Operation and Performance of Austrian Wastewater and Sewage Sludge Treatment as a Basis for Resource Optimization

Arabel Amann [1,*,†], Nikolaus Weber [1,†], Jörg Krampe [1], Helmut Rechberger [2], Ottavia Zoboli [1] and Matthias Zessner [1]

1   Institute for Water Quality and Resource Management, Research Unit of Water Quality Management, TU Wien, Karlsplatz 13/E226-1, 1040 Vienna, Austria; nikolaus.weber@tuwien.ac.at (N.W.); joerg.krampe@tuwien.ac.at (J.K.); ozoboli@iwag.tuwien.ac.at (O.Z.); matthias.zessner-spitzenberg@tuwien.ac.at (M.Z.)
2   Institute for Water Quality and Resource Management, Research Unit of Waste and Resource Management, TU Wien, Karlsplatz 13/E226-1, 1040 Vienna, Austria; helmut.rechberger@tuwien.ac.at
*   Correspondence: arabel.amann@tuwien.ac.at; Tel.: +43-1-58801-22632
†   These authors contributed equally to this work.

Abstract: Recent years came with a paradigm shift for wastewater treatment plants (WWTPs) to extend the sole purpose of contaminant removal to an additional function as resource recovery facilities. This shift is accompanied by the development of new European legislation towards better inclusion of resource recovery from wastewater. However, long operational lifespans and a multitude of treatment requirements demand thorough investigations into how resource recovery can be implemented sustainably. To aid the formulation of new legislation for phosphorus (P) recovery specifically, in 2017 we conducted a survey on Austrian WWTP-infrastructure, with a focus on P removal and sludge treatment, as well as disposal and sludge quality of all WWTPs above 2000 population equivalents (PE). Data were prepared for analysis, checked for completeness and cross-checked for plausibility. This study presents the major findings from this database and draws essential conclusions for the future recovery of P from wastewater. We see results from this study as useful to other countries, describing the current state of the art in Austria and potentially aiding in developing wastewater treatment and P recovery strategies.

Keywords: wastewater survey; phosphorus removal; phosphorus recovery; sludge stabilization; sludge disposal; sludge quality; sludge production

1. Introduction

While conventional wastewater treatment with the activated sludge system (mechanical, biological and chemical) was well established by the end of the 1990s [1], new issues have risen to the center of attention in the 21st century. The reduction of energy consumption [2,3], stricter effluent quality requirements for carbon and nutrients [4], the recovery of resources [5,6], wastewater reuse [7,8] and the removal of contaminants of emerging concern [9] and of antibiotic resistant genes [10,11] are only some of the new challenges to be addressed. This multitude of new tasks will in some cases not support themselves economically and might have negative side effects on the environment due to higher energy and/or material demands [12]. In addition, the primary function of wastewater treatment plants (WWTPs), the safe and cost-effective removal of major contaminants, should not be impeded by future task expansions or treatment changes. New legal requirements and plant configurations, therefore, need careful and thorough planning, best based on detailed information about the current status, infrastructure and performance of WWTPs.

The Federal Ministry of the Republic of Austria for Climate Action, Environment, Energy, Mobility, Innovation and Technology is currently underway in designing a new directive for the recovery of phosphorus (P) from municipal wastewater [13]. Eligibility of

WWTPs for P recovery is dependent on a multitude of design parameters and on sludge treatment infrastructure. Of high importance for the P recovery potential is the type of P removal (enhanced biological P removal (EBPR), or chemical with iron and aluminum) affecting its bio-availability in sludge [6] and sewage sludge ash [14]. For on-site recovery through struvite precipitation, higher P-concentrations in sludge will further positively affect the efficiency of the recovery units [15]. Recovery of P from sewage sludge ash is mainly competing with high concentrations in P raw materials (phosphate rock) of generally above 9% P [16]. Since incineration will leave only inorganic sludge components, sludge-P concentrations alone will not be a good predictor for sludge ash-P concentrations in these cases. Information on the inorganic sludge content is, therefore, a necessity. Next to P, heavy metal concentrations will also determine the suitability of sludge ash for P-recovery [17,18].

For WWTPs that are better suited for the extraction of P from sewage sludge ash (e.g., with no EBPR), recovery will in most cases require prior mono-incineration of their sewage sludge, a practice that is yet uncommon in Austria [13] and generally more expensive than co-incineration [19]. Therefore, additional costs due to legal changes towards P recovery requirements will depend largely on the current costs for sludge disposal, on sludge transport distances as well as on sludge production amounts.

To aid in the formulation of new legislation, in 2017 we commenced an extensive survey of the Austrian wastewater and sludge treatment system. We then established a comprehensive database through the help of a multitude of WWTP operators and federal state authorities. In accordance with the mentioned factors affecting P recovery, it covers information on wastewater treatment design, P removal and sludge stabilization methods, sewage sludge production, water content and quality, as well as on sludge disposal. This study presents the major findings from this database and draws essential conclusions for the future recovery of P from wastewater. We see results from this study as useful to other countries, describing the current state of the art in Austria and potentially aiding in developing wastewater treatment and P recovery strategies.

2. Materials and Methods

In 2017, a survey (see Supplementary Material 1) on Austrian WWTP-infrastructure and P-removal, as well as on sludge treatment, disposal and quality was sent out through the Austrian Water and Waste Management Association to all WWTPs with a design capacity above 2000 population equivalents (PE, 1 PE equals cumulative oxygen demand (COD) load of 120 g per day). WWTPs below 2000 PE were excluded from the analysis, as information on these small-scale plants is generally scarce due to less stringent or non-existent effluent criteria [20]. On the other hand, as WWTP above 20,000 PE treat approximately 95% of the COD load and 86% of the P-load in Austria (see Table 2), obtaining results from these plants was set as a priority. In addition, all nine federal states authorities of Austria were contacted for auxiliary data to advance the completeness of the data basis.

Data on WWTP loads were obtained from the national Austrian emission inventory on surface water bodies [21] based on the data collection requirements of the Austrian water act [22]. At the time of survey preparation in 2017 the general year of interest was set for 2016, because it was the year for which the most updated and complete information would be likely available. However, in some cases, data from 2015 and 2017 were included as well if no data were available for 2016. All data were grouped based on WWTP design capacity PE, although for comparison of resource use and production, actual yearly PE loads were used.

Data were then prepared for analysis, checked for completeness and cross-checked for plausibility by comparing values of each parameter within its group and with other available parameters (e.g., sludge stabilization method). For data analysis, the GUI *R Studio* and programming language *R* was used (Version 4.1.1). Data distributions are plotted using R package *ggplot*. The middle horizontal line gives the median, the upper and lower "hinges" correspond to the 25th and 75th percentiles. Upper and lower whiskers reach to

the highest and lowest values within a distance of 1.5 times the interquartile range (distance of 25th and 75th percentile) starting from the upper and lower hinges, respectively.

Pearson correlation method was used to check for linear correlation between two normally distributed variables. A two-way ANOVA was applied to test for significant differences in means of a quantitative dependent variable according to the levels of a related categorical independent variable. Statistical tests were considered significant at $p < 0.05$.

3. Results and Discussion

3.1. Data Availability

Data availability for WWTPs with a design capacity bigger than 20,000 PE was high and generally exceeded 50% availability for most parameters (see Table 1). Only sludge quality data (loss on ignition (LOI) and P content) was available for less than 50% of these plants. As WWTPs between 2000 and 20,000 PE were not the primary focus of this study and plant count is generally much higher, data availability for these plants was much lower. For all parameters, only 35% of plants or lower provided any data. However, since most wastewater is treated in WWTPs above 20,000 PE (Table 2) and a decent number of observations was still achieved for WWTPs below 20,000 PE, this dataset is seen as highly representative of the current Austrian status quo.

Table 1. Recovered data and data availability after data curation for wastewater treatment plants (WWTPs) bigger than 20,000 and smaller than 20,000 PE.

Data Availability in % of N	<20,000 PE (N = 439)	>20,000 PE (N = 194)
EBPR yes/no	11	78
Chemical P-removal yes/no	32	86
Flocculating agent demand	7	52
Primary clarifier yes/no	14	86
Sludge stabilisation method	35	99
Sludge production (dry matter)	35	100
Sludge production (wet)	32	96
Loss on ignition (LOI) of sewage sludge	4	42
Sludge dewatering yes/no	23	92
Type of sludge dewatering unit	18	86
Sludge dry matter content	3	76
Sludge drying yes/no	15	84
Type of sludge dryer	1	84
P content of sewage sludge	3	38
Heavy metal content of sewage sludge	30	60

Table 2. Share of treated COD and P, and distribution of precipitating agents according to the size class of WWTPs in percent of population equivalent (PE) loads.

	2000 to 20,000 PE	>20,000 to 50,000 PE	>50,000 PE 100,000 PE	>100,000 PE
Total number of WWTPs	439	125	33	36
Share of treated COD in %	5	19	10	66
Share of treated P in %	14	19	9	58
Precipitating agent; n = in % of PE load	114	97	25	28
Aluminium	22	20	9	6
Iron	49	47	59	89
Aluminium/Iron Mix	19	28	27	4
Others	10	5	6	0

3.2. Phosphorus Removal

Table 2 provides an overview on the obtained data for P and COD removal. Information is given separately for four size groups of WWTP design capacities: (1) smaller than 20,000 PE, (2) between 20,000 and 50,000 PE, (3) between 50,000 and 100,000 PE, and (4) larger than 100,000 PE. The 36 largest plants of more than 100,000 PE treat the major share of COD (66%) and P (58%).

Austrian WWTPs remove 90% of all P from wastewater influent [23]. All surveyed WWTPs (n = 308) have chemical P removal in place. In addition, around 30% state that they also have an anaerobic tank for EBPR installed. Iron is the primarily used precipitating agent (77% of total PE load) in all size groups (Table 2), and its probability of use increases with larger WWTP sizes. Aluminum is applied to treat around 10% of PE load. It is more commonly used in WWTPs below 50,000 PE (~21%). Aluminum–iron mixes treat 11% of PE load and are used below 100,000 PE (19–27%). Other agents like lime are rarely applied (2% of PE load), and if, only at plants <100,000 PE.

For P-recovery or direct use of sludge in agriculture, EBPR would be the preferred method of choice, enhancing the bio-availability of P in sewage sludge [24]. It is, however, questionable if iron-P-removal can be fully replaced, especially in larger WWTPs with anaerobic digestion, as it additionally functions in sulfide (odor) control in anaerobic digestors [25]. This is additionally supported by higher use of iron in WWTPs with anaerobic sludge treatment (92%) than aerobic treatment (80% of PE load; see Table S1).

The fact that all Austrian WWTPs with EBPR use at least little additional chemical dosing shows that EBPR-performance is limited in Austrian treatment plants. Therefore, chemical precipitant use might be reduced with the application of EBPR, but never fully redundant. A switch to EBPR will also come with additional operational tasks for plant operators, most importantly the prevention of scaling of pipes through uncontrolled P precipitation as struvite [26]. Controlling this struvite precipitation would be beneficial, as it would create more easily accessible P forms for recovery both on-site as well as from ash [27], but it is unclear if plant operators can be persuaded to take on these additional challenges.

For a better understanding of precipitating agent use, the declared demand by WWTP operators calculated as mol per year was plotted against the theoretical demand (Figure S1). Theoretical demand was estimated from P inflow minus P effluent loads, subtracting P demand for biomass production (1% of biological oxygen demand loads) and P removal if a primary clarifier is present. Assumed β values (mol of agent dosed per mol of P) for precipitation were set at 1.2 for effluent limits of 2 mg L^{-1}, at 1.5 for a limit of 1 mg L^{-1} and at 2.5 for precipitation after the secondary clarifier if stringent limits of 0.5 mg L^{-1} are set. In general, the derived theoretical precipitating agent demand from P and BOD WWTP loads is a good predictor for actual agent use (r = 0.83, p-value < 0.005). We further analyzed the calculated ß-values from plants with a limit p value of 1 mg L^{-1} (n = 120) and with and without EBPR (Figure 1). Median demand per mol of P precipitated was significantly different for those two groups (t (113) = −2.6589; p = 0.009), with plants with EBPR having an mean reduction in demand of 18%.

3.3. Primary Clarification, Sludge Production and Stabilization

Sewage treatment in Austria occurs mainly by the activated sludge system. Conventional aerobic sludge stabilization is mostly used in WWTPs smaller than 20,000 PE, while anaerobic sludge stabilization via digestion is the more frequently applied method in WWTPs larger than 20,000 PE (Table 3). Other designs are rare, with most cases being sequencing batch reactors in smaller WWTPs (largest SBR plant with 60,000 PE). Out of the plants with aerobic sludge stabilization, simultaneous stabilization (sludge age of more than 25 days) is most common, comprising ~73% of aerobically stabilized sludge. Separate aerobic stabilization has its highest share (28% of aerobically stabilized sludge) in plants between 20,000 and 100,000 PE.
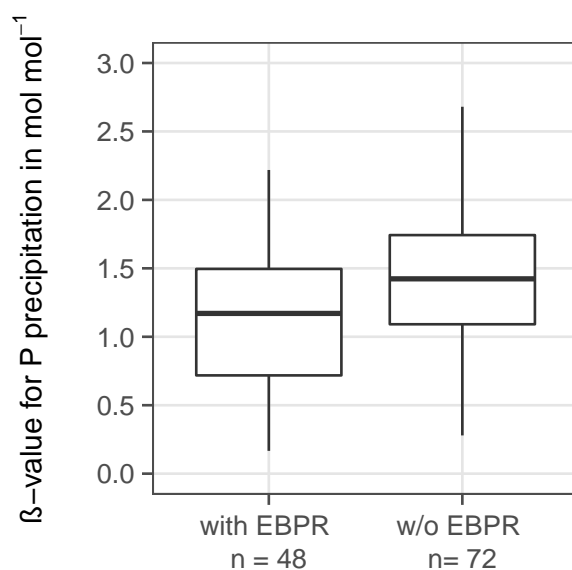
Figure 1. Comparison of ß-values for P-precipitation with and without additional enhanced biological phosphorus removal (EBPR) in mol precipitation agent per mol P. Statistical values are provided in Table S2.

Primary clarifiers are mostly installed in larger plants (89% for >100,000 PE) and less abundant in WWTPs smaller than 20,000 PE (24%). This size dependency is generally derived from the combined use of primary clarifiers with anaerobic sludge stabilization. Approximately 95% of clarifier capacity is installed on sites with anaerobic digestors. The remaining 5% capacity are installed in aerobic plants, mostly in combination with simultaneous sludge stabilization (80%).

Table 3. Primary clarifier abundance and use of various sludge stabilization methods in different size groups of Austrian wastewater treatment plants.

Shares Given in % of PE Treated	2000 to 20,000 PE	>20,000 to 50,000 PE	>50,000 PE 100,000 PE	>100,000 PE	Total
Primary clarifiers					
n	61	104	29	33	227
Occurrence	24	68	82	89	82
Sludge stabilisation					
n	155	124	33	36	358
anaerobic	11	66	79	97	85
aerobic, of which...	89	34	21	3	15
... simultaneous	74	70	59	100	73
... separated	18	28	28	0	21
... unknown	8	2	13	0	5

A thorough knowledge on sludge production is the basis for developing new sewage sludge mono-incineration plant concepts in Austria. Through the ongoing transformation of Austrian WWTPs towards anaerobic digestion and better stabilization, sludge amounts have been decreasing for years [23]. WWTPs with sludge utilization in agriculture often add legitimate amounts of lime for hygienization. If these plants switch to mono-incineration instead, a further decrease in sludge production is expected, since adding inorganic material to sludge will only further reduce P ash concentrations and thereby hinder recycling. To estimate sludge amounts for incineration, data on total sludge yield per PE (organic and inorganic) were analyzed according to their primary sludge treatment method (aerobic/anaerobic) and lime addition (Figure 2). Anaerobic treatment achieves

the lowest total sludge yield of 37 g PE^{-1} d^{-1} (Table S3). Simultaneous aerobic treatment produces more sludge (52 g PE^{-1} d^{-1}) than separated aerobic treatment (45 g PE^{-1} d^{-1}). Addition of lime resulted in around 54 to 57% higher yield with median values of 57 g for anaerobic and 77 g PE^{-1} d^{-1} for aerobic treatment.

-- Aerobic treatment   -- Aerobic treatment + lime

-- Anaerobic treatment   -- Anaerobic treatment + lime
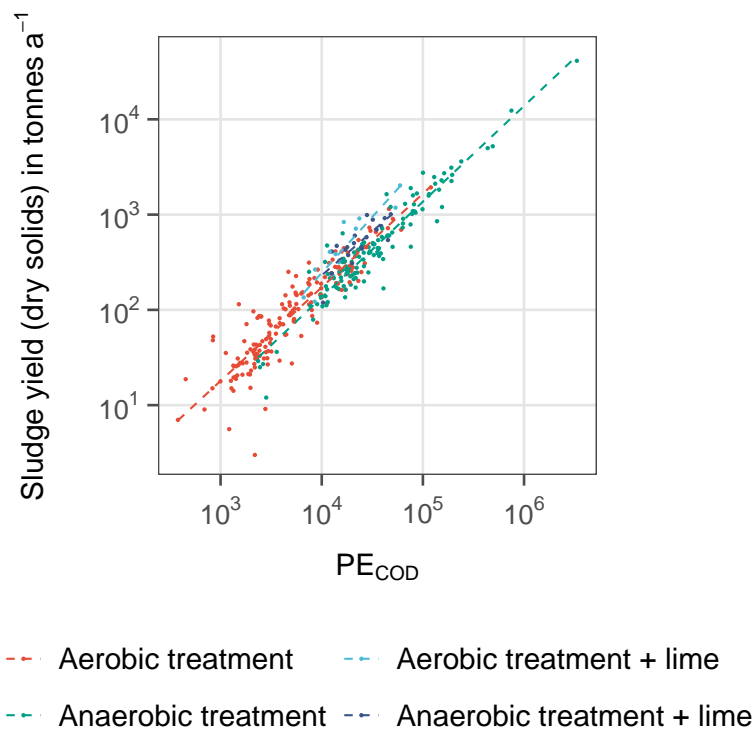
Figure 2. Total sludge yield (as dry matter) in tons per year as a function of treated population equivalents (PE; derived from COD with 120 g COD PE^{-1} d^{-1}) and dependent on the primary sludge stabilization method as well as on potential lime addition. Statistical values are provided in Table S3.

Derived values are in the range of observed and modeled values from literature, however, large variations were found for total sludge yield (27–82 g PE^{-1} d^{-1} [28,29]). Data on loss of ignition (LOI; Figure 3) can put sludge yield into context with the degree of sludge stabilization. Anaerobic treatment generally achieves the best stabilization and lowest LOI (59%), followed by separated aerobic treatment (64%). Simultaneous stabilization shows highest LOI of 71%. With the addition of lime (inorganic matter) LOI in sludge decreases considerably to 34–35%.

Previous detailed analysis on Austrian sludge production found a volatile suspended solids (VSS) yield of 16 to 20 g PE^{-1} d^{-1} for anaerobic and separated aerobic stabilization [30]. Non-sufficiently stabilized sludge from simultaneous aerobic stabilization showed a VSS yield of 20 to 35 g PE^{-1} d^{-1}. If LOI values are taken into account, total sludge yield can be estimated from VSS production according to Equation (1).

$$\text{Total sludge yield } [\text{g PE}^{-1}\ \text{d}^{-1}] = \text{VSS } [\text{g PE}^{-1}\ \text{d}^{-1}]/\text{LOI } [\%] \tag{1}$$

Assuming a sludge LOI of 60% for anaerobic or separated aerobic treatment and 71% for simultaneous stabilization (Table S4), the corresponding total sludge yields would be in the area of 26–33 g (anaerobic/separated) and 35–50 g PE^{-1} d^{-1} (simultaneous), respectively. In comparison, the higher median total sludge yields derived by this study (>37 g PE^{-1} d^{-1}) suggest a slightly incomplete stabilization and a VSS yield after stabilization of commonly above 20 g PE^{-1} d^{-1}.
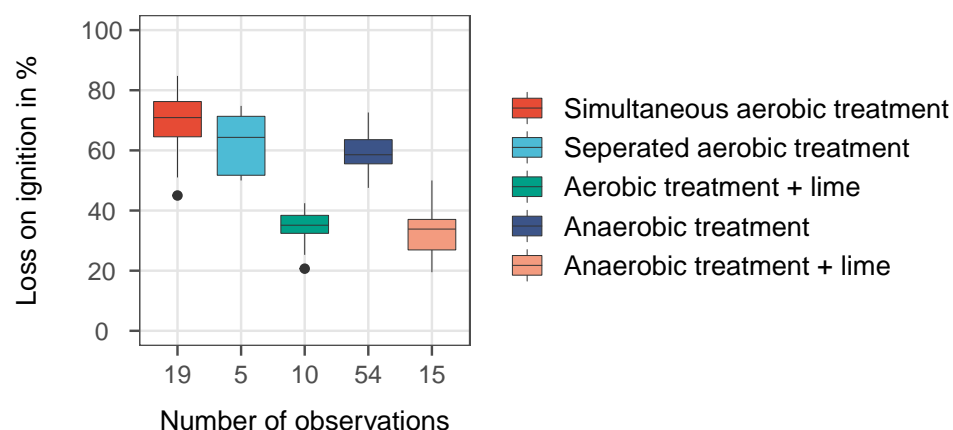
Figure 3. Loss on ignition (LOI) in percent as a function of the primary sludge stabilization method. Statistical values are provided in Table S4.

### 3.4. Sludge Processing: Dewatering, Drying and Hygienization

Types of dewatering units are unequally distributed between the different size groups (Table 4). Filter presses are most common in smaller WWTPs (39%), while above 20,000 PE, centrifuges are the standard method of choice (37–76%). Screw presses are more abundant in WWTP below 50,000 PE and have occupied the market only in recent years. Data from a German inquiry on sewage sludge treatment in 2003 showed no occurrence of screw presses for dewatering at that time [31]. Belt presses are rarely in use. Some WWTPs use mobile dewatering, however this is mostly common for smaller plants. Addition of lime for hygienization is done in 10–38% of the plants, correlated to the primary sludge disposal method (agricultural and composting). Sludge drying on-site is rarely implemented in Austria, with around 19 known installations, namely, 11 solar dryers, two belt dryers and four convection dryers.

Table 4. Summary table of dewatering units and use of lime according to the size group of the respective wastewater treatment plants.

	2000 to 20,000 PE	>20,000 to 50,000 PE	>50,000 PE 100,000 PE	>100,000 PE
Dewatering units				
n	74	103	28	33
Centrifuge in %	26	37	45	76
Belt press in %	6	8	10	2
Filter press in %	39	25	25	10
Screw press in %	24	26	19	12
Others in %	5	4	1	0
Hygienisation with lime				
n	5	76	21	21
Share in %	21	24	38	10

Dewatering with the addition of lime achieved the highest dry matter content in sewage sludge (Figure 4). Arguably, this is partially from an increase of the related solid mass to water ratio, but calcium addition is also known to increase floc strength and dewaterability [32]. Out of the different types of dewatering units, filter presses showed the highest median dry matter content (28%) followed by centrifuges (25%), screw presses (24%) and finally belt presses with the lowest median value of 22%. Mobile units showed a high range from 21 to 34%. Values correspond well with data from DWA guideline M 366

for sludge dewatering (filter press 22–28%, with lime 30–40%, centrifuges 22–30%, screw presses 20–28% and belt presses 20–28% [33]).
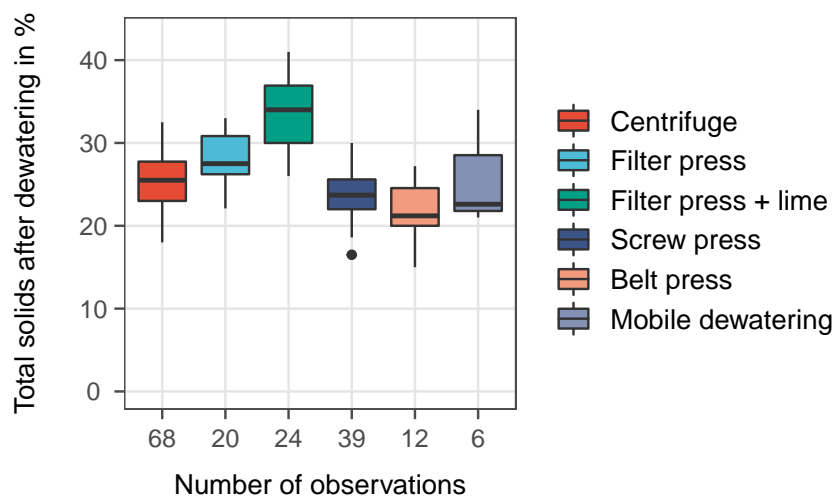
Figure 4. Dry matter content of dewatered sludge in percent according to the type of dewatering unit used and the (non-)addition of lime. Statistical values are provided in Table S5.

Further, a WWTP-size dependency was observed, with median dry matter content (dewatered and without lime addition) increasing from 23 to 25 to 26 to 26.7%, for size groups 1, 2, 3 and 4, respectively. While reasons for this cannot be derived from the data itself, it is assumed that the improved dewatering performance is achieved both by better aggregates as well as by better supervision and operation of larger plants.

Out of the installed dryers, the two convection dryers achieved the highest dry matter content of around 84%, with belt dryers and solar dryers at approximately 71–73% (Table S6).

3.5. Sludge Quality

For the development of sustainable P recovery strategies, information on sludge P concentrations and accompanying heavy metals is required. Not all treatment plants monitor their sludge quality; therefore, it will be necessary to resort to easily available data—e.g., recorded WWTP-P inflow and outflow loads—to determine P loads and concentration in sewage sludge. We plotted measured P concentrations from sludge analysis against theoretical P concentrations derived from yearly inflow/outflow loads and sludge quantities (Figure 5). As is depicted, theoretical P concentration is a good predictor for actual P concentrations ($t(103) = 9.9897$, $p < 0.001$), with a general deviation of smaller than 25%. Theoretical analysis might also give a better understanding of mean yearly concentrations, as measurements only represent a moment in time.

Observed P concentrations range from 9 to 63 g kg^{-1} (Figure 6a) and are in part connected to the degree of sludge stabilization, which is commonly well represented by LOI (Figure 3). Due to a lower LOI and a reduced sludge mass, P concentrations are higher in anaerobically treated sludge with 34 g kg^{-1}. As inorganic matter is added with lime addition, LOI is also lower. However, P is diluted by this treatment, leading to median P concentrations in the area of 22 g kg^{-1} only.

For recovery from sewage sludge ash, P concentrations in ash are of interest. We estimated P ash concentrations from LOI, sludge mass and P concentrations. As depicted (Figure 6b), Austrian ash P concentrations without added lime would rest around the 9% mark, which P rock is rarely falling short of [16]. For a better cost effectiveness of P recovery from ash, operators should try to reduce inorganic additives on-site, without compromising the effectiveness of wastewater treatment. Observed median levels of other nutrients in

sludge were 31 g nitrogen, 7 g magnesium, 1.4 g potassium and 0.9 g sodium per kg of sludge (Figure S2 and Table S11).
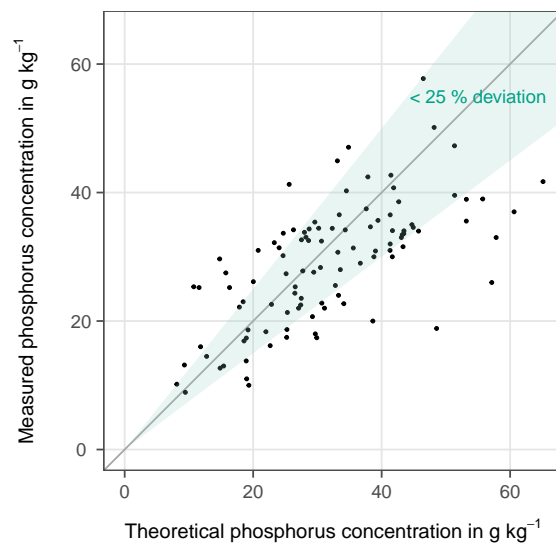

Figure 5. Comparison of measured sludge P concentrations from sludge monitoring data to theoretical sludge P concentrations derived from WWTP-sewage P-inflow minus P-outflow.
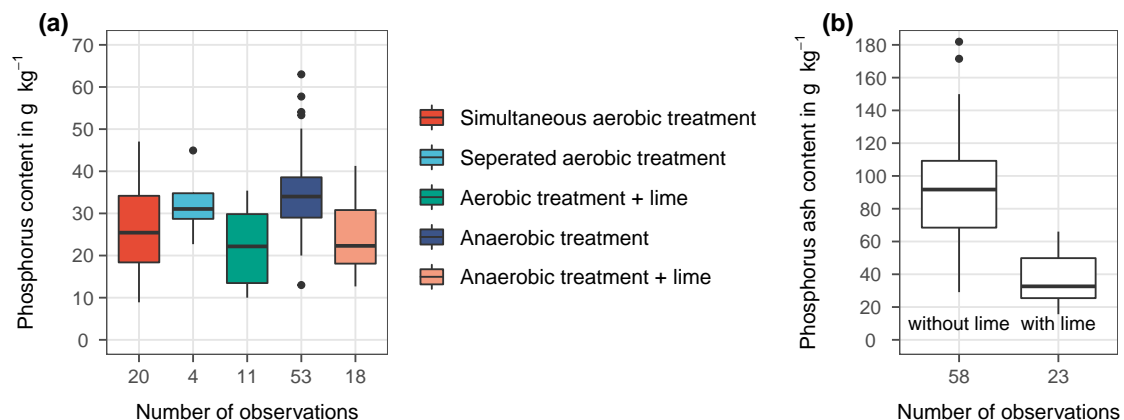
Figure 6. (**a**) P content in grams per kilogram as a function of the primary sludge stabilization method, and (**b**) derived P ash content as a function of lime addition. Statistical values are provided in Table S10.

While it is known that WWTPs partially remove heavy metals from the liquid stream, information on the impact of different operating conditions on heavy metal removal is scarce and sometimes contradictory [34,35]. Sludge heavy metal concentrations will further relate to the abundance of pollutant sources (municipal, industrial) in the WWTP drainage area [36]. As current German legislation obliges only certain WWTP size groups to recover P, it is of interest if there exists a correlation of heavy metal concentrations in sludge with WWTP sizes. Figure 7 shows Austrian heavy metal concentrations in sewage sludge as a function of the size group of the WWTP. It can be seen, that most heavy metals are similarly distributed across all WWTP groups. Exceptions are chromium and nickel, which show comparatively high values for WWTPs larger than 100,000 PE. Accordingly, sludge quality in Austria shows for the most part no size-dependency. The majority of WWTPs can fulfill even the more stringent Austrian heavy metal limits in sludge for use in agriculture (limits shown as dashed lines, taken from the work in [37]). Further analysis of the data, considering the influence of different treatment schemes and of the type of

drainage area, should be performed for a better understanding of final metal concentrations in sewage sludge.
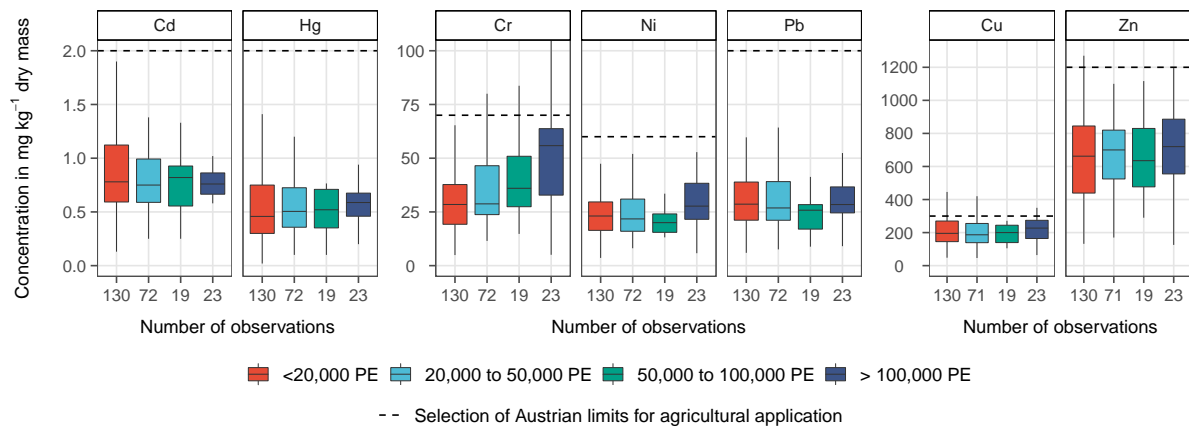
Figure 7. Heavy metal (cadmium, mercury, nickel, lead, chromium, copper and zinc) sewage sludge concentrations in milligrams per kilogram. Dashed lines give the most common Austrian heavy metal limits for the application of sludge in agriculture [37]. Statistical values are provided in Table S12.
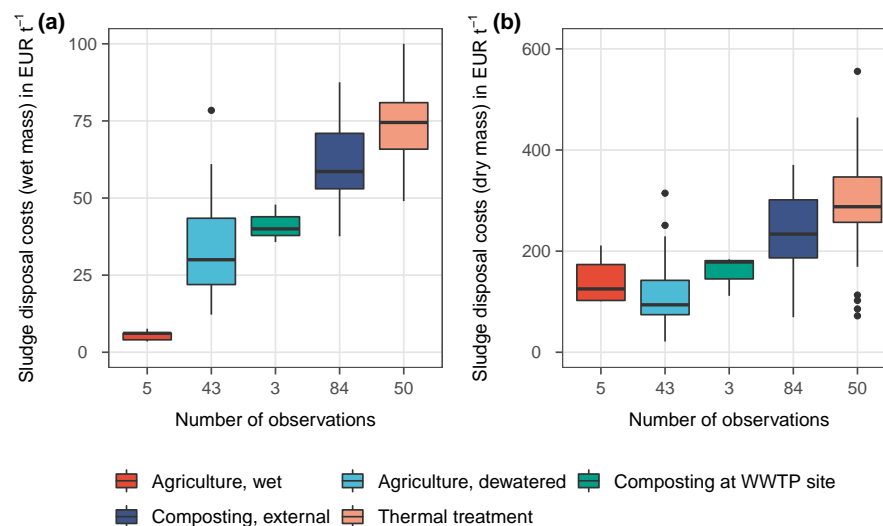
3.6. Sludge Disposal

For 91% of Austrian sewage sludge, disposal routes could be successfully tracked (Figure 8). The largest amounts are treated via mono-incineration or external composting. Sludge disposal routes are diverse, with some states (Vienna = W) with 100% thermal treatment and others (Burgenland = B) with close to a 100% of soil-based sludge use. Therefore, changes to sludge disposal due to potential P recycling and increased incineration will have very different degrees of effect in different states. Currently, only direct agricultural disposal (wet or dewatered = 18%) reliably brings P to arable land. Informal talks with WWTP stakeholders confirmed that compost from sludge is often not used in agricultural land with high P demand, but for recultivation of landfills or for landscaping. Specific estimates could not be derived, since tracking of composted sludge proved highly time consuming or impossible.
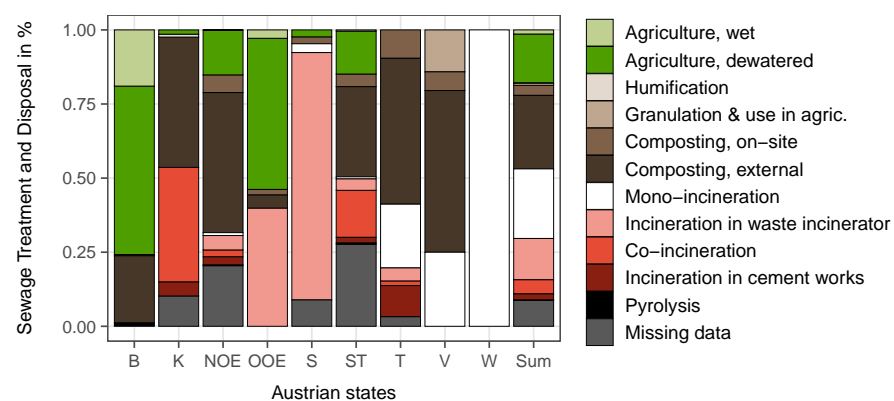
Figure 8. Sludge treatment and disposal routes in Austria in 2016 in % and for each federal state.

Sludge disposal costs in 2016 ranged from 3.5 to 100 EUR t^{-1} (wet mass) or 21 up to 560 EUR t^{-1} (dry mass) (Figure 9). These are well in line with published sludge disposal costs in Germany (160 to 480 EUR t^{-1} dry mass [38]). Disposal in agriculture and composting on-site was comparatively cheap with median values between 6 to 40 EUR and 94 to 180 EUR t^{-1} for wet and dry mass, respectively. External composting through 3rd party contractors and incineration was correlated to higher costs with median values

of 59 to 75 EUR t^{-1} (wet) and 230 to 290 EUR t^{-1} (dry mass). Compared to the cost of mono-incineration (280 to 480 EUR t $^{-1}$ dry mass [38]), an increase in costs for sludge disposal is likely if P recovery from ash is pursued.

Figure 9. Cost of sludge disposal in Euro per ton as a function of their disposal route (**a**) based on sludge wet mass and (**b**) based on sludge dry mass. Statistical values are provided in Tables S7 and S8.

Similarly to cost, transport distances were highest for external composting and thermal treatment, with some plants transporting over 530 km (one-way; Figure 10). Median values for thermal treatment were more than twice as high (120 km) than external composting (50 km). Agricultural disposal and composting at the WWTP site rarely exceeded 20 km with median values of 15 and 0.25 km, respectively. A move towards more incineration might result in longer transport distances. Careful evaluation of strategic locations for mono-incineration sites will, consequently, be decisive to limit future emissions from sludge transport.
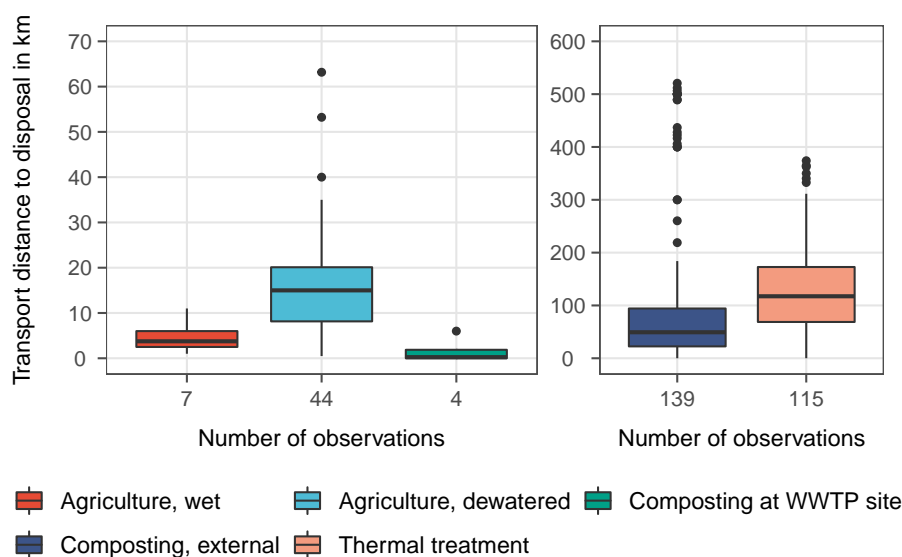
Figure 10. Distance of sewage sludge transport to disposal as a function of their disposal route. Statistical values are provided in Table S9.

4. Conclusions

This study presented the main findings from a carefully developed database on wastewater treatment and sludge disposal in Austria for the years 2015–2017. Data availability was high for most analyzed parameters, and the database posed a good basis for further deliberations on changes in sludge P management. Austrian wastewater treatment plants perform well, and have seen proper updates to state of the art technology, with a high share in anaerobic stabilization and well functioning dewatering units. Though EBPR would be the preferred P removal method of choice for P recovery, chemical precipitants are still vital for the secured removal of P from wastewater. Even WWTPs with EPBR installed use valid amounts of iron and aluminum, which will likely remain so in the years to come.

Sludge quantities are expected to decrease further with a shift from agricultural valorization to incineration of sludge. In turn, it is estimated that sludge disposal costs and transport distances will increase, from lower costs for agricultural disposal to higher costs for mono-incineration. To keep P concentrations high for an efficient recovery of P, efforts should focus on reducing inorganic additives to sludge as much as possible without inhibiting the treatment process. Heavy metals in sludge generally do not exceed the more stringent limits posed by Austrian legislators. Nevertheless, caution should be exercised, as with an increased recycling of sludge, sludge ash or of sludge derived products, total heavy metal loads to agriculture will increase if metals in sewage sludge ash for P recovery are not (partially) removed.

Before legislation for P recovery is implemented, care should be taken to further analyze different options of P recovery for their cost and environmental efficiency in the Austrian context. As mono-incineration of sludge is still rare in Austria, additional planning should focus on finding sustainable and strategically well-placed locations for new plants, in order to reduce impacts from sewage sludge transport.

Abbreviations

The following abbreviations are used in this manuscript:

BOD	Biological oxygen demand
COD	Chemical oxygen demand
EBPR	Enhanced biological phosphorus removal
LOI	Loss on ignition
P	Phosphorus
PE	Population equivalents based on COD (1 PE = 120 g COD d^{-1})
VSS	Volatile suspended solids
WWTP	Wastewater treatment plant

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