Science of the Total Environment 846 (2022) 157401

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Systematic data-driven exploration of Austrian wastewater and sludge treatment - implications for phosphorus governance, costs and environment

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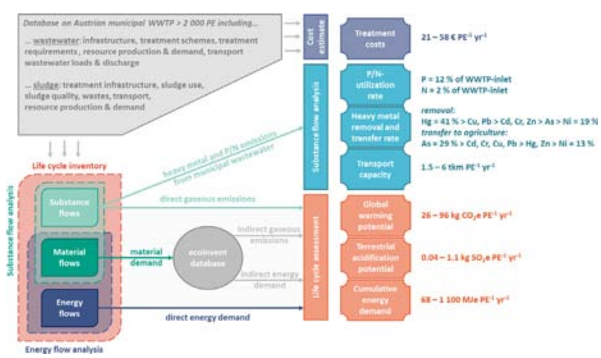
HIGHLIGHTS

- Only ca. 21 % of WWTP-inlet P and 3 % of N are currently directed to agriculture
- The transfer of heavy metals from the WWTP inlet to a final sink lies at about 32 %
- Wastewater and sludge treatment constitute about 0.3 % of Austria's total CED
- GWP emissions are sensitive to N_2O in off-gas from sludge incineration
- Landscaping is correlated with the highest transport volume (6 tkm PE^{-1} yr^{-1})

GRAPHICAL ABSTRACT



ARTICLE INFO

Editor: Huu Hao Ngo

Keywords:
Material flow analysis
Life cycle assessment
Cost estimation
Phosphorus
Wastewater
Sewage sludge

ABSTRACT

Within the new policy framework shaped by the EU Green Deal and the Circular Economy Action Plans, the field of wastewater and sludge treatment in Europe is subject to high expectations and new challenges related to mitigation of greenhouse gas emissions, micropollutant removal and resource recovery. With respect to phosphorus recovery, several technologies and processes have been thoroughly investigated. Nevertheless, a systemic and detailed understanding of the existing infrastructure and of the related environmental and economic implications is missing. Such basis is essential to avoid unwanted consequences in designing new strategies, given the long lifespan of any infrastructural change. This study couples a newly collected and highly detailed database for all wastewater treatment plants in Austria bigger than 2000 population equivalent with a combination of analyses, namely Substance Flow Analysis with focus on nutrient and metal distribution in different environmental and anthropogenic compartments, Energy Flow Analysis, Life Cycle Assessment and cost estimation. The case study of Austria is of special interest, given its highly autonomous administration in federal states and its contrasting traits, ranging from flat metropolitan areas like Vienna to low-populated alpine areas. The significant impact of electricity demand of wastewater treatment on the overall Cumulative Energy Demand (CED) shows the importance of optimization measures. Further, the current system of wastewater and sludge disposal have a low efficiency in recovering nutrients and in directing pollutants as heavy metals into final sinks. Sludge composting with subsequent use in landscaping does not only show an unfavorable environmental balance, but it is the only relevant route leading to additional CED and Global Warming Potential emissions and to the highest transport volume. Altogether, the outcomes of this study provide a sound basis to further develop national strategies for resource recovery aimed to optimize trade-offs between different economic and environmental objectives.

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http://dx.doi.org/10.1016/j.scitotenv.2022.157401
Received 25 May 2022; Received in revised form 11 July 2022; Accepted 12 July 2022
Available online 22 July 2022
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1. Introduction

Since the discovery of the activated sludge process in 1913 (Ardern and Lockett, 1914), wastewater treatment plants (WWTPs) have been improving their performance and adapting regularly to mounting pressures from problematic substances. The establishment of the EU Green Deal (EU, 2022b) and the Circular Economy Action Plans of 2015 and 2020 (EU, 2022a) brought new challenges and opportunities for WWTPs. Next to the ongoing removal of contaminants, the implementation of recovery and/or reuse of wastewater resources (energy, carbon, nutrients, …) will require firm action from legislators and WWTP-providers in the coming years. We expect that three major challenges have to be addressed for a sustainable development of wastewater treatment infrastructure:

First, wastewater and sludge treatment is a highly complex system interacting with a multitude of environmental compartments. Depending on the applied treatment processes, it can have impacts on the atmosphere (Prata et al., 2021; Pahunang et al., 2021; Valkova et al., 2021), on surface water resources through effluents (Dickenson et al., 2011; Agus et al., 2012; Suess et al., 2020) or on plants, soil and groundwater via soil-application of sludge (Singh and Agrawal, 2008). Second, the current status of partially inadequate, undetailed and unreliable data on wastewater treatment and sewage sludge management in EU member states (Bianchini et al., 2016) proves challenging in terms of conducting in-depth research of this field. And third, WWTP-infrastructure has a long lifespan, with the potential for technological lock-in. This has been uncovered in the case of wastewater treatment decentralization, where an existing centralized collection and treatment system with long remaining lifespans will drastically hinder the economic efficiency of a technological change towards better source separation (Garrido-Baserba et al., 2018).

We argue that economic efficiency and environmental success of future regulations towards the EU Green Deal can therefore only be guaranteed if detailed analyses are performed on the existing infrastructure and treatment concepts beforehand. In this light, the case study of Austria is of special interest, as it is a country with a highly autonomous administration in the nine respective federal states. Further, Austria has an extremely diverse population dynamic, with large flat metropolitan areas like Vienna in contrast to low-populated mountain areas. This resulted in varying requirements, sizes and options for wastewater and sludge treatment and created a non-uniform stock of treatment infrastructure (Überreiter et al., 2018), linked to different environmental and economic costs.

While environmental and economic assessments of wastewater and sludge treatment in different countries have been performed manifold in the past (see e.g. Lederer and Rechberger (2010); Niero et al. (2014); Fang et al. (2016); Polruang et al. (2018)), models are often based on one or few reference WWTPs with set and linear characteristics, sometimes low level of detail (Lam et al., 2020), little attention to uniformity of local conditions (Bradford-Hartke et al., 2015; Teoh and Li, 2020), spatial distribution of WWTPs, plant sizes and other infrastructure. This clear differentiation will, however, become evermore important as sludge treatment, recovery, and subsequent use of wastewater resources can happen off-site with a potential for long transport (Trimmer and Guest, 2018) and in even larger and more centralized plants with wide-reaching socio-technical impacts (Jedelhauser and Binder, 2018).

As a basis for analysing the potential of circular strategies for phosphorus (P) recovery from wastewater, and their impacts specifically, we established an in-depth model of the current Austrian wastewater treatment system (excluding wastewater collection). A thorough survey and spatial evaluation of every WWTP bigger than 2000 population equivalents (PE) enabled a detailed breakdown of the current system. Related environmental and economic costs were established via substance flow analysis (SFA) and energy flow analysis (EFA), life cycle assessment (LCA) and cost estimation. Due to the broad review of a multitude of different wastewater and sludge treatment systems, we believe that the variability seen in these impacts per PE can inform legislators and researchers alike. This case-study further displays the importance of holistic evaluations based on detailed databases and existing infrastructure. While this evaluation was trimmed to address P recovery specifically, results presented are seen as an informative basis for other reforms towards sustainable wastewater treatment as well.

2. Study area

Around 98 % of Austria's municipal wastewater (based on COD: Chemical Oxygen Demand) is monitored regularly and treated in 635 WWTPs (Fig. 1) bigger than 2000 PE (Überreiter et al., 2018). Plants mainly use advanced treatment via the activated sludge process and nutrient (N and P) removal (Amann et al., 2021). The average removal rate for P is 90 %, which is primarily achieved via chemical precipitation using iron-based agents. A small share of plants are also equipped with anaerobic tanks for enhanced biological P removal, but such process are always complemented by chemical precipitation. Loads to WWTPs and WWTP-sizes are rather non-homogeneous across Austrian federal states (Table 1). While some states (B, ST) treat more than 50 % of their PE with many WWTPs of a design capacity smaller than 50,000 PE, others treat the majority in a few plants with a capacity bigger than 50,000 PE. Similarly, the ratio of $PE_{treated}$ per inhabitant varies vastly across federal states; a consequence of regional tourism and industry.

Sludge treatment occurs in and across federal states and national borders via drying (granulation), composting, industrial co-incineration (e.g. paper industry), mono-incineration, municipal waste incineration, cement works and in few cases through pyrolysis (Amann et al., 2021). Locations of external treatment sites are shown in Fig. 1. At the time of the study, no technological recovery of P from sewage sludge was taking place.

3. Materials and method

As a basis for this study, we performed an in-depth survey on Austrian WWTPs and on sludge treatment and infrastructure (statistical results and response rates in Amann et al., 2021). Based on this survey, a database on Austrian WWTPs was established. The database includes information on wastewater treatment processes, P removal as well as sludge treatment, disposal and quality. We then conducted a SFA and EFA as the foundation for further economic and ecological evaluations, accounting for flows of goods, substances and energy in our defined system. Finally, analysis of the chosen assessment criteria (Table 2) was performed and results were edited for better insight and comparison to other studies.

3.1. System definition, criteria and functional units

For this analysis we included the treatment of Austrian municipal wastewater in WWTPs with a minimum design capacity of 2000 PE, and the drying, treatment and disposal or application of the resulting sewage sludge in the year 2016 (Fig. S 1). Wastewater collection was excluded, as 98 % of Austrian citizens are connected to combined and separate sewers (Überreiter et al., 2018), a strategy that will likely not change in the next decades. The possible intermediary or permanent sinks for goods and substances in the system are the material stock, agriculture, landscaping, groundwater, atmosphere, surface waters, landfills or food production (only considered for nutrients).

The following criteria were chosen as relevant indicators for the economic and environmental assessment of the system: The criteria P-utilization rate was selected to give an overview on current P use. This criteria takes into account that not all P applied via sludge is plant-available and might not taken up by plants in the short-term (Kratz et al., 2019). Deeper analyses of other major macro-nutrients, like N and K, were neglected due to a limited importance of sludge-N and -K for agriculture (Tanzer et al., 2018).

Heavy metal flows from wastewater are evaluated through the heavy metal removal rate (sink: landfill) and the rate of heavy metals directed to agricultural soils, both rates based on WWTP influent loads. Landfills are permanent sinks, are safe and allow the permanent removal of metals

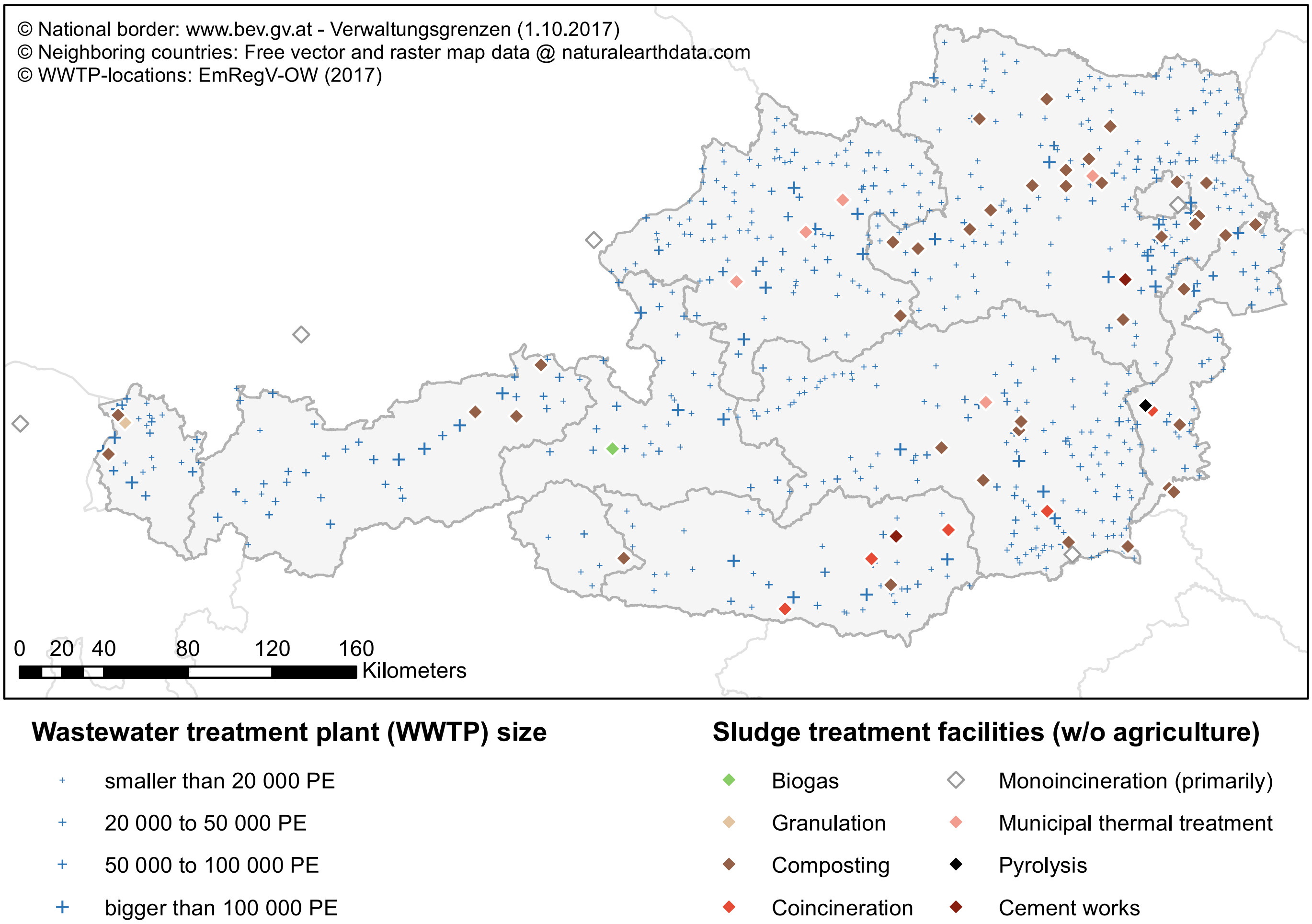

Fig. 1. Map of Austrian WWTPs locations including sewage sludge treatment sites (Data: EmRegV-OW, 2017).

from the environment. Agricultural soils are, however, only intermediary sinks and have the potential to accumulate heavy metals and harm the environment. Organic pollutants were excluded from the analysis due to (i) a limited availability of concentration data, (ii) a lack of knowledge on organic pollutant transport and transformation and (iii) the fact that increasing sludge incineration will automatically reduce organic sludge pollutants via the pathway of thermal destruction.

Freight transport in tkm is a transport indicator for wastewater treatment and considers all freight transport of goods or wastes in the system. It was chosen to observe the relevance of changes in sludge management on transport distances and the overall global warming potential.

LCA is used to evaluate the mid-point indicators global warming potential (GWP), terrestrial acidification potential (TAP) and cumulative energy demand (CED). We excluded other commonly used mid-point criteria like eutrophication and human or aquatic toxicity potential due to, first, the fact that they cannot incorporate actual impacts, but effects are comparatively dependent on local conditions (Hepp et al., 2022) and are therefore less informative. Second, freshwater eutrophication or aquatic toxicity from wastewater treatment are mainly related to the quality of treated effluents, which are or could be efficiently regulated through discharge limits. Third, avoided fertilizer effects on eutrophication, as considered in some LCA studies (Lam et al., 2020), are attributed with high uncertainty (Schmidt Rivera et al., 2017).

Due to their nature, effects of gaseous emissions are globally relevant, hence, GWP (expressed as CO_2 equivalents (CO_2e)) and TAP (expressed as SO_2 equivalents (SO_2e)) were chosen to analyse CO_2, CO, CH_4, N_2O, SO_2, NH_3 and NO_x emissions. GWP Energy demand for different treatment chains is highly varied, therefore CED_{total} was chosen as an indicator to analyse the energetic efficiency of the systems. CED is further divided into fossil/nuclear ($CED_{f,n}$) and renewable energy (CED_r).

Table 1
Share of load in % (based on PE_{120} = 120 g COD d^{-1}) treated by specific WWTP size classes as well as ratio of PE to inhabitants in different Austrian federal states.

Federal state	smaller than 20,000 PE	20,000 to 50,000 PE	50,000 to 100,000 PE	bigger than 100,000 PE	Rate of PE per inhabitant
Burgenland (B)	20	46	8.5	26	1.5
Styria (ST)	26	27	12	35	1.3
Tyrol (T)	6.2	33	20	40	1.8
Lower Austria (N)	25	25	5.2	45	1.4
Upper Austria (O)	15	17	13	55	1.4
Salzburg (SB)	7.5	25	11	56	1.8
Carinthia (K)	9.7	15	10	65	1.5
Vorarlberg (V)	3.4	8.8	15	72	2.3
Vienna (W)	0	0	0	100	1.8

Table 2
Overview on chosen economic and environmental criteria for a systematic assessment.

Criteria	Unit	Method
P utilization rate: share of plant-available P in agriculture on WWTP inlet P	%	SFA
Heavy metal removal rate: share of heavy metals in landfill (endpoint) on WWTP inlet concentrations	%	SFA
Heavy metals in agricultural soils: share of heavy metals in agriculture (endpoint) on WWTP inlet concentrations	%	SFA
Freight transport (mass-distance) in tonne-kilometre	tkm	SFA
Global warming potential 100a (GWP) IPCC method 2013 with 1 kg CO_2e = 1 kg CO_2 or 2.491/4.0624 (non-fossil/fossil) kg CO or 29.7 kg CH_4 or 264.8 kg N_2O	kg CO_2e	LCA
Terrestrial acidification potential 100a (TAP) ReCiPe midpoint with 1 kg SO_2e = 1 kg SO_2 or 0.56 kg NO_x or 2.45 kg NH_3	kg SO_2e	LCA
Total cumulative energy demand from fossil and nuclear resources (CED_{total})	MJe	LCA
Non-renewable cumulative energy demand from fossil and nuclear resources ($CED_{f,\ n}$)	MJe	LCA
Net cost of wastewater treatment and sludge disposal	EUR	CBA

The last indicator for analysis is the net cost of wastewater treatment and sludge disposal, to contrast costs and benefits (e.g. goods production) of different treatment systems.

Results were related to both the treatment of 1 population equivalent per year (1 PE^{-1} yr^{-1}), as well as to the treatment of wastewater of one inhabitant per year (1 inh^{-1} yr^{-1}). Since the ratio of $PE_{treated}$ per inhabitant can vary due to local conditions (Table 1), for better comparability, results are presented only per PE and year. Mean results are given as the sum of total impacts divided by the total amount of $PE_{treated}$, hence an Austrian average. Variation of results is represented by the $p_{0.05}$ to $p_{0.09}$ quantiles of impacts per $PE_{treated}$ of individual WWTPs. Values are normalized to total impacts of one Austrian inhabitant, to give an indication for societal relevance of these impacts. At last, a sensitivity analysis is performed for a selection of parameters.

3.2. Database of WWTPs and sludge disposal

To set up the database on WWTPs for further analysis, the following information was collected: WWTP size, location and pollutant loads in influent and effluent; primary clarification (yes/no); chemical or enhanced biological P removal and flocculating agent type and demand; type of sludge stabilization; sludge production; sludge dewatering; sludge quality (total solids, ignition loss, nutrient and heavy metal content); sludge treatment, transport, distances and disposal (sites). Due to to high response rates of over 70 % for most parameters, the supplied data could then be analysed, and missing data for WWTPs with no information was imputed from the survey. A summary on how proxy values for missing data were determined is given in Table S 1 (associated with Table S 2) and an overview on final sludge disposal types is presented in Fig. S 2.

3.3. Substance and energy flow analysis

The final WWTP database was then used as a foundation for building the SFA model according to *ÖNORM S 2096* (Austrian Standards, 2005). SFA is a well-established method that enables the determination of all goods and substance flows into, within, or out of, a temporally and spatially defined system (Brunner and Rechberger, 2016). All flows of required resources, sludge, water and produced goods were included to establish the mass balances. 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 (DMVO, 2004; KOMP-VO, 2001; Fertilizer Product Regulation, 2019). The partitioning of substance flows from one process input to one or multiple outputs is evaluated using transfer coefficients (Table S 3) taken from Boesch et al. (2009); Egle et al. (2016); Diepold (2020); Suess et al. (2020). For all thermal sludge disposal processes (incineration), we further conducted an EFA (Suh, 2005) to contrast energy demand for auto-thermal incineration and heat and electricity production during the process.

3.4. Life cycle assessment

Direct/indirect emissions from and energy demand (e.g. from resource production) for the treatment of Austrian wastewater are evaluated through the method of LCA (ISO standard 14040, 2006). Whilst considering the system boundaries, this method supports the analysis of environmental impacts during, preceding or succeeding a treatment step. Though the input data is different for every WWTP, it was necessary to define reference treatment processes and sub-processes to determine the types of resources and emissions (not necessarily amounts) considered. Two processes of wastewater treatment and five processes, as well as seven sub-processes of sludge treatment were distinguished (Table S 4). Reference processes were based on prior analysis of treatment configurations in Austria, of which the most common were chosen as a reference.

Based on the WWTP-database, SFA and the reference processes, we then established a life cycle inventory of all related material and energy flows (calculation and sources see Tables S 5, S 7, S 8) and direct emissions (calculation and sources see Tables S 6, S 9, S 10 and Fig. S 6).

Indirect gaseous emissions and energy demand are estimated by correlating material flows from the inventory to emissions and energy demand per unit of product as defined in the ecoinvent database (ecoinvent, 2020). The chosen reference datasets per material are presented in Table S 11. Direct and indirect emissions are then summed up by compound for final results and are related to its respective LCA criteria via characterisation factors. These are $CO_2 = 1$, $CH_4 = 29.7$, $N_2O = 264.8$ for GWP (IPCC, 2013) and $SO_2 = 1$, $NH_3 = 2.45$ and $NO_x = 0.56$ for TAP (Goedkoop et al., 2009). Results are then normalized to the functional unit.

3.5. Cost estimation

Costs for the treatment of wastewater and sludge were derived based on two aspects: WWTP size and sludge disposal scheme. First, operation costs (excluding sludge disposal) were determined using WWTP sizes and according to Austrian benchmark data (Lindtner, 2018, Table S 12). Next, sludge disposal costs for each WWTP were calculated via original or mean survey data (Amann et al., 2021, Fig. 8). Capital and administrative costs were estimated from analyses based on national funding data (Assmann et al., 2019), resulting in 22 % and 12 % of total costs, respectively.

4. Results

4.1. Phosphorus and nitrogen utilization rate

Around 21 % of WWTP-inlet P and 3 % of N are directed to agriculture. Nitrogen mostly leaves the system to the atmosphere (71 %; Fig. 2) through denitrification to N_2 at the WWTP or over the effluent, whereas the majority of P is either landfilled (44 %) or used in landscaping (22 %) with little need for its fertilizing effect. Accounting for the fertilizing efficiency of these nutrients in sewage sludge (Table S 5), only around 12 % of P and 2 % of N are applied in a form that is plant-available short-term. P however especially, might be transformed to plant-available forms in subsequent agricultural years and taken up later (Ibrikci et al., 2005).

WWTPs with only agricultural sewage application achieve P utilization rates between 70 and 79 %, the maximum achieved for N utilization is 10 % of the WWTP inlet N.

4.2. Heavy metal removal and transfer to agriculture

Heavy metals are distributed unequally between the analysed compartments, due to their varying environmental behavior (Bradl, 2005 Fig. 2). The average transfer of heavy metals from the WWTP inlet to an inert

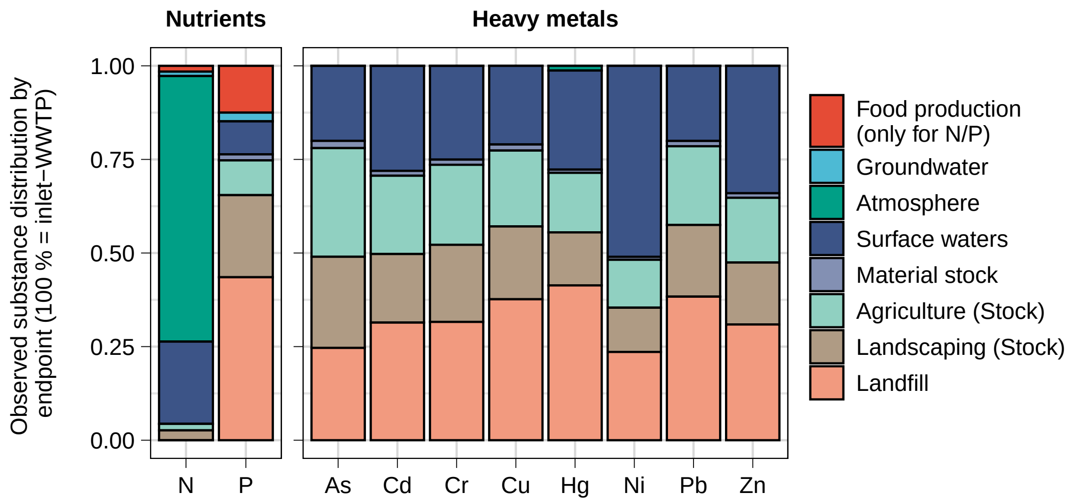

Fig. 2. Distribution of nutrients (N, P) and heavy metals from WWTP influent to natural or built-environment endpoints. Food production only accounted for in nutrient SFA.

form (i.e. landfilled) – and thereby removal from environmental compartments – lies around 32 %, with lowest values for Ni (24 %) and highest for Hg (41 %).

While heavy metal concentrations in wastewater and sludge have been decreasing for decades (KEK-1.2 Statistik, 2015), heavy metals from municipal wastewater are still relevant contributors to total heavy metal loads to agriculture, with shares between 1 and 6 % of the total load (Table 3; based on evaluations in Amann et al., 2016). Share and loads are naturally bigger for soils that are only fertilized with sewage sludge. Currently, on average around 20 % of the total heavy metal load from wastewater is transferred to agricultural soils, with shares from 13 % (Ni) to 29 % (As). If all sludge heavy metals were applied to soil, the subsequent total load would be limited to a further increase of maximum 9 % (Cu, Hg, Pb).

4.3. Global warming potential

The GWP for the current Austrian status quo of wastewater and sludge treatment is 38 kg CO_2e PE^{-1} yr^{-1}, with individual WWTPs ranging from 26 up to 96 kg CO_2e PE^{-1} yr^{-1}. This constitutes about 0.7 % of Austria's total GWP (= 9100 kg CO_2e inh^{-1} yr^{-1}; Anderl et al., 2021a). Though comparison to other studies proves difficult due to varying system boundaries, functional units and local conditions (Corominas et al., 2013a), results fall in the large range of literature values from 15 to 138 kg CO_2e PE^{-1} yr^{-1} (Corominas et al., 2013b; Risch et al., 2015; Lorenzo-Toja et al., 2016). Overall, electricity demand, indirect emissions from resource production, and direct atmospheric emissions are the biggest sources for GWP (Fig. 3).

Wastewater treatment is the dominant contributor to GWP with mean 39 kg CO_2e PE^{-1} yr^{-1} with aerobic (WWTP aerob. stabil.) and 32 kg with anaerobic (WWTP anaerob. stabil.) sludge treatment. Increased direct N_2O and CH_4 emissions with anaerobic sludge treatment (Valkova et al., 2021; Tauber et al., 2019) can in some cases offset the general advantage from energy production over aerobic treatment. Although direct CO_2 emissions from sludge are reduced via carbon sequestration (8 % of TOC; Lampert et al., 2011), use of composted sludge in landscaping (LSCP) is generally attributed with the highest GWP, as neither fertilizer nor energy is substituted by this treatment. Through application directly or as compost in agriculture (AGR), net-GWP can be reduced, as the fertilizing function of sludge is better utilized.

Due to a high thermal efficiency, treatment in waste and co-incinerators (THERM) is associated with the lowest GWP (mean 2.5 kg CO_2e PE^{-1} yr^{-1}). Mono-incineration (MONO) drops behind typical waste incinerators since sludge drying is needed for auto-thermal combustion, and the increased recycling value of mono-incinerated ash (i.e. no dilution; Schnell et al., 2020) is neither considered nor can it be accounted for, as long as recovery of resources is not implemented. Nevertheless, GWP emissions per PE and year express a large range even with similar sludge treatment. This coincides well with the review of Lam et al. (2020), showing that, dependent on the study design and assumptions, agricultural application of sludge can have a higher or lower impact on GWP than thermal treatment of sludge.

Table 3
Share of and maximum further increase of heavy metal inputs from agricultural application of sludge on total loads of heavy metals to Austrian agricultural soils.

	As	Cd	Cr	Cu	Hg	Ni	Pb	Zn
Average input in g ha^{-1} yr^{-1}	0.2	0.04	2.0	10	0.03	1.3	1.7	34
Share on total input (from Table S 14)	6 %	1 %	5 %	5 %	3 %	4 %	5 %	4 %
maximum further increase of total loads to agriculture	+5 %	+2 %	+8 %	+9 %	+9 %	+6 %	+9 %	+7 %

4.4. Terrestrial acidification potential

The TAP for the current Austrian status quo of wastewater and sludge treatment is 0.14 kg SO_2e PE^{-1} yr^{-1}, with individual WWTPs ranging from 0.04 to 1.1 kg SO_2e PE^{-1} yr^{-1}. This constitutes about 0.8 % of Austria's total TAP (= 28 kg SO_2e inh^{-1} yr^{-1}; Anderl et al., 2021b) and is in the range of literature (e.g. 0.27 kg SO_2e PE^{-1} yr^{-1}; Risch et al., 2015).

In contrast to GWP, TAP is much more influenced by the type of sludge treatment, and less from wastewater treatment itself (Fig. S 3). Largest contributors to total TAP are direct NH_3 emissions from application of dewatered sludge or compost, with higher emissions attributed to dewatered sludge (EEA, 2016). Thermal treatment in co- or mono-incineration plants is associated with only 1/10 of TAP compared to disposal of sludge in agriculture and landscaping, with direct SO_2e emissions playing only a small role.

4.5. Cumulative energy demand

The average total CED for the current Austrian status quo of wastewater and sludge treatment is 260 MJe PE^{-1} yr^{-1}, with individual WWTPs ranging from 68 to 1100 MJe PE^{-1} yr^{-1}. This constitutes about 0.3 % of Austria's total CED (= 160 GJ inh^{-1} yr^{-1}; eurostat, 2022). Results are supported by values extracted from relevant literature of around 60 to 340 MJe PE^{-1} yr^{-1} (Remy et al., 2014; Gandiglio et al., 2017).

CED is largely dominated by electricity demand for wastewater treatment (Fig. 4). Composting and use of sludge compost in landscaping is the only treatment associated with a net demand in energy. Thermal treatment generally produces a net-reduction in energy demand, however, this

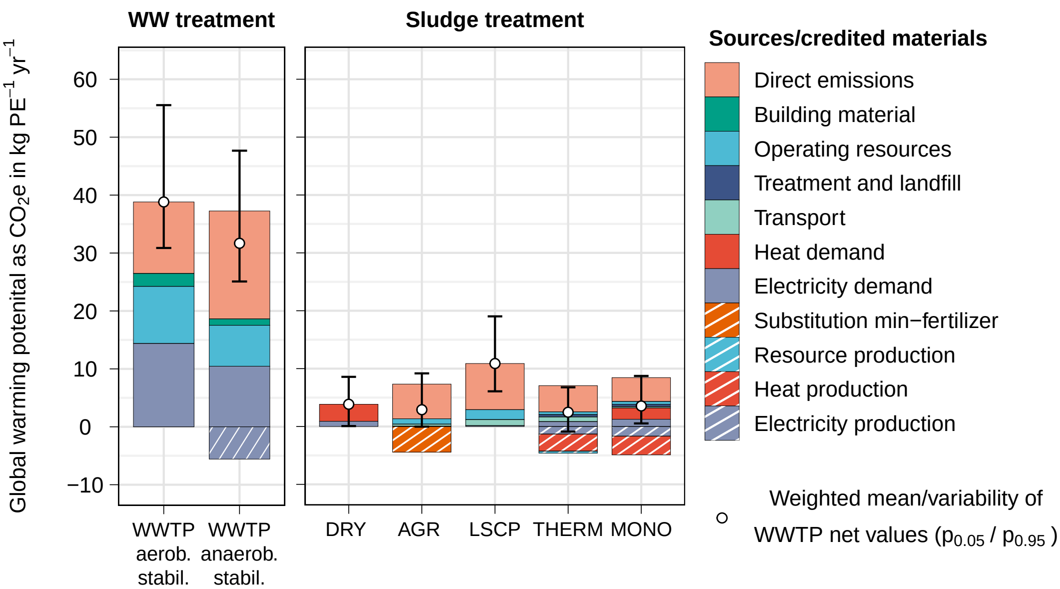

Fig. 3. Composition of GWP as kg CO_2e PE^{-1} yr^{-1} for different wastewater and sludge treatment technologies, as well as mean and variability ($p_{0.05}$ to $p_{0.95}$) of net-GWP of individual WWTPs. DRY: sludge drying, AGR: Agriculture, LSCP: Landscaping, THERM: Thermal treatment, MONO: Mono-incineration (Fig. S 1).

depends largely on the sludge heating value and transport distance to the next incineration plant. Since mineral N-fertilizer production is highly energy intensive (Kliopova et al., 2016), agricultural application of sludge can also have a net benefit through the substitution of primary N-fertilizer.

On average, 76 % of CED are in fossil and nuclear energy (Fig. S 4) with values between 43 and 840 MJe PE^{-1} yr^{-1}. Only 24 % of the system is currently run on renewable energy. Future changes in this share towards renewable CED are expected through the ongoing change of electricity production to renewable sources.

4.6. Transport

Around 60 million tkm yr^{-1} are covered by Austrian wastewater and sludge treatment, but with 0.1 % it makes up only a small share of Austria's total freight transport (= 55 billion tkm yr^{-1}; Statistik Austria, 2021). With 72 %, sludge transport is the biggest contributor to the total transport capacity, followed by chemical transport (20 %) and waste disposal (8 %).

As a result of some Austrian states transporting sludge comparatively far to external composting plants (Amann et al., 2021), composting of sludge and spreading in landscaping is correlated with the highest transport capacity per PE (6 tkm PE^{-1} yr^{-1}) as well as in total share (34 %; Table S 13). Only incineration in waste treatment plants is associated with similar values of average 5 tkm PE^{-1} yr^{-1}. Influenced by the fact that the biggest Austrian mono-incineration plant is situated right next to its primary sewage sludge source, mean transport capacity is only 1.5 tkm PE^{-1} yr^{-1} in comparison.

4.7. Costs

Estimated costs of Austrian wastewater treatment amounts to an average of 32 EUR PE^{-1} yr^{-1}. Costs are ranging from 21 to a maximum of 58

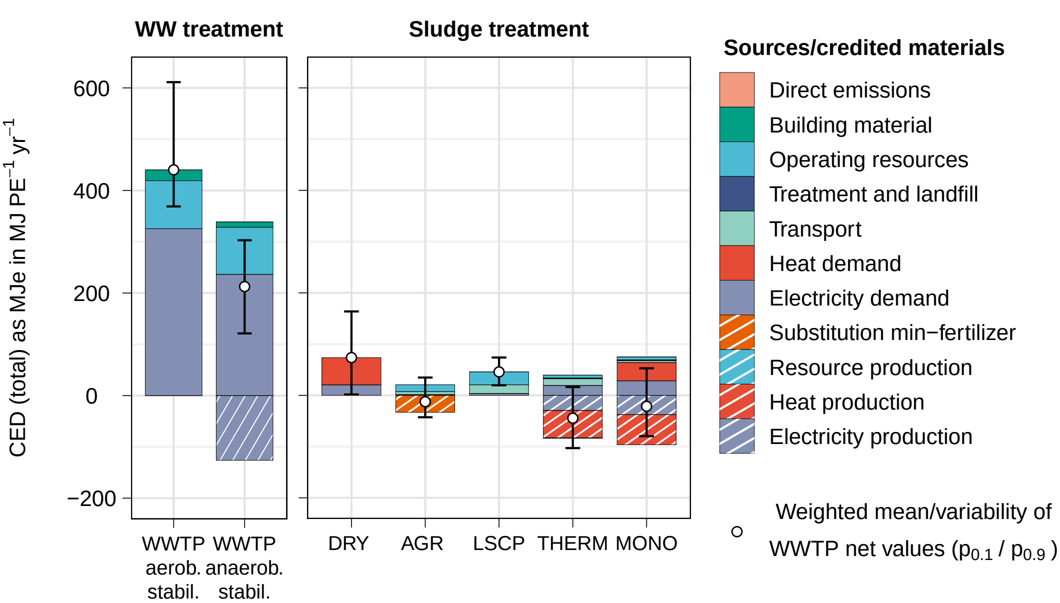

Fig. 4. Composition of total CED as MJe PE^{-1} yr^{-1} for different wastewater and sludge treatment technologies, as well as mean and variability ($p_{0.05}$ to $p_{0.95}$) of total net-CED of individual WWTPs. DRY: sludge drying, AGR: Agriculture, LSCP: Landscaping, THERM: Thermal treatment, MONO: Mono-incineration (Fig. S 1).

EUR PE^{-1} yr^{-1}, derived from a strong economy-of-scale effect in wastewater treatment (Friedler and Pisanty, 2006; Haslinger et al., 2016).

The biggest operational costs are related to personnel (= 26 of total cost, Haslinger et al., 2016). Other than that, capital cost make up the biggest share of total costs (Fig. 5). Costs for sludge management and disposal contribute from only 0.3 up to 21 % of total cost, with shares generally rising by increasing WWTP COD-load (Fig. S 5).

Differences in estimated mean costs per federal state are mostly influenced by their related WWTP size structure (see also Table 1) and in part by differences in sludge treatment (Fig. S 2).

4.8. Sensitivity

Due to the model-complexity, estimating the related uncertainty of the provided results would have proven too taxing for this analysis. Instead, a sensitivity analysis was performed for a number of selected parameters and assumptions.

Estimations of direct emissions from wastewater treatment have been systematically improved over recent years. For this analysis constant factors for N_2O and CH_4 emissions per PE and year were used. Valkova et al. (2021) found a variation in N_2O emissions of 0.1 to 80 g PE^{-1} yr^{-1}. If minimum and maximum values are applied in the sensitivity analysis, direct CO_2e emissions from wastewater treatment are either reduced by 50 % or increased by a factor of two, respectively. As suggested by Valkova et al. (2021) this high uncertainty can be reduced by relating N_2O emissions to the total nitrogen removal rate of WWTPs. Considering the variation between CH_4 emissions from 11 to 390 g PE^{-1} yr^{-1} (Schaum et al., 2015, 2016), total direct CO_2e emissions from wastewater treatment would be reduced by 27 % and increased by 23 %, respectively. The total system therefore shows a higher sensitivity to the variation of N_2O than CH_4 emissions.

According to our analysis, nitrous oxide emissions account for approximately 70 % of the GWP of mono-incineration. The basic assumption for emissions was set at 150 mg m^{-3} of flue gas. Nevertheless, direct measurements of German mono-incineration plants (German Environment Agency, 2018) have shown that values up to 800 mg m^{-3} are possible under unfavorable conditions. On the other hand, abatement measures could reduce emissions to 60 mg m^{-3} (Kraus et al., 2019). Assuming an average value of 300 mg m^{-3}, total GWP of the system would only increase by 2 %. Net-GWP for mono-incineration only would however increase by 0.9 up to 4.1 kg CO_2e PE^{-1} yr^{-1} % (mean = 73 %). An improved reduction of N_2O emissions down to 60 mg m^{-3} would in turn decrease net-GWP for mono-incineration by −0.5 to −2.5 kg CO_2e PE^{-1} yr^{-1} (mean = −44 %).

Another debate concentrates around the share of fossil carbon in conventional WWTPs (Ding et al., 2021). In EU climate reporting (e.g. Anderl et al., 2019), sludge-carbon is usually attributed with zero-carbon footprint, as it is argued that sources are 100 % of biogenic origin. In this study, we assumed in accordance with Parravicini et al. (2020) based on Law et al. (2013) that 10 % of carbon emissions are based on fossil resources. In comparison, total GWP of wastewater and sludge treatment would be reduced by 19 % if no fossil background was attributed.

Finally, the impact of the chosen energetic overall efficiency of mono-incineration plants was studied. The base assumption in the study was that 64 % of overall efficiency could be achieved. However, from a theoretical point of view efficiencies of up to 80 % are possible in fluidized bed incinerators (Böhmer et al., 2007). With a higher efficiency of 80 %, overall GWP, TAP and CED are reduced by 0.8, 0.1 and 2 %, respectively. As for the process of mono-incineration particularly values could decrease by 0.3–1.9 kg CO_2e and 6–35 MJe PE^{-1} yr^{-1} for GWP and CED, respectively.

5. Discussion and system recommendations

Most important contributors to total GWP are direct emissions and electricity demand on-site of WWTPs. Since associated GWP and CED_{fossil} from electricity use is varied in the EU due to inherent differences in energy sources (see evaluations in ecoinvent, 2020), values for wastewater and sludge treatment will vary slightly across the EU. As a further consequence, the renewable transformation of the EU electricity grid will automatically improve GWP and CED_{fossil} from wastewater treatment. In addition, benchmarking data can help WWTP operators to identify excess use of electricity (Haslinger et al., 2016; Gandiglio et al., 2017). Automatization (Gray et al., 2015), demand-side management (Zohrabian et al., 2021), designs improvements for agitators (Füreder et al., 2017) and improvement of digestor efficiency (Jenicek et al., 2013) can then reduce demand and increase electricity production where possible.

As argued previously in the literature (Law et al., 2013), accounting for 100 % biogenic carbon in sewage sludge might be misleading towards a more favourable net GWP for WWTPs. With 18 % of total GWP coming from sewage fossil carbon in this study, the importance of this parameter should not be understated. However, studies on variability and impact factors for fossil TOC content of sewage sludge are rare, likely a representation of the official disregard of CO_2 emissions in public accounting. For N_2O and CH_4 emissions, this analysis showed that direct emissions can vary highly and have a high impact on the total GWP of plants. Precise monitoring of these compounds is therefore recommended as complementary information in LCA studies of individual WWTPs and new treatment processes.

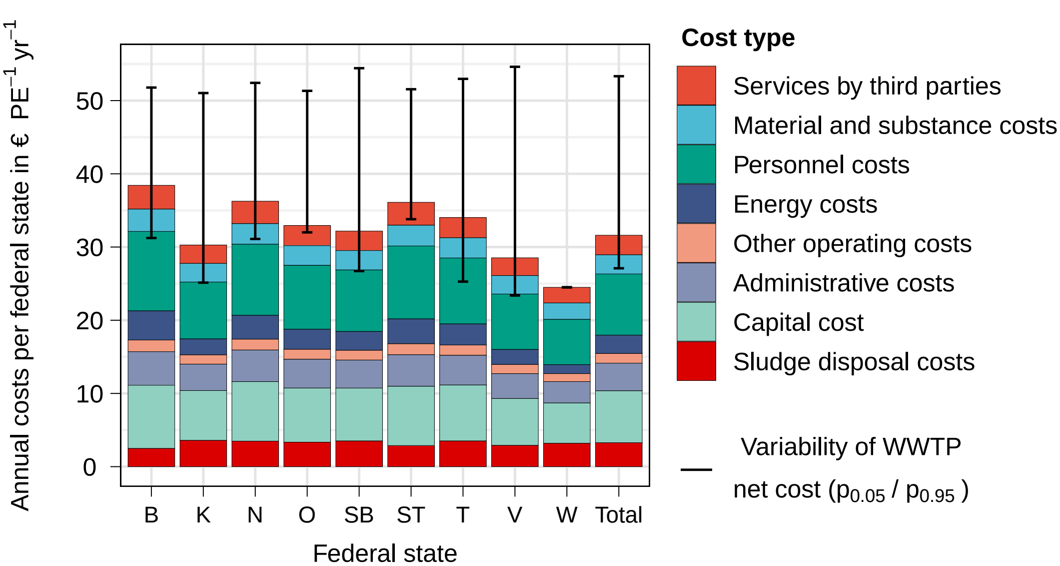

Fig. 5. Composition of total treatment cost as € PE^{-1} yr^{-1} for different federal states, as well as mean and variability ($p_{0.05}$ to $p_{0.95}$) of total net-cost of individual WWTPs.

Considering Austrian recommendations for increased mono-incineration of sludge with subsequent P recycling from sewage sludge ash (BMNT, 2018), the variability in N_2O emissions from combustion of sludge is a cause for closer governance, especially as factors for variability in emissions are plentiful. Due to high N-content, sewage sludge has a large potential for N_2O formation during combustion (Murakami et al., 2009). While rotary incinerators generally show lower formation in N_2O than fluidized bed technology (Gutierrez et al., 2005; Svoboda et al., 2006), sewage sludge is predestined for the latter due to optimized temperatures and lower NO_x and CO emissions by design. Looking closer at fluidized beds, stationary are the preferred choice over circulating beds (Werther et al., 1995; Werther and Ogada, 1999; Bonn et al., 1995). Higher O_2 excess (Grosso and Rigamonti, 2009), temperatures below 900 °C (Svoboda et al., 2006), and a low bed temperature (Mineur and Roschek, 2002) are other important factors related to high N_2O emission levels. If selective non-catalytic reduction (SNCR) is used, ammonia is the preferred chemical over urea, as N_2O formation is lowered by taking out possible intermediate reactions from urea to HCNO to N_2O (Grosso and Rigamonti, 2009). Further, the temperature ratio between bed, free-board and post-incineration should be coordinated properly for optimal N_2O and NO_x emission reduction.

A push for mono-incineration and P recycling will also increase prices of sludge disposal (Amann et al., 2021) and accordingly the total cost of wastewater treatment, but centralized concepts could maximize returns from marketed recycling products (Egle et al., 2016). As results show, size of WWTPs has, however, a much bigger impact on cost differences than sludge disposal costs. Highly centralized treatment of wastewater as in the state of Vienna (W; Fig. 5) can more than half total costs due to better efficiency in treatment and economy-of-scale effects.

6. Conclusion

Wastewater and sludge treatment are complex systems with large differences in resource and energy demand per PE. To capture these internal differences, we developed an extensive database for all WWTPs in Austria bigger than 2000 PE and applied a variety of state-of-the-art analyses like LCA and cost estimation. The combination of highly detailed data with these proven methods enabled a unique level of system understanding that to the best of our knowledge has never been achieved in wastewater treatment analyses before.

Hereby, we could identify the causes and main contributors to environmental impacts, metals diffusion in the environment and costs within the system. Furthermore, we could prove that environmental impacts and costs are subject to large variations, depending e.g. on WWTP infrastructure, WWTP size, sludge stabilization and sludge treatment.

Building on these analyses, it is now possible to carry out a well-founded scenario analysis to adequately assess which environmental and economic implications different strategies and future developments for P recycling would have on the whole system and on a regional level as well. Results from this study provide hints at which factors will or will not be important contributors for change. One example is sludge transport, which we determined was already quite substantial in Austria and is therefore not expected to play a large deciding role in ecological and economical decisions.

Future research in wastewater and sludge treatment should focus on including this variability of wastewater treatment systems into their analysis, to enable a better understanding of improvement potentials depending on the size and type of treatment plant.

Data availability and supplementary data

Restrictions apply to the availability of these data. Raw data and individual life cycle inventories per WWTP are not publicly available due to privacy issues and large data amounts. However, background data and all underlying assumptions and parameters presented in this study are available in the supplementary material or partly in Amann et al. (2021) as summarized statistical data. For more detailed, individual information please contact the corresponding author.

CRediT authorship contribution statement

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Conceptualization, A.A., H.R., O.Z. and M.Z.; Data curation, A.A.; Formal analysis, A.A., S.P., N.W. and O.Z.; Funding acquisition, A.A., H.R., O.Z. and M.Z.; Investigation, A.A., N.W. and O.Z.; Methodology, A.A., N.W., S.P., M.Z., O.Z.; Project administration, J.K., H.R., O.Z. and M.Z.; Resources, J.K. and H.R.; Software, A.A. and O.Z.; Supervision, J.K., H.R. and M.Z.; Validation, A.A., N.W., J.K., O.Z. and M.Z.; Visualization, A.A. and O.Z.; Writing—original draft, A.A.; Writing—review and editing, A.A., J.K., H.R., O.Z. and M.Z. All authors have read and agreed to the published version of the manuscript.

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.

Acknowledgements

This research was funded by the Federal Ministry of the Republic of Austria for Climate Action, Environment, Energy, Mobility, Innovation and Technology. We want to thank all treatment plant operators and the federal state authorities of Austria for their help with data curation in this project. We further acknowledge TU Wien Bibliothek for their financial support through its Open Access Funding Programme.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2022.157401.

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