage sludge (Fig. S 12). Plants that already predominantly use mono-incineration come off most favorably and can even achieve a reduction

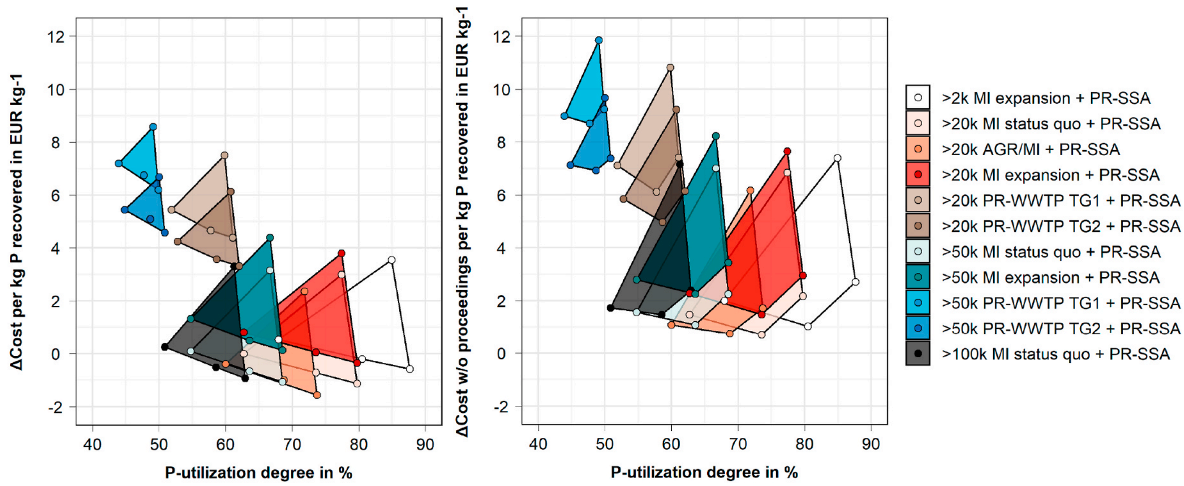

Fig. 5. Comparison of the P utilization degree in percent and the additional cost in EUR per kg of P recovered in relation to the *Status Quo*. The four dots per scenario mark the results for different ash recovery technologies (AshDec, EcoPhos, Direct integration, Phos4Life).

in costs if MI capacities are increased. The situation is similar for co-incineration and composting by a contractor. For most plants, the increase would be limited to less than 2.5 EUR inh^{-1} yr^{-1}. The plants that currently go the route of agricultural recycling or on-site composting will experience the highest cost increase because of the system change.

A sensitivity analysis, with unit prices adapted to Q2/2023, revealed a cost progression from 2020 to 2023 by 0.1–2.1 EUR inh^{-1} yr^{-1} or 0.3 to 3.2 EUR kg^{-1} P recovered (Table 5). Interestingly, scenarios with higher costs were subject to a lower cost increase in this period.

4. Discussion & conclusion

This study shows that agricultural application of sludge can outperform processes with P recovery from SSA at the individual WWTP level in the rate of P recycled to agriculture. Regarding the degree of actual P utilization however, this strongly depends on the availability of P in sludge or recovered products (Kratz et al., 2019). Overall, the potential of agricultural sewage sludge utilization in Austria is severely limited by application bans or restrictions and low acceptance (e.g. Oliva et al., 2009; Agrarmarkt Austria, 2023). This is in line with strategies of other European countries like Switzerland, Germany, and the Netherlands (Santos et al., 2021). An increase beyond the current level is therefore not realistic and factually controversial due to persistent organic pollutant levels and microplastics (Egle et al., 2023).

A strategy with a strongly expanded P utilization must therefore rely on recovery. Considering current P recovery rates of different processes, achieving up to 80 % recovery of $P_{influent}$ from WWTPs >2000 $PE_{capacity}$ via MI and SSA treatment appears to be realistic. This matches the current knowledge on achievable recovery rates by SSA recycling (e.g. Jama-Rodzeńska et al., 2021; Santos et al., 2021). If only WWTPs ≥100,000 $PE_{capacity}$ are included, this potential is reduced to 50 %. Processes that aim for recovery at the WWTP fall significantly behind SSA processes in terms of P recovered (compare e.g. to Yu et al., 2021).

Table 5
Comparison of scenarios costs of the price analysis in Q1/2020 and Q2/2023.

Scenarios with …	… mono-incineration + recovery from ash only (MI + PR-SSA)		… mono-incineration + ash recovery or recovery at the WWTP (PR-WWTP + PR-SSA)	
Year	EUR inh^{-1} yr^{-1}	EUR kg^{-1} P recovered	EUR inh^{-1} yr^{-1}	EUR kg^{-1} P recovered
Q1/2020	−0.8 to +2.6	−1.8 to +4.3	+1.8 to +3.6	+3.2 to +8.8
Q2/2023	−0.1 to +4.7	−0.2 to +7.5	+1.9 to +4.7	+3.5 to +10.9

Organic trace substances are largely destroyed during MI, and products from SSA are thus only slightly contaminated (Kraus et al., 2019a,b). The same applies to products recovered at the WWTP. The rate of heavy metal removal strongly depends on the selected P recovery process. In the worst case of direct use of SSA in the fertilizer industry without decontamination, heavy metal load to soils might strongly increase compared to the *Status Quo*.

Considering average costs of implementing a strategy for P recovery from municipal wastewater within the economic boundaries of 2020–2023, at best, savings up to 2 EUR kg^{-1} $P_{recovered}$ can be achieved. In unfavorable cases, additional economic costs of up to 12 EUR kg^{-1} $P_{recovered}$ are incurred. This at a fertilizer P price of about 2 EUR kg^{-1} $P_{recovered}$ (agrarheute.com, 2021). As stated by e.g. Jupp et al. (2021) these unfavorable economics require legislation and political support (Nedelciu et al., 2019) and the establishment of new value chains that valorize the added value of recycled fertilizers. Compared to the average cost of wastewater treatment (32 EUR PE^{-1} yr^{-1} or 51 EUR inh^{-1} yr^{-1}; Amann et al., 2022) the cost of P recycling with −0.8 to a maximum value of +4.7 EUR inh^{-1} yr^{-1} is rather low.

Some key cost factors determined in this study are (i) the choice of recovery process (difference of 3–4 EUR kg^{-1} of P recovered), (ii) the possibility of generating revenue from products (phosphoric acid, P fertilizer) (difference of 2–4 EUR kg^{-1} of P recovered), (iii) the prevailing interest rate (difference of 0.5–1.0 EUR kg^{-1} of P recovered), and (iv) the extent to which existing incineration capacity is utilized (difference of approximately 1 EUR kg^{-1} of P recovered). Recovery on-site of WWTPs is expected to be more expensive, especially if high recovery rates should be achieved.

By including smaller WWTPs in Austria, costs per kg of $P_{recovered}$ can be decreased. Inclusion of these WWTP sizes would result in lower costs for the larger WWTPs due to a higher exploitation of existing MI infrastructure and following traditional economy of scale arguments (e.g., Jama-Rodzeńska et al., 2021).

Implementing a P recovery strategy in Austria has the potential to slightly reduce the GWP of wastewater treatment and sludge disposal. Savings in emissions tend to rise with higher P recovery and use of P in agriculture. The CED of wastewater and sludge treatment can increase or decrease if a recovery strategy is implemented. Main influencing factor here is the choice of recovery process. Suitable processes at the WWTP can also lead to a reduction in required energy.

The total transport volume when implementing P recovery in Austria is strongly influenced by the overall strategy regarding incineration. The expansion of capacities at regionally strategic sites would keep the transport volume roughly the same as in the *Status Quo*. A strategy that relies predominantly on existing incineration capacities could significantly increase the transport volume. Corresponding, the same can be said for the environmental criteria GWP and CED.

To reduce potential negative effects of an enhanced P recovery strategy on the environment, it is important to limit nitrous oxide emissions (German Environment Agency, 2018; Egle et al., 2023) and to implement efficient heat utilization concepts during sludge incineration, as well as the use of recovery by-products in other processes (Amann et al., 2018). Concepts for decentralized drying will only be an environmentally friendly alternative where transport distances are high, and large parts of the drying energy required can be provided from surplus heat.

In conclusion, the following recommendations are made for the case of P recovery in Austria.

1. Of the concepts considered, P recovery from ash shows the highest potential regarding the utilization of P from wastewater and should thus be the basis of the chosen P recovery strategy. Requirements for potential recovery on-site of WWTPs should be kept high to avoid an outcome with low recovery.
2. A higher P utilization from wastewater should rely on processes that decontaminate products, otherwise heavy metal loads to agricultural soils might increase.
3. Recovery to the extent of 60–85 % of $P_{influent}$ of WWTPs >2000 $PE_{capacity}$ has expected average additional costs of −0.8 to +4.7 EUR inh^{-1} yr^{-1}. Key factors to be considered for costs are the choice of recovery process, revenues from products, and the use of existing incineration infrastructure, as well as external economic boundaries.
4. P recovery can lead to a reduction of greenhouse gas emissions in Austria if nitrous oxide emissions are limited in sludge incineration and efficient heat utilization strategies are implemented. There is a trade-off in terms of environmental and economic costs in choosing an incineration strategy. While a strategy that uses the existing infrastructure as much as possible leads to a reduction in costs, this strategy is worse in terms of environmental costs, primarily due to an increase in transport volumes. Combining existing incineration infrastructure in the east of Austria, using railway transport of sludge as far as possible, and building additional regional incineration infrastructure e.g., in Upper Austria and the West of Austria, presents an ideal balance of both criteria.

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CRediT authorship contribution statement

A. Long: Writing – review & editing, Writing – original draft, Visualization, Validation, Software, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation. **N. Weber:** Validation, Methodology, Investigation, Formal analysis. **J. Krampe:** Writing – review & editing, Supervision, Resources, Project administration, Conceptualization. **S. Peer:** Methodology, Formal analysis. **H. Rechberger:** Writing – review & editing, Supervision, Resources, Project administration, Funding acquisition, Conceptualization. **M. Zessner:** Writing – review & editing, Validation, Supervision, Project administration, Methodology, Funding acquisition. **O. Zoboli:** Writing – review & editing, Visualization, Validation, Software, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Conceptualization.

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.

Data availability

Data will be made available on request.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2024.121339.

References

agrarheute.com, 2021. Product Query on the 16.2.2021 at 13:00. .
agrarheute.com, 2023. Product Query on the 14.08.2023 at 13:00. .
Agrarmarkt Austria, 2023. ÖPUL 2023 Einschränkung Ertragssteigernder Betriebsmittel Stand Oktober 2023 (ÖPUL Restriction of Yield-Enhancing Operating Resources in October 2023).
alibaba.com, 2021. *Product query on 15.2.2021 at 16:00*. Alibaba, com.
alibaba.com, 2023. *Product query on 14.08.2023 at 13:00*. Alibaba, com.
Amann, A., Zoboli, O., Krampe, J., Rechberger, H., Zessner, M., Egle, L., 2018. Environmental impacts of phosphorus recovery from municipal wastewater. Resour. Conserv. Recycl. 130, 127–139. https://doi.org/10.1016/j.resconrec.2017.11.002.
Amann, A., Weber, N., Krampe, J., Rechberger, H., Zoboli, O., Zessner, M., 2021. Operation and performance of austrian wastewater and sewage sludge treatment as a basis for resource optimization. Water (Switzerland) 13 (21), 2998. https://doi.org/10.3390/w13212998.
Amann, A., Weber, N., Krampe, J., Rechberger, H., Peer, S., Zessner, M., Zoboli, O., 2022. Systematic data-driven exploration of Austrian wastewater and sludge treatment - implications for phosphorus governance, costs and environment. Sci. Total Environ. 846, 157401 https://doi.org/10.1016/j.scitotenv.2022.157401.
Anderl, M., Bartel, A., Geiger, K., 2021a. *Klimaschutzbericht (Climate Protection Report)* (REP-0776). Austrian Environment Agency (Umweltbundesamt).
Anderl, M., Gangl, M., Haider, S., Heinfellner, H., Lampert, C., Perl, D., Poupa, S., Purzner, M., Schieder, W., Titz, M., Zechmeister, A., 2021b. Emissionstrends 1990-2019 (emission trends 1990-2019). *Ein Überblick über die österreichischen Verursacher von Luftschadstoffen (Datenstand 2021)* (REP-0770). Austrian Environment Agency (Umweltbundesamt).
Bauwens, T., Hekkert, M., Kirchherr, J., 2020. Circular futures: what will they look like? Ecol. Econ. 175, 106703 https://doi.org/10.1016/j.ecolecon.2020.106703.
Brownlie, W.J., Sutton, M.A., Heal, K.V., Reay, D.S., Spears, B.M., 2022. Our Phosphorus Future. UK Centre for Ecology and Hydrology.
Böhmer, S., Kügler, I., Stoiber, H., Walter, B., 2007. Abfallverbrennung in Österreich. Statusbericht 2006 (REPORT No. Umweltbundesamt AT, Wien.
Brownlie, W.J., Sutton, M.A., Cordell, D., Reay, D.S., Heal, K.V., Withers, P.J.A., Vanderbeck, I., Spears, B.M., 2023. Phosphorus price spikes: a wake-up call for phosphorus resilience. Front. Sustain. Food Syst. 7. https://www.frontiersin.org/articles/10.3389/fsufs.2023.1088776.
Brunner, P.H., Rechberger, H., 2016. Handbook of Material Flow Analysis, second ed. CRC Press.
businessanalytiq, 2023. Query on 14.08.2023 at 10:00. *businessanalytiq.com/*. https://businessanalytiq.com/.
Buttmann, M., 2021. *Personal Communication on 1.2.2021* [Personal communication].
chemanalyst, 2023. Pricing overview. Query on 14.08.2023 at 10:00. https://www.chemanalyst.com/Pricing/Pricingoverview.
Chrispim, M.C., Scholz, M., Nolasco, M.A., 2019. Phosphorus recovery from municipal wastewater treatment: critical review of challenges and opportunities for developing countries. J. Environ. Manag. 248, 109268 https://doi.org/10.1016/j.jenvman.2019.109268.
Conesa, J.A., 2021. Sewage sludge as inhibitor of the formation of persistent organic pollutants during incineration. Sustainability 13 (19), 10935. https://doi.org/10.3390/su131910935.
Conesa, J.A., Nuñez, S.S., Ortuño, N., Moltó, J., 2021. PAH and POP presence in plastic waste and recyclates: state of the art. Energies 14 (12), 3451. https://doi.org/10.3390/en14123451.
dos Santos Gonçalves, P.V., Campos, L.M.S., 2022. A systemic review for measuring circular economy with multi-criteria methods. Environ. Sci. Pollut. Control Ser. 29 (21), 31597–31611. https://doi.org/10.1007/s11356-022-18580-w.
durchblicker.at, 2021. Query of the Official Austrian Gas Price on 15.2.2021 at 10:00. .
e-control, 2023. Strompreis und Gaspreisentwicklung für Industrie und Gewerbe (Electricity and gas price trends for industry and commerce). *Query on 14.08.2023 at 10:00*. https://www.e-control.at.
Egle, L., Zoboli, O., Thaler, S., Rechberger, H., Zessner, M., 2014. The Austrian P budget as a basis for resource optimization. Resour. Conserv. Recycl. 83, 152–162. https://doi.org/10.1016/j.resconrec.2013.09.009.
Egle, L., Rechberger, H., Krampe, J., Zessner, M., 2016. Phosphorus recovery from municipal wastewater: an integrated comparative technological, environmental and

economic assessment of P recovery technologies. Sci. Total Environ. 571, 522–542. https://doi.org/10.1016/j.scitotenv.2016.07.019.

Egle, L., Zoboli, O., Amann, A., Krampe, J., Rechberger, H., Zessner, M., 2018. Comparison of Technologies for Phosphorus Recovery – Identification of an Ideal Solution? IWA Publishing. https://doi.org/10.2166/9781780408361_457.

Egle, L., Marschinski, R., Jones, A., Yunta Mezquita, F., Schillaci, C., Huygens, D., 2023. Feasibility Study in Support of Future Policy Developments of the Sewage Sludge Directive (86/278/EEC). Publications Office of the European Union. https://data.europa.eu/doi/10.2760/305263.

European Comission, 2020. Communication from the commission to the EUROPEAN parliament, the council, the EUROPEAN economic and social committee and the committee of the regions. Critical Raw Materials Resilience: Charting a Path towards Greater Security and Sustainability. COM (2020) 474 final).

eurostat, 2022. Austrian Gross Inland Energy Consumption. https://ec.europa.eu/eurostat/databrowser/bookmark/17b29e74-0548-43b7-8404-9a1fe6f8567c?lang=en.

Faragò, M., Damgaard, A., Madsen, J.A., Andersen, J.K., Thornberg, D., Andersen, M.H., Rygaard, M., 2021. From wastewater treatment to water resource recovery: environmental and economic impacts of full-scale implementation. Water Res. 204, 117554 https://doi.org/10.1016/j.watres.2021.117554.

Gandon-Ros, G., Nuñez, S.S., Ortuño, N., Aracil, I., Gómez-Rico, M.F., Conesa, J.A., 2021. A win–win combination to inhibit persistent organic pollutant formation via the Co-incineration of polyvinyl chloride E-waste and sewage sludge. Polymers 13 (5), 835. https://doi.org/10.3390/polym13050835.

German Environment Agency, 2018. Evaluation und Minderung klimarelevanter Gase aus Abfallverbrennungsanlagen (Evaluation and abatement of climate relevant gases from waste incineration plants). German Environment Agency (Umweltbundesamt). https://www.umweltbundesamt.de/publikationen/evaluation-minderung-klimarelevanter-gase-aus.

Goedkoop, W., Heijungs, R., Huijbregts, M.A.J., de Schryver, A., Struijs, J., van Zelm, R., 2009. ReCiPe 2008, A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. Ministerie Van Volkshuisvesting.

Grafström, J., Aasma, S., 2021. Breaking circular economy barriers. J. Clean. Prod. 292, 126002 https://doi.org/10.1016/j.jclepro.2021.126002.

Haberkern, B., Maier, W., & Schneider, U. (2008). Steigerung der Energieeffizienz auf kommunalen Klaranlagen (Increasing energy efficiency at municipal wastewater treatment plants). (Report 205.26). Umweltbundesamt.

Haslinger, J., Lindtner, S., Krampe, J., 2016. Operating costs and energy demand of wastewater treatment plants in Austria: benchmarking results of the last 10 years. Water Sci. Technol. 74 (11), 2620–2626. https://doi.org/10.2166/wst.2016.390.

Hukari, S., Hermann, L., Nättorp, A., 2016. From wastewater to fertilisers — technical overview and critical review of European legislation governing phosphorus recycling. Sci. Total Environ. 542, 1127–1135. https://doi.org/10.1016/j.scitotenv.2015.09.064.

Jacobi, N., Haas, W., Wiedenhofer, D., Mayer, A., 2018. Providing an economy-wide monitoring framework for the circular economy in Austria: status quo and challenges. Resour. Conserv. Recycl. 137, 156–166. https://doi.org/10.1016/j.resconrec.2018.05.022.

Jama-Rodzeńska, A., Sowiński, J., Koziel, J.A., Białowiec, A., 2021. Phosphorus recovery from sewage sludge ash based on cradle-to-cradle approach—mini-review. Minerals 11 (9), 985. https://doi.org/10.3390/min11090985.

Jupp, A.R., Beijer, S., Narain, G.C., Schipper, W., Slootweg, J.C., 2021. Phosphorus recovery and recycling – closing the loop. Chem. Soc. Rev. 50 (1), 87–101. https://doi.org/10.1039/D0CS01150A.

Kratz, S., Vogel, C., Adam, C., 2019. Agronomic performance of P recycling fertilizers and methods to predict it: a review. Nutrient Cycl. Agroecosyst. 115, 1–39. https://doi.org/10.1007/s10705-019-10010-7.

Kraus, F., Seis, W., 2015. *P-REX Project: Sustainable sewage sludge management fostering phosphorus recovery and energy efficiency* (Deliverable 9.1: quantitative risk assessment of potential hazards for humans and the environment: quantification of potential hazards resulting from agricultural use of the manufactured fertilizers). Download. http://p-rex.eu/index.php?id=11.

Kraus, F., Zamzow, M., Conzelmann, L., Remy, C., Kleyböcker, A., Seis, W., Miehe, U., Hermann, L., Hermann, R., Kabbe, C., 2019a. Okobilanzieller Vergleich der P-Ruckgewinnung aus dem Abwasserstrom mit der Dungemittelproduktion aus Rohphosphaten unter Einbeziehung von Umweltfolgeschaden und deren Vermeidung (Ecological comparison of P recovery from wastewater with fertilizer production from raw phosphate rock under consideration of environmental secondary damages and their abatement) (p. 393). German Environment Agency.

Kraus, F., Zamzow, M., Hoffmann, C., Bessai, A.-K., Fischinger, S., Muskolus, A., Kabbe, C., 2019b. Einsatzmöglichkeiten für Nährstoffrezyklate im Ökolandbau (Possible applications for nutrient recyclates in organic farming; nurec4org). Kompetenzzentrum Wasser Berlin, Bioland Beratung; Institut für Agrar- und Stadtökologische Projekte; P-REX Environment. Financed through the Deutsche Bundesstiftung Umwelt, p. 393.

Krüger, O., Fattah, K.P., Adam, C., 2016. Phosphorus recovery from the wastewater stream—necessity and possibilities. Desalination Water Treat. 57 (33), 15619–15627. https://doi.org/10.1080/19443994.2015.1103315.

Law, K.P., Pagilla, K.R., 2019. Reclaimed phosphorus commodity reserve from water resource recovery facilities—a strategic regional concept towards phosphorus recovery. Resour. Conserv. Recycl. 150, 104429 https://doi.org/10.1016/j.resconrec.2019.104429.

LfU, 2015. Rückholbarkeit von Phosphor aus kommunalen Klärschlämmen. Abschlussbericht (Recoverability of phosphorus from municipal sewage sludge. Final report). Bayerisches Landesamt für Umwelt.

Mbah, R.E., Wasum, D.F., 2022. Russian-Ukraine 2022 War: a review of the economic impact of Russian-Ukraine crisis on the USA, UK, Canada, and Europe. Advances in Social Sciences Research Journal 9 (3), 144–153.

Mehr, J., Hellweg, S., 2018. Studie zum ökologischen Vergleich der Produktion von Phosphorsäure aus Klärschlammasche mittels Phos4life-Verfahren mit der Primärproduktion von P-Säure aus Rohphosphat (Study on the ecological comparison of the production of phosphoric acid from sewage sludge ash using the Phos4life process with the primary production of P-acid from rock phosphate. ETH Zürich 52 https://awel.zh.ch/internet/baudirektion/awel/de/abfall_rohstoffe_altlasten/rohstoffe/rohstoffe_aus_abfaellen/naehrstoffe/_jcr_content/contentPar/downloadlist_3/downloaditems/_kobilanz_bericht_zu.spooler.download.1527839087587.pdf/oekobilanz_bericht_phos4life.pdf.

Morf, L., 2018. *Schlussbericht: Verfahrenstechnische Marktanalyse für die Phosphor-Rückgewinnung aus dem Abwasserpfad* (*Final report: Process engineering market analysis for phosphorus recovery from the wastewater path*. Kanton Zürich, Baudirektion, Amt für Abfall, Wasser, Energie und Luft (AWEL) 100.

Nedelciu, C.-E., Ragnarsdóttir, K.V., Stjernquist, I., 2019. From waste to resource: a systems dynamics and stakeholder analysis of phosphorus recycling from municipal wastewater in Europe. Ambio 48 (7), 741–751. https://doi.org/10.1007/s13280-018-1097-9.

Oliva, J., Berhardt, A., Reisinger, H., Domenig, M., Kramer, H.J., 2009. *Klärschlamm - Materialien zur Abfallwirtschaft (Sewage sludge - Materials for waste management)* (REP-0221). Umweltbundesamt.

Önorm, S., 2024. 2005 01 01 - Stoffflussanalyse, ÖNORM S 2096 2005. https://shop.austrian-standards.at/action/de/public/details/187852/OENORM_S_2096-1_2005_01_01.

OpenStreetMap contributors, 2022. Planet dump. retrieved from. https://planet.osm.org. https://www.openstreetmap.org.

Orner, K.D., Smith, S., Nordahl, S., Chakrabarti, A., Breunig, H., Scown, C.D., Leverenz, H., Nelson, K.L., Horvath, A., 2022. Environmental and economic impacts of managing nutrients in digestate derived from sewage sludge and high-strength organic waste. Environmental Science & Technology 56 (23), 17256–17265. https://doi.org/10.1021/acs.est.2c04020.

Parravicini, V., Böhler, S., Emile, V.E., Amann, A., Svardal, K., Krampe, J., 2020. Ein nachhaltiges Konzept für die kommunale Abwasserreinigung der Zukunft (A sustainable concept for municipal wastewater treatment of the future). TU Wien, Institute for Water Quality and Resource Management, p. 136.

Regulation (EU), 2019. 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 and amending Regulations (EC) No 1069/2009 and (EC) No 1107/2009 and repealing Regulation (EC) No 2003/2003, PE/76/2018/REV/1. https://eur-lex.europa.eu/eli/reg/2019/1009/oj.

Remy, C., Jossa, P., 2015. Sustainable sewage sludge management fostering phosphorus recovery and energy efficiency. Deliverable D 9.2 Life Cycle Assessment of selected processes for P recovery from sewage sludge. Sludge Liquor or Ash. EU P-Rex Project.

Rufí-Salís, M., Brunnhofer, N., Petit-Boix, A., Gabarrell, X., Guisasola, A., Villalba, G., 2020. Can wastewater feed cities? Determining the feasibility and environmental burdens of struvite recovery and reuse for urban regions. Sci. Total Environ. 737, 139783 https://doi.org/10.1016/j.scitotenv.2020.139783.

Schlumberger, S., 2019. Phosphor-Mining aus Klärschlammasche. Abschlussbericht zu Handen der Baudirektion des Kantons Zürich (Phosphorus mining from sewage sludge ash. Stiftung Zentrum für nachhaltige Abfall- und Ressourcennutzung (ZAR) 24. *Final report for the Building Department of the Canton of Zurich)*.

Schoumans, O.F., Bouraoui, F., Kabbe, C., Oenema, O., van Dijk, K.C., 2015. Phosphorus management in Europe in a changing world. Ambio 44 (2), 180–192. https://doi.org/10.1007/s13280-014-0613-9.

Smol, M., Preisner, M., Bianchini, A., Rossi, J., Hermann, L., Schaaf, T., Kruopienė, J., Pamakštys, K., Klavins, M., Ozola-Davidane, R., Kalnina, D., Strade, E., Voronova, V., Pachel, K., Yang, X., Steenari, B.-M., Svanström, M., 2020. Strategies for sustainable and circular management of phosphorus in the Baltic Sea region: the holistic approach of the InPhos Project. Sustainability 12 (6), 2567. https://doi.org/10.3390/su12062567.

Stadt Wien, 2023. Wasserpreis (Water price). Query on 14.08. *2023 at 10:00*. wien.gv.at.

Statistik Austria, 2018. Verkehrstatistik 2017 (Transport Statistic 2017), Verlag Österreich GmbH, Vienna, Austria.

Statistik Austria, 2023a. Baupreisindex (Building cost index). Query on 14.08.2023 at 10:00. https://www.statistik.at/statistiken/volkswirtschaft-und-oeffentliche-finanzen/preise-und-preisindizes/grosshandelspreisindex.

Statistik Austria, 2023b. Großhandelspreisindex Technische Chemikalien (Wholesale price index for technical chemicals). Query on 14.08.2023 at 10:00. https://www.statistik.at/statistiken/volkswirtschaft-und-oeffentliche-finanzen/preise-und-preisindizes/grosshandelspreisindex.

Stenzel, F., Jung, R., Wiesgickl, S., Dexheimer, K., Eißing, M., Mundt, M., 2019. Arzneimittelrückstände in Rezyklaten der Phosphorrückgewinnung aus Klärschlämmen (Pharmaceutical residues in recyclates from phosphorus recovery from sewage sludge). TEXTE 31/2019 Fraunhofer-Institut für Umwelt-, Sicherheits-, und Energietechnik UMSICHT, p. 106. WESSLING GmbH, im Auftrag des Umweltbundesamtes Deutschland. https://www.umweltbundesamt.de/publikationen/arzneimittelrueckstaende-in-rezyklaten-der.

Stoiber, T., Evans, S., Naidenko, O.V., 2020. Disposal of products and materials containing per- and polyfluoroalkyl substances (PFAS): a cyclical problem. Chemosphere 260, 127659. https://doi.org/10.1016/j.chemosphere.2020.127659.

VDI, 2015. VDI Standard 4600 - Cumulative Energy Demand. VDI, Berlin, Germany.

Verordnung des Bundesministers für Land- und Forstwirtschaft, 2001. Umwelt und Wasserwirtschaft über Qualitätsanforderungen an Komposte aus Abfällen (Kompostverordnung; Compost Ordinance) StF: BGBl. II Nr. 292/2001.
Verordnung des Bundesministers fur Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft, mit der Bestimmungen zur Durchfuhrung des Dungemittelgesetzes 1994 erlassen werden (Dungemittelverordnung 2004; Fertilizer Ordinance 2004). StF:BGBl. II Nr. 100/2004. Version of 17.7.2014: BGBl. II Nr. 181/2014, (2004).
Winchell, L.J., Ross, J.J., Wells, M.J.M., Fonoll, X., Norton Jr, J.W., Bell, K.Y., 2021. Per- and polyfluoroalkyl substances thermal destruction at water resource recovery facilities: a state of the science review. Water Environ. Res. 93 (6), 826–843. https://doi.org/10.1002/wer.1483.
WKO, 2023. Transportkostenindex. Indexentwicklung der Kostenstruktur (transport cost index). Query on 14.08.2023 at 10:00. https://www.wko.at/branchen/transport-verkehr/gueterbefoerderungsgewerbe/Transportkostenindex.html.
Yang, L., Wang, X.-C., Dai, M., Chen, B., Qiao, Y., Deng, H., Zhang, D., Zhang, Y., Villas Bôas de Almeida, C.M., Chiu, A.S.F., Klemeš, J.J., Wang, Y., 2021. Shifting from fossil-based economy to bio-based economy: status quo, challenges, and prospects. Energy 228, 120533. https://doi.org/10.1016/j.energy.2021.120533.
Yazdani, M., Gonzalez, E.D.R.S., Chatterjee, P., 2019. A multi-criteria decision-making framework for agriculture supply chain risk management under a circular economy context. Manag. Decis. 59 (8), 1801–1826. https://doi.org/10.1108/MD-10-2018-1088.
Yu, B., Xiao, X., Wang, J., Hong, M., Deng, C., Li, Y.-Y., Liu, J., 2021. Enhancing phosphorus recovery from sewage sludge using anaerobic-based processes: current status and perspectives. Bioresour. Technol. 341, 125899 https://doi.org/10.1016/j.biortech.2021.125899.
Załuszniewska, A., Nogalska, A., 2022. The effect of meat and bone meal (MBM) on phosphorus (P) content and uptake by crops, and soil available P balance in a six-year field experiment. Sustainability 14 (5), 2855.
Zhou, K., Remy, C., Kabbe, C., Barjenbruch, M., 2019. Comparative environmental life cycle assessment of phosphorus recovery with different generations of the AirPrex® systems. Int. J. Environ. Sci. Technol. 16 (5), 2427–2440. https://doi.org/10.1007/s13762-018-1881-x.
Zoboli, O., Zessner, M., Rechberger, H., 2016. Supporting phosphorus management in Austria: potential, priorities and limitations. Sci. Total Environ. 565, 313–323. https://doi.org/10.1016/j.scitotenv.2016.04.171.