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Evaluation of conceptual model and predictors of faecal sludge dewatering performance in Senegal and Tanzania

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ABSTRACT

Unpredictable dewatering performance is a barrier to the effective management and treatment of faecal sludge. While mechanisms of dewatering in sludges from wastewater treatment are well understood, it is not clear how dewatering of faecal sludge fits into the framework of existing knowledge. We evaluate physical-chemical parameters, including EPS and cations, and demographic (source), environmental (microbial community), and technical factors (residence time) as possible predictors of dewatering performance in faecal sludge, and make comparisons to the existing conceptual model for wastewater sludge. Faecal sludge from public toilets took longer to dewater than sludge from other sources, and had turbid supernatant after settling. Slow dewatering and turbid supernatant corresponded to high EPS and monovalent cation concentrations, conductivity, and pH, but cake solids after dewatering was not correlated with EPS or other factors. Faecal sludges with higher EPS appeared less stabilised than those with lower EPS, potentially a result of inhibition of biological degradation due to high urine concentrations. However, distinct microbial community compositions were also observed in samples with higher and lower EPS concentrations. Higher EPS faecal sludge was comparable in dewatering behaviour and EPS content to anaerobically digested and primary wastewater sludges. However lower EPS faecal sludges had different dewatering behaviour than wastewater sludges and may be governed by different mechanisms.

1. Introduction

One third of the world's population relies on onsite sanitation facilities like pit latrines and septic tanks, and in low-income countries, less than 10% of urban areas are served by sewers (World Health Organization and UNICEF, 2017; Peal et al., 2014). The majority of cities in low-income countries do not have adequate faecal sludge management, with faecal sludge defined as what accumulates in onsite sanitation systems (Strande et al., 2014). In low-income countries, the majority of faecal sludge is discharged untreated into the urban environment, placing a huge burden on public and environmental health (Blackett et al., 2014; Cairncross and Feachem, 2019). Efficient treatment and management systems are needed to safely manage these quantities of faecal sludge, however unreliable solid-liquid separation is a major bottleneck (Gold et al., 2016; Cofie et al., 2006). Knowledge is needed to be able to predict and improve dewatering performance of faecal sludge prior to implementation of management solutions such as decentralized transfer stations, and to increase the capacity of existing faecal sludge treatment plants (FSTPs) (Gold et al., 2016; Strande et al., 2018). Solutions for improved dewatering performance are desperately needed to increase access to improved sanitation and make progress towards achieving the Sustainable Development Goals (SDGs).

Relatively little research has been conducted on faecal sludge treatment processes, as non-sewered sanitation has only recently been acknowledged as a long-term sustainable solution (USEPA, 2005; Strande et al., 2018; Strande et al., 2014). In contrast, centralized treatment processes such as activated sludge treatment have been researched for over one hundred years (Stensel and

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Makinia, 2014). Many expectations about mechanisms governing dewatering of faecal sludge have in the past been derived from centralized wastewater treatment (Gold et al., 2018a). However, faecal sludges are quite different from wastewater sludges; for example, faecal sludge can be comprised of any range of fresh excreta to products of anaerobic digestion from storage in containment, and can include soil, sand, and municipal solid waste (Van Eekert et al., 2019; Strande et al., 2014). In contrast, primary sludge is relatively fresh, not stabilised, easily settleable solids from raw wastewater (Tchobanoglous et al., 2014), and activated sludge is mainly composed of bacterial cells and metabolic products generated during aerobic secondary treatment (Nielsen et al., 2004). The metabolic products include high concentrations of extracellular polymeric substances (EPS) that are produced during biological growth (Bala Subramanian et al., 2010). EPS presents challenges for dewatering, as it is highly charged and binds water (Forster, 1983; Flemming et al., 1996). EPS has the secondary effect of reducing turbidity, as these charged polymer chains can also bridge particles together and form flocs when present in sufficiently high fractions (Mikkelsen and Keiding, 2002; Christensen et al., 2015). In wastewater sludges, solution properties that influence particle surface charge and EPS bridging, like pH and dissolved salts, play an important role in determining floc integrity and dewatering performance (Neyens et al., 2004).

The current state of knowledge for understanding dewatering behaviour in primary, activated and anaerobically digested (AD) wastewater sludges is that floc formation and disintegration are the major mechanisms governing supernatant turbidity, resistance to filtration and dewatering time (Christensen et al., 2015; Jørgensen et al., 2017; Novak et al., 2003). Degree of flocculation is generally highest in activated sludge with high fractions of EPS in the total suspended solids and low pH and monovalent cation concentrations (Christensen et al., 2015). Increasing monovalent cation concentrations can lead to floc disintegration via disruption of divalent cation bridges, and increasing pH makes EPS and sludge surfaces more electronegative, generating electrostatic repulsion within flocs (Christensen et al., 2015). Floc disintegration releases organic matter, including EPS, into bulk solution; as a result, soluble and loosely bound EPS concentrations have been correlated to slow dewatering (Yu et al., 2008; Lei et al., 2007; Novak et al., 2003). In sludges with high EPS fractions, like activated sludges, EPS binds water in the flocs through electrostatic interactions and hydrogen bonds, making it difficult to achieve high cake solids after dewatering (Neyens et al., 2004). As EPS in activated sludge is degraded (e.g. via anaerobic digestion or thermal/chemical hydrolysis), floc strength generally weakens, increasing turbidity and dewatering time (Novak et al., 2003), however destruction of EPS reduces bound water, allowing higher cake solids to be achieved (Mikkelsen and Keiding, 2002; Neyens et al., 2004). As a result, EPS fraction in the suspended solids remains the best predictor of how much water can be removed during dewatering, along the entire range of wastewater sludges produced from different digestion regimes (Skinner et al., 2015). Therefore, an emphasis in wastewater sludge treatment processes has been identifying conditions where sludge flocculates well during secondary treatment, followed by steps to reduce water-binding EPS concentrations in the sludge (e.g. anaerobic digestion) (Skinner et al., 2015; Mikkelsen and Keiding, 2002; Christensen et al., 2015; Neyens et al., 2004; Katsiris and Kouzeli-Katsiri, 1987).

Efforts to transfer technologies such as conditioners and mechanical dewatering from wastewater to faecal sludge have been largely unsuccessful due to highly variable and erratic performance (Moto et al., 2018; Heinss et al., 1999; Whitesell, 2016; Taylor, 2016; Ziebell et al., 2016). This is because faecal sludge has a wide range of stabilization, can be much more concentrated (0.5–20% TS), and is up to two orders of magnitude more variable in all characteristics (Gold et al., 2018a). Faecal sludge is highly variable due to differences in individual patterns of usage (e.g. flush toilet, grey water addition), the wide range of containment technologies (e.g. lined or unlined), and emptying frequencies (e.g. from days to years) (Strande et al., 2014). In addition, faecal sludge is collected individually, batch-wise from onsite containments (e.g. households, public toilets, or commercial enterprises) and transported to treatment, whereas wastewater is relatively more homogenized during transport in a sewer (Strande et al., 2014; USEPA, 1984). As a result of higher influent variability, faecal sludge dewatering performance is more variable compared to wastewater primary, activated, and AD sludges (Gold et al., 2018a). Solid-liquid separation technologies are currently primarily limited to settling-thickening tanks and drying beds, which have relatively large footprints and can take up to weeks or months to dewater faecal sludge to 60% TS (Strande et al., 2014).

Empirical and qualitative field observations indicate that public toilet sludge takes longer to settle and dewater than faecal sludge from households, and has higher effluent turbidity (Cofie et al., 2006; Heinss et al., 1999). It has been suggested that this is due to differing degrees of stabilization (i.e. the extent of biodegradation of organic material), although it is not yet clear how stabilization is linked to dewatering performance in faecal sludge. However, it has also been observed that type of containment (i.e. pit latrine or septic tank), is a stronger predictor of physical-chemical characteristics than source (i.e. household or public toilet) (Strande et al., 2018). Relationships between surface charge and conductivity to dewatering time have been observed in faecal sludge (Gold et al., 2018a). These relationships could also be influenced by EPS concentrations, although characterization of EPS in faecal sludge has not been reported in the literature.

The objective of this research was to evaluate how dewatering of faecal sludge fits into the state of knowledge of wastewater sludges based on EPS and physical-chemical properties, and to evaluate whether demographic (e.g. source), environmental (e.g. microbial community), and technical factors (e.g. residence time) contributing to variability of faecal sludge can be used as predictors of dewatering performance.

2. Materials and methods

2.1. Sample collection

A total of 25 faecal sludge samples were collected: 20 from Dakar, Senegal and 5 from Dar es Salaam, Tanzania. A variety of typical faecal sludge sources was represented, including 7 households, 6 schools, 5 public toilets, 3 offices, 2 restaurants, and 2 houses of worship. Containments in need of emptying were preselected for sampling. Specific vacuum trucks were arranged to empty only the containment at the selected site to avoid sampling a mixture of sludge from different containments. Members of our team accompanied the trucks during emptying, transport, and discharge at the FSTP. 4 L composite samples were then collected during discharge from each vacuum truck: 1 L at the start of discharge, 2 L in the middle, and 1 L near the end following the sampling protocol outlined in Bassan et al., (2013). Following collection, samples were immediately stored in a cooler with ice before being transported to the laboratory for analyses. Samples were stored for maximum 1 week at 4 °C between collection and analysis, with the exception of 3 weeks maximum storage time before total solids quantification. Microbial community samples were preserved with RNAlater and stored at −20 °C before analysis.

Questionnaires were administered to the person responsible for the containment system, and to the vacuum truck operators.

Questions about the containment type, construction and age, toilet flush type and use patterns, number of users per day, additional inputs to the containment (e.g. greywater, solid waste), and emptying frequency were collected. The time since last emptied was used to estimate the residence time of the faecal sludge in containment. In Dakar, all containments included in the sampling campaign were identified as “septic tanks”, multi-chambered tanks lined with concrete or brick, but designed without effluent outflow in response to local regulations. In Dakar, all respondents reported using pour-flush and/or flush toilets and practicing anal cleansing with water. Only the two restaurants reported toilet paper as a secondary anal cleansing option. Containments have no or limited liquid drainage, and most systems required emptying every several weeks to several months. Schools were the exception, emptied less than once per year. In Dar es Salaam, containment type was varied: two pit latrines, one cesspit, one septic tank, and one septic tank effluent storage pond. A full account of questionnaire questions and responses are available in SI.

2.2. Sample analysis

2.2.1. Physical-chemical characteristics

Samples were characterized for total and volatile solids (TS and VS), total and volatile suspended solids (TSS and VSS), electrical conductivity (EC), and pH according to the standard methods (APHA, 2005). Concentration (mg/L) of soluble mono- and divalent cations (Na^+, K^+, and Mg^{2+}, and Ca^{2+}) was determined using ICP-OES of the filtered supernatant (0.45 μm) after centrifugation at 3,345×*g* for 10 min (Park et al., 2006b). Prior to analysis, samples were acidified using 65% nitric acid. Monovalent/divalent (M/D) cation ratio was calculated based on the measured ion concentrations as $(Na^+ + K^+)/(Mg^{2+} + Ca^{2+})$.

2.2.2. Extracellular polymeric substances (EPS)

Polymeric substances were extracted from faecal sludge samples by sonicating 40 mL at 30 W (0.75 W/mL) in an ice bath for 2 × 2 min with a 30 s rest period using a Bandelin Sonopuls HD3100 with an M76 probe based on the procedure described by (D'abzac et al., 2010; Ras et al., 2008). Sonication intensity and duration was optimized to maximize polymeric substance extraction and minimize cell lysis, to avoid extraction and characterization of intercellular material. Lysis was monitored at various sonication settings using soluble ATP measurements (Promega BacTiter-Glo assays, luminometer) (Hammes et al., 2010). Samples were filtered (0.45 μm) and diluted with nanopure water (18.2 MΩ cm Milli-Q) prior to analysis. The extracted polymeric substances can be compared to what is termed “soluble and loosely-bound” EPS in most wastewater sludge and biofilm studies (Comte et al., 2006; Lin et al., 2014).

EPS concentration and fractionation were determined using size-exclusion chromatography-organic carbon detection-organic nitrogen detection (LC-OCD-OND) following the procedures outlined in (Stewart et al., 2013; Huber et al., 2011; Jacquin et al., 2017). Compounds were separated according to their size into five fractions with a size exclusion column (250 × 20 mm Toyopearl TSK HW-50S). Chromatograms were achieved using a phosphate buffer as eluent with a flux of 1 ml/min. Interpretation of the fractions was done using customized Fiffikus software (Huber et al., 2011). The column had a separation range of 100 Da to >20 kDa according to the supplier Tosoh Bioscience. Calibration with polysaccharides and proteins of different sizes showed that the separation range for polysaccharides was from 0.1 to 18 kDa, while for proteins the separation ranged from 0.5 to 80 kDa (Stewart et al., 2013). Assuming that nitrogen comprises 19% of the molecular weight of proteins (Torabizadeh, 2011), the carbon/nitrogen ratios indicated that the biopolymeric fraction of organic carbon was essentially entirely composed of proteins. Following the analysis procedure of (Jacquin et al., 2017), EPS was fractionated into protein-like substances (biopolymer peak), and humic-like substances (humic acids and building blocks peaks). Total EPS was reported as the sum of protein-like and humic-like substances. Data was adjusted from mg C/L to mg protein/L and mg humic acid/L using the following conversions: 0.53 g C/g protein (Rouwenhorst et al., 1991) and 0.54 g C/g humic acid (Allard, 2006), in order to allow for comparison with EPS data from wastewater sludges.

LC-OCD-OND was selected over more common colorimetric assays for the characterization of EPS due to the well-documented interferences that can be caused by the presence of likely-to-occur environmental compounds (including urea, uric acid, and humic acid) which are expected to appear in different concentrations in different faecal sludge samples (Le et al., 2016; Le and Stuckey, 2016).

2.2.3. Microbial community analysis

DNA extraction, amplification, reading, data cleaning, and analysis with 16S rRNA gene amplicon sequencing targeting bacterial and archaeal variable region V4 was conducted by DNASense ApS (Aalborg, Denmark). DNA extraction and library preparation yielded between 15672 and 39905 reads after quality control and bioinformatic processing. Relative abundance values were not adjusted by total cell count or extracted DNA concentration, as the variability in these values was lower than variability in solid-liquid separation performance, and minimization of false positive correlations was desired over the suppression of false negatives. Similarities of samples at the community level were visualized using non-metric multidimensional scaling (NMDS) of Bray-Curtis dissimilarity (Bray and Curtis, 1957) using the shinyapps.io data analysis toolkit provided by DNASense. Operational taxonomic units (OTUs) not present in more than 0.1% relative abundance in any sample were removed from the dataset prior to analysis. Phyla and genera most responsible for differences were identified using differential abundance analysis between groups using the shinyapps.io data analysis toolkit provided by DNASense. Prior to analysis, taxa not present in higher than 1% relative abundance in any sample were removed from the dataset. Differential abundance analysis was performed using a significance threshold of 0.01.

2.2.4. Solid-liquid separation performance

Supernatant turbidity after extended settling (3 weeks in refrigerated 50 mL centrifuge tubes) was evaluated in order to compare performance amongst the different sludges. Images of supernatant were taken using a standardized setup with reference colours to ensure comparability of individual photographs. Supernatant turbidity of individual samples was then ranked as clear, cloudy, or turbid based on visual assessment of the photographs. Photographs and turbidity rankings are included in the SI. Compressibility of the settled sludge was monitored by calculating the sludge volume index (SVI (mL/g)) after 30 min of settling in Imhoff cones in accordance with standard methods for activated sludge and biological suspensions (APHA, 2005). These results are not included in the text, but all results can be found in the SI.

Dewatering time, the time it takes for free water to filter through the sludge and filter paper, was measured using capillary suction time (CST (s)) according to standard methods (APHA, 2005). A Triton 319 Multi-CST apparatus with 18 mm funnel was used. CST values were normalized based on TS in the sample (sL/gTS), in order to compare results across samples with different solids concentrations (Peng et al., 2011, APHA, 2005).

Dewatered cake solids was defined as the total dry solids in the dewatered sludge cake after centrifugation. 30 mL faecal sludge

Table 1

Summary statistics for faecal sludge organized by source. For categories where n = 2, the values are reported for both samples (s1 and s2) in place of mean, median, and standard deviation (std). Literature values shown are a range of mean values from published characterization of faecal sludge and wastewater sludge: [a](USEPA, 1984), [b](Lowe et al., 2009), [c](Henze et al., 2008), [d](Gold et al., 2018a), [e](Strande et al., 2018), [f](Tchobanoglous et al., 2014), [g](Gold et al., 2018b), [h](Turovskiy and Mathai, 2006), [i](Arnaiz et al., 2006), [j](Miron et al., 2000), [k](Mikkelsen and Keiding, 2002),[m](Citeau et al., 2012), [n](Asztalos and Kim, 2015), [o](Park et al., 2006a), [p](Chiu et al., 2006), [q](Zorpas et al., 1998).

Source		n	pH	EC (mS/cm)	TS (g/L)	VS (% TS)	TSS (g/L)	VSS (g/L)	Total EPS (mg/L)	Total EPS (mg/gTSS)	Protein-like substances (mg/gTSS)	Humic-like substances (mg/gTSS)	Na^+ (mg/L)	K^+ (mg/L)	Mg^{2+} (mg/L)	Ca^{2+} (mg/L)	M/D cation ratio
Household	mean	7	7.5	3.5	34.9	53.9	17.0	10.5	96.7	15.5	6.2	9.4	196.0	67.7	24.9	8.4	8.2
	median		**7.4**	**3.2**	**11.0**	**56.9**	**7.4**	**6.3**	**86.2**	**11.0**	**4.0**	**6.9**	**184.7**	**63.6**	**24.7**	**6.9**	**8.2**
	std		0.1	1.3	45.7	15.0	18.7	9.3	48.4	15.8	5.5	10.5	86.4	22.2	8.4	6.7	1.9
School	mean	6	7.7	7.0	15.7	48.3	10.0	6.6	111.6	34.2	14.4	19.8	573.8	153.7	25.0	9.1	25.3
	median		**7.8**	**6.0**	**13.6**	**52.7**	**8.5**	**5.3**	**91.1**	**15.6**	**4.3**	**10.7**	**463.5**	**111.5**	**18.0**	**7.3**	**21.8**
	std		0.6	3.7	10.5	17.7	8.0	5.2	76.9	54.1	26.0	28.2	350.0	126.9	16.5	5.5	18.6
Public toilet	mean	5	7.9	15.4	19.0	56.9	9.2	7.6	366.5	71.2	15.0	56.2	577.2	474.7	5.9	62.3	17.4
	median		**7.8**	**13.5**	**13.0**	**56.7**	**4.9**	**3.5**	**340.5**	**69.7**	**16.2**	**53.5**	**456.5**	**413.0**	**4.5**	**64.6**	**18.5**
	std		0.2	3.5	14.5	12.9	8.2	6.8	109.3	51.2	7.8	43.6	234.0	139.4	2.2	24.1	7.8
Office	mean	3	7.7	4.2	7.7	60.3	5.2	3.8	67.2	15.2	5.2	10.0	207.0	93.5	21.6	8.9	9.4
	median		**7.7**	**4.3**	**9.0**	**60.4**	**5.9**	**4.1**	**71.4**	**12.1**	**3.4**	**8.8**	**263.9**	**87.7**	**20.8**	**4.9**	**11.4**
	std		0.2	2.2	4.2	4.3	3.3	2.2	29.7	5.7	3.3	2.4	116.0	55.6	5.6	8.2	4.0
Restaurant	s1	2	7.0	1.4	2.8	61.4	1.7	1.3	36.7	21.5	7.6	13.9	72.4	24.4	14.3	18.8	2.9
	s2		6.3	3.3	22.5	84.9	18.3	16.9	151.2	8.3	3.5	4.7	236.9	72.3	26.4	2.1	10.9
Place of worship	s1	2	7.6	4.4	5.8	53.2	4.2	2.7	58.4	14.0	3.9	10.1	209.4	79.4	19.7	5.6	11.4
	s2		7.7	5.3	16.7	61.8	11.4	8.2	120.6	10.6	4.2	6.4	233.2	113.5	19.3	16.9	9.6
Literature values																	
Faecal sludge			6.9–8.5[a,c]	2–18[d]	1–52[b,d]	43–74[d]	0.2–30[b,d]	3–19[e]	–	–	–	–	–	–	–	–	–
Wastewater primary sludge			5–8[f]	2.6–3.1[g]	20–70[h]	60–85[f]	45[i]	15–18[j,i]	–	75[k]	33[k]	36[k]	–	–	–	–	–
Wastewater activated sludge			6.5–8[f]	0.8–1.3[m,n]	4–15[h]	60–85[f]	4–18[h,i]	15[i]	–	130[k]	76[k]	42[k]	68–1087[o]	10–116[o]	6–44[o]	24–339[o]	0.8–18.4[o]
Wastewater anaerobically digested sludge			5.9–7.6[p,m]	1.5–16[q,p]	30–60[h]	50–60[h]	–	–	–	78[k]	40[k]	31[k]	70–1120[o]	45–135[o]	4.5–50[o]	23–167[o]	1.8–24.8[o]

samples were centrifuged in 50 mL centrifuge tubes at 3,345×g for 10 min. After centrifugation, supernatant was decanted and dry solids (% ds) in the cake measured using standard methods (APHA, 2005). Dry cake solids after centrifugation is a laboratory measurement to predict dewatering performance at scale (Kopp and Dichtl, 1998; Gold et al., 2018a).

Measurement replicates for parameters were performed according to recommended quality assurance and quality control (QA/QC) measures stipulated in standard methods (APHA, 2005). Reported values are averages of measurement replicates, and error bars in figures represent the standard deviation of the replicates. Statistical analysis and regressions were performed using the Statsmodel 0.9.0 module in Python (Seabold and Perktold, 2010). Plots were produced using Matplotlib 3.0.3 2D graphics package in Python (Hunter, 2007). For boxplots, the middle line represents the median, and the boundaries of the box represent the first and third quartiles (Q1 and Q3). The upper whisker extends to the last data point less than Q3 +1.5 * (Q3 − Q1), and the lower whisker extends to the first data point greater than Q1 − 1.5 * (Q3 − Q1). Outside of the whiskers, data are considered outliers and plotted individually as open circles.

3. Results

3.1. Characterization

Results of faecal sludge characterization are reported, grouped by source, and compared with existing literature values in Table 1. The physical-chemical characteristics in the current study are similar to reported values for faecal sludge from other studies. TS was on the lower end of published values, however, in the current study the samples were mainly from “septic tanks” that are analogous to cesspits with no overflow pipe, with pour-flush or cistern-flush toilets, and a community that uses water for anal cleansing.

Based on our review, values for EPS and cations in faecal sludge have not previously been reported in the literature. In the current study, the total EPS fraction in faecal sludge was an order of magnitude lower than reported values for activated sludge, and 5–7 times lower than primary and mesophilic AD sludge. The faecal sludge from public toilets was an exception, containing comparable amounts of EPS to primary and AD wastewater sludges. Humic-like substances contributed substantially to the total EPS in faecal sludge, making up a larger fraction of total EPS (0.56–0.76) compared to in wastewater sludge (0.32–0.48). Soluble cation concentrations in faecal sludge were comparable to reported values in activated and AD sludge, although K^+ from public toilets was higher.

Greater differences were observed for EPS and cation concentrations by source, than for more conventionally measured parameters (TS, TSS, VSS, VS fraction). Public toilet sludge had notably higher EPS, K^+, Ca^{2+}, and lower concentrations of Mg^{2+} compared to other sources. Higher salt concentrations could be due to different user behaviours at public toilets. For example, users are potentially more likely to defecate at home than during the day at marketplace public toilets, and men less likely to urinate in the open in a crowded marketplace than in discrete places where it is more acceptable. This would mean a higher fraction of urine enters containment, resulting in a higher ratio of urine to faeces and correspondingly higher cation concentrations.

EPS is known to be broken down during anaerobic digestion of primary wastewater sludge (Miron et al., 2000). Hence, it is logical that EPS would also be reduced during onsite storage of faecal sludge in facultative or anaerobic conditions (Philip et al., 1993; Nwaneri et al., 2008; Couderc et al., 2008), resulting in faecal sludge containing lower fractions of EPS than activated sludge, and being more comparable to primary or AD sludge. Comparable amounts of EPS in the faecal sludge from public toilets and reported values for primary sludge, also suggest that faecal sludge from public toilets is less stabilised than other sources, indicating that very little degradation of organic material is occurring prior to collection. This was corroborated by field observations during sample collection, as samples from public toilets were light-brown in colour and highly odorous. Also, all of the faecal sludge samples were liquid and unconsolidated; none appeared to have flocs. Based on these results, source of faecal sludge appears to be a predictor of cations and EPS concentration, most likely due to the differing management

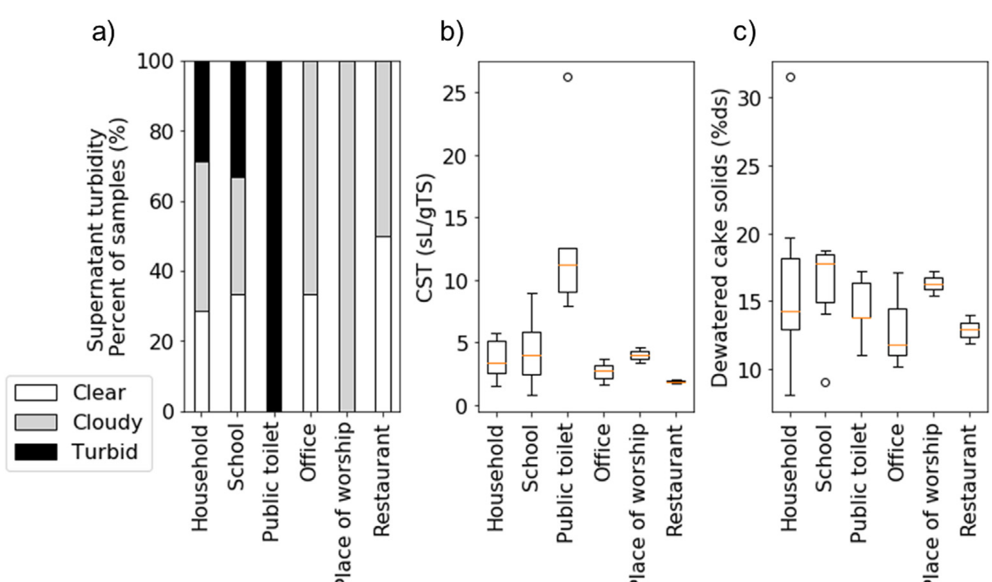

Fig. 1. Solid-liquid separation performance metrics broken down by faecal sludge source (household (n = 7), school (n = 6), public toilet (n = 5), office (n = 3), place of worship (n = 2), and restaurant (n = 2)). a) Stacked bar graph illustrating percentage of samples in each source category that fall under each supernatant turbidity classification (white bars = clear, grey bars = cloudy, black bars = turbid). Sample numbers (n) for each source category indicated above bars. b) and c) Box plots illustrating distribution of CST and dewatered cake solids by source.

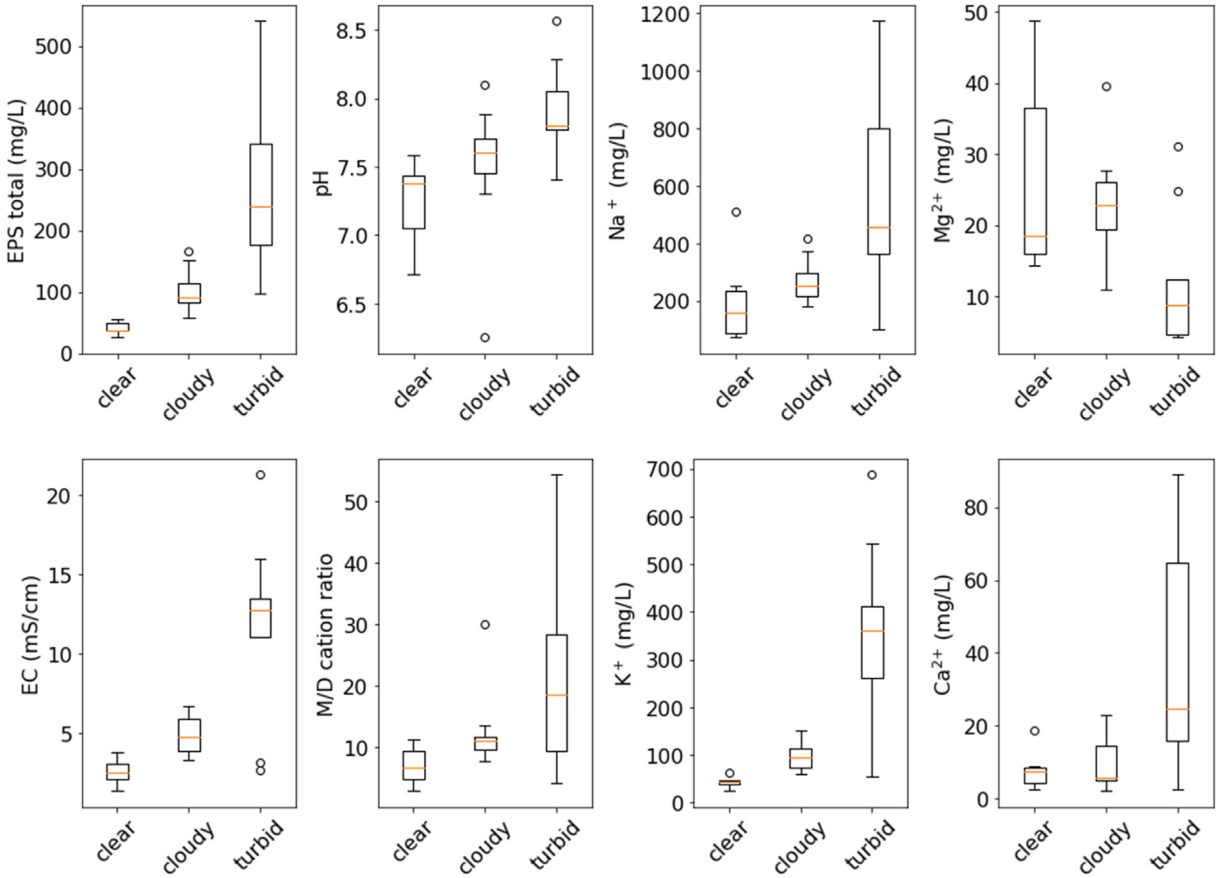

Fig. 2. Boxplots showing the relationship between a sample's supernatant turbidity after prolonged settling (clear (n = 6), cloudy (n = 10), turbid (n = 9)) and EPS total, EC, pH, M/D cation ratio, and monovalent (Na^+ and K^+) and divalent (Mg^{2+} and Ca^{2+}) cation concentrations.

practices.

3.2. Solid-liquid separation performance

Results of metrics of solid-liquid separation performance by source of faecal sludge are presented in Fig. 1.

One third of the faecal sludge had turbid supernatant after prolonged settling. The public toilet sludge all had turbid supernatant, while the supernatant turbidity from schools and households had widely variable turbidity, from clear to turbid. The other sources had lower variability, but there were also fewer samples in these categories. Turbidity was assessed using a qualitative method, but has relevance for treatment performance. In addition, all of the faecal sludge demonstrated compact settling as measured by SVI (presented in SI). Faecal sludge from public toilets took longer to dewater (indicated by higher CST values) compared to other sources. There was not a notable difference in dewatered cake solids between sources. While faecal sludge coming from public toilets was a predictor of high turbidity and long dewatering times, it was not a predictor of final cake solids after dewatering. In comparison to the research detailed in Strande et al. (2018), public toilets and other sites had the same type of containment. The differences in dewatering performance between public toilet and other sources is likely related to physical-chemical and/or biological differences between the sludges based on demographic, environmental, and technical factors that can affect faecal sludge qualities. It is important to note, that “public toilet” should not be assumed to be a universal predictor of dewatering behaviour. Factors such as containment technology, usage patterns, and emptying frequency, will most likely be more relevant (Strande et al., 2018). The most appropriate predictors will potentially vary by location, as well as accepted definitions of the terms “public toilet” and “septic tank”.

3.2.1. Settling

The influence of physical-chemical parameters and EPS on settling performance in terms of supernatant turbidity after prolonged settling is presented in Fig. 2. Turbid supernatant corresponded to higher EPS concentrations, higher EC, pH, K^+, Na^+, and Ca^{2+}, and lower Mg^{2+} concentrations, whereas clear and cloudy supernatant had higher Mg^{2+} concentrations and lower EPS, EC, pH, K^+, Na^+, Ca^{2+}, and M/D cation ratios. It is likely, based on the relationships illustrated in Fig. 2, that EPS along with other soluble and colloidal organic matter is contributing to turbidity.

3.2.2. Dewatering time

EPS was further fractionated into humic-like and protein-like compounds to evaluate whether these fractions were associated with dewatering time and turbidity in faecal sludge. Slower dewatering (i.e. higher CST) corresponded to higher turbidity (Fig. 3a) and higher concentrations of EPS (Fig. 3d), and specifically humic-like substances (Fig. 3b). High concentrations of soluble or easily extractable polymers contribute to clogging of filters and pores within the sludge cake, resulting in slower dewatering; this is a primary contributor to poor filtration performance for activated and anaerobically digested wastewater sludges (Yu et al., 2010; Lei et al., 2007; Novak et al., 2003). As noticed previously with field observations (Cofie et al., 2006; Heinss et al., 1999), level of

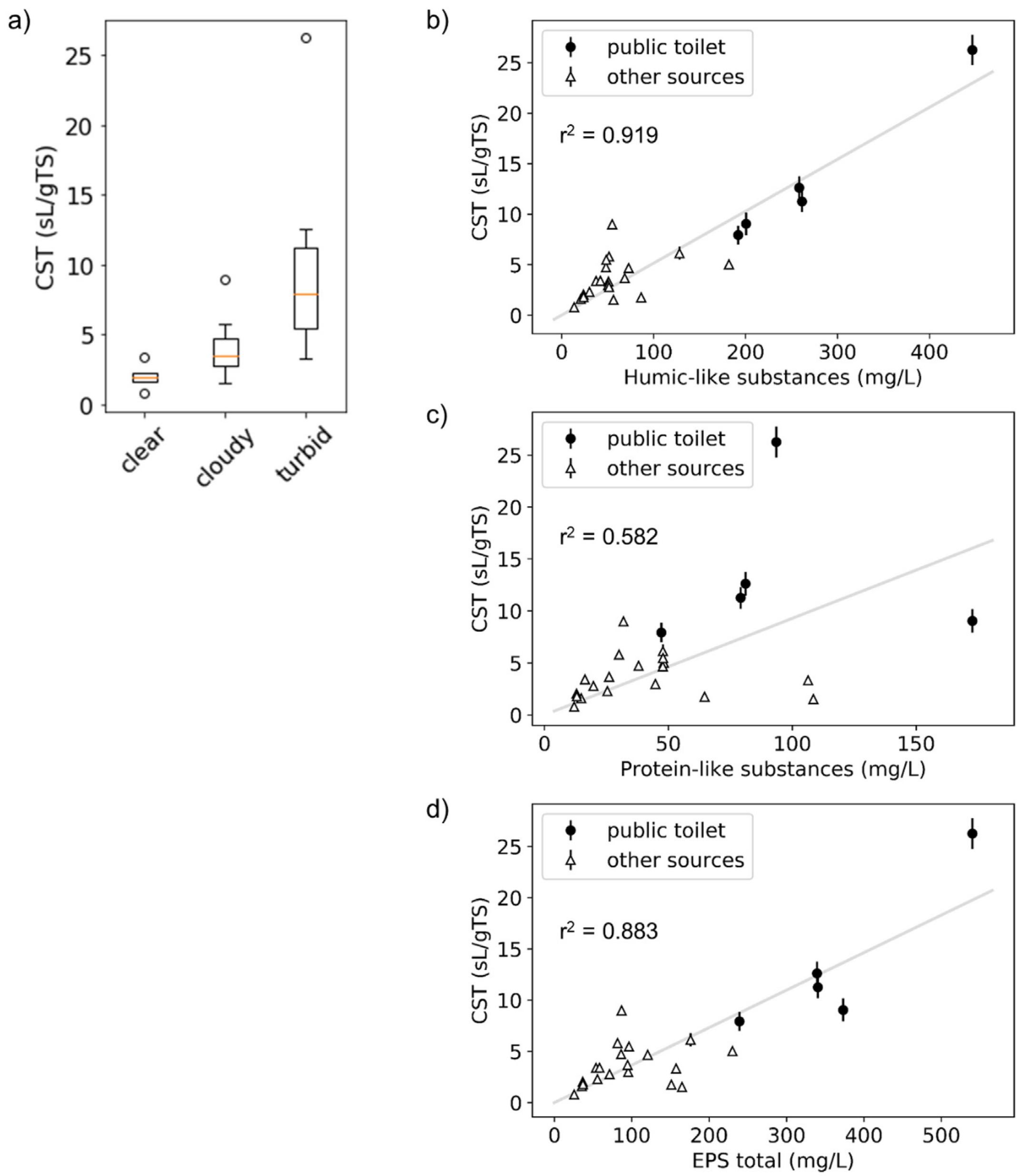

Fig. 3. a) Box plot showing CST values for samples with clear, cloudy, and turbid supernatant. b) Scatterplot illustrating the relationship between CST and the concentration of humic-like substances, (c) protein-like substances, and (d) EPS total. Linear trend lines and r^2 values are included. In scatterplots, samples from public toilets are represented by filled circles, and samples from all other sources (household, school, office, place of worship, and restaurant) are represented by open triangles.

stabilization appears to be a good predictor of dewatering time for faecal sludge. It is interesting to note that public toilet sludges, which appeared to be the least stabilised, contained the highest concentrations of humic-like substances. It is possible that more stabilised faecal sludge may dewater faster because it has lower concentrations of soluble and suspended EPS to clog filters and pores.

As depicted in Fig. 4, dewatering time (CST) increased with increasing EC, pH, K^+, Na^+ and Ca^{2+}, decreasing Mg^{2+}, and was not clearly associated with M/D ratio, similar to the patterns observed for supernatant turbidity. Based on the linear relationship observed between EPS and dewatering time (Fig. 3), it would be logical that the observed correspondence between electrochemical solution properties and dewatering time are due to their underlying relationships with soluble and colloidal EPS. For example, samples with high EC exhibited high EPS concentrations, and thus, took longer to dewater. The difference in the relationships between the divalent cations (Ca^{2+} and Mg^{2+}) and CST is unexpected, and could be an indicator of a species-specific interaction driving coagulation, or of struvite precipitation, which has been shown to occur in wastewaters with high concentrations of urine (Udert et al., 2003). We hypothesize that high EPS concentrations are observed in the faecal sludge samples with high EC, pH, and cation concentrations due to a combination of environmental and microbiological factors affecting degradation of organic material, and electrochemical factors affecting particle surface charge and coagulation properties.

3.2.3. Dewatered cake solids

Illustrated in Fig. 5, dewatered cake solids were generally higher in faecal sludge with lower VSS fraction, although the correlation

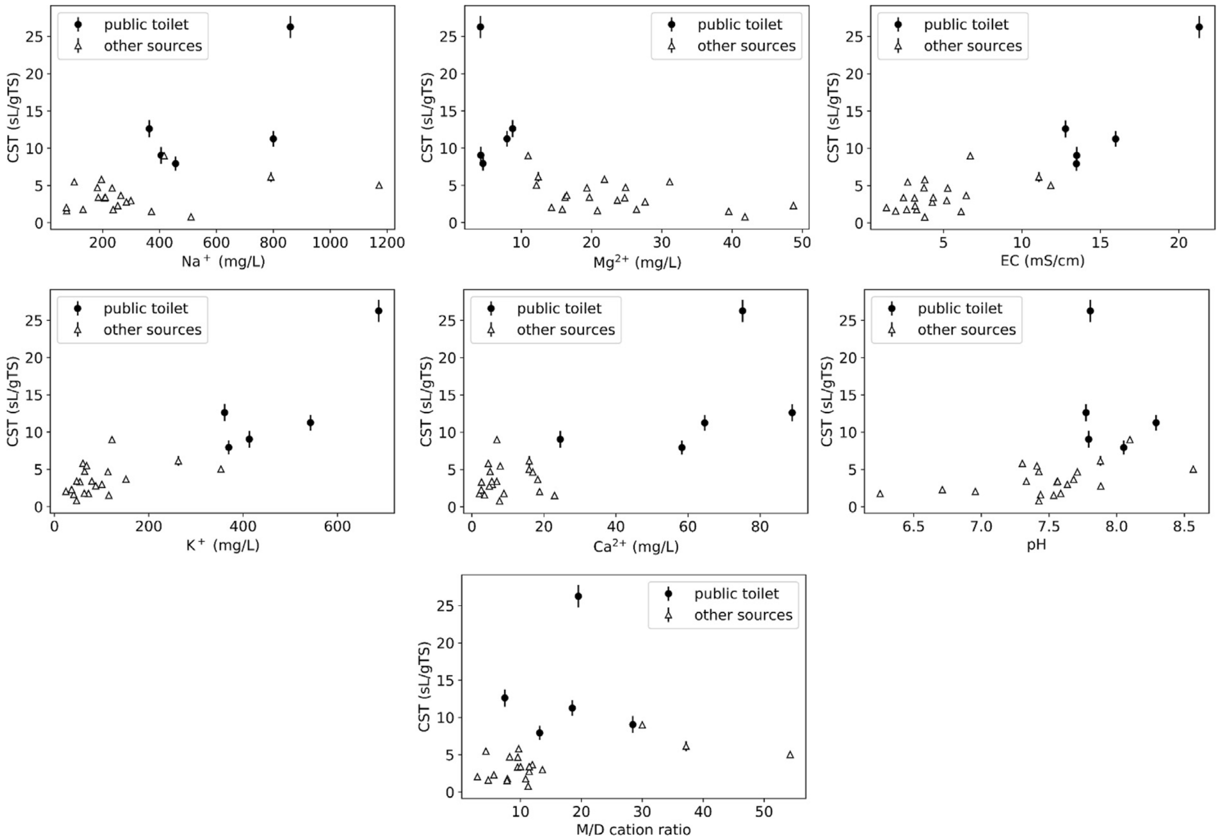

Fig. 4. Scatterplots illustrating the relationship between dewatering time (CST (sL/gTS)) and the concentration of soluble Na^+, K^+, Mg^{2+}, and Ca^{2+}, M/D cation ratio, EC, and pH. Samples from public toilets are represented by filled circles, and samples from all other sources (household, school, office, place of worship, and restaurant) are represented by open triangles.

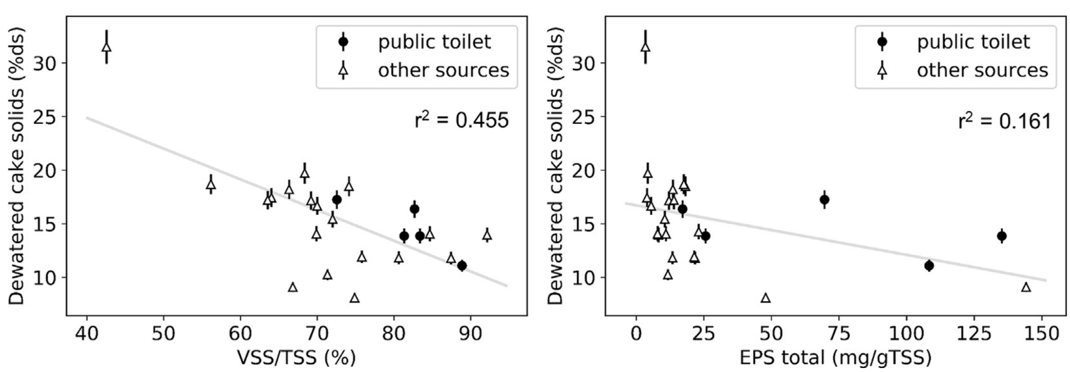

Fig. 5. Scatterplots plotting dewatered cake solids against VSS/TSS (%) and mass fraction of EPS total. Linear trend lines and r^2 values are included. Samples from public toilets are represented by filled circles, and samples from all other sources (household, school, office, place of worship, and restaurant) are represented by open triangles.

was not strong. There does not appear to be a linear relationship between EPS mass fraction and how much water can be removed from the sludge cake, although these parameters have been shown to correlate strongly with cake solids in wastewater sludges (Skinner et al., 2015; Cetin and Erdincler, 2004). We hypothesize that EPS and VSS were not as strong predictors of dewatering behaviour due to the influence of large inorganic particles (e.g. sand) that may be present in faecal sludge.

3.3. Microbial community

Microbial community composition was evaluated to determine whether it could be a predictor of dewatering performance. This was selected, as microbial community has been linked to properties such as EPS concentration and stabilization (Bala Subramanian et al., 2010; Marcus et al., 2013). Illustrated in Fig. 6, the top three most abundant phyla represented in the faecal sludge samples were *Firmicutes*, *Proteobacteria*, and *Bacteroidetes*. Results are in alignment with other reported studies, for example these phyla were the most prevalent in faecal sludge sampled from pit latrines in Tanzania and Vietnam (Torondel et al., 2016), pour-flush pits in South Africa (Byrne et al., 2017), and model septic tanks (Marcus et al., 2013). Full microbial community dataset is available in SI.

A correlation was not observed between the relative abundance

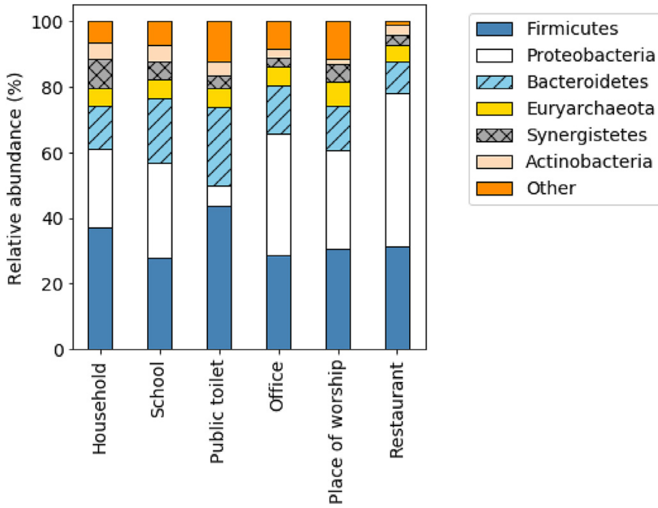

Fig. 6. Relative abundance of six most abundant phyla in the faecal sludge samples, broken out by source.

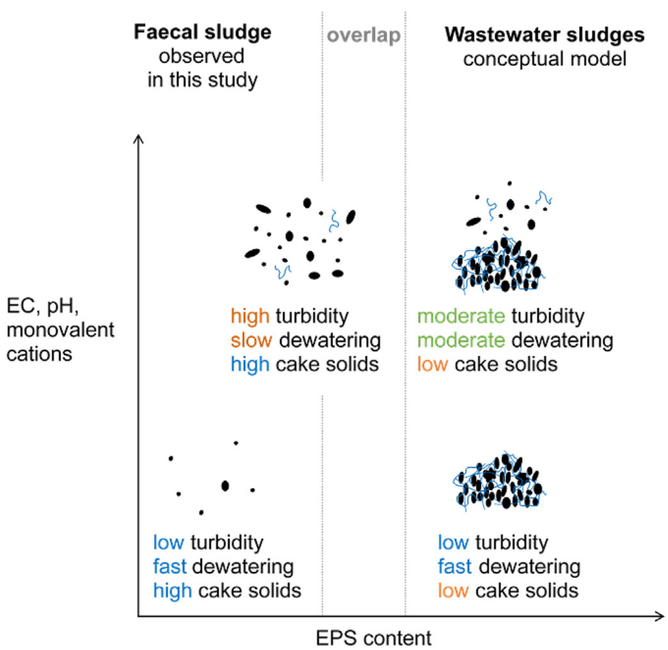

Fig. 7. Schematic depicting the relationships between EPS content and physical-chemical parameters (conductivity, pH, and monovalent cation concentration) for faecal sludge from this study, and the currently accepted conceptual model for dewatering behaviour in wastewater sludges (Christensen et al., 2015; Liu and Fang, 2003; Mikkelsen and Keiding, 2002). The right side of the figure depicts the existing understanding for wastewater sludges, and left side the faecal sludge observed in this study. The area within the grey lines represents the overlap of the observations in this study with the conceptual model for wastewater sludges. EPS is depicted as blue lines. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

of specific OTUs and the metrics of dewatering performance. However, by grouping faecal sludge samples into categories based on their dewatering performance, we could identify genera that were most responsible for community-wide differences using differential abundance analysis (grouping and full results in SI). With respect to supernatant turbidity after settling, samples with clear supernatant had significantly higher abundance of the phylum *Proteobacteria*, and turbid samples had higher abundance of *Euryarchaeota*. The genera (or most specific level) most responsible for the differences in microbial community between the samples were: the *Gammaproteobacteria Pseudomonas*, *Aeromondales*, and *Tolumonas*, and the *Bacteroidetes Porphyromonadaceae, Bacteroides*, and *Macellibacteroides*, which were present in higher relative abundance in clear samples. Turbid samples had higher relative abundance of the *Firmicutes Family XI*, and *Ruminococcaceae*. When faecal sludge samples were separated by their dewatering time (grouped into categories based on CST), the phylum *Proteobacteria* were present in higher abundance in fast dewatering (CST < 2.3 sL/gTS) sludges compared to those that dewatered slowly (CST > 6.1 sL/gTS). At the genus level, fast dewatering samples had higher abundance of the *Gammaproteobacteria*, *Tolumonas*, and slow dewatering samples had higher amounts of the *Euryarchaeota*, *Candidatus Methanogranum*. Sludge cake solids following dewatering was also associated with microbial communities at the phylum and genus level. Sludges with low cake solids (<11.9% ds) had higher relative abundance of *Proteobacteria* than sludges with high cake solids (>17.3% ds). Specifically, sludges with low cake solids had more of the *Betaproteobacteria*, *Rhodocyclaceae*. The presence of different populations of microorganisms associated with metrics of dewatering performance could indicate the importance of microbiological processes in faecal sludge dewatering behaviour.

4. Discussion

It is valuable to compare the dewatering behaviour of faecal sludge to that of primary, activated, and AD sludges from wastewater treatment, to determine what knowledge can be transferred to faecal sludge and what cannot. An overview of the current state of knowledge of the dewatering performance of wastewater sludges based on EPS, monovalent cations, and pH was presented in the introduction. This information can also be summarized in a conceptual model, as presented in Fig. 7. The dewatering behaviour of faecal sludges analysed in this study (shaded in light grey) are partially outside of the accepted conceptual model for wastewater sludges (shaded in dark grey). Activated sludges with high EPS fractions (bottom right) form flocs, which reduces supernatant turbidity and promotes faster dewatering, but also binds water resulting in low cake solids. Flocculation can be disrupted by high monovalent cation concentrations or pH (top right), or by reducing EPS fraction in the sludge (top centre) (Christensen et al., 2015; Liu and Fang, 2003; Mikkelsen and Keiding, 2002). Higher EPS faecal sludges had similar EPS fractions to primary or AD wastewater sludges (top centre) (see Table 1), occurred at high conductivity and pH, and exhibited high turbidity and slow dewatering; similar to digested sludges at high pH or high monovalent cation concentrations where flocculation is inhibited (Christensen et al., 2015). Faecal sludges with lower EPS (bottom left) have lower fractions of EPS than wastewater sludges. Lower EPS faecal sludges exhibited low turbidity, fast dewatering, and high cake solids, and had low conductivity, monovalent cation concentrations, and pH. Lower EPS faecal sludges had different dewatering behaviour than wastewater sludges and may be governed by different mechanisms. Faecal sludges have been observed to become less difficult to dewater as they are stabilised (Cofie et al., 2006; Heinss et al., 1999), which is consistent with observations in this study if we consider higher EPS faecal sludges to be “fresher” and less degraded, and lower EPS sludges to be more stabilised. This distinguishes higher EPS faecal sludges from primary wastewater sludges, which have large particle size and low conductivity, and have been found to dewater more quickly and thoroughly than stabilised wastewater sludges (Tchobanoglous et al., 2014; Gold et al., 2018a).

EPS appears to play a different role in faecal sludge dewatering performance than in activated sludge, at least partly because it is present in lower amounts. In this study, dewatering time and turbidity correlated with the concentration of extractable EPS, but not with the EPS fraction in the solids. This indicates that within the range of EPS content observed in faecal sludges, higher EPS fractions do not promote flocculation as they have been shown to do in activated sludges (Christensen et al., 2015). This fits with our observation that M/D cation ratio does not correspond to differences in dewatering time and turbidity. While higher concentrations of monovalent cations cause increased turbidity and poor filtration in activated sludges, this is because increasing the M/D cation ratio contributes to the destruction of divalent cation bridges between EPS and sludge particles (Christensen et al., 2015; Sobeck and Higgins, 2002). Jørgensen et al. (2017) observed that interactions between polyvalent cations and EPS are only relevant for floc formation when there are high enough fractions of extractable EPS in the sludge (>100 mg/gTSS). Considering that the median EPS fraction extracted from public toilet sludge was only 70 mg/gTSS, it is possible that the EPS content even in unstabilised faecal sludges may be too low to promote floc formation. EPS concentrations, instead of contributing to flocculation, were strongly positively correlated with dewatering time and turbidity. This suggests that EPS in faecal sludge may be more accurately described as colloidal and suspended organic matter that contributes to filter blinding and turbidity, as opposed to polymeric glue that binds flocs together.

However, if we consider faecal sludge as a suspension of particles without long polymer chains, we would expect that EC, pH, and cations, which influence surface charge in accordance with DLVO theory (Mikkelsen and Keiding, 2002), would be related to coagulation of colloidal particles and reduction of turbidity. If this were the case, however, we would expect to see the opposite trend with respect to EC, cations, and turbidity. Higher concentrations of cations should shield particle surface charge, reducing the electrostatic barrier for particles to agglomerate, instead of correlating with an increase in turbidity and dewatering time.

One possible unifying explanation of the results is that solution properties (EC, pH, cations) are directly related to the concentration of EPS in faecal sludges because they are related to stabilization processes. High concentrations of EPS could be present in sludges with high pH and EC because those conditions are less favourable for biological degradation of EPS during storage in onsite containment. While none of the measured cations are present at concentrations inhibitory to anaerobic digestion (Parkin and Owen, 1986), it is likely that other substances present in high quantities in urine or faeces (e.g. ammonia) could reach concentrations high enough to inhibit anaerobic bacteria and decomposition, meaning that it would take longer to degrade EPS and other organic material. This would also fit with our previous observation about usage behaviour in public toilets – high fractions of urine have been shown to lead to extremely high ammonia concentrations, well above the inhibitory limit (Englund et al., submitted, Heinss and Strauss, 1999; Rose et al., 2015). Measured cation concentrations should correlate with ammonia concentrations if they are indeed due to high urine concentration. This idea is supported by information on emptying frequency and uses per day collected at the toilet sites with questionnaires (SI). Public toilets reported similar emptying frequencies and uses per day to offices and toilets of large households, so the effective residence times in containment are comparable. The lower levels of stabilization in public toilet faecal sludge even over long residence times would be logical if sludge in public toilets is undergoing inhibition due to ammonia toxicity. Although ammonia was not quantified in this study, Gold et al. (2018) observed that $NH4^{+}$ concentrations correlated strongly with longer dewatering times in faecal sludge from septic tanks and

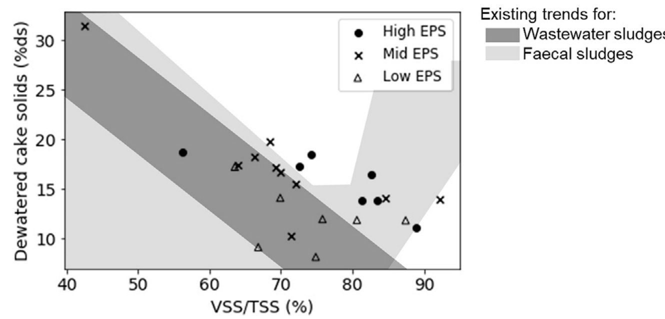

Fig. 9. Scatterplot illustrating the relationship between dewatered cake solids and VSS/TSS (%). Points represent data from this study, divided into groups based on EPS concentration (circles = upper 25% of EPS concentrations, x's = middle 50%, and triangles = lower 25%). The dark grey filled area represents the regime of behaviour from wastewater sludges from various aerobic and anaerobic digestion processes (Skinner et al., 2015), and the light grey filled area represents faecal sludges from septic tanks and lined pits (Gold et al., 2018a).

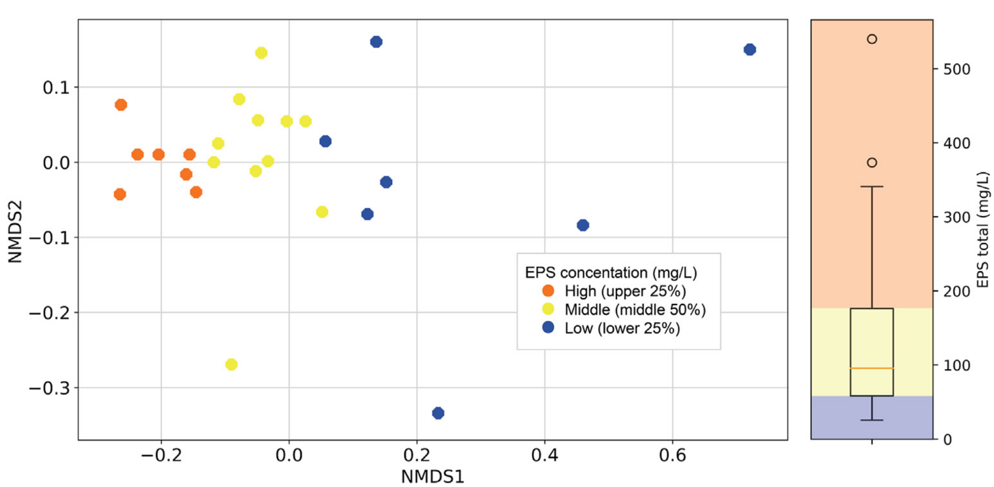

Fig. 8. Left: Non-metric multidimensional scaling plot of OTU compositions in each sample. Colours denote the EPS concentration in the sample, broken into orange (upper 25% of EPS concentrations), yellow (middle 50%), and blue (lower 25%). Right: Box plot of EPS concentration distribution with colours corresponding to the figure on the left. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

lined pit latrines. Ammonia toxicity could also explain the correlation between K^+ and EPS, turbidity, and dewatering time. K^+ is released into the bulk as a microbial stress response to toxic compounds, and K+ efflux has been observed as a toxicity response in activated wastewater sludge. (Bott and Love, 2002; Henriques and Love, 2007).

The idea that there is a biological component tying together dewatering behaviour and physical-chemical conditions in containment is supported by microbial community data. Depicted in Fig. 8, there is a notable difference between the microbial communities in samples with the highest (upper 25%) and lowest (lower 25%) concentrations of EPS, based on Bray-Curtis distance and NMDS clustering. Samples with high EPS concentrations appear to form a cluster, indicating that their microbial communities may be more similar to each other than the other categories. It is not clear whether specific populations of microorganisms are themselves contributing to differences in EPS concentration, or whether their abundance is determined by environmental factors that also influence stabilization or degradation kinetics, like cation concentrations.

Although EPS and its microbial degradation appear to play an important role in filtration of faecal sludge, these do not appear to be as important for determining the amount of moisture remaining in the sludge cake following dewatering. Illustrated in Fig. 9, the light grey shaded area depicts the trend observed for faecal sludge in Gold et al. (2018), the dark grey shaded area depicts the trend observed for wastewater sludges in Skinner et al. (2015), and data points for this study are included for comparison. Dewatered cake solids do not increase with decreasing VSS fraction, in contrast to the observed behaviour in wastewater sludges (Skinner et al., 2015). However, observations do fit within the existing behaviour of faecal sludges from septic tanks and lined pit latrines (Gold et al., 2018a). The expected relationship between VSS fraction and cake solids does not hold for faecal sludge, even though it remains consistent through a range of wastewater sludges generated through a variety of different treatment processes. The explanation for this likely lies with the heterogeneity of the solids fraction in faecal sludges. The relationship between VSS and dewatered cake solids is strong in wastewater sludges because the VSS fraction is representative of the EPS fraction (Skinner et al., 2015). Variable quantities of sand, soil, and other inorganic materials can be introduced into faecal sludge during daily toilet use, or during emptying (Seck et al., 2015) – because of this, even if there is quite a high concentration of volatile organic material, the VSS/TSS fraction would still be low. In addition, we could not detect a correlation between EPS concentration or fractionation and dewatered cake solids in faecal sludge. It is probable that at the EPS fractions present in faecal sludge, other factors, such as soil content, may play a more important role in determining the solids fraction of dewatered faecal sludge.

5. Conclusions

Based on the observations in this study, the key conclusions are:

- EPS is important for faecal sludge dewatering performance observed in this study. Higher concentrations of soluble and colloidal EPS are likely to contribute to clogging of sand drying beds, filters, and other dewatering technologies. However, EPS fractions (mg/gTSS) do not measurably contribute to flocculation or cake moisture content, as is observed in activated and anaerobically digested wastewater sludge.
- The observed relationships between EC, pH, supernatant turbidity, and dewatering time could be further developed and applied in online monitoring of faecal sludge. This would be relatively quick and inexpensive to implement, and could predict dewaterability at treatment plants, or be used for dosing of conditioners for enhanced dewatering.
- For planning of community-to city-wide faecal sludge management, including the design of transfer stations and treatment plants, relationships between demographic factors (e.g. source) and physical-chemical characteristics of faecal sludge could provide a relatively low-cost way to help pre-determine or predict dewatering performance.
- Faecal sludges behave differently than wastewater sludges. There will not be one reference sludge that is appropriate to serve as a proxy for faecal sludge, based on the vast differences in redox conditions, biomass, nutrients, salts/ions, stabilization, particle size, EPS, undigested plant fibers, etc. Hence this emerging research topic needs to be approached in different ways and cannot be solved with just a direct transfer of wastewater knowledge. Looking to other fields of dewatering, for example pulp and paper, sediment dredging, and food science, could provide fresh insights for meeting this challenge.

Declaration of competing interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.watres.2019.115101.

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