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Abstract

This review compiles information on sidestream characteristics that result from anaerobic digestion dewatering (conventional and preceded by a thermal hydrolysis process), biological and primary sludge thickening. The objective is to define a range of concentrations for the different characteristics found in literature and to confront them with the optimal operating conditions of sidestream processes for nutrient treatment or recovery. Each characteristic of sidestream (TSS, VSS, COD, N, P, Al^{3+}, Ca^{2+}, Cl^-, $Fe^{2+/3+}$, Mg^{2+}, K^+, Na^+, SO_4^{2-}, heavy metals, micro-pollutants and pathogens) is discussed according to the water resource recovery facility configuration, wastewater characteristics and implications for the recovery of nitrogen and phosphorus based on current published knowledge on the processes implemented at full-scale. The thorough analysis of sidestream characteristics shows that anaerobic digestion sidestreams have the highest ammonium content compared to biological and primary sludge sidestreams. Phosphate content in anaerobic digestion sidestreams depends on the type of applied phosphorus treatment but is also highly dependent on precipitation reactions within the digester. Thermal Hydrolysis Process (THP) mainly impacts COD, N and alkalinity content in anaerobic digestion sidestreams. Surprisingly, the concentration of phosphate is not higher compared to conventional anaerobic digestion, thus offering more attractive recovery possibilities upstream of the digester rather than in sidestreams. All sidestream processes investigated in the present study (struvite, partial nitrification/anammox, ammonia stripping, membranes, bioelectrochemical system, electrodialysis, ion exchange system and algae production) suffer from residual TSS in sidestreams. Above a certain threshold, residual COD and ions can also deteriorate the performance of the process or the purity of the final nutrient-based product. This article also provides a list of characteristics to measure to help in the choice of a specific process.

Contents

1. Introduction

Wastewater characteristics have been studied for decades because they are key to design and optimise the operation of wastewater treatment processes. The usual characteristics of urban wastewater (including total solids, organic matter, total nitrogen, total phosphorous and organic matter biodegradability) from different countries is well-documented, especially in view of process modelling (Rieger et al., 2012), and experimental methods to characterise the composition of wastewater have been established and benchmarked (Gillot and Choubert, 2010; van Loosdrecht et al., 2016). At the same time, there is a growing interest in the characterisation of both the solid and the liquid phase of sewage sludge. This interest is primarily motivated by the need to reduce sludge volumes and associated sludge management costs (Zhen et al., 2017) as well as to maximise their reuse as fertiliser (Kacprzak et al., 2017). One of the current concerns is the concentration of heavy metals and emerging contaminants when sludge disposal route is land application (Appels et al., 2010; Steele et al., 2022).

As part of sewage sludge management in water resource recovery facilities (WRRFs), sludge thickening and dewatering units result in the production of different types of sidestreams, also called “reject water”, “centrate”, “supernatant” or “filtrate”, recycled into the main wastewater treatment line. The major concern about these streams arose with the overall tightening of WRRFs effluent standards (Preisner et al., 2020) and the development of anaerobic digestion (AD) that generates a nutrient-rich supernatant (Gourdet et al., 2017). In the context of a wastewater identified as a resource rather than a waste stream, AD has proved to be an essential technology as it reduces sludge volume, stabilises sludge and more importantly recovers energy as methane (Appels et al., 2011). Sidestreams in WRRFs equipped with AD can contain up to 25% of the total nitrogen load and 10% of the total phosphorus load to the facilities (Couturier et al., 2001; Grulois et al., 1993). This phosphorus load can be even higher (up to 30%) when enhanced biological phosphorus removal (EBPR) is implemented in the water line (Ueno and Fujii, 2001). Also, to boost AD and dewatering performance, several sludge pre-treatments have been developed (Carrère et al., 2010). Among them, the thermal hydrolysis process (THP) is the most applied technology and existing full-scale references have so far reported an enhancement of AD performance after THP pre-treatment (Kor-Bicakci and Eskicioglu, 2019).The main drawback of THP is that it generates streams with high concentration of refractory compounds and an increase in sidestream ammonia concentration (Barber, 2016; Bougrier et al., 2008; Dwyer et al., 2008b).

High nutrient content in sidestreams can lead to increased energy consumption and degradation of effluent quality (Cullen et al., 2013; Janus and van der Roest, 1997), especially when the facility operates close to full-scale capacity. In such conditions, sidestream processes are good opportunities for upgrading treatment capacity without needing to expand existing works (van Loosdrecht and Salem, 2006). The main biological processes for the treatment of nitrogen in sidestreams includes: nitritation-denitritation, nitrification-denitrification, partial nitrification-anammox, bioaugmentation and algae production (Eskicioglu et al., 2018).

Sidestream processes also offer an excellent opportunity for nutrient recovery, essential for sustaining the food production industry. Currently, nitrogen-based fertilisers are mainly produced by the energy-intensive Haber–Bosch process, while rock phosphate is the main raw material in phosphorus-based fertilisers (Nancharaiah et al., 2016; Shaddel et al., 2019). The implementation of these production routes on the long-term is questioned because (1) phosphorus depletion is expected by 2100 (Van Vuuren et al., 2010) and (2) fertilizers production currently accounts for more than 1% of the world's emissions of greenhouse gases (GHG) (Kehrein et al., 2020). Municipal wastewater is thus an interesting nutrient-source as its nitrogen and phosphorus content accounts respectively for 14% and 7% of the global fertilizer demand (Qadir et al., 2020). Several reviews present in detail the technologies available for nutrient recovery from wastewater. They are mainly based on physical, physicochemical, and bio-electrochemical mechanisms (Guilayn et al., 2020; Vaneeckhaute et al., 2017; Ye et al., 2018). Struvite precipitation, ammonia stripping, membrane filtration, electrodialysis, bio-electrochemical system, ammonia and phosphate sorption are the most investigated processes. Performances of such processes depend on sidestream characteristics. Their efficiency, the capacity for nutrient removal, the energy and chemical consumption and the quality of recovered products have to be specified. The nutrient-based products must also comply with the current N & P fertilizer characteristics and legislation. In addition, to complete COD, N and P fractionation, the detailed ionic strength of sidestreams is necessary to correctly design and model treatment or recovery processes as they are mainly based on physicochemical reactions. This information is especially required as input of the new advanced plant wide models that couple biokinetics with physicochemical framework (Flores-Alsina et al., 2015; Kazadi Mbamba et al., 2015; Lizarralde et al., 2015; Solon et al., 2015; Vaneeckhaute et al., 2018b).

Despite the growing interest in the implementation of sidestream processes, a complete characterisation of this stream is relatively scarce and sparse in the literature. Published data include phosphorus (Martí et al., 2017), nitrogen (Kassouf et al., 2020), COD fractionation (Noutsopoulos et al., 2018) and ionic composition (Bhuiyan et al., 2009). However, no synthesis compares sidestream characteristics from different WRRFs, considering a large number of components (solids concentration, COD, biodegradability, N, P, ions, heavy metals and ionic composition) that constitute wastewater (Metcalf & Eddy. Inc et al., 2003).

This review article compiles and analyses literature data of sidestream characteristics from different locations in WRRFs. The aim is to bring knowledge on sidestream characteristics to assist in selection, design and modelling of sidestream processes. Data are questioned in order to choose the most optimal operating conditions of the main sidestream processes and to identify potential limits of application. Implications in terms of plant wide modelling are also mentioned, in order to improve sidestream description and nutrient recovery options. Finally, the need for data on characterisation of sidestreams are highlighted to support the development of processes that can improve environmental and economic impacts of WRRFs.

2. Literature Data Compilation

The set of data used in this study results from the compilation of information from 87 documents (peer-reviewed and grey literature). The characteristics found have been classified according to the source of sidestreams:

- biological sludge for sidestreams resulting from the thickening of biological sludge
- primary sludge for sidestreams resulting from the thickening of primary sludge
- anaerobic digestion for sidestreams resulting from the dewatering of digested sludge
- THP anaerobic digestion for sidestreams resulting from the dewatering of digested sludge preceded by a thermal hydrolysis process (THP).

The collection and description of the data can be found in the associated data paper (Devos et al., n.d.) and the dataset is available on the french repository "DataGouv" (https://doi.org/10.57745/FOHRHY). Only papers with clear information on sidestream sources were selected. Sidestreams from full-scale measurements only were included in the dataset.

All figures presented in this document have been generated with RStudio software version 4.1.2. Most of the data is presented in the form of a violin plot overlaid on a boxplot. Outliers have been detected with the Bonferroni test (Bretz et al., 2010) for each boxplot when the p-value is below 5%. They are indicated in the figures (in grey) but are not included in the calculation of the median nor in the count of the total number of values. The different sources of sidestreams were statistically compared using the Kruskal-Wallis test. The pairwise comparison using the Wilcoxon test was also used to compare the impact of different WRRF configurations. In addition, correlations between different parameters were identified using the Spearman test.

3. Sidestream Characteristics

In the following, sidestream characteristics are first described in terms of major pollutants (TSS, COD and nutrients): concentrations and mass loads are analysed. The ionic composition of sidestreams are then described, as well as pH, alkalinity and temperature. Collected data mainly refer to streams from the anaerobic digestion of sewage sludge. A few data on sidestreams from primary or biological sludge thickening are also presented, when available.

3.1 Major pollutants in sidestreams

3.1.1 Concentration of Total Suspended Solids (TSS) and Volatile Suspended Solids (VSS)

Figure 1 presents the TSS concentrations in sidestreams from anaerobic digestion preceded or not by a THP. TSS concentrations are highly variable from one study to another. This variability is partly due to the sampling methodology as most of the values have been obtained on grab samples. There is no significant difference between both types of sidestreams which means that TSS concentration in sidestreams is mainly driven by other factors than the presence of a THP, such as the dewatering unit performance. Interestingly, most of the data compiled on anaerobic digestion sidestreams came from centrifuges. The centrifuge has a solid capture rate between 95% and 99% depending on sludge conditioning (Metcalf & Eddy. Inc et al., 2003). Considering a TSS concentration in digested sludge of 25 g/L, the TSS concentration in sidestreams from an anaerobic digester would not exceed 1500 mg/L. Outliers in Figure 1 are much higher than this value, indicating that the centrifuge may sometimes underperform. Besides, variability of water content and particle size of sludge induced by sludge transport and storage can impact the demand for polymers, and therefore the performance of the dewatering unit (Andreoli et al., 2007; Henze and Comeau, 2008). Anaerobic digestion sidestream has a median VSS to TSS ratio of 70% (Figure 1 B). A value in the range of 55 – 75% is expected (Bowden et al., 2015) which corresponds to volatile solids (VS) range for digested sludge (Andreoli et al., 2007).

3.1.2 Concentration of Chemical Oxygen Demand (COD)

The concentrations of total COD and soluble COD are significantly different between anaerobic digestion and THP anaerobic digestion sidestreams (Figure 2). As for TSS, there is a large variability of COD concentration which is explained by the significant correlation between total COD and TSS. The difference of total COD concentration between both types of sidestreams is mainly due to the higher soluble COD concentration in THP anaerobic digestion sidestreams, induced by a higher solubilisation rate (Barber, 2016; Devos et al., 2020). Part of this additional soluble COD has been reported to be refractory compounds produced through the Maillard and Amadori reaction that also impacts the effluent COD of the WRRF. The amount of refractory compounds produced is dependent on the temperature of THP, and becomes significant at a temperature higher than 160°C (Toutian et al., 2020). For six facilities in Berlin (without sidestream processes) and for different THP temperatures the increase in effluent soluble COD was estimated to be in the range of 2 – 15 mg/L (Toutian et al., 2020). Another study reported that THP implementation in five facilities led to a 3 – 8 mg/L increase of effluent COD concentration, depending on the quantity of primary sludge versus the quantity of biological sludge (Svennevik et al., 2020).

The concentration of total COD in biological sludge sidestreams and in the treated water are usually close except in the case of thickening unit malfunction. The concentration of COD in primary sludge sidestreams is similar to wastewater and is in the range of 250 – 800 mg/L (Constantine, 2006). However, higher values of 823 mg/L (Roldán et al., 2020) and up to 4244 mg/L have been encountered probably due to sludge loss following a rain event in the gravity thickener (Noutsopoulos et al., 2018).

3.1.3 Organic matter biodegradability in sidestreams

Table 1 shows COD biodegradable fractions in sidestreams based on literature review. For a given type of sidestreams, results show a large variability explained on the one hand by varying methods employed to characterise the biodegradability and on the other hand by differences in operating conditions of AD and THP (T°C, sludge concentration, retention time). The fractionation of COD into different classes of biodegradability (slowly biodegradable, rapidly biodegradable, inert soluble and particulate) is essential for design, operation purposes and modelling (Gillot and Choubert, 2010). However, more data are required to compare the sludge biodegradability after anaerobic digestion and THP anaerobic digestion sidestreams. Primary sludge thickening sidestreams has a biodegradable fraction similar to wastewater generally between 54 – 88 % (Gillot and Choubert, 2010) of total COD. The biodegradable fraction of biological sludge thickening sidestreams is similar to the one of treated wastewater which is generally 70 to 80% lower than the untreated wastewater.

3.1.4 Concentration of nitrogen species

For conventional mesophilic anaerobic digesters, the resulting ammonium (N-NH_4) concentration in sidestreams typically ranges from 400 mg/L to 1 300 mg/L (Metcalf & Eddy. Inc et al., 2003) which is in line with present data (Figure 3).

Variability of N-NH_4 can be linked to the quantity of wash water used in the dewatering unit (Metcalf & Eddy. Inc et al., 2003) and to AD operating conditions (sludge concentration, VS removal). N-NH_4 is significantly higher in anaerobic digestion with THP compared to AD without THP, which is explained by the fact that THP favours (1) an increase in the sludge concentration in the digester due to reduced viscosity (Urrea et al., 2015), (2) a higher solubilisation rate (Dwyer et al., 2008a; Wilson and Novak, 2009) and (3) an increase in biodegradation of organic matter and therefore of proteins content (Bougrier et al., 2008). Likewise, increased release of ammonium in the digester is expected. It was reported that total ammonium release per mass of volatile solids removed is equivalent for conventional mesophilic AD and THP whatever the sludge type (Wilson et al., 2011). Soluble nitrogen in AD dewatering sidestreams is mainly in the form of ammonium and soluble organic nitrogen accounts for a maximum of 10% of soluble TKN. Of this soluble organic nitrogen fraction, approximately 50% is considered non-biodegradable. The soluble organic nitrogen is believed to be produced through cell metabolism and decay of the anaerobic bacteria and waste activated sludge. Therefore, typical digester sidestream will add approximately 0.2 mg/L of refractory dissolved organic nitrogen to the WRRF effluent (Metcalf & Eddy. Inc et al., 2003). A recent survey on soluble organic nitrogen content in treated wastewater reported an average final concentration of 0.93 mg/L with a range of 0 – 2.5 mg/L (Galvagno et al., 2016). Therefore, the non-biodegradable fraction brought by AD dewatering sidestreams will account for approximatively 25% of final organic nitrogen. This quantity can be higher with THP pre-treatment (Ahuja, 2015).

As expected, N-NH_4 concentration in biological sludge and primary sludge sidestreams is significantly lower compared to anaerobic digestion sidestreams. However, the formation of anaerobic zones in thickening units such as gravity thickener or dissolved air flotation can lead to sludge hydrolysis thus favouring ions release, which can explain outliers of 50 mg/L for biological sludge sidestreams and 123 mg/L for primary sludge sidestreams (not clearly visible in Figure 3 due to the scaling).

3.1.5 Concentration of phosphate

Phosphate (P-PO_4) concentration for different sources of sidestreams is shown in **Erreur ! Source du renvoi introuvable.** 4. The concentration of phosphate in biological sludge and primary sludge sidestreams is significantly lower compared to anaerobic digestion sidestreams. Outliers in Figure 4 with very high concentrations of phosphate (up to 180 mg/L) for biological sludge sidestreams was attributed by Barat et al. (2009) to the formation of anaerobic zones in the thickener. This phenomenon is unlikely to happen in fast thickening processes such as centrifuge, rotary drum or belt press (Wild et al., 1997). In addition, as THP increases phosphorus solubilisation, a higher phosphate content in AD dewatering sidestreams was expected compared to conventional digestion (Khunjar et al., 2019). Surprisingly this was not supported by literature data. Whilst the intracellular phosphorus can be released during THP, this phosphorus can be directly immobilized by metallic ions such as Mg^{2+}, $Fe^{2+/3+}$, Ca^{2+} and Al^{3+} (Han et al., 2020). Consequently, the phosphorus content in the THP return liquor does not differ from the one of conventional AD. This result was found without considering the different types of applied phosphorus treatment because more data is needed to complete this comparative analysis.

To investigate the high phosphate concentration variability in anaerobic digestion sidestreams, Figure 5 shows phosphate concentration according to the phosphorus treatment type installed in the water line: enhanced biological phosphorus removal (EBPR), chemical phosphorus removal, a combination of both biological and chemical phosphorus removal and no specific phosphorus treatment. Only WRRFs operated with EBPR lead to higher phosphate in anaerobic digestion sidestreams compared to chemical phosphorus removal and no specific phosphorus treatment. The high phosphate concentration range for biological phosphorus removal and the combination of biological with chemical phosphorus removal can be attributed to different level of precipitation inside the anaerobic digester and different iron dosage.

3.1.6 Contribution to the inlet mass flows

The flow of the different sources of sidestreams accounts for less than 5% of the total flow at the WRRF inlet (Figure 6). Mass flow of total COD and TSS also represent less than 5% of the total mass flow at WRRF inlet for biological sludge and anaerobic digestion sidestreams. Primary sludge can excess this 5% threshold, especially after a rain event. As expected, the highest sidestream contribution for total phosphorus and total nitrogen comes from anaerobic digestion sidestreams. Regarding the nitrogen mass flow, sidestreams from anaerobic digestion contribute on average to 17% of the nitrogen mass flow at the WRRF inlet, which is much higher than the contribution of biological sludge sidestreams (1%) and primary sludge sidestreams (3 – 8%). The variability of the nitrogen mass flow is due to the the wide range of $N\text{-}NH_4$ concentration found in anaerobic digestion sidestreams and the different load of nitrogen at the WRRF inlet. Phosphorus mass flow depends on the type of phosphorus treatment implemented in the water line similarly to the concentration of phosphate in anaerobic digestion sidestreams (§3.1.5). For WRRFs with EBPR, anaerobic digestion sidestreams can contribute up to 34% of the total phosphorus mass flow. Mass flow of total phosphorus from primary sludge sidestreams and biological sludge sidestreams can exceed 5% especially when the thickening unit favours the formation of anaerobic zones (gravity thickener, dissolved air flotation).

3.2 Ionic composition of sidestreams

Figure 7 shows the concentration of different ions in sidestreams from anaerobic digestion. The concentration of aluminium (Al^{3+}) and iron ions ($Fe^{2+/3+}$) is very low because these metals precipitate easily with phosphorus or sulphur (Wilfert et al., 2015). Only a high salt dosage can lead to residual iron or aluminium in the soluble phase. The addition of salts lead to a large variability of chloride (Cl^-) concentration from one study to another. This variability can be explained by different dose of salts or chemicals applied within the WRRF but also during disinfection of potable water or in sewers (Howe et al., 2012). Calcium (Ca^{2+}), magnesium (Mg^{2+}), potassium (K^+), sodium (Na^+) and sulphate (SO_4^{2}) concentrations depend on tap water characteristics that result from the distribution network, water source and treatment, geographical location and geology (Hori et al., 2021). The higher the concentration at the WRRF inlet, the higher the concentration in sidestreams. A study on tap water characteristics also indicated a positive correlation between Na^+ & Cl^- and between Ca^{2+}, Mg^{2+} & alkalinity (Banks et al., 2015). Positive correlations between Ca^{2+} & Mg^{2+} and Na^+ & Cl^- have also been found in anaerobic digestion sidestreams with the dataset used in the present study (p value < 5%).

Table 2 shows typical ion concentrations for domestic wastewater. The concentration range are systematically more extensive in sidestreams with values that can be very different compared to wastewater. Sulphate concentration is lower in sidestreams compared to wastewater because sulphur is usually removed from wastewater to prevent H_2S formation in the digester and it can also be stripped as H_2S gas. For the other ions, the concentration in sidestreams may results from (1) the addition of industrial wastewater to be treated in the urban WRRF, (2) their accumulation in sludge and subsequent reaction of precipitation / dissolution in the digester and in the thickening or dewatering unit and (3) the addition of chemicals such as lime. For example, calcium concentration in wastewater can increase up to 500 – 1500 mg/L with industrial wastewater (Arabi and Nakhla, 2008). Besides, calcium, potassium and magnesium are present in wastewaters as organic or inorganic forms.

These ions can accumulate in sludge and Ca^{2+} and K^+ removal of 23% and 38% in wastewater has been observed in a WRRF in Iran (Hosseinipour Dizgah et al., 2018). Therefore, during anaerobic digestion, there is a release of ions which can then lead to precipitation as phosphates and carbonates of calcium and/or magnesium (Marti et al., 2008). Consequently, even if wastewater characteristics have an impact on the quantity of ions in sidestreams, the concentration of ions throughout the plant can vary. As an example, a WRRF performing nutrient removal reported Ca^{2+}, Mg^{2+} and K^+ concentrations of respectively 63, 12 and 284 mg/L in sidestreams; whereas, the concentration in wastewater was respectively of 131, 26 and 27 mg/L (Martí et al., 2017). In the latter example, Ca^{2+} and Mg^{2+} have a lower concentration in sidestreams compared to WRRF inlet but the opposite occurs for K^+ because K^+ do not precipitate in high extent in digester (Barat et al., 2009).

3.3 Alkalinity, pH and Temperature

Figure 8 presents alkalinity in sidestreams from anaerobic digestion and THP anaerobic digestion. A significant difference between both types of sidestreams is observed. The high alkalinity concentration in THP anaerobic digestion is due to the retention of carbon dioxide in the digester bulk liquid to balance the positively charged ammonium ion at the typical pH range of the digesters (Metcalf & Eddy. Inc et al., 2003). The correlation between alkalinity and ammonia has been confirmed with the set of data (p value < 5%). In addition, the mass ratio alkalinity:N-NH_4 is similar between anaerobic digestion (0.24 ± 0.12) and THP anaerobic digestion (0.27 ± 0.2). The alkalinity in primary sludge and biological sidestreams (699 mg/L and 409 mg/L, respectively) are lower than anaerobic digestion sidestreams because of the lower ammonia concentration (§3.3).

The pH values from anaerobic digester sidestreams (not shown) ranged from 6.6 to 8.6 (median value of 7.8). This pH is in the high range of typical pH of digesters (6.5 – 7.5) (Paul and Liu, 2012). The low pH values can be due to CO_2 stripping in dewatering units (van Rensburg et al., 2003). pH in primary sludge and biological sludge sidestreams ranged from 6.3 to 7.6.

The temperature in anaerobic digestion sidestreams is comprised between 18°C and 27°C. This reflects the cooling between the outlet of the anaerobic digester (35 - 38°C) and the dewatering unit.

3.4 Heavy metals, micro-pollutants and pathogens

Sidestreams valorisation through a nitrogen or a phosphorus based-product is possible only if the latter complies with the regulations with reference to trace elements, including heavy metals, micropollutants and pathogens (Rey-Martínez et al., 2022). Indeed, heavy metals can be incorporated into the crystal, in case of struvite recovery, reducing the purity of the product (Muys et al., 2020; Uysal et al., 2010). In Table 3, heavy metal concentrations in sidestreams are compared with their concentrations in wastewater. The concentrations in sidestreams can be higher than in wastewater due to the accumulation of metals in sludge. In terms of fluxes, a study reported that AD dewatering sidestreams load contributed to 10%, 10%, 15%, 10%, 5%, 10%, 10% of the load entering the WRRF for Cd, Cr, Cu, Hg, Ni, Pb and Zn, respectively (Yoshida et al., 2015).

For organic micropollutants, information of their concentration in sidestreams have been found in the work of Yoshida et al. (2015) and Uysal et al. (Uysal et al., 2010) for the following species: bis(2-ethylhexyl)phthalate (DEHP), polychlorinated biphenyl (PCBs) and polycyclic aromatic hydrocarbons (PAHs)). The AD dewatering sidestreams load contribute to 15%, 5%, 30% of the load entering the WRRF for DEHP, PCBs and PAHs, respectively.

4. How do sidestream characteristics impact the choice of a treatment/valorisation process?

4.1 Struvite precipitation

Struvite ($MgNH_4PO_4{\cdot}6H_2O$) is a white crystalline substance precipitating in a theoretical molar ratio of 1:1:1 (NH_4:PO_4:Mg) (Le Corre et al., 2009). Struvite precipitation is usually used to recover phosphorus in AD dewatering sidestreams when P-PO_4 concentration is above 50 mg/L (Wu and Vaneeckhaute, 2022). This process was implemented first to help reduce struvite clogging issues in pumps and pipes (Kleemann et al., 2015). The different technologies available on the market generally use a fluidised bed column reactor due to different solid and hydraulic retention time as well as to facilitate the recovery of struvite (Ghosh et al., 2019).

Even if the performance of P recovery through struvite in sidestreams is over 90% (Jaffer et al., 2002; Münch and Barr, 2001; Parsons et al., 2001; Yoshino et al., 2003), the overall plant-wide efficiency is lower than 50% (Muys et al., 2020). The recovered struvite can be applied directly to the field as a slow-release fertiliser if permitted and proven to be a favourable option for agricultural use (Melia et al., 2017). The pH at which struvite may precipitate is one of the main factors influencing the crystallisation process (Le Corre et al., 2009). The pH in a struvite precipitation reactor is usually between 7.5 (Liu and Qu, 2017) and 9.5 (Daneshgar et al., 2019) with optimal conditions around 8.5 (Münch and Barr, 2001). As the pH in AD dewatering sidestreams is from 6.6 to 8.9 (§3.6), adding sodium hydroxide to adjust the pH above 8 can be required. Because Mg:P ratio from 1 to 2 enhances the degree of supersaturation (Desmidt et al., 2013), Mg is generally added to the precipitation reactor as magnesium chloride or magnesium oxide (Münch and Barr, 2001; Xavier et al., 2014). Consumption of high quantities of magnesium can limit the economic interest of struvite recovery technologies (Astals et al., 2021). Likewise, low-cost magnesium source have been envisaged such as magnesium from nano filtration of seawater (Shaddel et al., 2020). However, this option also brings others ions so a good knowledge of sidestreams is paramount before adding new sources of impurities (Lahav et al., 2013).

Other parameters can affect the performance of struvite reactors. TSS above a concentration of 1000 mg/L (Barnes et al., 2007) has been reported to interfere with crystal growth by reducing the aggregation of crystals, hence their final size (Muys et al., 2020). TSS, even at a low concentration under 20 mg/L, can also adsorb to the surface of struvite crystals and decrease struvite purity (Desmidt et al., 2015; Ping et al., 2016). Not only TSS but VSS content can be responsible of a low phosphate removal efficiency because organic material can react with ions both on the media and on the surface of crystal nucleus (Ping et al., 2016; Tong and Chen, 2007). Other ions play a significant role in the purity of the final product. Indeed, amorphous calcium phosphate, brucite, magnesium phosphate, calcite, newberyite, K-struvite can precipitate in sidestreams (Musvoto et al., 2000).

The impact of Ca^{2+} in the purity of struvite depends on both Ca^{2+}:P-PO_4 and Ca^{2+}:Mg^{2+} molar ratio as well as the initial P-PO_4 concentration. For a Ca^{2+}: Mg^{2+} molar ratio over 0.5, struvite is heavily impacted by calcium and co-precipitated with amorphous calcium phosphate and calcium carbonate (Korchef et al., 2011; Tao et al., 2016), even sometimes for a Ca^{2+}:Mg^{2+} molar ratio between 0.2 and 0.5 (Yan and Shih, 2016) and from ratio over 1 no struvite can be formed (Le Corre et al., 2005). If initial concentration of phosphate is higher than 60 mg/L, there is no influence of calcium on struvite precipitation for a Ca^{2+}:Mg^{2+} molar ratio lower than 0.2. However, at concentration of phosphate lower than 40 mg/L, struvite was affected by the presence of calcium for every ratio of Ca^{2+}:Mg^{2+}. Consequently, calcium phosphate precipitation will be more interesting than struvite precipitation for wastewaters with low phosphate concentrations (Desmidt et al., 2013).

In addition, even if the heavy metals concentrations in sidestreams are very low (§3.7), they can accumulate in the minerals precipitated. A concentration of mercury in sidestreams of 0.426 mg/L can result in a concentration of 4.23 mg/kg dry matter in struvite (Uysal et al., 2010) exceeding the limit of 1 mg/kg dry matter (European Commission, 2019). As far as pathogens are concerned, struvite crystallisation selectively seem to exclude them, leaving them in the anaerobic digestion sidestreams (Muys et al., 2020).

4.2 Partial nitritation – anaerobic ammonium oxidation (PN-anammox)

The PN-anammox process is the most innovative worldwide applied technology for nitrogen removal in WRRFs in recent years. The technologies used at full-scale are single-stage or separate-stage systems that can be divided into 3 goups: moving bed biofilm reactor, granular sludge processes or sequencing bed reactor (Lackner et al., 2014). In comparison to conventional nitrification - denitrification and to nitritation - denitritation, this process does not require supplemental carbon addition, consumes less oxygen (1.9 kg O_2/kg N instead of 4.6 kg O_2/kg N for nitrification-denitrification) and has lower sludge production (Van Hulle et al., 2010). The process transforms a mix of NH_4^+ and NO_2^- into N_2 and a small quantity of NO_3^- according to the following equation (Ahn, 2006):

$$NH_4^+ + 1.32\ NO_2^- + 0.66\ HCO_3^- + 0.13\ H^+ \rightarrow 0.066\ CH_2O_{0.5}N_{0.15} + 1.02\ N_2 + 2.03\ H_2O + 0.26\ NO_3^-$$

Under anoxic conditions, anaerobic ammonium oxidising bacteria (anammox) can oxidize ammonium to molecular nitrogen, using nitrite as the final electron acceptor and CO_2 as carbon donor. The doubling time of anammox bacteria is about 10-12 days at 35°C (Talan et al., 2021); therefore reactors providing high biomass retention time such as biofilm reactors are often used. This process has been applied worldwide particularly in sidestreams from anaerobic digestion because these warm effluents with high nitrogen and low carbon content comply with optimal growth conditions of anammox bacteria (Kartal et al., 2010). However, process instability has been noted in connection with high or varying TSS concentration. Increased TSS load affects sludge withdrawal and consequently active biomass content in the reactor (Lackner et al., 2014).

Inhibitions by soluble, colloids and particulate COD has also been reported in literature (Arora et al., 2021; Jin et al., 2012, 2016; Lackner et al., 2014; Talan et al., 2021). A low COD:N-NH_4 ratio, generally lower than 2 (Lackner et al., 2014), is recommended for the operation of the PN-anammox process. Indeed, for a COD:N-NH_4 ratio higher than 2, inhibition has been reported in a laboratory-scale reactor with particulate COD concentration as low as 300 mg/L (Chamchoi et al., 2008) which is expected in sidestreams from anaerobic digestion (§3.2). Particulate COD and colloidal COD were identified as the main inhibitory parameters that decreased aerobic ammonium oxidising bacteria (AOB) rate. Under high levels of colloidal matter, oxygen transfer efficiency decreased, resulting in limited dissolved oxygen availability and consequently a poor nitrification performance. This was resolved by improving the dewatering process through an optimised polymer dosing to capture the colloidal fraction. No decrease of anammox activity was observed during operation of the reactor as long as the soluble COD concentration remained below 2500 mg/L (Zhang et al., 2016). This value corresponds to a COD:N-NH_4 ratio of 2.4 and creates inhibition of anoxic ammonium oxidizing bacteria due to competition with denitrifiying bacteria. Such competitions between heterotrophic bacteria that outcompete both AOB and anammox bacteria has been described by different authors, especially with THP and high COD content (Baeten et al., 2019; Molinuevo et al., 2009). To avoid such phenomena, a dilution was suggested to maintain a constant soluble COD concentration in the process and to decrease toxicity effects from refractory compounds especially for anaerobic digestion sidestreams preceded by THP that can reach elevated COD concentrations (§3.2) (Driessen et al., 2020). The potential drawback of a higher dilution is the temperature drop (Zhang et al., 2016) below anammox optimal temperature growth rate of 30 – 40 °C (Shao et al., 2019).

A pH of 7–8 was reported to be suitable for anammox activity (Talan et al., 2021) and in range for avoiding inhibition by free ammonia and free nitrous acid. The inhibition at high pH is caused by the increase of free ammonia; however, a low pH value enhances free nitrous acid inhibition. A free ammonia concentration of 20 mg/L (Fernández et al., 2012) or even lower during process startup (Jung et al., 2007) can cause instability of the process (Figdore et al., 2011). This concentration is expected at a temperature of 27 °C, pH = 7.8 and a concentration of N-NH_4 of 810 mg/L (§3.3 and §3.7). Dilution and pH control is one solution to stabilise the operation of deammonification processes (Lackner et al., 2014; Ochs et al., 2021). A gradual start up for biomass acclimation is nevertheless possible up to 150 mg N-NH_3/L (Aktan et al., 2012).

Inhibition of Anammox activity by phosphate was reported for a wide range of concentration: 57.6 mg/L (Jin et al., 2012); 235 mg/L (Yang et al., 2019), 310 mg/L (Arora et al., 2021), 475 mg/L (Eskicioglu et al., 2018). However, the underlying mechanisms are still under debate and are likely to vary depending on many parameters such as: phosphate concentration, aggregate state (flocculated or densified biomass), pH conditions; degree of acclimation of the biomass and duration of the inhibition test (short vs long-term), among others. According to Zhang et al. (2017), the formation of dihydrogen phosphate ion, under weakly basic conditions and high phosphate concentration, may be responsible for the inhibition of the enzymes of the anammox reaction. This effect seems less pronounced in granules due to their multi-layer structure and higher extracellular polymeric substances levels that act as a protective layer for anammox bacteria. Biologically induced precipitation of calcium phosphate was confirmed in P/NA granular sludge and could be an additional explanation for the higher tolerance to phosphate stress of granules compared to anammox flocs (Johansson et al., 2017).

Although essential nutrients (Ca, K, Fe, Mg, Mn, Co, Cu, Mo, Ni, Zn) are usually sufficiently available in digested sewage sludge reject liquors (Burgess et al., 1999), fulvic and humic-like organic substances generated by the THP process are known for binding metal-ions, possibly reducing the bioavailability of essential trace elements (Zhang et al., 2018). Table 4 shows that Fe, Cu, Al and Zn content in sidestreams do not always meet the minimum requirements for biomass growth. To ensure optimal biological activity and growth of the biomass micronutrients, essential trace metals are sometimes dosed to the anammox reactor, especially for anaerobic digestion sidestreams with THP (Driessen et al., 2020). Besides, it has been reported that the specific anammox growth rate could be significantly enhanced by adding ferrous oxide (Zhang et al., 2022). High concentrations of heavy metals can inhibit anammox activity as it is reported in Table 4 but such elevated concentrations are not likely to be encountered in municipal WRRF sidestreams.

Reduction of sulphate to H_2S often occurs in anaerobic digestion processes (Forouzanmehr et al., 2022) and in anammox-based system (Arora et al., 2021) inducing the presence of sulphide in sidestreams. In addition, one should also mention that sulphate can be biologically reduced to sulphide (Bi et al., 2020). The concentration of sulphide is mitigated by the formation of insoluble metal sulfide complexes in the anaerobic digester (Forouzanmehr et al., 2021). The intermediate sulphide produced biologically was reported in a review to inhibit anammox activity starting from a concentration of 32 mg/L (Jin et al., 2013).

4.3 Ammonia stripping

Stripping of ammonia lies on the liquid-gas equilibrium where ammonia from the liquid phase is transferred to a gas phase in a packed tower. The ammonia gas is then sent to an air scrubber for ammonia absorption to an acid, generally sulphuric acid, in order to recover a solution of ammonium sulphate ($(NH_4)_2SO_4$) (Boehler et al., 2015). This process has been applied at full-scale but it is generally not favourable from the energy and chemicals consumption point of view except for a niche market (Fernández-Arévalo et al., 2017; Shaddel et al., 2019). The main bottlenecks of this process are scaling and fouling of the packing material, and the consequent high energy and chemical requirements. To avoid that, removal of Ca, Mg, carbonates and TSS is required (Gopalakrishnan et al., 2000). For a concentration of TSS higher than 1000 mg/L, a separation solid-liquid is required before entering the process (Vaneeckhaute et al., 2018a). The minimum alkalinity is 4000 mg/L as $CaCO_3$ to satisfy the pH requirements by stripping out CO_2 without the use of chemicals as NaOH (Vaneeckhaute et al., 2018c). The quantity of Cl^- above 20 kmol/m^3 (which corresponds to 564 mg/L) negatively impacts the NH_3 removal efficiency because it decreases the pH, while increasing the ionic strength of the solution (Vaneeckhaute et al., 2018c). As discussed in §3.1, §3.5 and §3.6 these threshold levels can be reached in anaerobic digestion sidestreams and pre-treatment step is required when the conditions are not met for the correct operation of the process. Moreover, the minimum ammonia concentration for this process is 1000 mg/L to be economically viable (Wu and Vaneeckhaute, 2022). One study shows lower disadvantages by applying a thin film evaporator directly on digested sludge (Costamagna et al., 2020).

4.4 Emerging processes

Three emerging technologies are briefly presented in the following section, together with the main sidestream characteristics that may impact their performance according to literature data.

4.4.1 Membrane

Hollow-fiber membrane contactor is a promising technology for N recovery. In this system, ammonia passes through a microporous hydrophobic membrane and a sulfuric acid solution is used as draw solution to recover N as a valuable product (Robles et al., 2020). This technology has been applied at full-scale in only one WRRF (Seco et al., 2018; Richter et al., 2020) but the presence of suspended solids and colloidal materials can make the use of membrane-based technologies for separate treatment of AD sidestream difficult due to membrane fouling (Eskicioglu et al., 2018; Metcalf & Eddy. Inc et al., 2003; Wäeger-Baumann and Fuchs, 2012). Nevertheless, the optimisation of the materials used and recent works have shown an application of membranes for nitrogen recovery directly in digested sludge (Rivera et al., 2022). Regarding the final nitrogen-based product obtained, it can be contaminated by other ions present in the original substrate; therefore membranes should be consider with additional steps (TSS, COD and foreign ions removal) to obtain a pure ammonia product (Darestani et al., 2017; Beckinghausen et al., 2020). Application of forward osmosis membrane or membrane distillation have not been found for sidestream application because of excessive fouling (Vu et al., 2019).

4.4.2 Electrodialysis, bioeletrochemical system & ion exchange resin

Electrodialysis process uses an electric current to migrate ions to the cathode or anode and trap them on ion exchange membranes. A concentrated ammonia or phosphate solution is obtained (Ward et al., 2018). In bioelectrochemical systems, the oxidation of organics produces electrons used as energy for the migration of NH_4^+ ions from the anode to the cathode in order to maintain charge neutrality (de Fouchécour et al., 2022). In the cathode chamber, NH_4 is transformed into NH_3 thanks to the high pH value to be recovered (Nancharaiah et al., 2016). Ion exchange systems use resins which can exchange an ion adsorbed on the resin surface with a specific cation or anion in the centrate (Huang et al., 2020). Performance of these systems depend on electrode, membrane and resin fouling because high calcium, magnesium, TSS and carbonate can lead to significant deposit as calcium carbonate, struvite or accumulation of colloidal particles (Beckinghausen et al., 2020; Feng et al., 2017; Mondor et al., 2009). For the bioelectrochemical system, the low COD content in sidestreams limits its development at full scale and this technology is best suitable for wastewater rather than centrate (Al-Sahari et al., 2021). Phosphorus adsorption is also highly dependent of pH value because it affects the surface charges of the absorbent. The co-existing of different ions such as SO_4^{2-}, NO_3^- and Cl^- may inhibit P adsorption due to ions completion for the vacant adsorption sites (Song and Li, 2019; Ye et al., 2017).

4.4.3 Algae production

Microalgae-based wastewater treatment systems can be used for the removal of organic and inorganic carbon as well as for nutrients from wastewater (AlMomani and Örmeci, 2020). The main interest of this process lies in the production of a high growth rate algae biomass for biogas or biofuel production (Romero Villegas et al., 2017). The algae biomass has low carbon requirement which can be attractive for anaerobic digestion sidestreams treatment (Peralta et al., 2019). However, high content of TSS can negatively impact the growth of microalgae biomass because algal biomass can then compete with other bacteria for N, P and alkalinity (Marazzi et al., 2019). Another limit of this process is the design of the algae culture system as harvesting the biomass produced is still a challenging step (Zhao et al., 2018).

5. Discussion

The selection of processes for nutrient treatment/recovery of anaerobic digestion sidestreams have been discussed based on current published information on their operation. Although of high importance, current literature review revealed that it is challenging to define operational limits of the processes with regards to sidestream characteristics based on full-scale data. Published data are indeed scarce, and most of the time only limited characteristics were evaluated or investigated. In addition, it is likely that reported threshold values for sidestream characteristics leading to a decrease of process performance or to an inhibition embed the effect of other operational parameters. It is therefore not excluded that process configuration, the way it is operated as well as the exposition time and the acclimation of the biomass to an inhibitor highly affects the range of reported values. This is especially critical when limited data are available.

Based on literature data, Figure 9 presents the most important parameters to be measured before choosing a specific process. Comparison of the ranges of concentrations found and the list of sidestream characteristics shows that the range of concentrations from literature is large in comparison to the inhibition mentioned in the previous sections. Consequently, installation of a pretreatment step for TSS or COD removal for example can be required to ensure stable process performance.

Nutrient recovery in sidestreams, and especially in anaerobic digestion sidestreams seem promising but a well-defined product with high purity is required if recovery as fertiliser is considered. For example, organic farmers have a need for a pure nitrogen fertilizer, rather than a combination that includes P or K (Beckinghausen et al., 2020). As seen in previous paragraphs other ions and organic matter can impact the purity of the final recovered product or the efficiency of the process to obtain these products. Therefore, more investigations have to be carried out on the feasibility of producing a product with higher purity considering the full ionic composition of AD dewatering sidestreams (Shaddel et al., 2019). THP can, in addition to improving the performance of anaerobic digestion, improve the possibilities of recovering nitrogen because the N-NH_4 content is higher in sidestreams compared to a conventional digester sidestream. However, this does not apply for P which is released as phosphate upstream of the AD through the THP but then precipitates in the digester with Ca^{2+} and Mg^{2+}. The potential for P recovery upstream of AD needs to be more expanded as only a few examples exist to date (Bouzas et al., 2019). Nutrient recovery from primary sludge and biological sludge sidestreams has not been extensively explored. Indeed, the concentrations of phosphorus and nitrogen are lower than in the anaerobic digestion sidestreams. However, nutrient content in these sidestreams can be interesting for the recovery in some specific cases; especially when there is a thickening unit with high retention time as the release of ions is stimulated. In such specific context, sidestreams can be joined together and the process installed on this sidestream combination (Latimer et al., 2016). Overall, there is not always local demand for nitrogen or phosphorus fertiliser (Kehrein et al., 2020; Robles et al., 2020) and nutrient recovery technologies need a large quantity of energy and chemicals which induces higher environmental impacts (Pradel and Aissani, 2019).

Future research should focus on the definition of evaluation criteria that take into account the WRRF performance, effluent quality and operation costs, but also the environmental impacts, the efficiency of the recovery process and purity and the destination of the final product. To do so, there is an urgent need to develop shared databases with updated information on recovery processes including performance and operating conditions because there is a lack of information on recovery processes in real conditions (Puchongkawarin et al., 2015). Plant-wide modelling could also help in the choice and comparison of different routes to treat or valorise sidestreams. Even if some models already include precipitation as struvite (Lizarralde et al., 2019) or ammonia stripping (Vaneeckhaute et al., 2018b), they still need to be validated not only for the targeted nutrient but also considering the different compounds that can interact (Mg^{2+}, Ca^{2+}, $Fe^{2+/3+}$, SO_4^{2-}, K^+, Cl^-). Access to data from full-scale measurement campaigns is therefore essential in order to understand the full ionic distribution in different locations in the WRRF and to integrate new mechanisms into existing models.

6. Conclusions

Sidestream processes are increasingly being optimised to mitigate their effects on the water treatment line but also to recover nutrients in water resource recovery facilities. The implementation of such processes depends on sidestream characteristics. Ranges of concentrations of the main components observed in sidestreams at full scale have been discussed. To aid in the development, design and optimum operation of sidestream processes, this critical review identified the following key points:

1) Anaerobic digestion sidestreams contribute significantly to the nitrogen (17%) and phosphorus (11%) mass flows at the WRRF inlet. The quantity of phosphate in sidestreams depends on the type of applied phosphorus treatment with a median value of 33 mg/L for chemical phosphorus removal and of 167 mg/L for enhanced biological phosphorus removal.

2) The concentration of COD, N and alkalinity are higher in THP anaerobic digestion sidestreams than conventional anaerobic digestion. However, there is no difference in the phosphorus content because the phosphate release during THP is directly immobilised by others ions (Ca^{2+}, Mg^{2+}, $Fe^{2+/3+}$). Phosphorus recovery before anaerobic digestion should be considered in the presence of THP, and the quantity of Ca^{2+}, Mg^{2+} and $Fe^{2+/3+}$ should be quantified to assess the potential for P recovery.

3) The variability of ion concentrations (Al^{3+}, Ca^{2+}, Cl^-, $Fe^{2+/3+}$, Mg^{2+}, K^+, Na^+, SO_4^{2-}) depends on: (1) wastewater characteristics in particular the presence of industrial wastewater, (2) the use of chemicals such as iron chloride or lime and (3) dissolution and precipitation mechanisms in thickening or dewatering unit. The latter should be further investigated to better assess the impact of the full ionic composition on nutrient treatment or recovery.

4) All the characteristics previously mentioned can have an impact on sidestream processes. However, the definition of a concentration range or threshold value to ensure the successful operation of these processes is not a straightforward task. Indeed, the information about inhibitions is sparse in the literature and depends on a lot of different parameters (reactor configuration, scale, biomass acclimatisation, operating conditions). This review provides a list of characteristics to be measured in order to select the most suitable sidestream process for each specific application.

5) Future research should focus on further data acquisition, especially on the concentration of the different ions, to better assess the potential for nutrient recovery and to minimise the economic and environmental impact of WRRFs.

Credit author statement

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Declaration of Competing Interest

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

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Tables

Table 1: COD fractionation in sidestreams based on literature data

Method	Type of sidestream	Biodegradable fraction (%total COD)	Reference
Ultimate BOD test	Conventional AD	55%	(Akhiar et al., 2017)
Respirometry	Conventional AD	14%	(Im and Gil, 2019)
N/A	Conventional AD	15%	(Liu et al., 2014)
Respirometry	Conventional AD	57%	(Noutsopoulos et al., 2018)
Continuous aeration of centrate	THP + AD	27%	(Gupta et al., 2015)
Respirometry	THP + AD	40%	(Noutsopoulos et al., 2018)
Respirometry	Primary sludge	82%	(Noutsopoulos et al., 2018)
Respirometry	Biological sludge	15%	(Noutsopoulos et al., 2018)

Table 2: Typical ion concentrations in wastewater

Ions (mg/L)	Concentration range in wastewater (mg/L)	Reference
calcium	20 – 120 but with industrial wastewater 500 – 1500 mg/L	(Arabi and Nakhla, 2008)
magnesium	5 – 74 mg/L	(Barat et al., 2009; Wilfert et al., 2016)
potassium	11 – 32 mg/L	(Metcalf & Eddy. Inc et al., 2003)
sodium	50 – 250 mg/L	(Arienzo et al., 2009)
sulphate	24 – 72 mg/L	(Metcalf & Eddy. Inc et al., 2003)

Table 3: Heavy metals concentration in anaerobic digestion sidestreams compared to concentrations in wastewater

	Ranges in sidestreams	References	*maximum values in wastewater from 64 WRRFs* *(Vriens et al., 2017)*	*average values in wastewater from 6 WRRFs* *(Choubert et al., 2011)*
Cadmium (Cd)	0 – 0.017	(Ebbers et al., 2015; Karwowska et al., 2016)	0.000365	0.00020
Copper (Cu)	0.025 – 0.1477	(Bohutskyi et al., 2015; Ebbers et al., 2015; Karwowska et al., 2016; Ledda et al., 2015; Romero Villegas et al., 2017; Shao et al., 2019; Zhao et al., 2018)	0.052	0.054
Nickel (Ni)	0.01 – 0.2	(Ebbers et al., 2015; Karwowska et al., 2016; Shao et al., 2019; Zhao et al., 2018)	0.067	0.0103
Lead (Pb)	0.02 – 0.11	(Ebbers et al., 2015; Karwowska et al., 2016; Zhao et al., 2018)	0.0019	0.0065
Zinc (Zn)	0.014 – 0.28	(Bohutskyi et al., 2015; Ebbers et al., 2015; Karwowska et al., 2016; Ledda et al., 2015; Romero Villegas et al., 2017; Shao et al., 2019; Uysal et al., 2010; Zhao et al., 2018)	0.059	0.137
Mercury (Hg)	0.426	(Uysal et al., 2010)	-	0.00040
Chromium (Cr)	0.01 – 0.0834	(Shao et al., 2019; Uysal et al., 2010; Zhao et al., 2018)	0.0036	0.00109

Table 4: Comparison of trace elements requirements to inhibiting concentrations and concentrations found in anaerobic digestion sidestreams

Trace elements	Requirements *(Burgess et al., 1999)*	Inhibiting concentrations *(Zhang et al., 2022)*	Concentrations in sidestreams from Table 2 and Table 3
Ca	0.4 – 1.4	-	10 – 321
K	0.8 – 3	-	17 – 626
Fe	0.1 – 0.4	-	0.04 – 32
Mg	0.5 – 5.0	-	1 – 94
Mn	0.01 – 0.05	-	-
Cu	0.01 – 0.05	4.2	0.01 – 0.1477
Al	0.01 – 0.05	-	0.1 – 16
Zn	0.1 – 1	6.76	0.014 – 0.28
Mo	0.1 – 0.7	-	-
Co	0.1 – 5	-	-
Cd	-	7	0 – 0.017
Cr	-	8.96	0.010 – 0.198
Hg	-	2.3	0.42
Ni	-	3.6	0.005 – 0.2
Pb	-	4.3	0.03 – 0.11

Figures

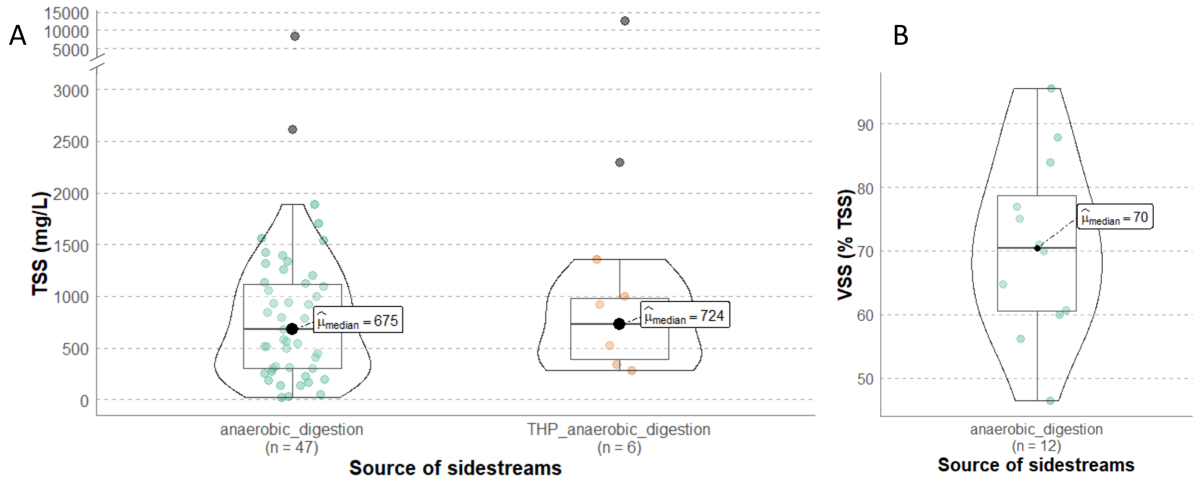


Figure 1: Literature data compilation of TSS (A) and VSS (B) in sidestreams from anaerobic digestion and from anaerobic digestion preceded by a THP

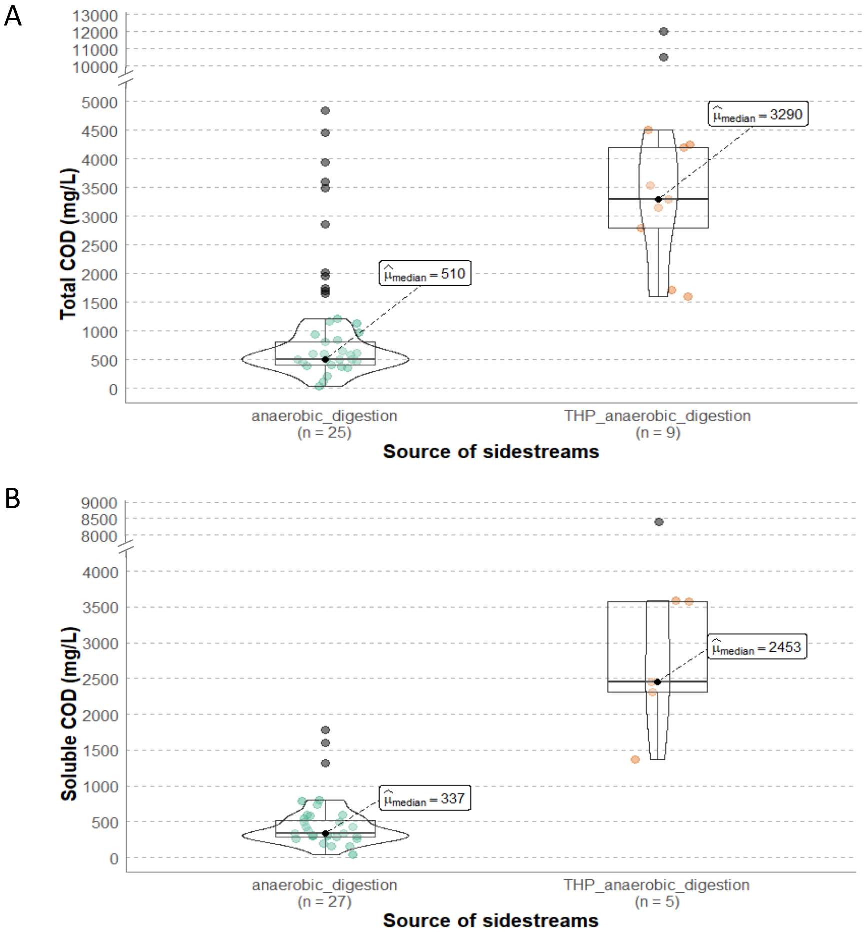


Figure 2: Literature data compilation of Total COD (A) and Soluble COD (B) in sidestreams from anaerobic digestion and from anaerobic digestion preceded by a THP

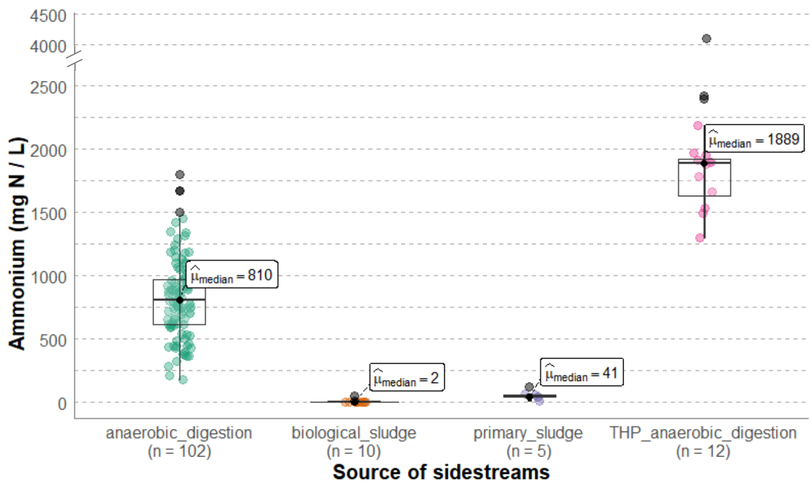


Figure 3: Literature data compilation of ammonium ion concentrations in sidestreams from different sources of sidestreams

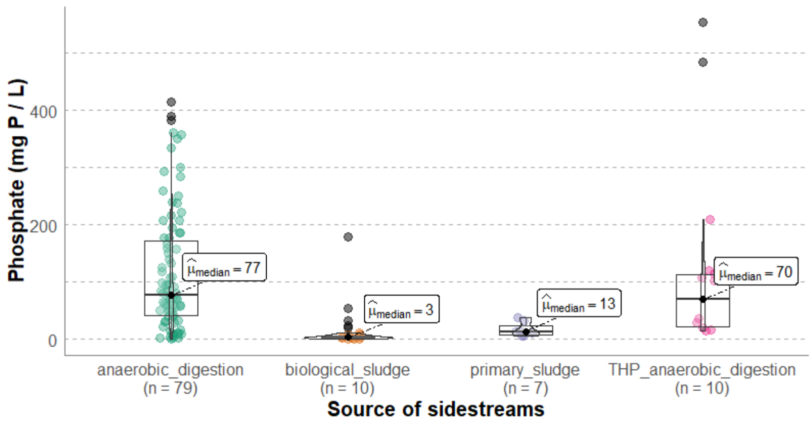


Figure 4 : Literature data compilation of phosphate concentrations in sidestreams from different sources of sidestreams

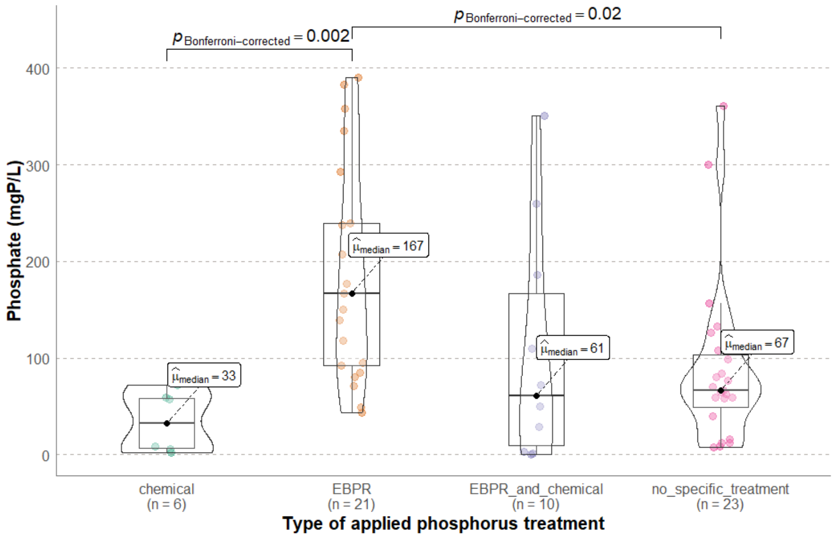


Figure 5: Literature data compilation of phosphate concentrations in anaerobic digestion sidestreams for different type of phosphorus treatment

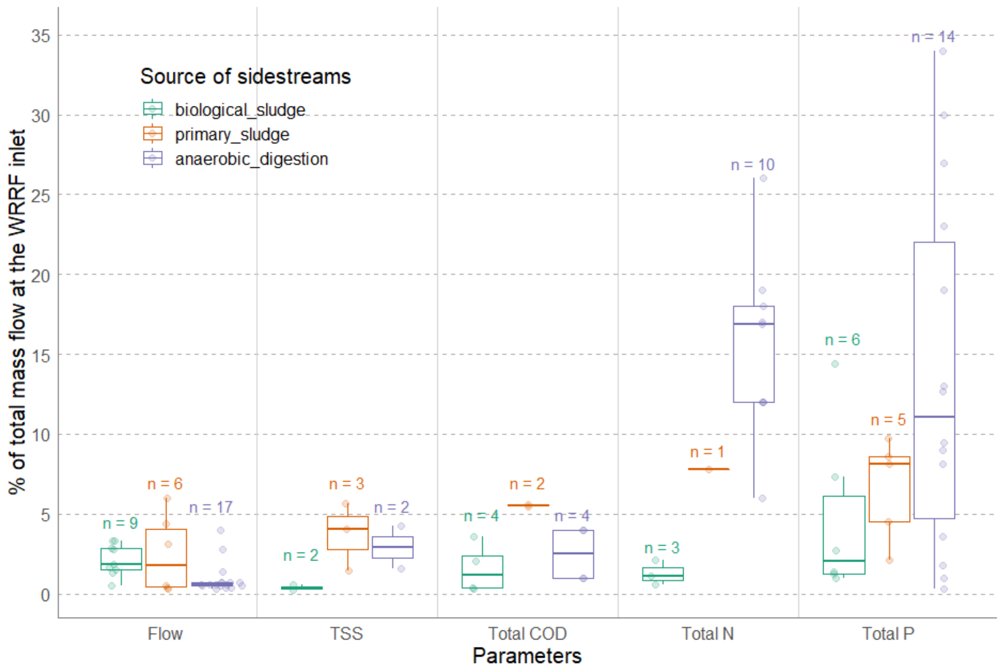


Figure 6: Literature data compilation showing the contribution of the different sidestreams to the WRRF inlet mass flows

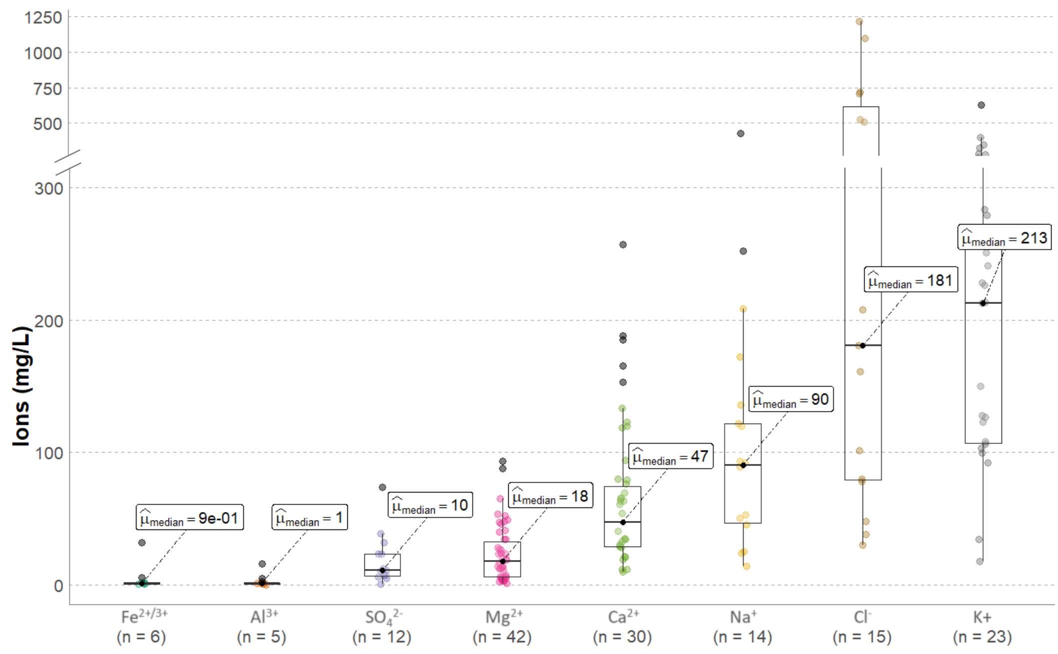


Figure 7 : Literature data compilation of the different ions in anaerobic digestion sidestreams

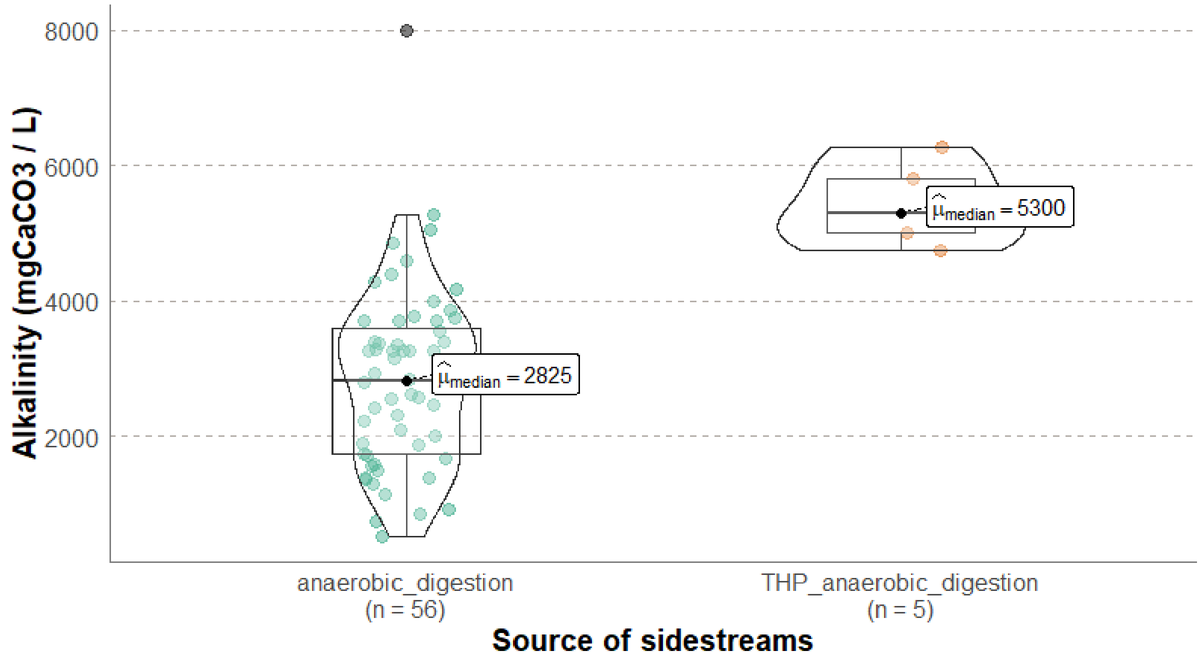
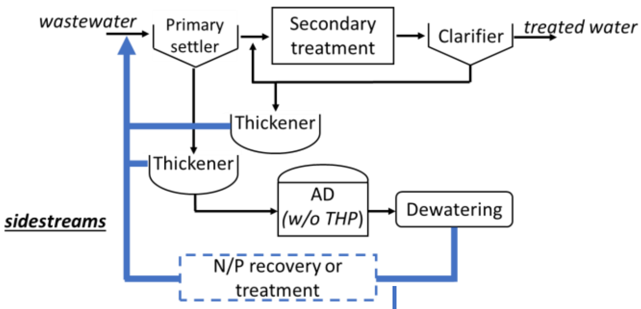


Figure 8: Literature data compilation of alkalinity in sidestreams from anaerobic digestion and THP anaerobic digestion



- TSS = [18 – 12600] mg/L
- pH = [6,6 – 8,9]
- T°C = [18,4 – 26,8] °C
- sCOD = [40 – 8400] mg/L
- $P\text{-}PO_4$ = [0,14 – 554] mg/L
- Alkalinity = [530 – 8000] mg $CaCO_3$/L
- Cl^- = [30– 1217] mg/L
- $N\text{-}NH_4$ = [175 – 4100] mg/L

List of characteristics to be measured before implementation of the process

Process	Characteristics
Struvite precipitation	TSS, pH, $P\text{-}PO_4$, Ca^{2+}:Mg^{2+}
Partial Nitritation / Anammox	COD:$N\text{-}NH_4$, $P\text{-}PO_4$, pH, T°C
Ammonia stripping	Alkalinity, Cl^-, $N\text{-}NH_4$
Membrane	TSS, $N\text{-}NH_4$, $P\text{-}PO_4$, Ca^{2+}, Cl^-, Mg^{2+}, K^+, Na^+, SO_4^{2-}
Electrodialysis, BES & IEX resin	TSS, Alkalinity, $P\text{-}PO_4$, $N\text{-}NH_4$, Ca^{2+}, Mg^{2+}
Algae production	TSS, $N\text{-}NH_4$, $P\text{-}PO_4$

Figure 9: Main characteristics of anaerobic digestion sidestreams and list of key characteristics for sidestream processes implementation