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Settling and Dewatering of Fecal Sludge: Building Fundamental Knowledge for Improved Global Sanitation

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Summary

More than 1/3 of the global population relies on onsite sanitation systems, and in some cases entire urban areas in low- and middle-income countries are not served by sewers. Improved options for the management of fecal sludge that accumulates in these onsite systems are required immediately, as the majority of it is currently discharged untreated into the urban environment, placing a huge burden on public and environmental health. One of the greatest obstacles to establishing more reliable and accessible fecal sludge treatment is inconsistent and unpredictable solid-liquid separation. Extreme influent variability in influent fecal sludge poses operational problems with settling and dewatering, reducing the capacity of existing centralized and semi-centralized treatment facilities and hindering the transfer of low-footprint technologies which could extend decentralized and community-scale sanitation coverage in high-density urban areas.

In order to develop robust and reliable treatment solutions for fecal sludge, we need to understand the factors governing solid-liquid separation. In this thesis, the influence of extracellular polymeric substances (EPS), solution properties, particle size distribution, and stabilization in onsite containment on solid-liquid separation performance in fecal sludge was investigated, and a conceptual model for fecal sludge settling and dewatering was developed. Based on this research, field predictors of fecal sludge characteristics and dewatering performance were identified and predictive models and an app were developed that can use photographs and probe measurements to predict dewatering performance of influent sludge. The aim of this thesis was to understand fundamental drivers of solid-liquid separation in fecal sludge and use knowledge gained to inform transfer of treatment technologies, develop methods for rapid characterization of influent, and predict dewatering performance to facilitate responsive process control for dewatering at treatment facilities.

In chapter two, the relationships between physical-chemical parameters including EPS and cations and settling and dewatering performance from fecal sludge field samples were evaluated. Higher concentrations of EPS appeared to contribute to turbid supernatant and worsened filtration by clogging pores but were not associated with differences in bound water in dewatered sludge. Fecal sludge had different physical-chemical characteristics and displayed different dewatering and settling behavior compared to wastewater sludges, and the existing conceptual model for sludge dewatering may need to be edited and expanded to include fecal sludge.

In chapter three, the results of fundamental research in chapter two was applied to advise practitioners on how to develop, transfer, and scale-up dewatering and drying technologies for fecal sludge treatment.

In chapter four, low-cost and simple field measurements and questionnaire data were used to predict fecal sludge influent characteristics and solid-liquid separation performance with different types of empirical models. Color and texture information from photographs and conductivity and pH measurements from probes were good predictors of total solids, ammonium concentration, settling efficiency, and dewatering time when combined with linear and machine learning models. Accuracy of models based on photos and probe measurements could be sufficient for estimating conditioner dosing for dewatering technologies.

In chapter five, the results of fundamental research in chapter four were applied to develop the prototype Sludge Snap app. The app automates image processing of field photographs and runs the predictive models developed in chapter 4 to make real-time predictions of influent characteristics and dewatering performance for use by researchers and practitioners in the field.

In chapter six, field samples and controlled anaerobic storage experiments were used to determine the relationships between (1) stabilization and time in onsite containment, (2) stabilization and particle size distribution, and (3) particle size distribution and dewaterability. The common perception that stabilization and dewatering of fecal sludge are linked to storage time in onsite containments did not hold up to scientific investigation. However, although time was not a predictor, particle size and dewatering performance were related to stabilization. Particle and aggregate size distribution, especially the concentration of small particles <10 µm, was a driver of dewatering performance.

This thesis provides insight into the fundamental drivers of fecal sludge solid-liquid separation and informs suggestions about how to improve and adapt treatment technologies. Overall, suspended small particles were identified as responsible for poor dewatering and settling in fecal sludge, which suggests that treatment efforts should focus on technologies that remove small particles. This currently cannot be reliably achieved during storage in containment, but could be accomplished with treatment options that promote flocculation (e.g. conditioners) or destruction of small particles (e.g. hydrolysis pretreatment followed by controlled anaerobic digestion). Predictive models based on photographs and probe measurements could help to facilitate adaptive process control to allow for these treatment technologies to successfully function with highly variable influent fecal sludge.

Kurzfassung

Mehr als ein Drittel der Weltbevölkerung ist auf dezentrale Abwassersysteme angewiesen, und ganze Stadtgebiete in Ländern mit niedrigem und mittlerem Einkommen sind nicht an die Kanalisation angeschlossen. In diesen Systemen sammelt sich Fäkalschlamm an. Verbesserte Lösungen zur Behandlung von Fäkalschlamm sind sofort erforderlich, da derzeit ein Grossteil davon unbehandelt in die städtische Umwelt entsorgt wird, was eine enorme Belastung für die öffentliche Gesundheit und die Umwelt darstellt. Eines der größten Hindernisse auf dem Weg zu einer zuverlässigeren und leichter zugänglichen Fäkalschlammbehandlung ist die inkonsistente und unberechenbare Fest-Flüssig-Trennung. Die hohen Schwankungen des Fäkalschlamms im Zulauf führen zu betrieblichen Problemen bei der Absetzung und Entwässerung, wodurch die Kapazität bestehender zentraler und semizentraler Kläranlagen verringert wird. Zusätzlich behindern die Schwankungen die Einführung von Technologien mit geringem Platzbedarf, die die dezentrale und kommunale Abwasserentsorgung in städtischen Gebieten mit hoher Bevölkerungsdichte ausweiten könnten.

Um robuste und zuverlässige Behandlungslösungen für Fäkalschlamm zu entwickeln, müssen wir die Faktoren verstehen, die die Fest-Flüssig-Trennung bestimmen. In dieser Arbeit untersuchten wir, wie sich extrazelluläre polymerer Substanzen (EPS), Lösungseigenschaften, die Stabilisierung im Vor-Ort-Containment und die Partikelgrößenverteilung auf die Entwässerunsleistung von Fäkalschlamm auswirken. Wir entwickelten ein konzeptionelles Modell für die Absetzung und Entwässerung von Fäkalschlamm. Auf der Grundlage dieser Forschung haben wir Prädiktoren für Fäkalschlammeigenschaften und Entwässerungsleistung identifiziert. Zusätzlich haben wir Vorhersagemodelle und eine App entwickelt, die Fotos und Sondenmessungen des Schlamms zur Vorhersage der Entwässerungsleistung nutzen kann. Ziel dieser Arbeit war es, die grundlegenden Faktoren für die Fest-Flüssig-Trennung in Fäkalschlamm zu verstehen und die gewonnenen Erkenntnisse zu nutzen, um Behandlungstechnologien zu transferieren, Methoden zur schnellen Charakterisierung des Zulaufs zu entwickeln und die Entwässerungsleistung vorherzusagen. Das ist die Grundlage für eine reaktionsfähige Prozesssteuerung für die Entwässerung in Kläranlagen.

Im zweiten Kapitel haben wir die Beziehungen zwischen physikalisch-chemischen Parametern, einschließlich EPS und Kationen, und der Absetz- und Entwässerungsleistung von Fäkalschlamm-Feldproben untersucht. Höhere EPS-Konzentrationen scheinen zu einem trüben Überstand beizutragen und die Filtration durch Verstopfung der Poren zu verschlechtern, stehen aber nicht in Zusammenhang mit Unterschieden im gebundenen Wasser im entwässerten Schlamm. Im Vergleich zu Abwasserschlämmen hatte Fäkalschlamm andere physikalisch-chemische Eigenschaften und zeigte ein anderes Entwässerungs- und Absetzverhalten, und wir kamen zu dem Schluss, dass das bestehende konzeptionelle Modell

für die Schlammentwässerung möglicherweise überarbeitet und erweitert werden muss, um Fäkalschlamm mit zu berücksichtigen.

Im dritten Kapitel haben wir die Ergebnisse der Grundlagenforschung aus dem zweiten Kapitel angewandt, um Praktiker und Praktikerinnen bei der Entwicklung, dem Transfer und dem Scale-up von Entwässerungs- und Trocknungstechnologien für die Fäkalschlammbehandlung zu beraten.

Im vierten Kapitel bewerteten wir wie kostengünstige und einfache Feldmessungen und Fragebogendaten die Eigenschaften des Fäkalschlamms und die Fest-Flüssig-Trennleistung vorhersagen. Dafür wurden verschiedene empirische Modelle verwendet. Farb- und Texturinformationen aus Fotos sowie Leitfähigkeits- und pH-Messungen aus Sonden waren gute Prädiktoren für Gesamtfeststoffe, Ammoniumkonzentration, Absetzleistung und Entwässerungszeit, wenn sie mit linearen und maschinellen Lernmodellen kombiniert wurden. Die Genauigkeit von Modellen, die auf Fotos und Sondenmessungen basieren, könnte ausreichen, um die Dosierung von Konditionierungsmitteln für Entwässerungstechnologien zu bestimmen.

Im fünften Kapitel haben wir die Ergebnisse der Grundlagenforschung aus dem vierten Kapitel angewandt, um einen Prototyp der Sludge Snap App zu entwickeln. Die App automatisiert die Bildverarbeitung von Feldaufnahmen und führt die von uns entwickelten Prognosemodelle aus, um Echtzeitvorhersagen über die Eigenschaften des Zuflusses und die Entwässerungsleistung zu erstellen, die in der Forschung und der Praxis vor Ort genutzt werden können.

Im sechsten Kapitel haben wir Feldproben charakterisiert und kontrollierte anaerobe Lagerexperimente durchgeführt, um folgende Beziehungen zu bestimmen: (1) Stabilisierung und Zeit im Vor-Ort-Containment, (2) Stabilisierung und Partikelgrößenverteilung und (3) Partikelgrößenverteilung und Entwässerungsfähigkeit. Dass die Stabilisierung und Entwässerung von Fäkalschlamm mit der Lagerungszeit im Vor-Ort-Containment zusammenhängt, ist zwar die allgemeine Auffassung, hielt aber unserer wissenschaftlichen Untersuchung nicht stand. Obwohl die Lagerungszeit kein Prädiktor war, hingen die Partikelgröße und die Entwässerungsleistung mit der Stabilisierung zusammen. Die Partikel- und Aggregatgrößenverteilung, insbesondere die Konzentration kleiner suspendierter Partikel <10 µm, war ein Einflussfaktor für die Entwässerungsleistung.

Diese Arbeit gibt einen Einblick in die grundlegenden Faktoren der Fest-Flüssig-Trennung von Fäkalschlamm und ermöglicht es, Vorschläge für die Verbesserung und Anpassung von Behandlungstechnologien zu machen. Insgesamt haben wir festgestellt, dass kleine suspendierte Partikel für eine schlechte Entwässerung und Ablagerung von Fäkalschlamm verantwortlich sind, was darauf hindeutet, dass bei der Auswahl von Technologien sichergestellt werden sollte, die kleinen Partikel zu entfernen. Dies kann derzeit nicht zuverlässig während der Lagerung im Containment erreicht werden, könnte aber mit Behandlungsoptionen erreicht werden, die die Ausflockung (z.B. Konditionierer) oder die Zerstörung kleiner Partikel (z.B. Hydrolyse-Vorbehandlung gefolgt von kontrollierter anaerober Gärung) fördern. Modelle auf der Grundlage von Fotos und Sondenmessungen könnten die adaptive Prozesssteuerung erleichtern, damit diese Behandlungstechnologien auch bei stark schwankendem Fäkalschlammzufluss erfolgreich funktionieren.

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List of Abbreviations

FSM	fecal sludge management
FSTP	fecal sludge treatment plant
EPS	extracellular polymeric substances
COD	chemical oxygen demand
sCOD	soluble COD
pCOD	particulate COD
TSS	total suspended solids
VSS	volatile suspended solids
CST	capillary suction time
SVI	sludge volume index
TOC	total organic carbon
TKN	total kjeldahl nitrogen
TN	total nitrogen
TS	total solids
VS	volatile solids
NH_4^+- N	ammonium nitrogen
EC	electrical conductivity
TGA	thermogravimetric analysis
DTA	differential thermal analysis
DSC	differential scanning calorimetry
SEC	specific energy consumption
DS	dry solids

Spelling Note

Spelling conventions are either American or British English depending on the chapter, because of the different publishing language requirements of specific journals and textbooks in which the chapters were originally published.

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Chapter 1

Introduction

1.1 Onsite sanitation and fecal sludge treatment

State of global sanitation and fecal sludge management

More than 1/3 of the global population, 3.1 billion people, rely on onsite sanitation (WHO and UNICEF 2019). Shockingly, despite its prevalence, non-sewered sanitation has been historically ignored as a research topic and was not considered a viable long-term sanitation solution (Gambrill et al. 2020). The recent paradigm shift toward city-wide inclusive sanitation acknowledges the importance of onsite and decentralized solutions in addition to traditionally prioritized sewered and centralized systems (Gambrill et al. 2020, Schrecongost et al. 2020). This has drawn attention to the seriously inadequate infrastructure available for managing the waste that accumulates in these systems, especially in rapidly growing urban areas of Sub-Saharan Africa and South- and Central Asia. This is especially alarming as these areas will experience a population explosion of an additional 2.25 billion city dwellers by 2050 (Klinger et al. 2019, UN DESA 2018). In a review of shit-flow diagrams (SFD) generated for 39 cities in low- and middle-income countries, only 30% of excreta collected in onsite systems was successfully treated and not released back into the environment (Peal et al. 2020). This pervasive lack of safe sanitation has serious public health consequences, contributing to 875,000 deaths (e.g., from diarrheal diseases, malnutrition, etc.) in 2016 alone (UNICEF and WHO 2020). Onsite containment and treatment of blackwater is also becoming increasingly relevant for high-income cities attempting to address the capital costs and climate impacts of centralized sanitation (Etter et al. 2016, Hyde-Smith et al. 2022).

What is fecal sludge?

Fecal sludge is everything that accumulates in onsite containment systems and subsequently needs to be emptied and transported to a treatment facility. Onsite systems vary tremendously in their technical design specifications; descriptors of onsite systems include septic tanks, pit latrines, cesspits, and container-based systems. Depicted in Figure 1.1, fecal sludge contains urine and feces (excreta), but can also include anything else that enters the onsite containment, including flush water, anal cleansing materials, menstrual hygiene products, greywater from bath or kitchen, municipal solid waste, and soil (Strande et al. 2021b). The term “sludge” evokes images of thick paste, but fecal sludge is typically more than 95% water (Gold et al. 2016). However, illustrated in Figure 1.2, fecal sludge field samples have a high range of variability in solids concentration, and can be anywhere from a liquid, slurry, semi-solid or solid (Velkushanova and Strande 2021). If the reader is able to think of food while reading this thesis: fecal sludge can resemble anything from weakly brewed tea to chocolate milk to brownie batter.

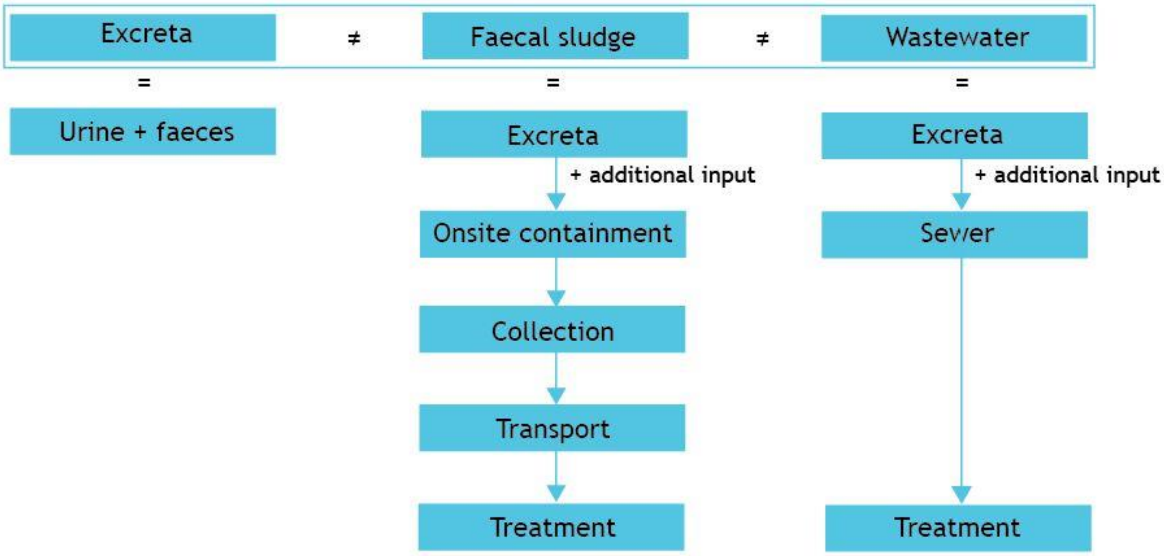


Figure 1.1. Comparison of excreta, fecal sludge, and wastewater, including the sanitation service chains for fecal sludge and wastewater. Reprinted from (Strande et al. 2021a).

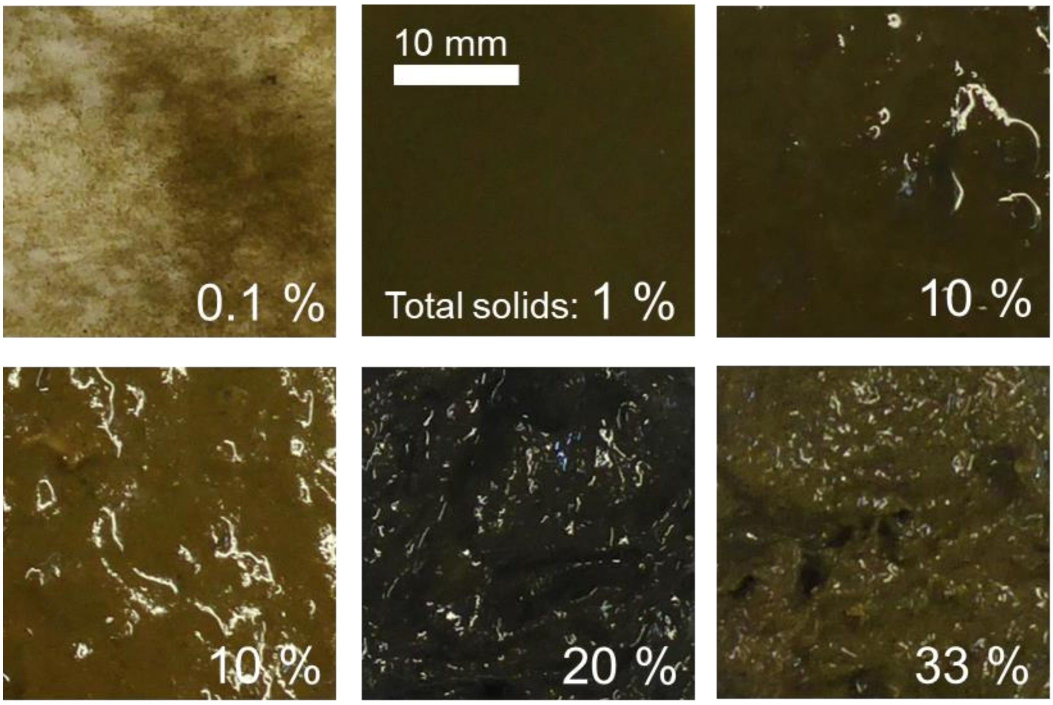


Figure 1.2. Examples of fecal sludge appearance, including liquid, slurry, semi-solid, and solid samples, with total solids concentration included (Ward et al. 2021a).

Fecal sludge is not like wastewater

Fecal sludge is distinct from wastewater in several important ways which affect its treatment. Instead of being homogenized during conveyance in a sewer, fecal sludge is periodically collected from individual containments and each batch is delivered separately to treatment facilities. Illustrated in Figure 1.3, influent fecal sludge has up to two orders of magnitude higher chemical oxygen demand (COD) concentrations and follows a different distribution compared to influent wastewater arriving for treatment at the same facility. Fecal sludge also displays more variability in level of stabilization (shown by the ratio of volatile suspended solids to total suspended solids (VSS/TSS) in Figure 1.4). The high variability in fecal sludge characteristics are the result of the diversity of onsite containment technologies, usage patterns, addition of household greywater, kitchen waste, or municipal solid waste, number of users, frequency of emptying, and batchwise delivery to treatment facilities (Strande and Brdjanovic 2014).

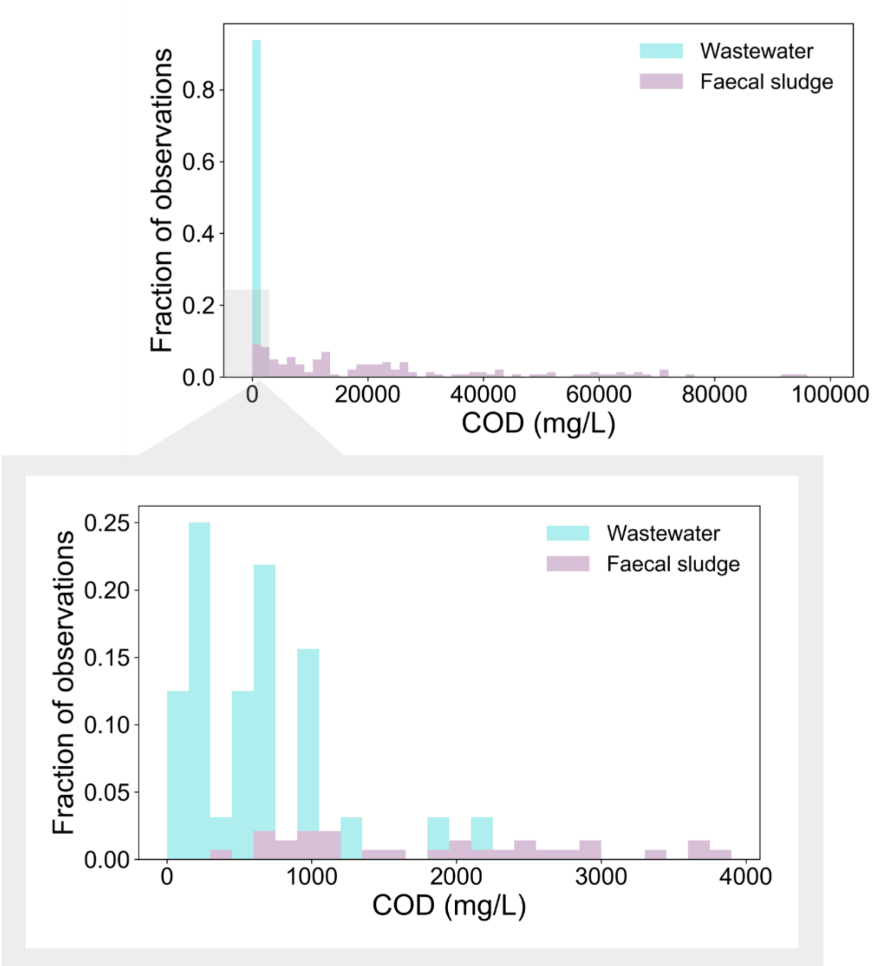


Figure 1.3. Histograms showing the chemical oxygen demand (COD) of influent at the Lubigi Wastewater and Fecal Sludge Treatment Plan in Kampala, Uganda. The bottom histogram shows a zoomed in view of the x-axis from COD of 0-4000 mg/L. (Englund and Strande 2019)

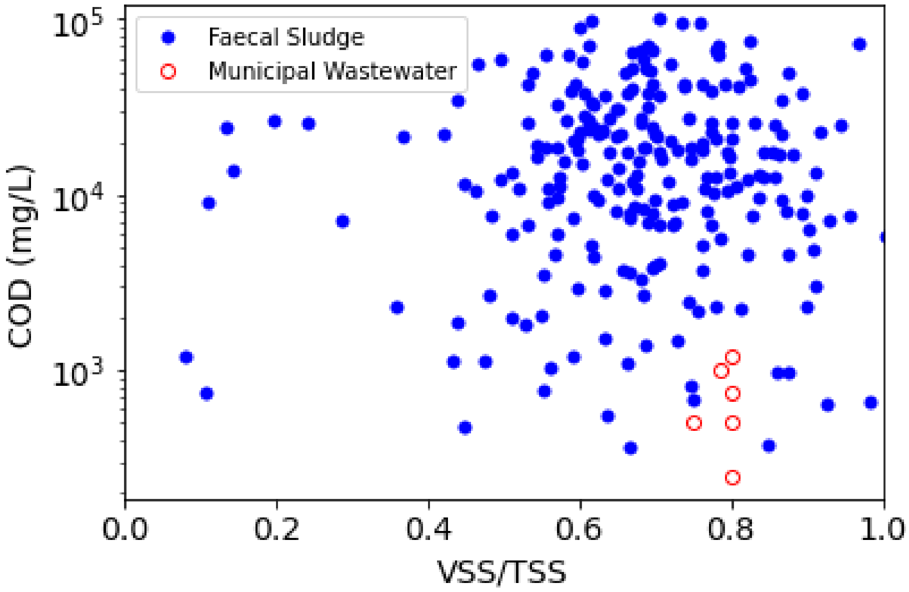


Figure 1.4. Comparison of organic loading and volatile solids fractions in influent at a municipal wastewater treatment plant (Henze et al. 2008, Tchobanoglous et al. 2014) and fecal sludge delivered to treatment facilities or sampled during vacuum truck discharge (Englund et al. 2020, Gold et al. 2018a, Shaw et al. submitted, Ward et al. submitted).

Current state of fecal sludge treatment

Historically, the treatment strategies favored for fecal sludge have been transferred from the wastewater sector (Bassan et al. 2014). Existing fecal sludge treatment facilities are scarce, and typically rely on large-footprint, passive technologies including settling-thickening tanks, drying beds, and stabilization ponds (Tayler 2018). Semi-centralized to centralized fecal sludge treatment plants necessitate the transport of heavy liquid sludge by truck for long distances, often through traffic, which can discourage emptiers and promote illegal dumping (Hossain et al. 2016, Semiyaga et al. 2015). Decentralized or community-scale treatment options could help address these issues in high-density urban areas, as they would reduce the need to transport heavy liquid sludge to a centralized treatment facility (Semiyaga et al. 2015). However, land is limited in urban areas, and low-footprint treatment technologies would need to be developed. Since fecal sludge is typically more than 95% water, a solid-liquid separation step is necessary in almost every treatment process train. However, attempts to transfer low-footprint solid-liquid separation technologies including geotextiles, mechanical presses, and mobile dewatering systems from use with wastewater sludge have been largely unsuccessful (Tayler 2016, Whitesell 2016), making solid-liquid separation one of the greatest technical barriers to implementing low-footprint community-scale fecal sludge treatment technologies.

Challenges in solid-liquid separation

The technical barriers to solid-liquid separation of fecal sludge are primarily associated with its high variability and unpredictable treatment performance, and the lack of fundamental knowledge of factors influencing settling and dewatering in fecal sludge. Although settling-thickening tanks and drying beds are established treatment technologies for fecal sludge, they often perform unpredictably due to intermittent and highly variable influent characteristics, requiring manual intervention by operators, and failing to adequately treat effluent before it is released back into the environment (Klinger et al. 2019). Most low-footprint options are filtration-based technologies (for example, geotextile membranes and mechanical presses) and require the addition of flocculation aids called 'conditioners'. Conditioner dose and selection are highly sensitive to deviations in influent sludge physical-chemical characteristics, which we currently have no means of monitoring (Gold et al. 2016, Novak and Park 2004). In addition, we know very little about the mechanisms or processes that govern the solid-liquid separation behavior of fecal sludge. To develop reliable solid-liquid separation technologies that work for fecal sludge, we must 1) understand what hinders and improves dewatering in fecal sludge, 2) be able to monitor variable influent, and 3) predict how to adjust process control based on influent characteristics.

1.2 Solid-liquid separation in wastewater and fecal sludge

Solid-liquid separation is defined here as the removal of particulate solids from liquids through settling or dewatering processes. Settling is the gravity-driven separation of free water from solids. Dewatering is defined here as the removal of free water and interstitial water that is loosely bound in pores and interstitial spaces in sludge particles and aggregates, illustrated in Figure 1.5. Dewatering technologies can include passive filtration or mechanical methods. The term "dewaterability" can refer to several different metrics, making the wastewater sludge dewatering literature appear contradictory at first glance. In this thesis, solid-liquid separation performance is assessed in the following ways:

- *Suspended solids removal*, by supernatant turbidity after centrifugation
- *Dewatering time*, by capillary suction time (CST). When we talk about "dewaterability" in this thesis, we are referring to this metric unless otherwise stated.
- *Maximum dryness achievable*, by dewatered cake solids after centrifugation

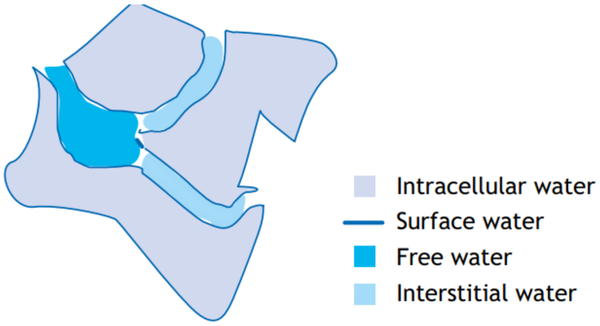


Figure 1.5. Representation of the different categories of water present in sludges (reprinted from (Ward et al. 2021b)).

Solid-liquid separation mechanisms

While we have established that fecal sludge and wastewater sludges have different physical-chemical characteristics, it is helpful to start from the existing body of knowledge about solid-liquid separation in wastewater sludges to understand what factors might be similar or different in fecal sludge. Floc formation and disintegration are the key mechanisms governing settling and dewatering in activated, and anaerobically digested wastewater sludges (Christensen et al. 2015, Jørgensen et al. 2017, Novak et al. 2003). Suspended small particles not incorporated in flocs are responsible for slow filtration and turbid supernatant after settling or centrifugation (Christensen et al. 2015, Karr and Keinath 1978, Lawler et al. 1986). Specifically, high concentrations of supracolloidal particles, between 1 and 100 µm in diameter, have been shown to directly contribute to poor filtration in primary, activated, and anaerobically digested wastewater sludges (Karr and Keinath 1978). Figure 1.6 depicts the size ranges of different particle fractions that are discussed in this thesis in terms of their influence on solid-liquid separation processes.

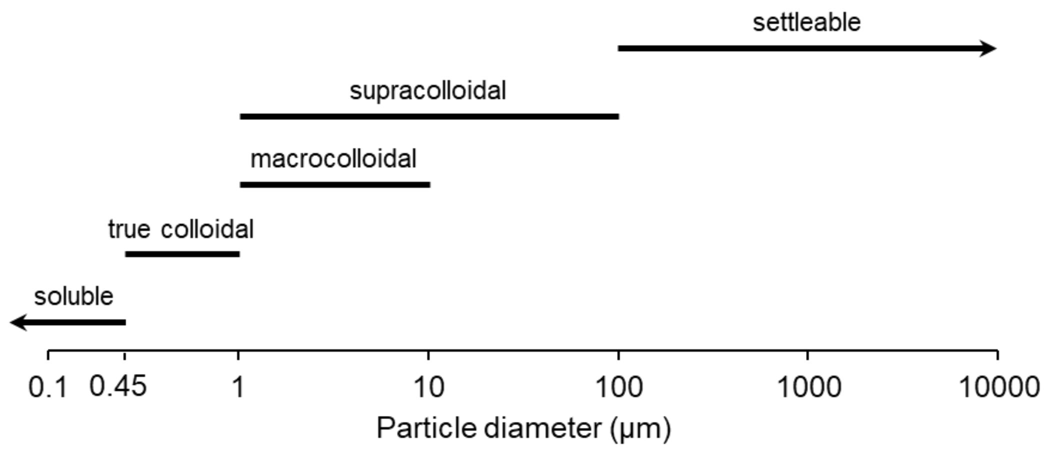


Figure 1.6. Size ranges of different particle fractions in sludges.

In contrast to fecal sludge, activated sludge is characterized by large flocs and high fractions of microbial biomass. Flocs in activated sludge are composed of single bacterial cells, bacterial microcolonies, filamentous bacteria, inorganic particles, organic fibers, and extracellular polymeric substances (EPS) (Nielsen et al. 2012). EPS is secreted by microorganisms, and comprises a large fraction of the organic matter in activated sludge flocs (40-60% of dry weight of flocs) (Christensen et al. 2015). It is made up of negatively charged biopolymers: proteins, polysaccharides, humic substances, lipids, nucleic acids (Christensen et al. 2015, Comte et al. 2006). In this thesis, we distinguish flocs in wastewater from aggregates in fecal sludge in order to avoid implying that they share similar characteristics.

Flocs in wastewater are held together by EPS and its interactions with solution properties (e.g., cations), which facilitate the floc's resistance to shear and determine floc size (Mikkelsen and Keiding 2002, Sponza 2003). The physical-chemical makeup of the EPS in sludge flocs influences floc characteristics and thus dewatering performance. For example, the ratio of proteins to polysaccharides in EPS and the molecular weight of EPS biopolymers influence floc cohesion (Ajao et al. 2021, Wilén et al. 2008, Ye et al. 2011). The microbial community present in the sludge is strongly associated with the characteristics of EPS present in the flocs, and therefore is critical in determining floc properties (Nielsen et al. 2012). There are several mechanisms by which EPS and solution properties contribute to holding flocs together, including bridging, physical entanglement, hydrophobic interactions, and DLVO forces (Kara et al. 2008, Sobeck and Higgins 2002).

Bridging – Divalent cations (Ca^{2+}, Mg^{2+}) allow bridges to form between negatively charged EPS functional groups and the negatively charged functional groups on the surfaces of particles. The bridging mechanism explains why it is often reported that high concentrations of monovalent cations (Na+, K+) and higher monovalent to divalent cation ratios lead to floc disintegration and poor dewatering in wastewater sludges

(Sobeck and Higgins 2002). It is generally agreed that divalent cation bridging is the mechanism that best explains the interactions between EPS and cations and how they contribute to floc properties in wastewater sludges (Christensen et al. 2015).

Physical entanglement – Long biopolymer chains interact sterically with one another and with particles to physically enmesh particles into flocs. Physical entanglement has been proposed as an important flocculation mechanisms for sludges with high molecular weight EPS (Wang et al. 2014). The alginate theory proposed by Bruus et al. (1992) fits into this concept: when alginate, a type of polysaccharide secreted by some microbial species present in activated sludge, is present along with Ca^{2+}, it forms an alginate gel which physically holds flocs together (Sobeck and Higgins 2002).

Hydrophobic interactions - EPS can also contribute to floc cohesion by affecting the hydrophobicity of the floc. More hydrophobic EPS is better for particle adhesion, and proteins are more hydrophobic while polysaccharides are more hydrophilic (Liu and Fang 2003). However, it has been suggested for activated sludge that hydrophobic effects are more relevant for microcolony-level adhesion and cation bridging mechanisms are most important for floc-level adhesion (Wang et al. 2014).

DLVO forces – These forces are also expected to contribute to floc formation or disintegration in wastewater sludges, although they are not explicitly tied to EPS properties and may be more relevant in sludges with low concentrations of EPS (Mikkelsen and Keiding 2002, Yin et al. 2004). DLVO forces encompass the influence of ions in the bulk liquid on surface charge and electrostatic repulsion of particles. Since particles in sludge are typically negatively charged, they will electrostatically repel one another and not agglomerate. If the ionic strength in the bulk solution is high enough then the electric double layer (the cloud of ions surrounding the particle) compresses. As the cloud of ions compresses the electrostatic repulsion reduces which allows particles to get close enough together for short-range attractive forces to take over and facilitate agglomeration (Sobeck and Higgins 2002). In systems where there is no EPS present (e.g., model kaolin clay suspensions), particle coagulation is governed by DLVO forces, although there are some extensions of DLVO theory necessary when considering the aqueous aggregation of particles of heterogeneous size, shape, and surface charge (Islam et al. 1995). Practically, in these low- or no-EPS systems, dissolved salts and other solution properties like pH are expected to determine particle agglomeration and the concentration of suspended particles in the bulk.

The maximum dryness achievable in dewatered sludge is determined not only by how well the free water can be separated (e.g., by settling or filtration), but also by how much water is physically and chemically bound within flocs. In wastewater sludges, although higher concentrations of EPS generally contribute to better settling and filtration, the tradeoff is that EPS also chemically binds water, and more EPS means less water is able to be removed during the dewatering process (Mikkelsen and Keiding 2002, Neyens et al. 2004). This binding of water is part of the reason that anaerobic digestion is employed in wastewater treatment, as it degrades EPS and reduces the water holding capacity of the sludge, allowing for more water to be removed from the dewatered digested sludge (Tchobanoglous et al. 2014).

Effects of anaerobic digestion and storage on floc properties and solid-liquid separation

Anaerobic digestion and storage of activated wastewater sludge leads to poorer filtration and settling performance, but higher maximum dryness in the dewatered sludge. This is due to the anaerobic hydrolysis of EPS, which reduces floc strength and leads to the release of small particles into the bulk, but at the same time releases water which was previously bound by EPS (Christensen et al. 2015, Skinner et al. 2015, Zhang et al. 2019). For activated sludge stored under anaerobic conditions, Rasmussen et al. (1994) saw an increase in free cells in the supernatant and in the particle size fraction from 0.45-10 µm, which they attributed to floc breakdown and associated with increases in supernatant turbidity and declines in filtration performance.

There is not a clear consensus in the literature on the influence of anaerobic digestion on the solid-liquid separation of primary sludge. Mahmoud et al. (2004) saw that filtration performance deteriorated after 10 days of anaerobic digestion at 25 °C, improved after 20 days, and then deteriorated again at 30 days, and linked poor filtration with increased numbers of free cells in the bulk and floc breakdown. Miron et al. (2000) saw an initial decline in filtration performance followed by an improvement, and Houghton and Stephenson (2002) saw a steady decline over the course of anaerobic digestion. The key to explaining this variability in dewatering performance could be the presence of stressful or inhibitory conditions during anaerobic digestion. Lawler et al. (1986) posited that if anaerobic digestion occurs under ideal conditions, it should result in improved settling and filtration. The particle size distribution might be smaller after readily biodegradable particulate matter has been hydrolyzed, due to the breakup of larger flocs, but it is the *concentration* of small particles that ultimately influences filtration and supernatant turbidity (Karr and Keinath 1978). The overall particulate solids concentration in the sludge should substantially decrease over the course of digestion, resulting in improved solid-liquid separation. However, if anaerobic digestion is inhibited in some way (e.g., by lack of substrate availability, presence of inhibitors, non-ideal temperature, or non-ideal redox conditions), complete hydrolysis of small particles cannot proceed. Lawler et al. (1986) saw that larger flocs were broken up under non-ideal temperature and substrate availability conditions, releasing smaller suspended particles, but that the small particles were not subsequently hydrolyzed. This led to a decrease in dewatering performance over the course of anaerobic digestion.

Extrapolating from wastewater literature to fecal sludge

Consistent with wastewater, the mechanisms governing dewatering and settling in fecal sludge will still depend on the same physical rules; for example, suspended small particles will still clog filters and resist settling. However, the physical composition of fecal sludge organic matter and particles could be substantially different from wastewater sludges, leading to different solid-liquid separation behavior.

Typically fecal sludge contains a *smaller fraction of EPS* compared to activated wastewater sludge, as anaerobic conditions are likely predominant in onsite storage (Shaw and Dorea 2021, van Eekert et al. 2019). In addition to facilitating the hydrolysis of EPS biopolymers (Wilén et al. 2008), anaerobic environments promote much slower microbial growth and subsequently less biomass production compared with aerated systems (Madigan et al. 2010). Our recent research supports this: EPS in fecal sludge was 2-10 times lower than in activated sludge (Ward et al. 2019), and Sam et al. (submitted) found that EPS in pit latrine sludge was lower than reported concentrations for wastewater sludges. EPS fractionation is also different in fecal sludge, with humic-like substances contributing to more than half of the total EPS, and the rest of the EPS comprising primarily proteins with undetectable levels of polysaccharides (Sam et al. submitted, Ward et al. 2019). Differences in EPS concentrations and characteristics do change how EPS relates to dewatering in fecal sludge compared to wastewater sludges. In fecal sludge, EPS was linked to poor filtration and high turbidity, but there was no evidence that EPS contributed to binding aggregates together. Associations between slow dewatering, turbid supernatant and high EPS concentrations were observed in field samples (Ward et al. 2019), and controlled studies investigating the addition of EPS to fecal sludge showed that dewatering got slower the more EPS was added (Sam et al. submitted).

Because of the lower concentration and different composition of EPS in fecal sludge, the mechanisms governing the aggregation of particles could differ from those observed in wastewater sludges. At the start of this thesis research, it was not even known whether or not particles in fecal sludge aggregated to any extent. However, microscopic images of field samples revealed that aggregates appear to be present in some but not all fecal sludge samples (Ward et al. submitted), Figure 1.7. Due to lower EPS concentrations in fecal sludge, divalent cation bridging may not be a relevant mechanism in holding aggregates together. The ratio of monovalent to divalent cations was not associated with dewatering time or supernatant turbidity, which supports this conclusion (Ward et al. 2019). Because of the lower concentrations of EPS, it was expected that DLVO forces related to ionic strength, pH, and their influence on surface charge were associated with aggregates and dewatering performance in fecal sludge. However, decreases in dewatering performance of fecal sludge corresponded to increases in cation concentrations and electrical conductivity, which is the opposite trend expected if DLVO theory were governing particle aggregation (Ward et al. 2021a, Ward et al. submitted, Ward et al. 2019). A possible explanation for these results is that high conductivity and poor dewatering and settling are both characteristic of unstabilized fecal sludge, and that transformations during the in situ stabilization process are what governs dewatering and settling in fecal sludge.

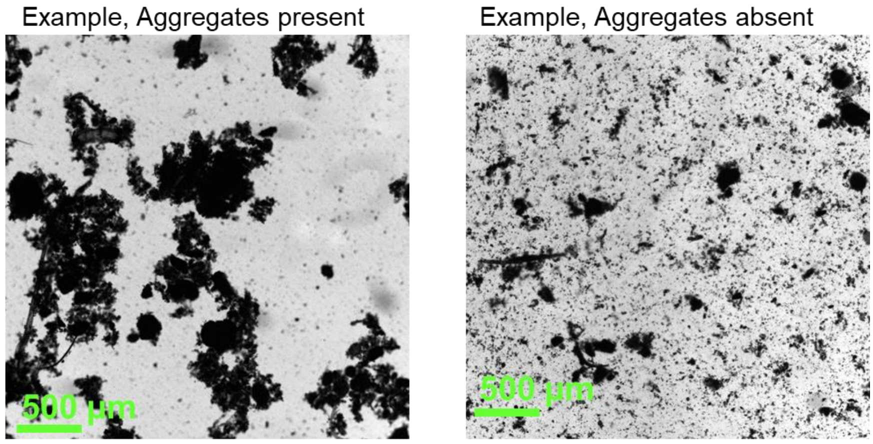


Figure 1.7. Example images from laser scanning confocal microscopy of fecal sludge field samples showing a sample with mostly large aggregates (left) and mostly individual suspended particles (right) (Ward et al. submitted).

How we expect fecal sludge dewatering to change with stabilization

Empirical and qualitative field observations by applied researchers and treatment facility operators indicate that more stabilized fecal sludge has better solid-liquid separation performance (better settling and faster dewatering with less pooling on drying beds) than fresh fecal sludge (Cofie et al. 2006, Heinss et al. 1999). These field observations were corroborated in our recent research, linking qualitative descriptors of sludge stabilization (color, odor) and quantitative stabilization indicators (VSS/TSS, C/N, pCOD/COD) to solid-liquid separation performance in fecal sludge field samples (Ward et al. 2021a, Ward et al. submitted, Ward et al. 2019). There are several possible explanations for what could be happening during stabilization in containment to affect dewatering and settling.

One possible explanation is that, as with primary wastewater sludge, the concentration of small suspended particles is decreasing as anaerobic hydrolysis is carried out during storage (Foxon and Buckley 2008). However, it is important to note that excreta and greywater is regularly added to onsite containments, which means “fresh” organic matter is constantly being replenished. Containments with “more stabilized” sludge could be those where the kinetics of particle hydrolysis are keeping pace with addition of new organic material (Byrne et al. 2017). Another factor associating level of stabilization with solid-liquid separation could be the characteristics of the soluble and colloidal organic material including EPS in fresh sludge. Even small concentrations of humic acids have been shown to strongly increase the electrostatic and stearic stability of colloidal suspensions when adsorbed to the surface of kaolinite clay, and this effect was especially strong for high ionic strength solutions (Kretzschmar et al. 1997). High ionic strength and high concentrations of humic-like substances corresponded to poor filtration and turbid supernatant in fecal sludge from Dakar, Senegal (Ward 2019). A third potential influencing factor could be that in situ biological processes are contributing not only to degradation of particles, but also to aggregation of particles. Changes in the composition of organic matter based on microbial degradation of specific substrates, for example, preferential hydrolysis of polysaccharides or proteins, can alter floc properties by changing the hydrophobicity and charge of biopolymers (Liu and Fang 2003, Sobeck and Higgins 2002). Alternatively specific microbial populations could be producing EPS which promote bioflocculation in situ. High abundances of the Gammaproteobacteria genus Pseudomonas in fecal sludge field samples corresponded to larger aggregates (Ward et al. submitted), faster dewatering, and less turbid supernatant (Ward et al. 2019). Species of Pseudomonas have been linked to bioflocculation and strong aggregate formation in wastewater sludges, especially under high NH_4^+-N conditions (Yang et al. 2021).

Notably, although dewatering performance was linked to level of stabilization, no association between the storage time in containment and the level of stabilization, the amount and fractionation of EPS, or dewatering performance was observed (Sam et al. submitted, Ward et al. submitted). This contradicts the common perception that the amount of time that fecal sludge is stored within containment is a predictor of dewatering performance, based on level of stabilization (Cofie et al. 2006, Lopez Vazquez et al. 2021). Stabilization and solid-liquid separation of fecal sludge in containment may be governed by environmental factors that either facilitate or inhibit the degradation of particles or the formation of aggregates, for example, nutrient limitations (Colón et al. 2015), high antibiotic concentrations (Bischel et al. 2015), high concentrations of recalcitrant organic matter (Krueger et al. 2021), or high NH_4^+-N concentrations (Ward et al. 2019).

Differences in solid-liquid separation based on SPA-DET factors

In addition to level of stabilization, practitioners and researchers have reported differences in fecal sludge characteristics and solid-liquid separation performance associated with spatially analyzable demographic environmental and technical (SPA-DET) factors. Examples of SPA-

DET factors include demographic data (type of establishment, income level, number of users), environmental data (geology, seasonal flooding), and technical data (containment type, emptying frequency, water connection)(Strande et al. 2021b). These relationships can be used to estimate quantities and qualities of fecal sludge accumulating in given areas for use in planning and designing fecal sludge treatment facilities (Strande et al. 2021b). SPA-DET factors that have been associated with dewatering and settling performance include containment type, i.e., pit latrine vs septic tank (Gold et al. 2018a, Ward et al. 2021a), lined vs unlined containment (Gold et al. 2018a, Semiyaga et al. 2017), establishment type, i.e., household vs public toilet (Cofie et al. 2006, Ward et al. 2019), and city where the sludge was collected (Gold et al. 2018a, Ward et al. submitted). Factors contributing to dewatering performance are likely linked to technical aspects or usage patterns represented by different SPA-DET categories. For example, sand content, which influences how easily water can be drained from fecal sludge, can be significantly different between lined and unlined systems (Semiyaga et al. 2017). Divalent cation concentrations, which can influence aggregation behavior, have been shown to be containment technology dependent in Durban, South Africa (Krueger et al. 2021) and establishment type dependent in Dakar, Senegal (Ward et al. 2019). The presence of inhibitors to stabilization are likely to be dependent on SPA-DET factors as well, for example, higher concentrations of NH_4^+-N in sludge from public toilets due to high urine fraction (Ward et al. 2019) or high concentrations of antibiotics in sludge from areas with high prevalence of HIV (Bischel et al. 2015). Differences in emptying practices between cities or containment technologies could also be expected to affect particle size distribution and therefore dewatering. High shear from emptying by vacuum trucks could disrupt flocs and contribute to poorer dewatering performance (Christensen et al. 2015), however this was not observed in samples collected by vacuum truck in Naivasha, Kenya (Ward et al. submitted).

Predicting fecal sludge characteristics for solid-liquid separation

Researchers and treatment facility designers and operators have made use of correlations between SPA-DET factors and dewatering performance to make informed decisions about process control based on influent sludge characteristics. Cofie et al. (2006) described combining influent fecal sludge from public toilets and households in a specific ratio to keep sludge from clogging and pooling on drying beds. For the application of conditioner dosing to facilitate low-footprint dewatering, we estimated that using assumptions about influent characteristics based on SPA-DET data (containment type, water connection, establishment type) was not accurate enough to reliably avoid overdosing (Ward et al. 2021a). Researchers at Sanivation in Naivasha, Kenya conducted pilot-scale experiments to select conditioner dose based on whether the influent sludge was from a pit latrine or a septic tank. While they found that this adjustment did improve polymer dosing, they still experienced incidences of overdosing, which clogged geotextile membranes and necessitated extensive manual cleaning efforts (Ward et al. 2021b). To avoid overdosing, other current efforts to pilot low-footprint dewatering technologies have had to conduct a series of jar test experiments to determine optimal conditioner dose for every new batch of influent fecal sludge (Kocbek et al. 2021). Neither of these approaches is a sustainable solution to dosing conditioner for low-footprint dewatering. As a solution, we proposed models based on sludge photographs (color and texture) and probe measurements (EC, pH) to predict TS for conditioner dosing among other characteristics of influent fecal sludge (Ward et al. 2021a) and developed a prototype app to allow researches and operators at treatment facilities to rapidly characterize influent fecal sludge (Ward et al. 2021c). Our models performed significantly better than the predictions based on SPA-DET data, and may be sufficiently accurate for online dosing of conditioners. Based on the relationships identified in Ward et al. (2021a), Shaw et al. (submitted) investigated the ability of rapid field measurements combined with linear regression models to predict the dose of chitosan for the building-scale dewatering of blackwater. They observed that probe measurements of conductivity and TSS performed very well in predicting the volume of chitosan to add for optimal dewatering performance. The necessity to accurately assess influent fecal sludge characteristics and predict conditioner dose will be critical to address as low-footprint dewatering technologies become integrated into community-scale and decentralized fecal sludge management plans. It is my hope that this thesis has made a contribution to our understanding of the fundamentals of solid-liquid separation in fecal sludge and advanced applied research to enable implementation of low-footprint treatment of fecal sludge.

1.3 Objectives and research questions

The aim of this thesis is to advance our understanding of fundamental drivers of solid-liquid separation in fecal sludge and develop methods for rapid field characterization in order to implement robust and reliable treatment solutions that can be adapted to work with highly variable influent. I will address the roles of EPS, solution properties, and particle size distribution in solid-liquid separation and investigate their relationship with stabilization. Based on these outcomes, I will propose field measurements that can be used together with empirical models to predict dewatering performance and influent characteristics at treatment facilities. In two applied chapters, I will provide guidance on developing, transferring, and scaling up dewatering and drying technologies, and will introduce an app that practitioners can be use to rapidly characterize influent fecal sludge to make process control decisions.

The main objective is divided into three fundamental research questions addressed in chapters 2, 4, and 6, and two questions for applying research outcomes in chapters 3 and 5:

Chapter 2: How does solid-liquid separation of fecal sludge fit into the state of knowledge of wastewater sludges based on EPS and solution properties?

Chapter 3: What do researchers and practitioners need to know to develop, transfer, scale-up, and optimize dewatering and drying technologies for fecal sludge treatment?

Chapter 4: Can predictive models using field measurements be used as a proxy for laboratory-based methods for fecal sludge characterization and solid-liquid separation performance?

Chapter 5: Can an app be developed to automatically integrate field measurements and predictive models for use by researchers and treatment operators?

Chapter 6: Is particle size distribution a determining factor in solid-liquid separation of fecal sludge, and to what extent are differences in particle size and dewatering driven by stabilization?

1.4 Thesis outline

This thesis investigates the drivers of variable solid-liquid separation performance in fecal sludge and evaluates field measurements for monitoring influent characteristics and models for predicting settling and dewatering performance. Chapters are ordered as research articles followed by the applied publications that resulted from the research.

Chapter 2 evaluates how physical-chemical parameters, including EPS and cations, are associated with settling and dewatering in fecal sludge field samples from Dakar, Senegal, and Dar es Salaam, Tanzania. It compares results with the existing conceptual understanding of dewatering for wastewater sludges.

Chapter 3 presents a comprehensive overview of experimental design for developing, transferring, and scaling-up dewatering and drying technologies for fecal sludge treatment. It introduces experimental design and dewatering fundamentals and presents case studies applying experimental design practices to optimize conditioner dosing, dewatering, and drying operations.

Chapter 4 assesses the use of cost-efficient and simple field measurements and questionnaire data to predict fecal sludge characteristics and solid-liquid separation performance when combined with different empirical models. It evaluates a large dataset from Lusaka, Zambia. We identify the best predictors and models and discuss their utility and limitations for different

FSM scenarios (e.g., predicting conditioner dose at a transfer station or predicting influent loadings to design a new FSTP).

Chapter 5 details the development of the Sludge Snap app. The app can be used by a researcher or FSTP operator to collect fecal sludge field data and run the predictive models developed in Chapter 4 to estimate influent characteristics for process control.

Chapter 6 investigates the relationship between particle size distribution and dewatering performance in fecal sludge. Field samples from Kampala, Uganda and Naivasha, Kenya are characterized and subjected to controlled anaerobic storage conditions to evaluate the influence of stabilization on particle size and dewatering performance.

Chapter 7 summarizes overarching conclusions from the entire thesis and makes recommendations for future research and for application of our research by designers and operators of fecal sludge treatment facilities.

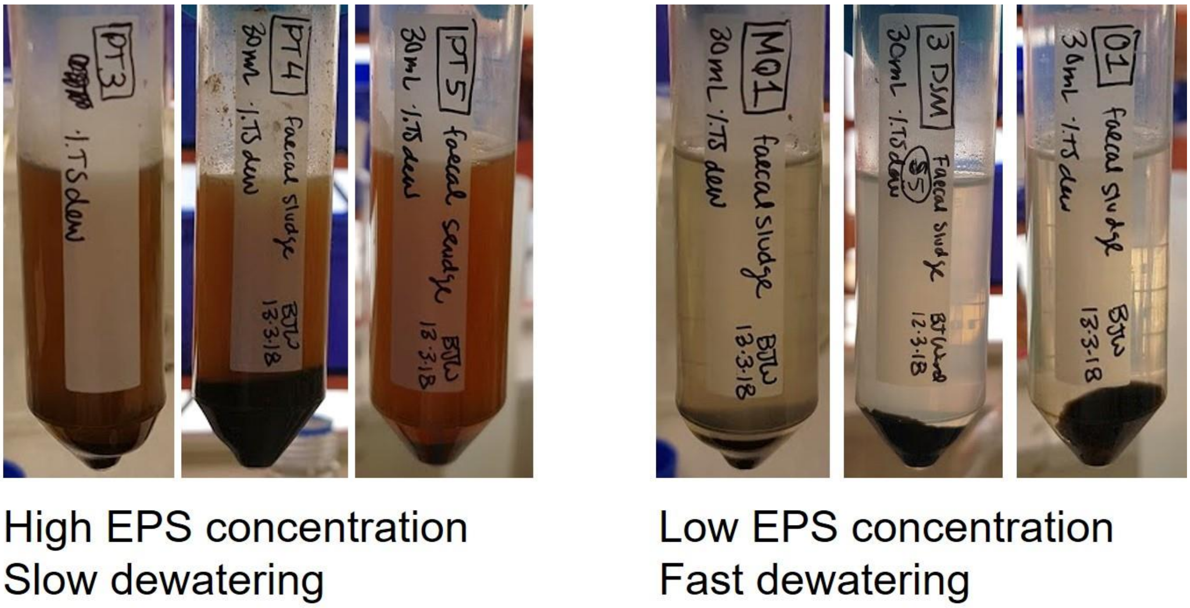
Chapter 2

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

This chapter was published in Water Research as:

Ward, B. J., Traber, J., Gueye, A., Diop, B., Morgenroth, E., & Strande, L. (2019). Evaluation of conceptual model and predictors of faecal sludge dewatering performance in Senegal and Tanzania. *Water Research*, *167*, 115101. DOI: 10.1016/j.watres.2019.115101

Graphical abstract



Highlights

- Dewatering time and turbidity are related to EPS concentration in faecal sludge.
- Dewatered cake solids not linked to EPS in faecal sludge, in contrast to wastewater.
- Higher EPS faecal sludge comparable to digested or primary wastewater sludge.
- EC, pH, and cations correlated with dewatering time and turbidity.
- Demographic data can be a predictor of dewatering time.

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.

2.1 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 (WHO 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 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.

2.2 Materials and methods

2.2.1 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 Appendix A.

2.2.2 Sample analysis

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 mm) after centrifugation at 3,345 x g for 10 min (Park et al., 2006a). 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 $(Na^+ + K^+)/(Mg^{2+} + Ca^{2+})$.

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 x 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 mm) 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 x 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).

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.

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 sludgesamples were centrifuged in 50 mL centrifuge tubes at 3,345 x 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.

2.3 Results

2.3.1 Characterization

Results of faecal sludge characterization are reported, grouped by source, and compared with existing literature values in Table 2.1. 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 cisternflush 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.

Table 2.1. 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., 2006b), [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]

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 practices.

2.3.2 Solid-liquid separation performance

Results of metrics of solid-liquid separation performance by source of faecal sludge are presented in Figure 2.1. 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”.

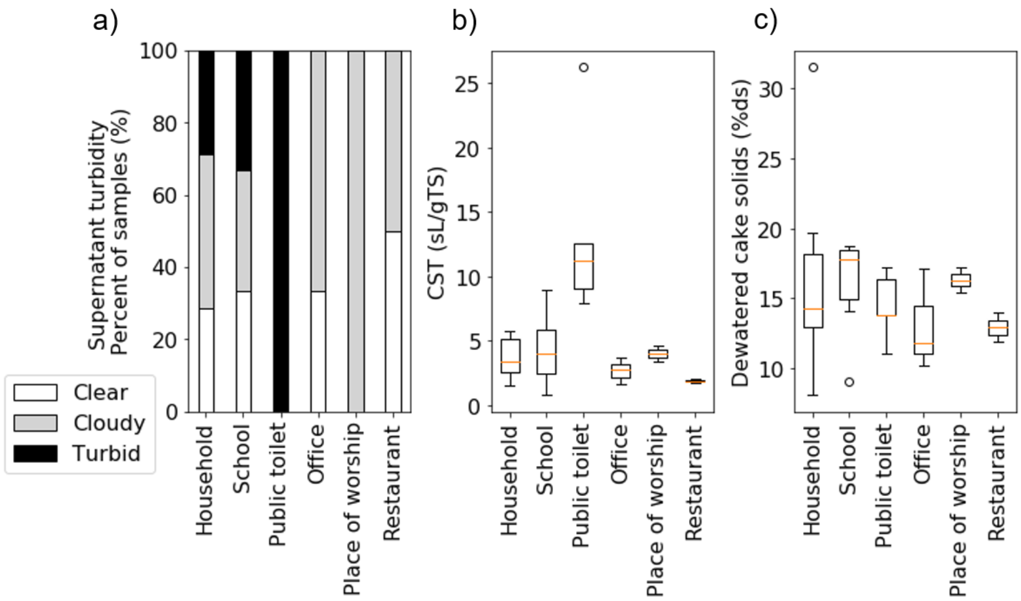


Figure 2.1. Solid-liquid separation performance metrics broken down by faecal sludge source (household (n = 7), school (n = 6), public toilet (n = 5), office (n = 3), place of worship (n = 2), and restaurant (n = 2)). a) Stacked bar graph illustrating percentage of samples in each source category that 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.

Settling

The influence of physical-chemical parameters and EPS on settling performance in terms of supernatant turbidity after prolonged settling is presented in Figure 2.2. 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 Figure 2, that EPS along with other soluble and colloidal organic matter is contributing to turbidity.

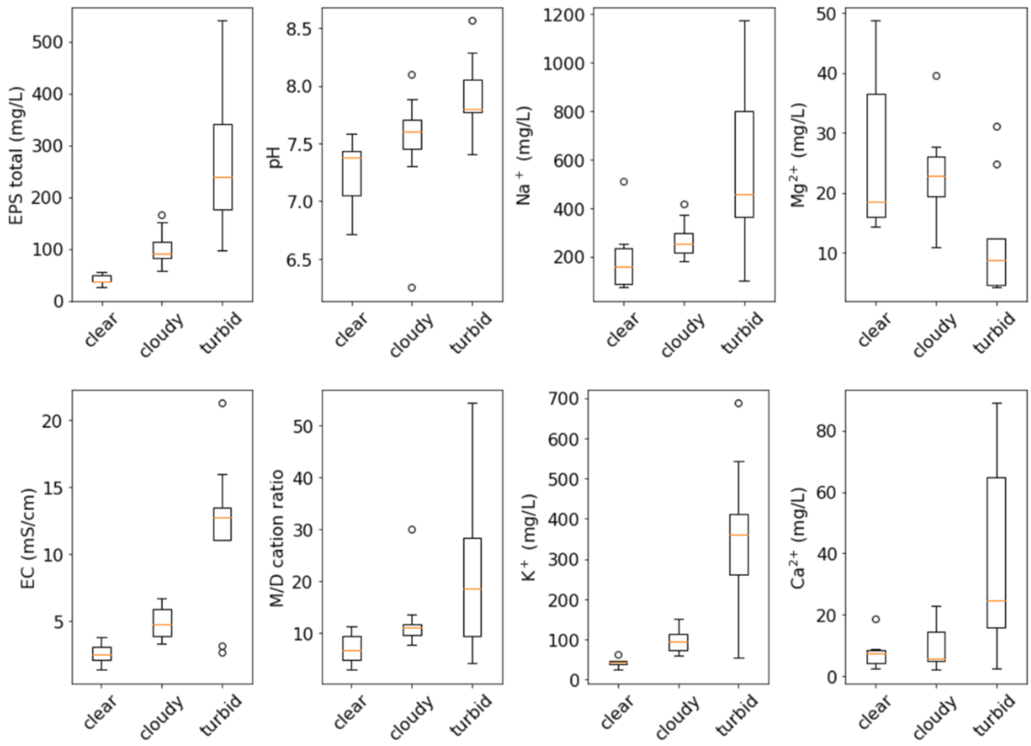


Figure 2.2. Boxplots showing the relationship between a sample's supernatant turbidity after prolonged settling (clear (n = 6), cloudy (n = 10), turbid (n = 9)) and EPS total, EC, pH, M/D cation ratio, and monovalent (Na^+ and K^+) and divalent (Mg^{2+} and Ca^{2+}) cation concentrations.

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 (Figure 2.3a) and higher concentrations of EPS (Figure 2.3d), and specifically humic-like substances (Figure 2.3b). 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 ofstabilization 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.

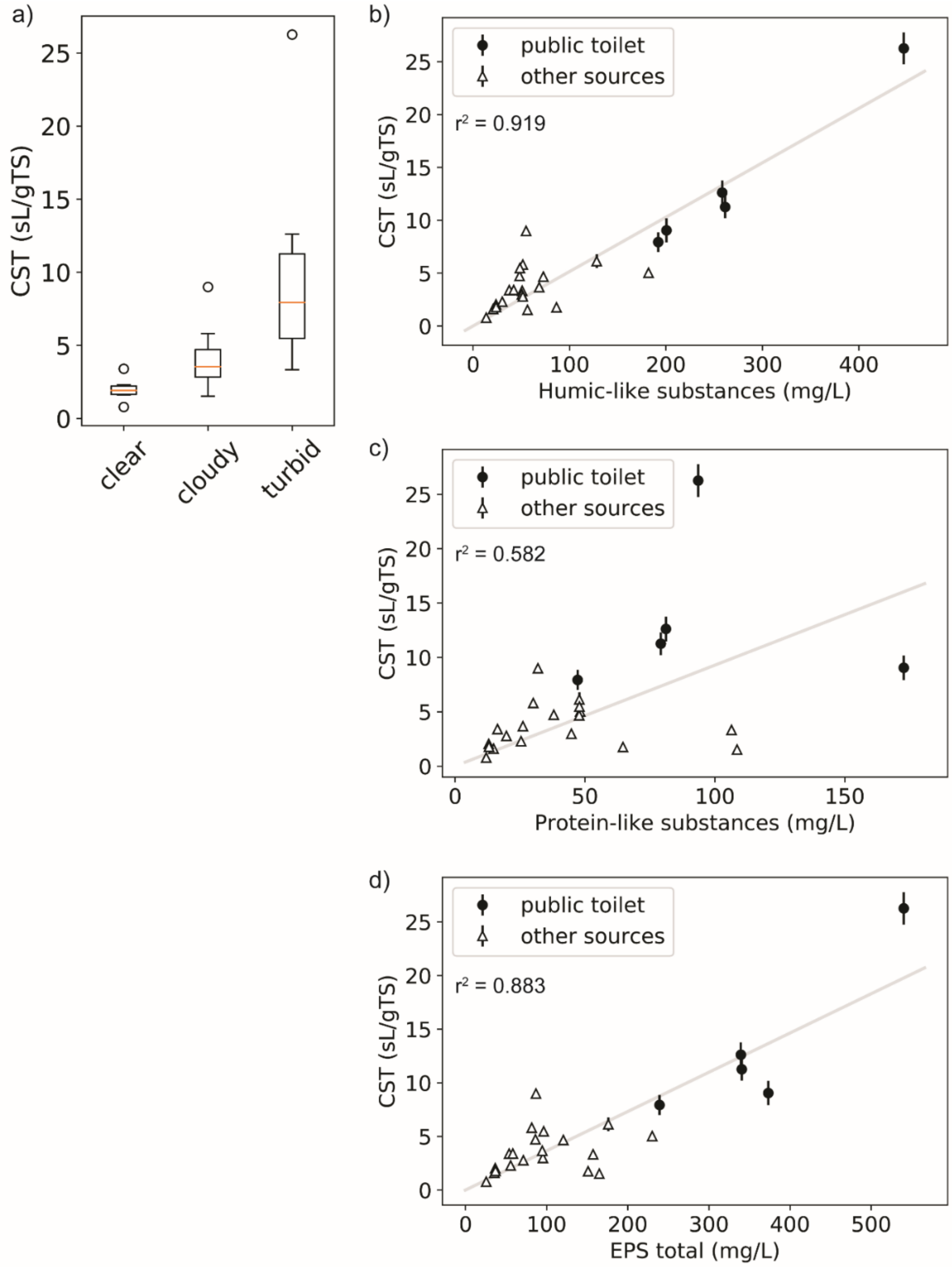


Figure 2.3. a) Box plot showing CST values for samples with clear, cloudy, and turbid supernatant. b) Scatterplot illustrating the relationship between CST and the concentration of humic-like substances, (c) protein-like substances, and (d) EPS total. Linear trend lines and r2 values are included. In 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.

As depicted in Figure 2.4, 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 (Figure 2.3), 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.

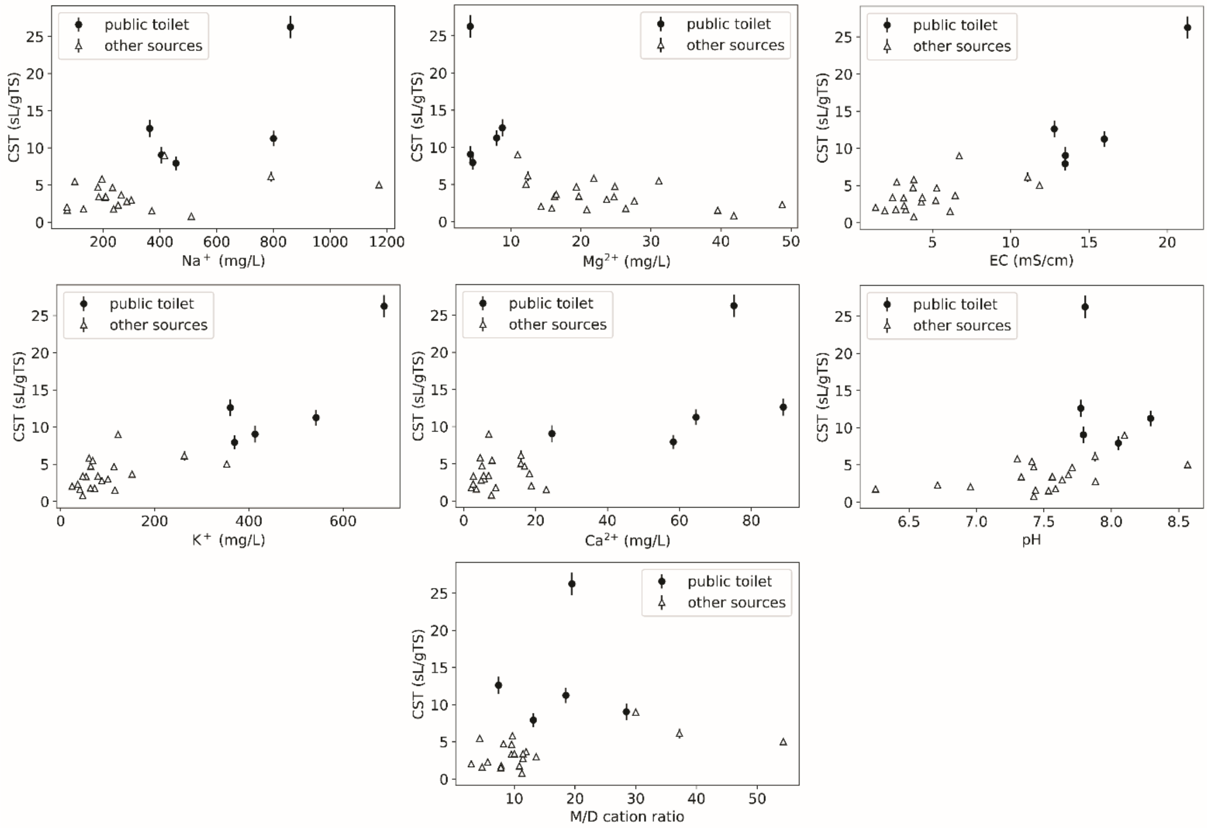


Figure 2.4. Scatterplots illustrating the relationship between dewatering time (CST (sL/gTS)) and the concentration of soluble Na+, K+, Mg^{2+}, and Ca^{2+}, M/D cation ratio, EC, and pH. Samples from public toilets are represented by filled circles, and samples from all other sources (household, school, office, place of worship, and restaurant) are represented by open triangles.

Dewatered cake solids

Illustrated in Figure 2.5, dewatered cake solids were generally higher in faecal sludge with lower VSS fraction, although the correlation 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.

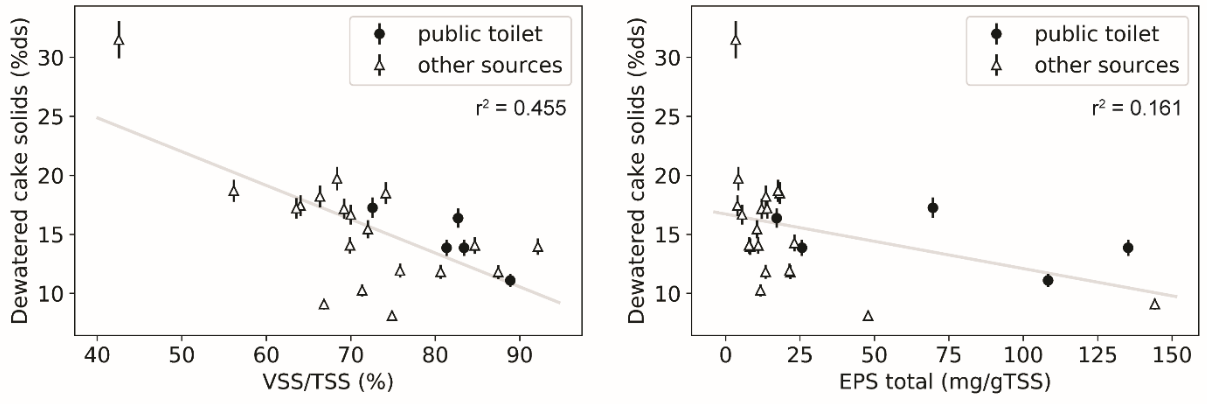


Figure 2.5. Scatterplots plotting dewatered cake solids against VSS/TSS (%) and mass fraction of EPS total. Linear trend lines and r^2 values are included. Samples from public toilets are represented by filled circles, and samples from all other sources (household, school, office, place of worship, and restaurant) are represented by open triangles.

2.3.3 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 Figure 2.6, 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 Appendix A.

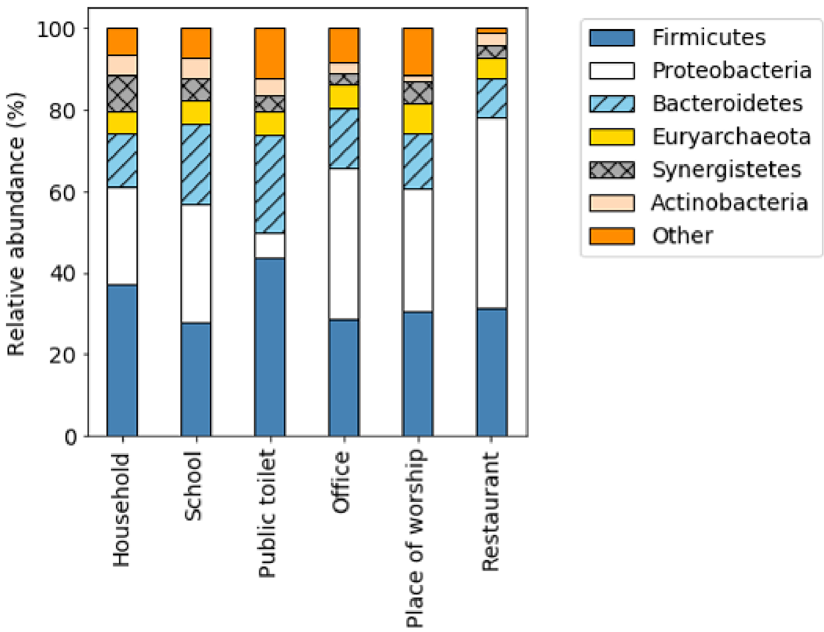


Figure 2.6. Relative abundance of six most abundant phyla in the faecal sludge samples broken out by source.

A correlation was not observed between the relative abundanceof 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 Appendix A). 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.

2.4 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 Figure 2.7. The dewatering behavior 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 2.1), 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).

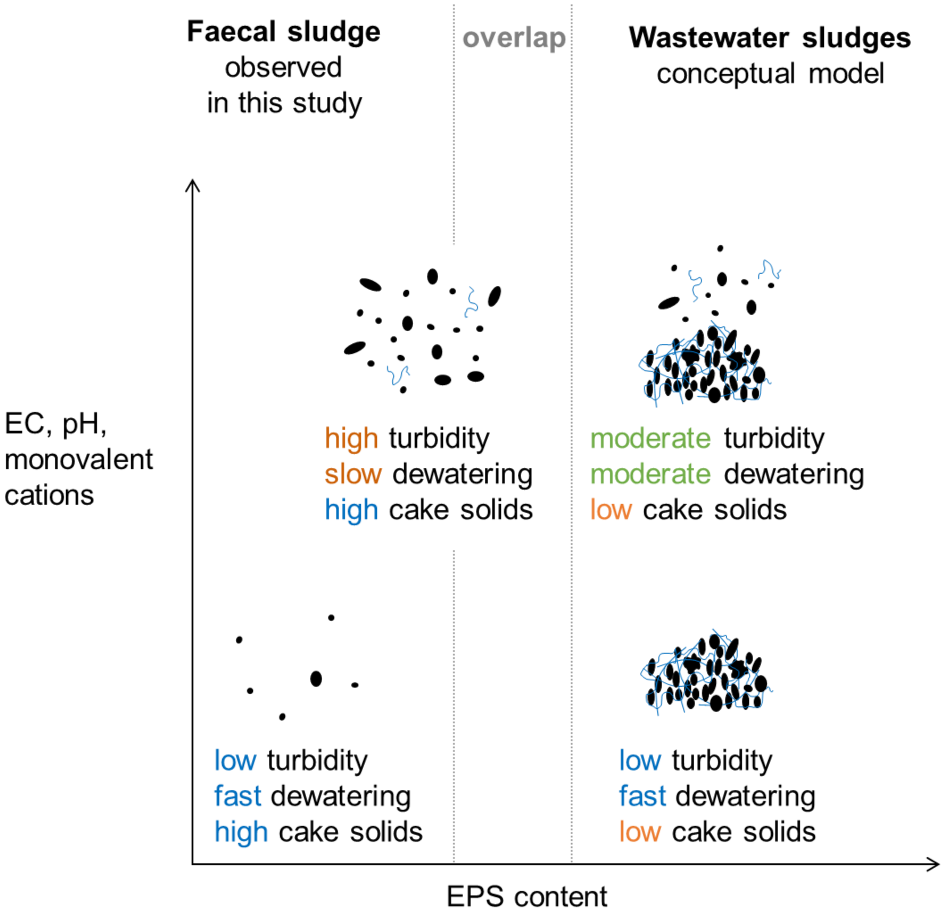


Figure 2.7. Schematic depicting the relationships between EPS content and physicalchemical parameters (conductivity, pH, and monovalent cation concentration) for faecal sludge from this study, and the currently accepted conceptual model for dewatering behaviour in wastewater sludges (Christensen et al., 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.

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., 2020; 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. (2018a) observed that NH_4^+ concentrations correlated strongly with longer dewatering times in faecal sludge from septic tanks and 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 Kþ 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 Figure 2.8, 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.

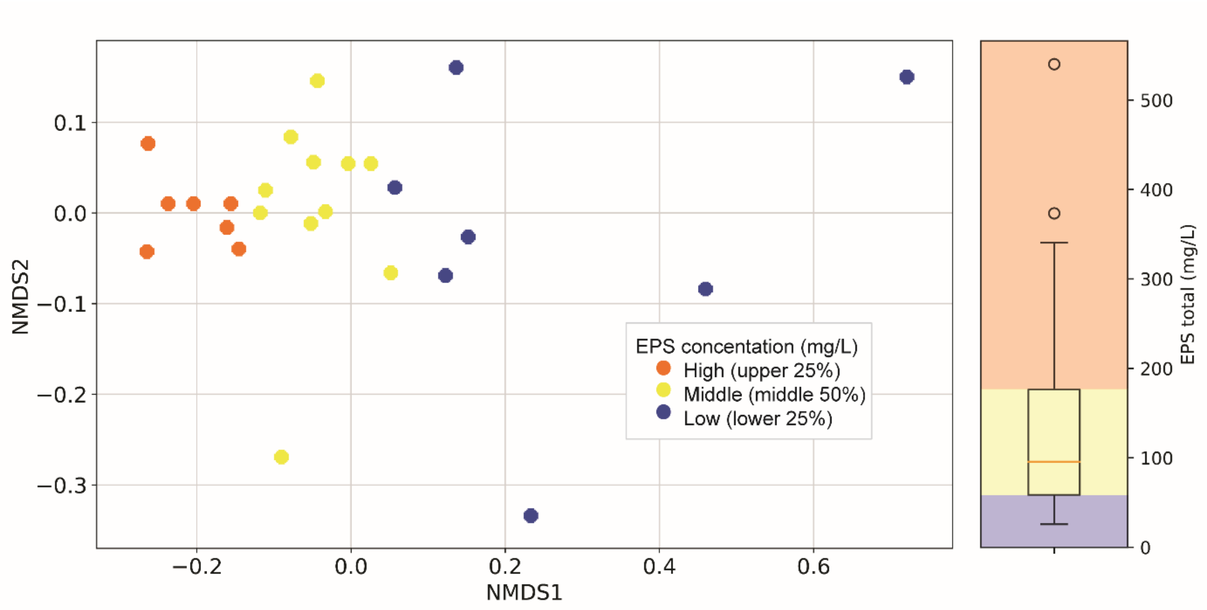


Figure 2.8. Left: Non-metric multidimensional scaling plot of OTU compositions in each sample. Colours denote the EPS concentration in the sample, broken into orange (upper 25% of EPS concentrations), yellow (middle 50%), and blue (lower 25%). Right: Box plot of EPS concentration distribution with colours corresponding to the figure on the left.

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 Figure 2.9, the light grey shaded area depicts the trend observed for faecal sludge in Gold et al. (2018a), 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 behavior 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.

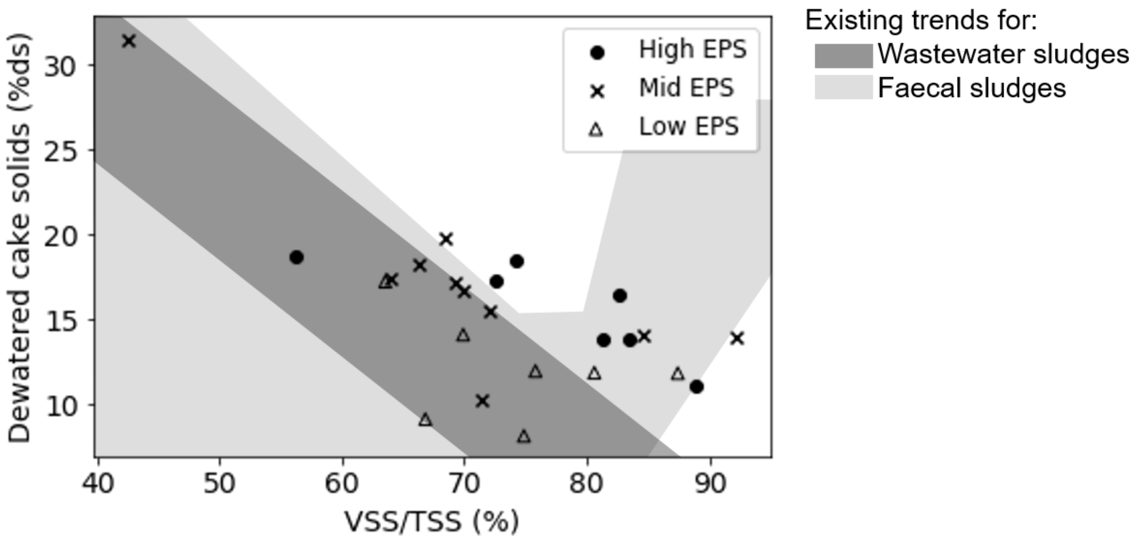


Figure 2.9. Scatterplot illustrating the relationship between dewatered cake solids and VSS/TSS (%). Points represent data from this study, divided into groups based on EPS concentration (circles = upper 25% of EPS concentrations, x’s = middle 50%, and triangles = lower 25%). The dark grey filled area 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).

2.5 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.

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methods, analysed EPS data, and provided insight for EPS discussion. Amadou Gueye supervised fieldwork and laboratory analysis in Dakar, and provided local context and insight during project conception and data interpretation. Becaye Diop supervised work in Dakar, contributed to development of sampling and survey methodology, and arranged for access to FSTPs and laboratory. Eberhard Morgenroth contributed to theory, and helped shape manuscript structure. Linda Strande conceived of the project idea, contributed to theory and writing, and supervised the project.

Supporting information and additional data

Supplementary information is included in Appendix A at the end of this document. The complete dataset is published at https://doi.org/10.17632/w5y55vf3cn.1.

Chapter 3

Experimental design for the development, transfer, scaling-up, and optimization of treatment technologies: case studies of dewatering and drying

This chapter was published in the textbook *Methods for faecal sludge analysis* as:

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3.1 Objectives

The objectives of this chapter are:

- To introduce scales of experimentation and experimental design for the development, transfer, scaling-up, and optimization of faecal sludge treatment technologies
- To provide examples of experimental approaches for scaling up conditioners for dewatering and drying for resource recovery
- To present case studies that address research questions at different scales of faecal sludge treatment processes and technology development and adaptation

3.2 Introduction

This chapter provides a methodology for experimentation in developing treatment technologies for faecal sludge management. A methodology helps to ensure that results are reproducible, reliable for application to design, and available for further interpretation. Experimentation is used to learn about how physical, chemical and biological principles can be employed to achieve defined objectives. In the field of sanitary engineering, an overarching goal is the protection of public and environmental health. With this in mind, sanitary engineers have been active in experimental work around centralized wastewater treatment for more than a century (Stensel and Makinia, 2014, Van Loosdrecht et al., 2016). Developments have included physical, biological, and chemical advances in wastewater treatment plants, first pertaining to removal of solids and organics, then nutrients, and now even micropollutants and trace contaminants. Experimental work has helped to understand fundamentals, develop technologies, and scale up and optimize process trains and treatment steps. More recently, there has been focus on faecal sludge management (FSM), also known as non-sewered sanitation (NSS). The importance of FSM has been gaining acknowledgement, and is recognized as a long-term sustainable solution. A major challenge now is to use experimental work to fill the comparative gap in knowledge, and to develop full-scale, operational solutions for FSM. This will require experimentation to determine how faecal sludge (FS) behaves with different treatment technologies, in order to scale up and design reliable full-scale treatment facilities.

The current state of knowledge in faecal sludge treatment covers technologies that are either *established, transferring*, or *innovative* (Strande, 2017; WHO, 2018). *Established technologies* are those where adequate knowledge exists on how to make recommendations for their full-scale design and operation to protect public and environmental health. Examples of established technologies include settling-thickening tanks, drying beds, co-composting, and stabilization ponds. Experimentation is important for established technologies in order to optimize their use and performance, to further understand treatment performance and mechanisms, and to monitor in order to ensure treatment performance is adequate. *Transferring technologies* are those that are already established in other applications, such as wastewater treatment, and appear promising for use in FSM. Their use has not yet been widely established in FSM, but ongoing research is helping to establish their use and effectiveness. Examples of transferring technologies include mechanical dewatering, conditioners, alkaline treatment, incineration, anaerobic digestion, pelletizing, geotextiles, and thermal drying. Research and experimentation are very important in the transfer of these technologies, because faecal sludge is highly variable and very different in composition from the mixed domestic wastewater that most biological wastewater treatment plants are designed for. *Innovative technologies* are new and emerging technologies that are still under development and not yet established. Due to the level of unknowns, the level of expertise required to design and operate these technologies in a fashion that adequately manages risks is much greater than with established technologies. As further research is carried out, many of them will also hopefully become established technologies. Examples of innovative technologies include, but are not limited to, the use of fly larvae, and ammonia treatment.

The four main treatment objectives that need to be addressed for sustainable faecal sludge management are (i) stabilisation, (ii) nutrient management, (iii) pathogen inactivation and (iv) dewatering/drying (Niwagaba et al., 2014). In this chapter, experimentation for the purpose of scaling up dewatering and drying experience are provided as examples of implementation of the presented methodology. Dewatering is defined here as removal of free water and water that is loosely bound in pores and interstitial spaces of sludge particles and flocs (Figure 3.1). Depending on the properties of faecal sludge, it can be dewatered to between 70 and 80 % moisture by weight, or 20 to 30 % dry solids. Drying is defined here as the further removal of water from the solids fraction following dewatering, for example water trapped within cells or bound to particle surfaces.

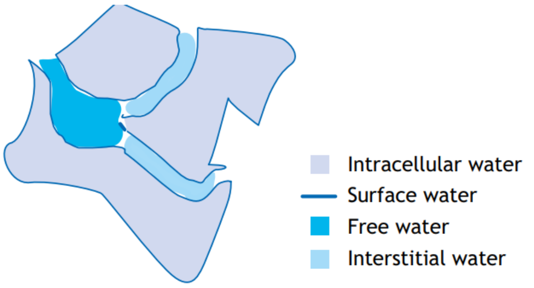

Figure 3.1. Representation of the different forms of water in a sludge floc (adapted from Bassan et al., 2014)

This chapter first discusses the general scales of experimental work and introduces a methodology for experimental design and how to apply these concepts to faecal sludge treatment processes. This is followed by background information that is necessary to apply the concepts for scaling up dewatering and drying, together with five case studies: two for conditioning for improved dewatering, and three for thermal drying for energy recovery. The background and case studies provide examples of how to implement the methods presented in Chapter 8 of Velkushanova et al. (2021b). Prior to conducting experiments at any scale, preliminary research must first be completed. This includes a literature review to learn from experience, and to ensure efforts are not unnecessarily replicated.

In this era of virtual communication, open access to many materials and online communities of researchers and practitioners has made it easier to share findings and obtain feedback and support. To put this advantage to good use, the sharing of research results and raw data is strongly encouraged.

3.3 Experimentation in FSM

Before starting experiments, it is important to become familiar with the key elements used in setting up experimental work. Experiments are a way to understand cause-and-effect relationships in a system, by deliberately changing conditions in a controlled fashion, and observing the changes in the system that are produced as a result of what has been altered. An experiment can be defined as '*a series of runs in which purposeful changes are made to the input variables of a process or system so that we may observe and identify the reasons for changes that may be observed in the output response*' (Montgomery, 2019). A run is one component of an experiment, conducted with a specific set of input variables. Tests are measurements of specific faecal sludge characteristics or properties, as described in Chapters 2 and 8 of Velkushanova et al. (2021b). Tests can be part of an experiment (e.g. measuring pathogen levels in treated faecal sludge), but can also be conducted outside of a planned experiment. For example, laboratory tests are used for routine characterization to monitor the performance of existing treatment systems.

3.3.1 Scales of experiments

In the development and scaling-up of treatment technologies, reasons for experimentation will include developing fundamental knowledge (e.g. mechanisms controlling dewaterability), designing and developing new processes or technologies (e.g. LaDePa), and transferring and optimizing existing technologies (e.g. geotextiles, pelletizers, and conditioners). To accomplish this, the different levels of laboratory-, pilot-, and full-scale experimentation are employed depending on the stage of development and specifically defined objectives.

Laboratory-scale experiments

Laboratory-scale experiments are conducted in a laboratory, often using existing conventional analytical equipment and SOPs. This is the smallest, bench-scale for experimentation, typically using low volumes of faecal sludge (i.e. mL to several L). Laboratory-scale experiments allow for controlled conditions when the experimenter wants to investigate the isolated effects of specific process parameters. This scale of experimentation lends itself well to comparisons with results from other researchers, as they should be replicable in other laboratories with the same setup. One caveat is variability in faecal sludge, which can be addressed through experiments that include simulant faecal sludge, as presented in Chapter 7 of Velkushanova et al. (2021b). Laboratory-scale experimentation is also often used for establishing proof-of-concept for a new technology, and answering questions about fundamentals and mechanisms involved with faecal sludge treatment.

Pilot-scale experiments

Pilot-scale experiments are regarded as a necessary step on the way from laboratory-scale research to full-scale process optimization and implementation (Wood-Black, 2014). Typical pilot-scale experiments operate at capacities between 50-2,000 L of faecal sludge per day. Pilot-scale experiments help to answer questions about practical operation and feasibility of the process. Reasons for piloting a treatment process can be predicting costs and energy requirements, establishing the needs for process control, understanding practical operating conditions, and anticipating any potential unforeseen impacts of adopting a new technology or process unit on the rest of a faecal sludge treatment plant (FSTP). Pilot-scale experiments can ultimately be used to determine whether it is feasible to implement a new technology at full-scale.

Full-scale experiments

Full-scale experiments are conducted at existing FSTPs that have been designed for treatment capacities ranging from 1,000 - 800,000 L of faecal sludge per day (Klinger et al., 2019). Experiments that take place at full-scale are used to optimize FSTP performance. The FSM sector is undergoing rapid change, so transitions from pilot- to full-scale application for many treatment processes is expected to happen with increasing frequency in the near future. However, while full-scale experiments are necessary to make faecal sludge treatment as effective, efficient, robust, and sustainable as possible, they must always be balanced with the responsibility of maintaining certain standards of treatment for the protection of public and environmental health.

Working with faecal sludge and with transferring or innovative treatment technologies inherently includes uncertainties and risks that need to be managed. The transitions between laboratory-, pilot-, and full-scale experiments may be iterative and will require time and dedication to achieve high-quality experimental design and execution. It is critically important to incorporate a research component into any faecal sludge treatment project from its inception. Risks can be mitigated by forming partnerships between municipalities and universities/research institutes, which can help guide experimentation from the start of the project to the optimization and monitoring of a full-scale FSTP (Strande, 2017).

3.3.2 Designing an experiment

After identifying the purpose, rationale, and scale of experimentation, the following guidelines adapted from the book Design and Analysis of Experiments (Montgomery, 2019) can be used to design experiments. Montgomery (2019) can also be consulted for detailed information about experimental design and statistical analysis for process engineering. In addition, a wealth of information on experimental methods is available in Van Loosdrecht et al., 2016 and on experimental data handling and analysis in Von Sperling et al., 2020a.

The experimental design guidelines are presented here, together with examples specific to helminth inactivation during drying (in italics):

1. Specify the research question.

   What is the optimum retention time for drying of faecal sludge in an infrared dryer to achieve complete helminth inactivation?

2. Select the response variable to measure.

   Mean percentage viability of helminths after drying.

3. Identify relevant design factors, levels, and ranges over which the experiment should operate.

   Infrared drying technology can operate over the range of 105-125 °C, so the retention time will be evaluated at 105, 115, and 125 °C. Retention times of 10, 30, 60, and 120 seconds will be evaluated at each temperature.

4. Identify factors that could influence the response variable, and evaluate if they can be kept constant during the experiment.

   Moisture content of air, characteristics of faecal sludge.

5. Identify laboratory methods and SOPs to measure the response variable, influencing factors, and operating conditions.

   See Section 2.8.2.1 Helminth counting methods in Velkushanova et al. (2021a).

6. Determine how many replicates to run to determine the uncertainty in your response variable.

   Triplicate runs for each combination of temperature and residence time.

7. Develop a QA/QC protocol to ensure meaningful results (e.g. standards, blanks, duplicates).

   Use Ascaris suum egg standards with known egg count and percentage viability as a positive control, and sludge simulant as a negative control. Prepare 3 positive controls by spiking sludge simulant with Ascaris suum egg standards. Once the drying *experiments have been carried out, test the negative control and positive controls along with the test samples, as per Section 2.8.2.1 Helminth counting methods in Velkushanova et al. (2021a).*

8. Perform the experiment.

   Carry out experiments as previously described; write down any deviations from the original plan.

9. Interpret the results.

   Visual inspection of data, graphs, statistical interactions, and empirical models.

10. Define the next steps based on conclusions and recommendations from interpretation of the results.

    All residence times tested at 125 °C yield complete helminth egg inactivation; 10 seconds is the recommended residence time based on these results. Conduct further tests on a broader range of sludges, and a cost-benefit analysis of the operating temperatures.

Presented in this chapter are five case studies for dewatering and drying of faecal sludge, together with adequate background information for understanding of the case studies.

3.4 Transferring technology: Conditioning to improve dewatering

Presented in this section is background information on the use of conditioners to improve dewatering of sludge, followed by two real-life case studies of experimental design for faecal sludge conditioning processes.

3.4.1 Introduction to faecal sludge dewatering with conditioners

Prior to dewatering, faecal sludge can be up to 99 % water by weight. Separation of solids and liquids is required in order to fully treat the liquid fraction before end use or discharge into the environment. It is also required before treatment of the solids fraction for disposal or end use, and enables more efficient transportation of the solids fractions.

Separating the solids and liquids in faecal sludge can be achieved through settling (e.g. settling-thickening tanks), filtration (e.g. drying beds or geotextiles), or mechanical methods (e.g. screw presses, filter presses, or centrifuges). Settling-thickening tanks (Figure 3.2) and drying beds (Figure 3.3) are widely-used, established technologies for faecal sludge treatment, however they require large areas of land and long residence times to sufficiently dewater sludge. Depending on its specific properties, sometimes sludge dewaters more quickly and thoroughly, and other times dewatering performance is quite poor. To address this, transferring technologies from wastewater treatment, such as geotextiles (Figure 3.10, Case study 3.2) or mechanical presses are being considered to increase throughput, treatment performance, and reduce footprint. However, these transferring technologies do not reliably or predictably perform without the addition of dewatering aids called ‘conditioners’.

Figure 3.2. Settling-thickening tanks at Lubigi FSTP in Kampala, Uganda (photo: Sandec).

Figure 3.3. Drying beds at Niayes FSTP, Dakar, Senegal (photo: Sandec).

Conditioners are chemicals that are used to improve dewatering and settling performance. They are well-established in wastewater and water treatment, food processing, and the pulp and paper industry, which have relatively more homogenous waste streams than faecal sludge. Empirical and observational knowledge is starting to be gathered about conditioning of faecal sludge at the laboratory- and pilot-scale, but very little fundamental knowledge is available. Further experimentation at all scales will be necessary to scale up conditioners.

Conditioners are mixed into a slurry or suspension, and added to sludge during treatment at optimal ‘doses’. Selection of the optimal conditioner and dose of that conditioner are based on physical and chemical characteristics of the sludge to be dewatered. Accurate dosing is required, as both under-dosing and over-dosing result in poor flocculation, which results in quickly clogged filters and slow or incomplete dewatering performance (i.e. increased organic loadings in the filtrate, clogged or blocked drying beds or geotextiles, and higher residual moisture in dewatered faecal sludge). Dosing needs to be frequently reassessed and varied in response to changes in influent characteristics, and is based on online monitoring, making the high variability in quantities and qualities of influent faecal sludge currently a barrier to implementing them at scale.

Recent research on the optimal dosing of conditioners for faecal sludge has been based on laboratory testing, which is too time- and labour-intensive to be scaled-up. However, it indicates that when the right conditioner and dose are applied, significant improvements to faecal sludge dewatering performance are possible; for example, faster dewatering on drying beds, and cleaner effluent from drying beds, settling-thickening tanks, and geotextiles (Gold et al., 2016). Research needs to be directed at developing methods to rapidly characterise influent faecal sludge quantities and qualities (Q&Q, see Chapter 5 of Velkushanova et al. (2021b)) to determine the conditioner dose (Gold et al., 2018a, Ward et al., 2019). In addition, considerations such as cost, availability, supply chain, chemical safety, and possible requirement of additional infrastructure (storage tank, dosing device, mixing tank) need to be taken into account when designing experiments and selecting conditioners and dosing processes to apply at pilot- and full-scale.

3.4.2 Types and mechanisms of conditioners

The following section has been adapted from Chapter 5 (Section 5.2) of the book Faecal Sludge Management: Highlights and Exercises (Ward and Strande, 2019), and provides additional background information on the use of conditioners to improve dewatering performance of faecal sludge, and methods for measuring performance.

Conditioners can be inorganic chemicals such as lime, ferric chloride or aluminium sulphate, or they can be charged polymers (‘polyelectrolytes’). Polymers can be locally produced from natural materials, such as chitosan or *Moringa oleifera*, or can be proprietary materials sourced from chemical companies. It is expected that cationic (positively charged) polyelectrolytes will work best with faecal sludge, as they will be more likely to interact with organic particles, which are negatively charged. Conditioners work by destabilizing small suspended particles to form larger aggregates (shown in Figure 3.4). This happens through coagulation, which is the initial destabilization and aggregation of colloidal particles. This is followed by flocculation, which is the formation of larger particles, or ‘flocs’, from smaller particles.

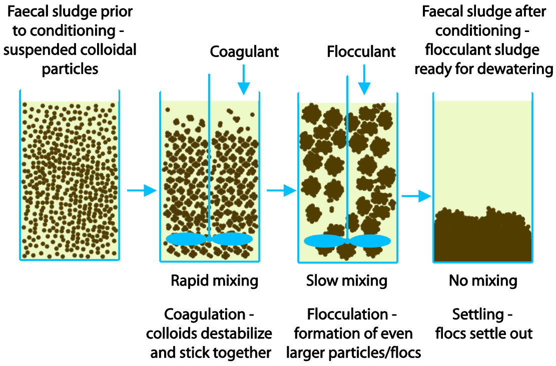

Figure 3.4. Above: steps in faecal sludge conditioning, coagulation, flocculation, and sedimentation; below left: flocculation of faecal sludge; below right: settling of faecal sludge flocs (figure adapted from Ward and Strande, 2019, photos: IHE Delft).

3.4.3 Key parameters for selection of conditioners and optimal dose

The selection of conditioners and the optimal dosage is specific to the faecal sludge properties, the dewatering technology, and the mixing conditions of the chemicals with the sludge.

- *Faecal sludge properties*: conditioners are commonly dosed as a function of total suspended solids, or with faecal sludge often total solids is used in the absence of total suspended solids (TSS) measurements. Other factors such as the electrical conductivity or the degree of stabilization may influence which type of conditioners work best.
- *Dewatering technology*: conditioners need to be compatible with technologies used for dewatering. For example, centrifuge dewatering requires conditioning with polymers that produce flocs that are resistant to high shear (i.e. very high molecular weight, and usually branched or structured polymers).
- *Mixing conditions*: complete mixing of faecal sludge with conditioners is necessary to make the particles collide and stick together (coagulate) and grow into flocs (flocculate); however, mixing speeds need to be selected to avoid floc destruction. Mixing for coagulation needs to be vigorous in order to cause many particle collisions. However, mixing for flocculation needs to be gentle to keep flocs from breaking up. This should also be considered during the selection of pumps for example, for the transfer of conditioned faecal sludge from a settling-thickening tank to a drying bed.

Use of conditioners will also impact the properties of the dewatered faecal sludge, which needs to be taken into account when designing further process steps. Conditioners can increase total solids production, and affect the rheology and water-binding behaviour of the conditioned sludge.

3.4.4 Laboratory- and pilot-scale testing

Laboratory-scale tests used to evaluate the suitability of conditioners in faecal sludge are included in the SOPs (Chapter 8 of Velkushanova et al. (2021b)):

- *Jar test*: a common method for testing conditioner performance at different doses. Faecal sludge is mixed with different doses or different types of conditioner. After mixing, the settling and/or dewatering performance of the conditioned faecal sludge is compared to unconditioned faecal sludge (Figure 3.5).
- *Sludge volume index (SVI)*: a metric for settling performance using Imhoff cones (Figure 3.6).
- *Chemical oxygen demand (COD)*: a metric for organic loading in the supernatant after settling, or in the filtrate after filtering.
- *Total suspended solids (TSS)*: a metric for particulate loading in the supernatant after settling, or in the filtrate after filtering.
- *Capillary suction time (CST)*: a metric for dewatering time (Figure 3.7).
- *Dewatered cake dryness*: a metric for dewaterability, determined by dewatering using a centrifuge or a lab-scale filter press. Dry solids fractions in the dewatered sludge cake are measured and compared.

Figure 3.5. Example of a jar test setup to test suitability of a conditioner (photo: Sandec).

Figure 3.6. Example of an SVI settling test setup with graduated Imhoff cones (photo: Sandec).

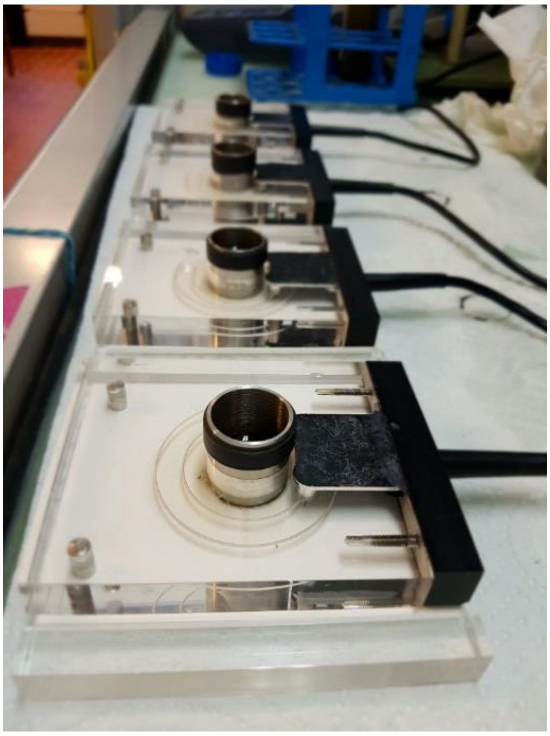

Figure 3.7. Example of replicates being measured in a CST test to determine sludge dewatering time (photo: Sandec).

At the pilot-scale, similar experiments can be conducted with settling-thickening columns, pilot-scale drying beds, or pilot-scale mechanical presses. Specific considerations when transitioning from laboratory experiments to pilot-scale conditioner trials include mixing conditions and sampling protocols. Replicating mixing speed and turbulence achieved during laboratory-scale jar tests is often difficult at pilot-scale. The shape and power of the mixer, and shape and aspect ratio of the mixing tank influence the completeness of mixing, and may therefore alter the optimal dose. Sampling protocols are another point to consider when scaling up. If the pilot-scale experiments require a comparison of faecal sludge properties before and after conditioning, mixing, and dewatering, the pilot facility should be designed to accommodate this.

3.4.5 Case studies – conditioning for improved dewatering

In the following case studies, examples are provided of (i) a laboratory-scale comparison of different conditioners followed by discussion of how to implement pilot-scale testing on drying beds, and (ii) an account of a pilot-scale study of online conditioner dosing combined with geotextile dewatering, with lessons learned for full-scale implementation.

Case study 3.1: Evaluating conditioners produced from locally-available materials for improved faecal sludge dewatering in Dar es Salaam, Tanzania

This case study is based on a two-year Master´s project by Nuhu Moto at the University of Dar es Salaam (UDSM), a collaborative project between Eawag and UDSM in Dar es Salaam, Tanzania (Moto et al., 2018). This project was motivated by the desire to increase the capacity of unplanted drying beds at an FSTP. Laboratory-scale experiments were conducted to find out whether conditioners could be a possible treatment option for faecal sludge in Dar es Salaam. Two conditioners, which could be produced from locally-available materials, were compared using jar tests, and conclusions were drawn about which conditioners and which doses to select for pilot-scale drying bed trials.

Research question

Which locally-available conditioners and at which doses should be selected for pilot-scale trials?

Response variables

- CST was used to quantify filtration time.
- TSS of the supernatant after settling was used to quantify particulate removal.

Factors, levels, and ranges

- Type of conditioner tested.
  Two types of conditioners that could be manufactured from locally-available natural materials were tested: chitosan and *Moringa oleifera.*
- Conditioner dose
  0, 1, 2, 3, 5, and 8 mg/gTS for chitosan and 0, 10, 50, 100, 250, 500, 750, 1,000 mg/gTS for *M. oleifera*

Factors that might influence the response variables

- Mixing speeds and durations and beaker size/shape can influence results of a jar test. To avoid interference from these factors, consistent mixing speeds, mixing durations, and beakers were used for all the jar tests.
- Physical-chemical characteristics of faecal sludge (TS, TSS, pH, conductivity) can affect how well a conditioner works. To account for this, one large faecal sludge

sample was used for every jar test, and care was taken to homogenize the sample well so that all the beakers contained representative sludge. To make sure that they were not selecting the best conditioner and dose for just one specific batch of sludge, jar tests were run with multiple faecal sludge samples.

- Faecal sludge processing procedures (e.g. homogenizing with a blender) can change the dewatering performance of a sludge. Blending can disrupt particles and flocs, which can change dewatering behaviour, so homogenization was done by hand mixing so as to not destroy particles.

Experimental design details

The number of replicates was based on suggestions in standard methods for specific SOPs. An optimal conditioner dose was defined as the lowest dose that achieves > 75 % reduction of CST (based on literature, explained in Ward and Strande, 2019).

Interpreting the results

To determine the optimal doses of chitosan and *M. oleifera*, jar tests were performed with the following concentrations of conditioners, and the capillary suction time (CST) and TSS of supernatant were measured. Results for CST are shown in Table 3.1 and Figure 3.8. Trends in TSS were similar to trends in CST, and are not shown.

Table 3.1. Results of jar tests to determine the effect of different doses of conditioners chitosan and M. oleifera on CTS reduction.

Conditioner	Dose (mg/gTS)	Reduction in CST (%)
Chitosan	0	0
	0.5	45
	1	60
	2	79
	3	88
	5	90
	8	92
M. oleifera	0	0
	10	13
	50	25
	100	33
	250	68
	500	83
	750	87
	1,000	80

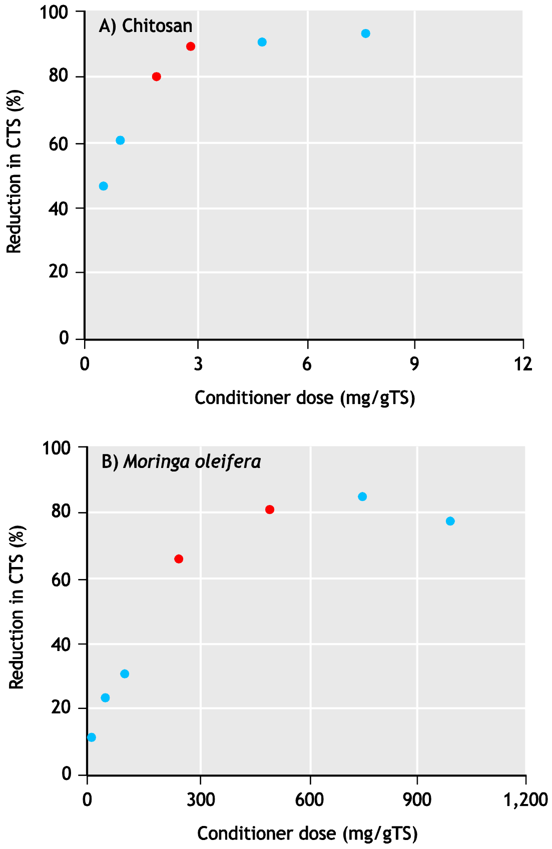

Figure 3.8. A) Results of jar tests with chitosan. B) Results of jar tests with *M. oleifera*. The red dots indicate the optimal dose of each conditioner.

The results indicated that for this sludge, the optimal dose of Chitosan is approximately 2-3 mg/gTS, and the optimal dose of *M. oleifera* is approximately 250-500 mg/gTS (the red dots on the graphs).

Scaling-up from laboratory to pilot-scale

Both conditioners that were tested achieved similar performance in terms of CST and TSS reduction, but the optimal doses for each were very different. In Dar es Salaam, chitosan was estimated to cost 15 US$/kg and *M. oleifera* 30 US$/kg. The cost of each conditioner at optimal dose for 1 tonne of faecal sludge (with TS of 10 g/L) would be 0.38 US$ for chitosan and 112 US$ for *M. oleifera* (see Ward and Strande, 2019 for full details). Because *M. oleifera* was prohibitively expensive at the optimal dose, only chitosan was chosen to proceed to the pilot-scale trials (Figure 3.9).

Figure 3.9. Left: mixing chitosan conditioner for pilot-scale trials. Right: the pilot-scale dewatering research facility at the University of Dar es Salaam, including the settling-thickening tanks, conditioner mixing tank, and six sand drying beds (photos: Sandec).

New research questions were developed for the pilot-scale experimentation, including:

- Does chitosan decrease residence time on unplanted drying beds?
- Can chitosan be used to condition every batch of incoming faecal sludge, or does it only work for sludge with certain physical and chemical characteristics?
- What is the economic value of decreased residence time on drying beds? Is the increased treatment capacity worth the cost of the conditioner?

For more information on the results, refer to Moto et al., 2018.

Case study 3.2: Scaling-up conditioner dosing for full-scale faecal sludge dewatering

This case study is based on research by Naomi Korir, Jonathan Wilcox, and Catherine Berner at Sanivation in Naivasha, Kenya. This pilot-scale research was done to inform the design of a full-scale dewatering process for a new FSTP in Naivasha, Kenya (capacity 4,000 tonnes FS per month, delivered by vacuum trucks from pit latrines and septic tanks). Requirements for the plant included a small treatment footprint for the dewatering step, and economic viability. Previous laboratory-scale research characterized hundreds of samples of faecal sludge from Naivasha and established the selection of polymer conditioner and the optimal dose for flocculation. Sanivation wanted to scale up dewatering with geotextiles. To do this requires experimentation for the online dosing, as presented in Section 3.3.1. Because of the iterative experimental approach, questions should be answered one at a time. Therefore the following experiments were carried out on the assumption that geotextiles would work. The pilot-scale setup was sized to process sludge from one vacuum truck at a time, and was designed to test different online conditioner dosing and mixing configurations followed by a subsequent dewatering step using geotextile skips suspended on metal supports (Figure 3.10).

Figure 3.10. A) a geotextile skip setup at the pilot facility; B) a geotextile skip being loaded with conditioned faecal sludge; C) dewatered sludge ready to be unloaded from a geotextile skip (photos: Sanivation).

Research question

What is the optimal configuration for online dosing and mixing of conditioners?

Response variables

Sanivation defined the 'optimal' dosing configuration as one that yields fast dewatering and low solids loading in the filtrate while requiring the lowest possible conditioner cost.

- Dewatering time was the amount of time it took for sludge to dewater in geotextile skips (residence time); sludge was considered 'dewatered' when it reached 15-20 % dry solids (75-80 % moisture). This benchmark was chosen as it is the required input dryness for Sanivation's heat treatment method, the next step in the treatment process.
- Filtration efficiency was used to quantify how well the geotextiles filtered solids from the incoming faecal sludge. Filtration efficiency was calculated using measured values of TSS of the influent faecal sludge (TSS_{FS}) and of the filtrate leaving geotextile skips ($TSS_{filtrate}$), using the following equation:

$$Filtration\,efficiency = \frac{TSS_{FS} - TSS_{filtrate}}{TSS_{FS}} \qquad (3.1)$$

Every batch of filtrate was also characterized for TS, COD, BOD, ammonia and nitrates, to understand the removal of different pollutants by the geotextiles, and the type of treatment that would be required to treat the liquid effluent to required standards (NEMA Standards).

- Cost of polymer per tonne faecal sludge was used to predict material costs for a full-scale process

Factors, levels, and ranges

- Dosing configurations: different numbers of dosing ports (one or multiple dosing ports) and different mixing conditions (no mixing, mixing with baffles, mixing with a mechanical stirrer) were tested (Figure 3.11). Figure 3.12 shows the actual setup.
- Conditioner doses: the laboratory-scale conditioner experiments indicated that the optimal polymer conditioner dose was 2 g polymer per kg FS; however, the Sanivation team suspected that due to different mixing conditions at the pilot-scale, the optimal dose for the scaled-up process could be different. Doses of 2-60 g polymer per kg FS were tested at the pilot-scale.
- Geotextile cleaning methods: geotextiles were cleaned to determine whether their lifetime could be extending by cleaning and reusing between faecal sludge batches. Three cleaning methods were investigated (detergent, detergent + salt, detergent + salt + high-pressure water rinse (Figure 3.12).

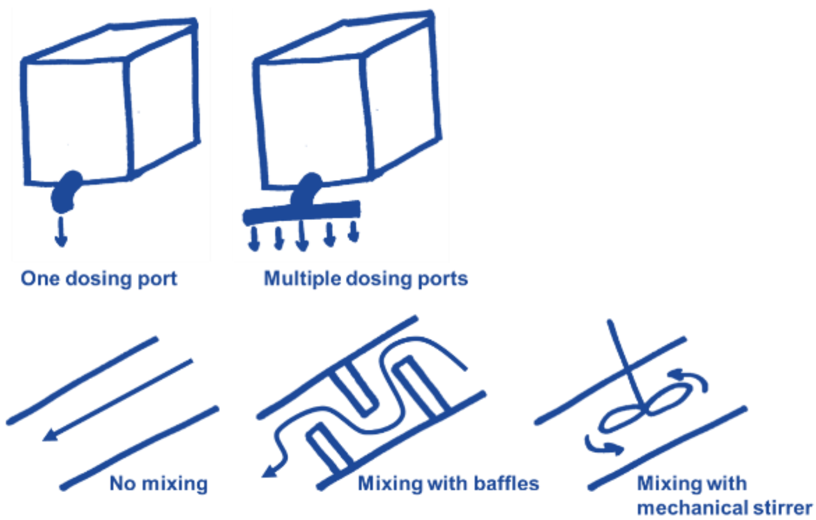

Figure 3.11. Diagram representation of the different conditioner dosing and mixing configurations evaluated by Sanivation.

Figure 3.12. Left: an example of a conditioner dosing configuration: one dosing port followed by mixing with baffles and right: a Sanivation employee washes detergent, salt, and particulate residue from a geotextile using a high-pressure water rinse (photos: Sanivation).

Factors that might influence the response variables

- Age of the geotextile/frequency of cleaning can affect dewatering time. New geotextiles dewater quickly (several minutes for septic tank faecal sludge, several hours for pit latrine faecal sludge), but older geotextiles require more time. To account for this, trials were carried out with three geotextile skips that were the same age and had undergone the same cleaning regimen.
- Weather: rain and humidity can affect how long it takes sludge to dewater, since geotextile boxes were open to the air and could gather rainwater. To account for this, the Sanivation team set up an evaporation trial with a tray of water exposed to the same conditions as the geotextiles. Rainwater did not accumulate in the tray for timescales significant to the study.
- Physical-chemical characteristics of sludge can change the optimal dose and dewatering speed. Every batch of incoming faecal sludge was characterized for TS, TSS, COD, BOD, ammonia and nitrates. Sanivation engineers designed different conditioner dosing flow rates for pit sludge and septic sludge to account for higher levels of observed TS in sludge from pit latrines and lower levels observed in septic tanks.

Experimental design details

Each dosing configuration and each geotextile cleaning method were typically trialled with at least one batch of pit latrine sludge, and one batch of septic tank sludge. If the first repetition was not successful, then further replicates were not completed. For promising configurations, more replicate testing was performed to determine the reproducibility and variability of performance.

Interpreting the results

The optimal conditioner dose was not directly transferable from lab-scale studies to pilot-scale. Different, less ideal mixing conditions at the pilot-scale called for increased doses of polymer to be used to account for incomplete mixing with sludge particles. Multiple dosing ports performed better than a single port, and the addition of both baffles and mechanical mixing led to the most thorough mixing of conditioner and subsequently the shortest dewatering times in the geotextile boxes (less than 5 days compared to 14 days with an incorrect conditioner dose) and highest filtration efficiency. With the optimal setup, polymer doses from 2-8 g/kg produced the best results.

Overdosing occurred at doses over 8 g/kg, resulting in immediate clogging of the geotextiles and a prolonged dewatering time. The team continued to experience issues with achieving precise dosing with respect to TS. Because of this, it was difficult to avoid overdosing even when doses <8 g/kg were targeted.

Geotextiles were able to be reused after employing the optimal cleaning method: detergent + salt + high-pressure washing. After cleaning, geotextiles were restored to about 30% of the performance of original unused geotextile at negligible cost increase. However, cleaning was labour-intensive and required 1.5 hours of work to clean every bag after every loading/unloading cycle.

Scaling-up from pilot to larger-scale FSTP

Based on their performance at pilot-scale, the Sanivation team decided not to scale up geotextile skips. This decision was based on the estimated land area required for dewatering using performance data from optimized dosing, mixing, and geotextile cleaning processes in place (with mechanical mixing, multiple dose ports, and cleaning between every load cycle). The average residence time in the geotextile skips at optimal conditions was 5 days per truckload. The full-scale FSTP is designed for a capacity of 20-25 truckloads per day, and the footprint of a geotextile skip is 8 m^2. In the best-case scenario involving constant operation 7 days/week and just one day to unload and clean a geotextile skip, 150 geotextile skips would be required, which means 8 $m^2 \cdot 150$ = 1,200 m^2 or 0.12 hectares of land would be required for dewatering (10% of the entire land allotment for the new FSTP). Labour costs were also a significant factor in the decision not to scale up geotextiles. Sanivation also identified that geotextiles can be reused for dewatering up to 10 times with washing in between loadings.

Sanivation is moving forward with the design and implementation of their full-scale FSTP, and will proceed with their optimal polymer dosing configuration. However, the team will switch to an alternative low-footprint dewatering technology, the screw press. The screw press technology is more resilient to overdosing and the team hopes it will not clog as easily as geotextiles. Screw presses operate continuously instead of being batch processes like geotextiles, allowing for a higher throughput of 20 m^3 sludge per hour. The allotted footprint of the full-scale dewatering process is 120 m^2, an order of magnitude lower than geotextiles would have allowed. Piloting experiments with screw presses are now planned in order to inform the FSTP design. New research questions can be asked, for example, ‘What are the optimal operation conditions of the screw press (hydraulic loading rate, conditioner dose, wash water flow rate)?’.

Fast, easy, and reliable methods for online measurements to adjust conditioner doses are still lacking. This is one of the key research topics that needs to be addressed in order to avoid overdosing and reduce conditioner costs. Research is actively being pursued to advance this knowledge (Ward et al., 2021a). When accurate methods for online dosing have been adequately developed, the use of geotextiles will be more readily transferable to faecal sludge. However, there are other cases where geotextiles are currently being successfully employed for dewatering, for example, the Dumaguete FSTP in the Philippines (Strande, 2017).

3.5 Transferring technology: Thermal drying for resource recovery of dried sludge for energy

Presented in this section is background information on thermal drying of sludge, followed by three real-life case studies of experimental design for thermal drying processes.

3.5.1 Introduction to resource recovery of faecal sludge as solid fuel

Producing value-added end products from faecal sludge can be an incentive for appropriate management and treatment. Revenue from resource recovery can be used to offset operational and maintenance costs at FSTPs, which can incentivize adequate collection and delivery of sludge to treatment plants and achievement of consistent treatment targets (Diener et al., 2014). A market-driven approach should be used to determine the revenue potential from possible end products of faecal sludge treatment (Schoebitz et al., 2016). In Accra, Ghana and in Kampala, Uganda, use as a solid fuel for manufacturing industries (e.g. brick and cement factories) was identified as a high-demand end product of faecal sludge (Diener et al., 2014). Many industries in these cities typically rely on wood and waste biomass, and struggle when availability of these fuels fluctuates. Solid fuels produced from faecal sludge can have comparable energy densities to these traditionally used fuels (Andriessen et al., 2019; Gold et al., 2017; Muspratt et al., 2014). The decision to target resource recovery allows FSTP designers to set treatment targets based on the requirements set by the consumers (e.g. moisture content, energy density, pathogens), and select appropriate treatment technologies accordingly.

3.5.2 Introduction to faecal sludge drying

Drying is a requirement for producing solid fuels from faecal sludge. In addition to increasing net energy gains (Muspratt et al., 2014; Septien et al., 2020), drying also reduces the mass, making it easier to handle and decreasing transportation costs. Drying can be achieved passively, for example with drying beds, but this requires a large footprint and long residence times (weeks to months). Hence, researchers are pursuing heat drying of dewatered faecal sludge as a transferring technology from the food processing industry. One example is the LaDePa process, developed by the eThewkini municipality and Particle Separation Systems (in Durban, South Africa). The LaDePa can be used at a full-scale treatment plant to dry and pasteurize sludge from ventilated improved pit latrines (VIPs) (see Case study 3.4 and Septien et al., 2018a). Another example is the Tehno Sanitizer® (also known as The Shit Killer®), based on microwave technology that has been used for food drying for years (e.g. pasta, fruit etc., see Case study 3.5). Requirements for how much moisture needs to be removed are dictated by the treatment process design and by the end-user requirements. Different technologies require different input moisture contents, and further drying may be necessary after sludge has been processed, as illustrated in Figure 3.13. In general, solid fuels do not perform well if they contain too much moisture, but this needs to be balanced with higher energy inputs or longer drying times.

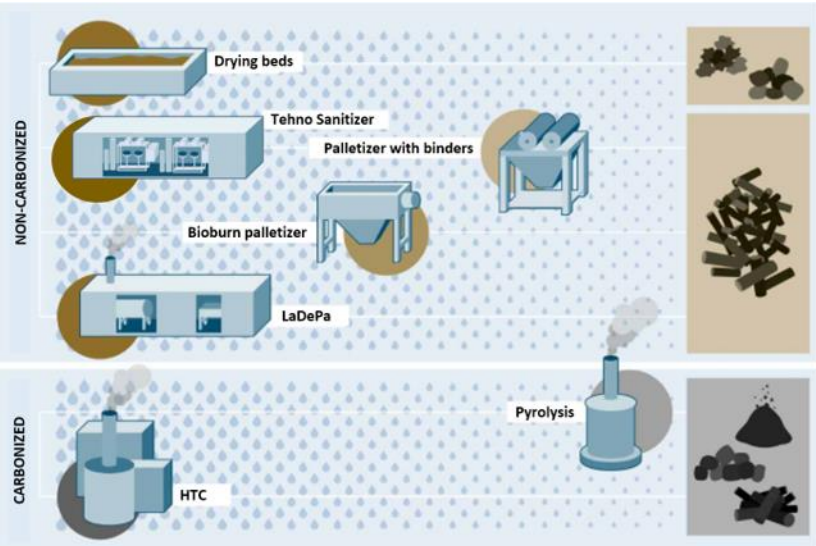

Figure 3.13. An overview of technology options for producing solid fuel, starting from dewatered faecal sludge at 80% moisture and ending at non-carbonized or carbonized solid-fuel end products. The position of the technology icons from left to right indicates the required dryness of the input sludge for each technology, as indicated by the size of the droplets, ranging from 80 % moisture on the left to 10 % moisture on the right (figure and caption adopted from Andriessen et al., 2019).

It has been difficult to adapt and scale up drying technologies to full-scale faecal sludge treatment processes. Drying technologies face many of the same challenges as any faecal sludge treatment process, for example high variability in quantities and qualities of the influent sludge. However, drying faecal sludge presents its own specific technical challenges as well. These include the high energy demand, the release of strong odours during drying, and the stickiness acquired by faecal sludge during the drying process. As with conditioning, more research on the mechanics of faecal sludge drying is required to generate a fundamental understanding of the process and to inform the development and adaptation of well-functioning drying processes.

3.5.3 Types and mechanisms of thermal drying (technical background)

Understanding the underlying physical, chemical, and biological processes supporting a technology is crucial for making informed decisions about adapting it to work with faecal sludge. During thermal drying, heat is transferred to the sludge from a heating source (e.g. hot fluid, heated wall, infrared radiation) or generated internally after conversion of another form of energy (e.g. microwave, dielectric radiation), leading to the movement of moisture to the sludge surface where it evaporates. The rate of drying depends on the temperature or irradiance from the heat source, humidity, flow rate, and pressure of the ambient air, and on the area, thickness, and thermal properties (i.e. heat capacity and thermal conductivity) of the exposed sludge surface. A schematic representation of heat drying of faecal sludge is shown in Figure 3.14.

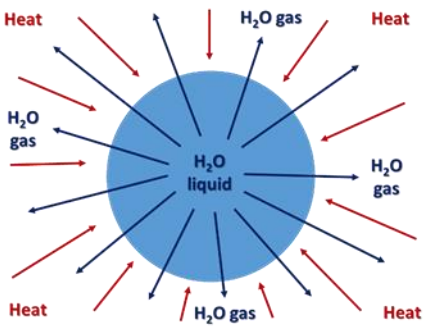

Figure 3.14. Schematic representation of drying faecal sludge. Red arrows represent heat transfer, blue arrows represent mass transfer of water (H_2O).

The most common way to classify thermal drying technologies is according to the heat transfer mode, which are convection (convective drying), conduction (contact drying), and radiation (radiative drying). Convective drying (or direct drying) works by passing hot air or gases directly through the sludge. Contact drying (or indirect drying) instead uses heat exchangers to heat a surface that the sludge is in contact with. Radiative drying provides the heat for moisture evaporation by solar, infrared, microwave, or dielectric radiation. Different types of drying modes can be combined for a given technology. Most drying systems include a ventilation or vacuum system to evacuate the evaporated moisture and avoid saturation of the air, which can inhibit the drying process. In passive drying systems such as drying beds, the sun and wind provide heat and air flow to promote evaporation. For more detailed information about drying mechanisms or types of industrial dryers used in other fields see Mujumdar (2014). Examples of convective, contact, and radiative drying technologies that have been used with faecal sludge at pilot- and laboratory-scales are presented in Figures 3.15 to 3.18.

Figure 3.15. Left: a rotary convection dryer operated by Pivot, in Kigali, Rwanda and right: waste cardboard is burned in a boiler and the hot gases produced are used as the heat source for drying. The sludge is pre-dried by a solar dryer before entering the convective dryer pictured here (photos: UKZN PRG).

Figure 3.16. Left: a contact dryer at the Omni-processor pilot plant at Niayes FSTP in Dakar, Senegal. In this plant, the sludge is incinerated leading to the generation of heat and electricity. Right: part of the heat from combustion is recirculated in the process for the drying of the sludge with this contact dryer unit (photos: UKZN PRG).

Figure 3.17. A pilot-scale solar dryer developed by Swansea University and tested in Durban, South Africa. Above: sludge is placed inside these sheds to dry. Below: the walls of the sheds absorb solar energy and transfer it to the sludge inside through a ventilation system (photos: UKZN PRG).

Figure 3.18. Left: the bench-scale Shit Killer microwave-based technology for sludge treatment developed by IHE Delft and Fricke und Mallah Microwave Technology GmbH, and right: the follow-up prototype – the pilot-scale Tehno Sanitizer developed by IHE Delft and Tehnobiro d.o.o. (photos: IHE Delft).

3.5.4 Key parameters when implementing thermal-drying technologies

When designing or implementing a drying technology, first the amount of time it takes the sludge to dry to the desired moisture content (i.e. the drying rate) needs to be determined, along with the amount of required energy. Optimal combinations of key process parameters will yield dry sludge with the desired moisture content at the lowest energy cost. Methods such as pre-treatment of the sludge with stirring, or techniques to enhance the heat and mass transfer such as mechanical vibrations or ultrasound can also be investigated to improve drying performance. The following factors will influence the drying rate and energy consumption of the faecal sludge drying process, and need to be taken into account during technology transfer and process optimization (Septien et al., 2018a):

- Intensity of the energy source used to heat the sludge influences the evaporation, heat and mass transfer rates, and energy consumption. Examples of how this is measured are air temperature for convective drying; temperature of the heated surface for contact drying; and irradiance of the radiation source for radiative drying.
- Residence time of sludge in the dryer influences the energy consumption and the final moisture content of the treated sludge. Optimal residence time is used to design for capacity of specific treatment technologies.
- Relative humidity and flow rate of the air stream influence the heat and mass transfer kinetics. Faster air flow and lower relative humidity promote faster and more complete evaporation, but also often require higher energy input.
- Physical and chemical characteristics of the faecal sludge influence how much moisture needs to be removed, for example different starting moisture contents and water-binding characteristics, which influence the required energy to remove moisture.
- Sludge volume and geometry influence the rates of heat and mass transfer during drying, for example pellets, bulk sludge, thin or thick layers. Configurations with a higher sludge surface area to volume ratio, such as pellets, promote faster drying, whereas thick layers of bulk sludge require more time.

3.5.5 Laboratory-scale and pilot-scale testing

Laboratory- and pilot-scale testing of drying needs to consider comparable drying temperatures, air-flow rates, energy sources, and humidity ranges to the pilot- and full-scale technologies. Pilot-scale testing should replicate full-scale conditions as closely as possible, using knowledge of scientific mechanisms to evaluate performance for scaling-up. For example, a pilot-scale drying technology should produce pellets of the same size and aspect ratio as the full-scale system.

As mentioned in the previous section, the performance of the drying process at any scale is measured through the evolution of the faecal sludge moisture content as a function of time. In an experimental setup, this can be done through different methods:

- Online or intermittent measurement of the mass of the sample over time, assuming that the mass loss is exclusively due to moisture removal;
- Measurement of the moisture content after sampling a fraction of the sludge at a given time interval;
- Online or intermittent measurement of the humidity at the air-flow outlet, after assuming that the gain of humidity in the air is due to moisture evaporation.

The determination of the drying rate under different conditions enables a better understanding of the process, and facilitates the development of kinetic models that can be used as tools for the design, operation and optimization of drying technologies.

Drying kinetics can be characterized at the laboratory-scale using commercially available instruments or custom-designed drying rigs. The commercially available thermogravimetric analyser (TGA) offers high-precision mass measurement during the thermal decomposition of materials, under controlled conditions. It can be coupled to a differential thermal analysis (DTA) or differential scanning calorimetry (DSC) unit, in order to determine the heat released or consumed during the transformation of the material. The main drawbacks of this method are the high cost of the TGA and DSC instruments, and the low sample weight that has to be used in experiments (i.e. milligrams), which can lead to reproducibility problems due to the heterogeneity of faecal sludge. The moisture analyser is a more affordable commercial instrument that can record the loss of mass of the sludge during drying. In this device, the sludge is heated by an infrared radiator and a ventilation system evacuates the evaporated moisture. A larger amount of sample can also be used (i.e. grams). However, the drying conditions cannot be controlled as well as in the TGA. Custom-designed drying rigs can be adjusted in size and complexity according to the needs and means of the experimenter, and can give a more tailored representation of the drying kinetic of a specific technology. Custom rigs can be as simple as a conventional oven where sludge is occasionally removed to track the mass loss, or a sophisticated experimental rig with high levels of instrumentation and an interface to log the data. Provided in Case study 3.3 is an account of the use of a custom experimental rig to measure faecal sludge drying kinetics under variable process settings.

The physical and chemical changes that the sludge undergoes during drying must be characterized in order to have a deeper understanding of the drying process. Periodic characterization of the sludge properties during drying also helps researchers to target drying processes to produce suitable end products. The properties of the dried sludge can be quantified with the methods described in the SOPs (Chapter 8 of Velkushanova et al. (2021b)):

- Total solids of dried sludge; measured gravimetrically by sludge weight before and after complete drying in a 105 °C oven.
- Calorific value is a measure of energy density, and is measured using a bomb calorimeter.
- Ash and volatile solids content of the sludge are measured gravimetrically with a 550 °C muffle furnace.
- Rheological properties, such as shear stress and viscosity under different shear rates, are measured with a rheometer or viscometer.
- *E. coli* or Helminth eggs can be monitored as indicator organisms for pathogen inactivation, if the end product is required to be pathogen-free.

3.5.6 Case studies – thermal drying for energy recovery

The following three case studies provide examples of (i) how to get useful kinetics data from laboratory-scale devices for the design and development of pilot-scale and full-scale dryers, and (ii) how to optimize the performance of a full-scale drying process using experiments conducted with laboratory-scale apparatus.

Case study 3.3: Determination of faecal sludge drying kinetics with a custom-designed experimental rig

This case study presents an example of how to determine faecal sludge drying kinetics in a laboratory-scale custom-designed experimental rig. This investigation was carried out by the Pollution Research Group (PRG) at the University of KwaZulu-Natal (UKZN) in Durban, South Africa. It was part of a MScEng project to learn about the rate at which pit latrine faecal sludge dries under different operating conditions (Makununika, 2017). A rig was custom-designed to study drying rates under different operational conditions in a convective dryer. In this rig, faecal sludge pellets were dried with hot air while their mass loss due to evaporation was measured in real time. The determination of the drying rates will aid in the development of drying technologies suitable for faecal sludge. Determination of kinetic data is an important step towards the design, development, optimization and scaling-up of drying technologies. It provides information that is used to size the dryer, to determine the optimum operating conditions, to fix the residence time (continuous mode) or holding time (batch mode), and to estimate the power consumption of the process.

> *Research question*
> What is the rate of faecal sludge drying with varying temperature, humidity, air velocity, and pellet diameter?
>
> *Response variable*
> Change in moisture content over time was characterized gravimetrically by a custom-designed convective drying rig. A photograph and schematic representation of the convective drying rig are displayed in Figure 3.19.

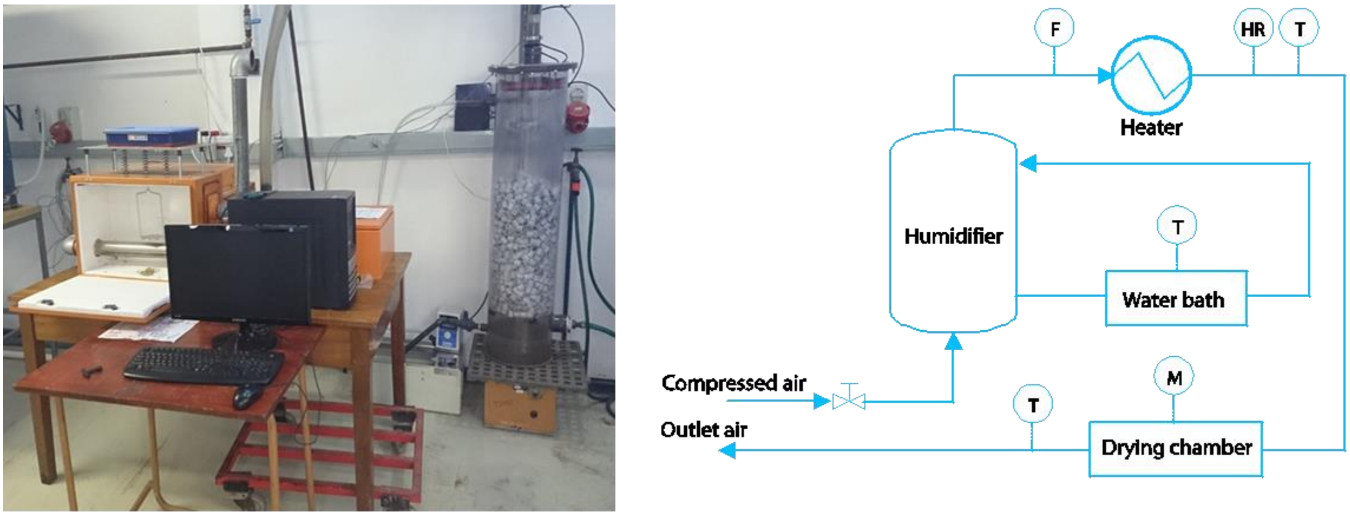

Figure 3.19. Left: the custom-designed convective drying rig (photo: UKZN PRG) and right: a schematic representation of the convective drying rig, F: air-flow measurement; T: temperature measurement; M: mass measurement; HR: relative humidity measurement.

During the experiments, dehumidified compressed air was fed into the drying rig. The air-flow rate was measured by a differential pressure measurement device and was controlled by a globe valve. The air stream was humidified in a packed column by counter-current contact with a water flow. The relative humidity of the air was adjusted by controlling the water temperature. The humidified air then passed through an electric heater to raise its temperature to the set value. The hot air stream was then introduced into the drying chamber where the faecal sludge sample was placed on a sample holder linked to a precision weighing strain gauge load cell with an accuracy of 0.01 g. The sample mass was measured online to track the change in mass with time. The air temperature and relative humidity were monitored at the inlet and outlet of the drying chamber. All the measurements were continually logged on a computer.

Factors, levels, and ranges

- Temperature: 40, 60, and 80 °C
- Relative humidity: 5, 15, and 25 %
- Air velocity: 0.1, 0.2, and 0.4 m/h
- Pellet diameter: 8, 10, 12, and 14 mm

Factors that might influence the response variable

Presence of rubbish: faecal sludge can contain considerable amounts of rubbish that can cause interferences and clogging during the drying experiments. In order to avoid this, the sludge samples were screened prior to the experiments, and large pieces of rubbish were removed.

Heterogeneity of faecal sludge: faecal sludge is highly heterogeneous, which can lead to inconsistent experimental results. In order to reduce heterogeneity and ensure repeatability, the sludge samples were thoroughly mixed prior to the experiments.

Experimental design details

Each run was performed in triplicate. Table 3.2 displays the runs performed in this study from all the possible runs. If all the possible combinations of the selected factors, levels, and ranges had been tested, 72 different runs would have been required. However, this was not feasible in terms of time and resources, so the most appropriate combination of runs was selected in order to study the influence of each variable. This was done by varying the value of a single variable while keeping the others constant at a reference value.

Table 3.2. Matrix with the different runs performed (marked with the symbol ■) out of all the possible combinations.

Temperature (°C)	Relative humidity (%)	Air velocity (m/h)	Pellet diameter (mm)			
			8	10	12	14
40	5	0.1	■	■		
		0.2		■		
		0.4		■		
	15	0.1		■		
		0.2				
		0.4				
	25	0.1		■		
		0.2				
		0.4				
60	5	0.1	■		■	■
		0.2				
		0.4				
	15	0.1				
		0.2				
		0.4				
	25	0.1				
		0.2				
		0.4				
80	5	0.1	■			
		0.2				
		0.4				
	15	0.1				
		0.2				
		0.4				
	25	0.1				
		0.2				
		0.4				

Interpreting the results

The results of the experiment are presented in Figure 4.20.

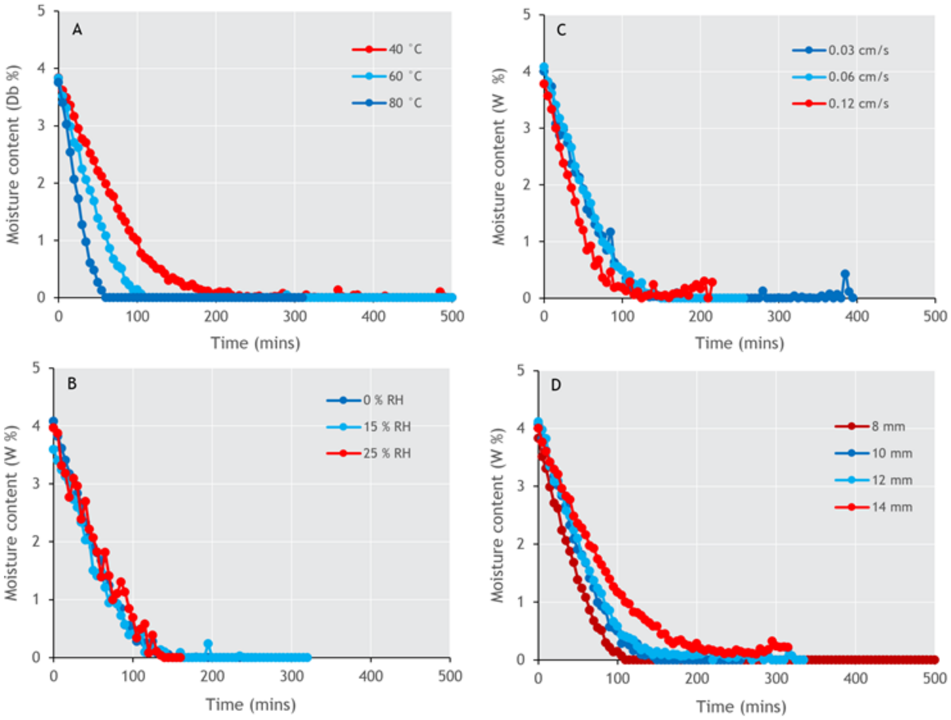

Figure 3.20. A) Drying rate as a function of air temperature, B) relative humidity, C) air velocity, and D) pellet diameter.

The main findings of this study were:

- Air temperature has a major influence on the drying rate. Increasing the temperature from 40 to 80 ºC decreased the drying time from 3 hours to 1 hour.
- The diameter of the sludge pellets also has an important influence on the drying rate. The 8 mm pellets were completely dried within 100 minutes, whereas the 14 mm pellets required drying times greater than 200 minutes.
- The relative humidity and air velocity had low or negligible influence on the drying kinetics under the explored conditions.

Scaling-up from laboratory to pilot-scale

According to the experimental results in this case study, the most critical parameters to optimize during drying are the air temperature and diameter of the sludge pellets.

The experimental data from this work was used to develop a mathematical model that could be inserted into reactor models as a tool for simulation to design new dryers, and can be used in process control for scaled-up systems (Makuninika, 2017).

Case study 3.4: Optimising the LaDePa process for infrared faecal sludge drying

This case study is based on a Master's thesis by Simon Mirara (Mirara, 2017). Further information can be found in Septien et al., 2018a, 2018b, and Septien et al., 2020. The motivation for this research project was to optimize the existing full-scale Latrine Dehydration Pasteurization (LaDePa) process. The LaDePa process was implemented in the eThewkini municipality in Durban, South Africa to treat the faecal sludge from ventilated improved pit (VIP) latrines through infrared drying, to produce dry, pathogen-free pellets for use as a soil amendment or solid fuel. The LaDePa process was developed by the eThewkini municipality and Particle Separation Systems as a transferring technology from the mining industry where it was used for drying minerals. Based on the treatment performance of the full-scale LaDePa, the municipality decided that it needed to be optimized to minimize energy consumption while maximizing the drying rate, pasteurization performance, and end-use potential in the treated sludge. In order to optimize drying in the LaDePa process and to develop a deeper understanding of the drying process, a 1:10 laboratory-scale replica of the full-scale LaDePa was constructed.

Research question

What process settings for faecal sludge drying with the LaDePa infrared dryer minimize energy consumption and maximize sludge drying rate?

Response variables

The laboratory-scale LaDePa was used to characterize the moisture content of the dried pellets, and energy consumption of the process, at different conditions (see factors, levels and ranges). The sludge was fed into the machine as pellets formed with a screw extruder, which were conveyed by a moving belt under two successive infrared emitters (providing heat for drying). An air stream was induced in the drying zone through an air suction box system installed below the belt to keep humidity low (Figure 3.21).

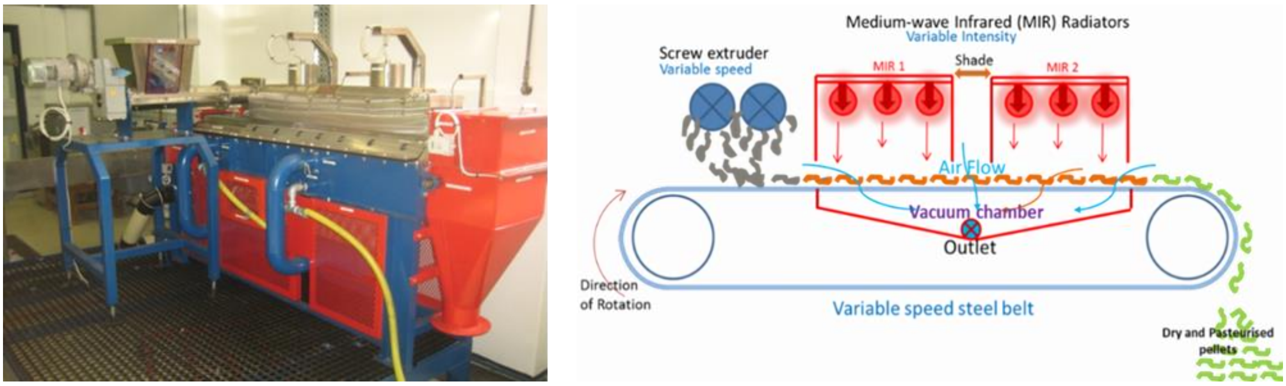

Figure 3.21. Left: the laboratory-scale LaDePa, and right: a corresponding schematic representation of the process (photo and schematic: PRG).

The dried pellets after processing were analysed to determine their moisture content, volatile solids content, pathogen content (Ascaris eggs) and physical and chemical properties (nutrient content, calorific value, thermal properties). The drying and pasteurization performance of the process were measured through the moisture content evolution and Ascaris egg viability. The end-use potential of the dried sludge was evaluated through the measurement of the physical and chemical properties.

Factors, levels, and ranges

- Emitter intensity (infrared irradiance): 6, 24, and 34 kW/m^2.
- Residence time: 4, 8, 12, 17, 26, and 39 minutes (varied by adjusting the speed of the belt).
- Distance between the belt and infrared emitters: 50, 80 and 115 mm (varied by adjusting the belt height).
- Suction air-flow rate: 11.1 and 18.3 m^3/s.
- Pellet diameter: 8, 10, 12 and 14 mm.

Factors that might influence the response variables

- Heterogeneity of sludge and presence of rubbish: as in Case Study 3.3, large pieces of rubbish were screened and removed from the sludge, and screened sludge samples were thoroughly homogenized prior to experimentation.
- Ambient temperature and humidity: ambient air is used for ventilation in the LaDePa, thus, the temperature and humidity of the suction air stream is dependent on ambient conditions. As the laboratory is climate-controlled, the ambient conditions are quite steady throughout the year and it was assumed that these parameters did not significantly change throughout the course of the study.
- Loading density of the pellets on the belt: this could have an influence on the performance of the process, as it could be expected that the drying of large sample loads would require a higher heat input. To address this, the loading density on the belt was kept consistently low in this investigation. Prior to scaling-up, higher loadings will be investigated.

Experimental design details

Due to available time and resources, the following runs, indicated with a ■ in Table 3.3, were determined to be the most relevant.

Table 3.3. Matrix with the different runs performed (the most relevant marked with ■) out of all the possible combinations.

Emitter irradiance (kW/m²)	Height emitter (mm)	Air-flow rate (m³/h)	Pellet diameter (mm)			
			8	10	12	14
6	50	11.1				
		18.1	■			
	80	11.1				
		18.1				
	115	11.1				
		18.1				
24	50	11.1				
		18.1	■	■	■	■
	80	11.1				
		18.1	■			
	115	11.1	■			
		18.1	■			
34	50	11.1				
		18.1	■			
	80	11.1				
		18.1				
	115	11.1				
		18.1				

Interpreting the results

Results of the experiment are presented in Figure 3.22.

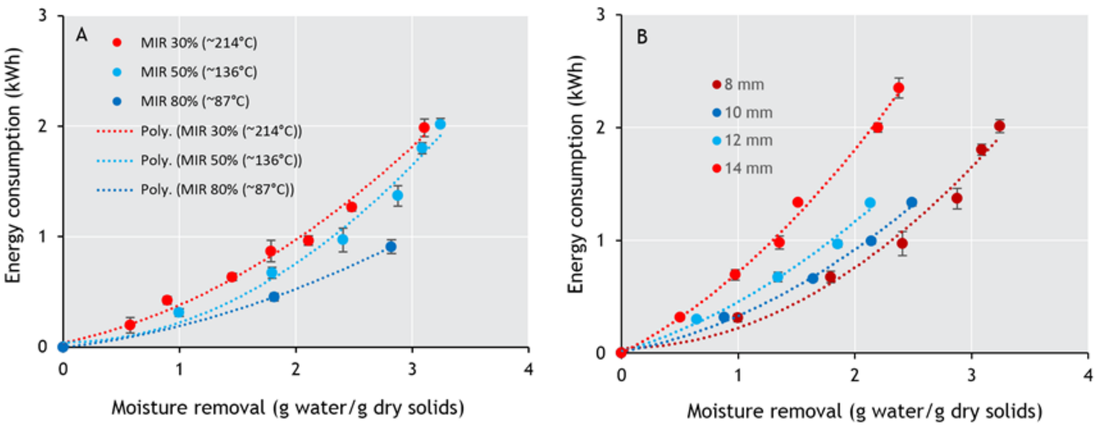

Figure 3.22. A) Plot of moisture removal vs energy consumption at varied medium-wave infrared intensities (MIR). MIR of 30, 50 and 80 % equals infrared irradiance of 6, 24, and 34 kW/m^2, respectively, and B) plot of moisture removal vs energy consumption at varied pellet diameters.

As expected, the rate of drying increased as the intensity of the infrared radiation increased and the distance between the pellets and the heating source decreased. Drying was faster for pellets with a smaller diameter. Increasing the suction air-flow rate caused a cooling effect on the sludge (negative for the process) but also enhanced the evacuation of moisture from the surface of the pellets (positive for the process). Under the explored conditions, these opposing effects counteracted each other and the overall drying rate was not affected by changing the air-flow rate. The pre-treatment of the sludge also did not affect the drying rate.

The energy consumption for moisture removal was determined from the kinetic data. Depicted in Figure 3.22, the drying process consumes less energy to remove a given amount of moisture when operating at higher infrared heating intensity and with smaller diameter pellets. However, it was observed that drying at too high a heating flux could induce thermal degradation of the sludge, which could lead to charring or burning. During the trials, drying at the highest infrared intensity (34 kW/m^2) resulted in the pellets starting to smoke.

Process optimization from laboratory- to full-scale

Based on the results of the laboratory-scale experiments, it is recommended to operate the LaDePa at the highest possible infrared radiation intensity that does not cause thermal degradation. During laboratory tests, in addition to monitoring energy consumption and drying time, Ascaris egg viability (Helminth) and net calorific value were measured. During tests, full deactivation of Ascaris eggs was achieved. It is not recommended to operate at the highest intensity setting, as the resulting thermal degradation could reduce the suitability of the dried sludge for reuse as a solid fuel. The distance between the infrared emitters and the belts should be minimized, in order to maximize the amount of radiation received by the pellets without the need of an increased power supply. Implementing these results will result in lower energy use and operating costs.

The faecal sludge should also be pelletized at the lowest diameter possible for a more efficient drying process. This will require experimentation with the full-scale extruder to determine the smallest diameter achievable at scale. After process changes are made, pellets produced at full-scale will need to be further evaluated for pathogens to ensure protection of public health during the end use.

3.6 Transferring technology: Microwave drying for resource recovery of dried sludge for energy

Microwave drying is a type of radiative drying where microwave radiation is used to heat the sludge. In the microwave drying process, microwave radiation heats the core of the sludge particles promoting the transport of water molecules from the inside to the surface; this results in a large amount of water molecules at the surface of the sludge that can be more easily evaporated compared to the water bound deeper within sludge particles. Due in part to this mechanism, microwave drying can offer energy savings compared to other thermal drying technologies.

Case study 3.5 Optimising the Tehno Sanitizer technology for microwave faecal sludge sanitization and drying

This case study is based on two PhD and several MSc studies carried out at IHE Delft Institute for Water Education in The Netherlands. It concerns the development of a novel microwave-based technology for sludge sanitization and drying. The new technology is an example of a development that has passed through all the Technology Readiness Levels (TRLs), (Héder M. 2017), starting from a small lab-scale setup using an adapted kitchen microwave (Mawioo et al., 2016a; Mawioo et al., 2016b), to a bench-scale unit (Mawioo et al., 2017) and finally, to a full-scale prototype (Kocbek et al., 2020, 2021). This technology, called the Shit Killer, was initially developed for decentralized faecal sludge treatment in emergency sanitation (Brdjanovic et al., 2015) and has evolved into a robust and efficient technology known nowadays as the Tehno Sanitizer (Figure 3.23). The Tehno Sanitizer prototype, recently tested in Jordan, is equipped with four technologically-independent but inter-connected functional components, namely: (i) microwave-based sludge treatment, (ii) liquid stream treatment, (iii) air treatment, and (iv) an energy-recovery system.

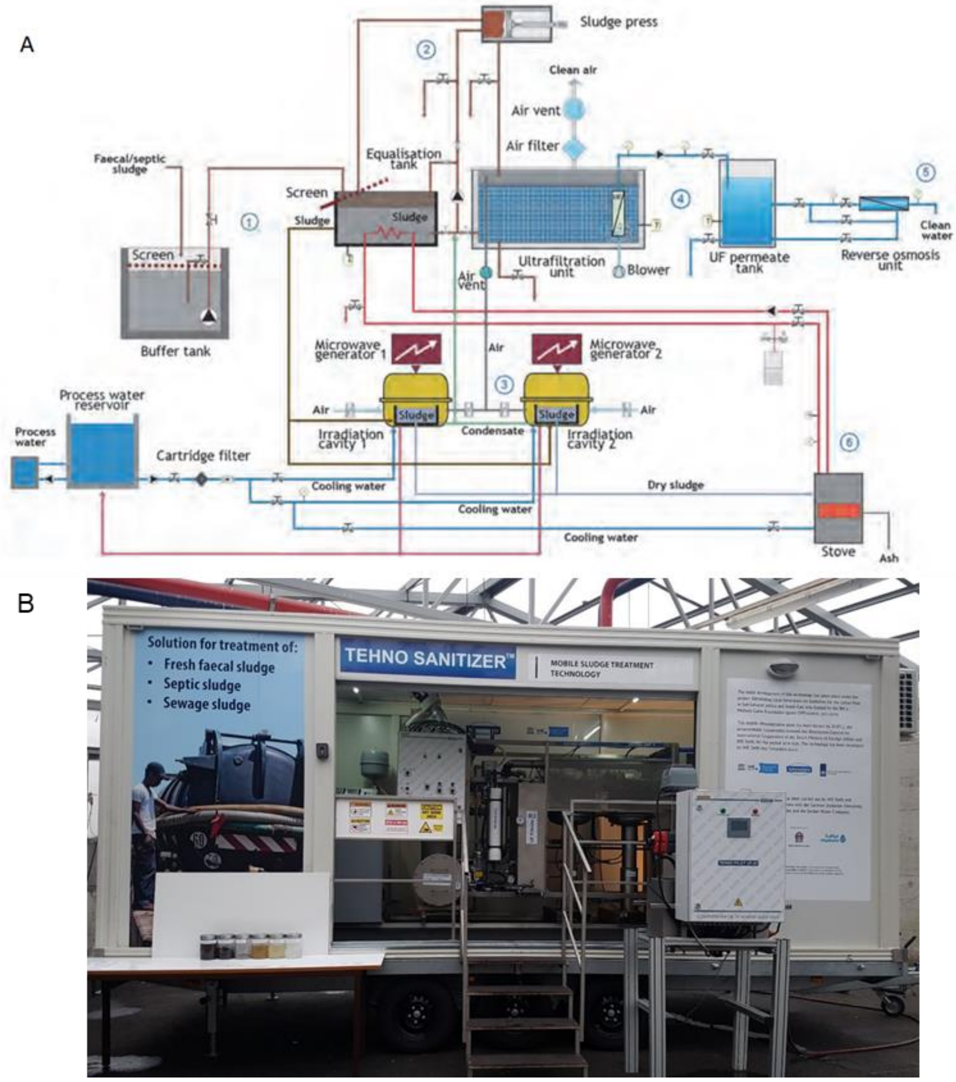

Figure 3.23. A): simplified process flow diagram of the Tehno Sanitizer: (1) sludge intake, (2) sludge pre-treatment, (3) the sludge sterilization and drying unit, (4) microfiltration, (5) reverse osmosis, and (6) the sludge energy recovery unit, and B): a full-scale Tehno Sanitizer prototype (source: Tehnobiro d.o.o.).

The bench-scale Shit Killer unit was successfully tested for pathogen removal and sludge drying in Slovenia. At that time, the specific energy consumption (SEC) (energy consumed per liter of evaporated water) was not the primary objective and thus was, as expected, sub-optimal. The main reasons for this were: (*i*) lack of thermal insulation, (*ii*) inefficient use of microwave energy, (*iii*) less efficient mixing at higher sludge densities, (*iv*) cold ambient temperature (5 °C), (*v*) poor extraction of the condensate from the cavity, (*vi*) unnecessary heating of the cavity, and (*vii*) absence of energy recovery features.

All of these shortcomings have been addressed and mitigated in the next generation full-scale prototype: the Tehno Sanitizer. This system (Figure 3.23) is a semi-decentralised and containerized mobile full-scale prototype designed for the treatment (drying, pathogen inactivation, and resource recovery) of diverse types of sludges such as fresh faecal sludge and waste activated sludge, with different water and dry solids contents. This mobile unit has the capacity to process 300 kg of wet sludge per day. The integration of the different technologies provides an attractive approach for treating sludge and wastewater streams generated while producing valuable resources that can be utilized in agricultural and domestic applications, with up to 95 % DS. The initial results obtained from studies focusing on pathogen indicator organisms (Mawioo et al., 2016a and 2016b), carried out at laboratory and bench scale set-ups, suggest that the Tehno Sanitizer could be an effective technology for sanitization of sludge.

The main challenge addressed in the development of the full-scale prototype was how to minimize the specific energy consumption (SEC) of the system from the value initially observed in the bench-scale unit of 4.0 kWh/L of evaporated water, to the target level of below 1.0 kWh/L of evaporated water.

Research question
Which microwave power output settings on the full-scale prototype achieve the target dryness (85% DS) while minimizing the SEC to below 1.0 kWh/L of evaporated water?

Response variables
The experimental setup was designed to measure the SEC (kWh/L) of the system. The SEC was calculated using the power output setting of the microwave generator, set at the desired value (kW). This value was multiplied by the time of the exposure and divided by the mass of water that had evaporated at that exposure time.

The mass of the sludge in the microwave cavity was continuously measured and the moisture content and the DS were calculated from the TS measurement of the sludge sample taken just before the start of the test. Also the sludge temperature was continuously measured by a sensor installed inside the cavity.

Factors, levels, and ranges
Microwave power output: 1.0, 1.5, 3.0, 3.25, 4.5 and 6.0 kW (adjusted manually)

Factors that influence the response variable
The factors that influence the SEC include:

- energy losses due to the lack of thermal insulation,
- frequencies at which the microwave energy is delivered,
- mixing conditions at the irradiation cavity,
- condensation of the evaporated water in the microwave cavity, and
- the microwave energy absorption capacity of the sludge (power density) at the evaluated microwave power outputs.

Experimental design details

Experiments were conducted using the full-scale prototype. The experimental setup (Figure 3.24) consisted of two stainless steel microwave cavities equipped with a rotating polypropylene turntable and an oval sludge-holding vessel, two microwave power supply units, and two microwave generators with a combined power output of 12.0 kW operated at a frequency of 2,450 GHz. An electromotor was used to rotate the sludge samples at a speed of 1 rpm to alleviate the effect of non-uniform sludge heating. Ancillary equipment included an air extraction and treatment unit and a microwave generator-cooling water-based system. In total six identical tests were executed (each at different power level) because only one cavity was equipped with a load cell to continuously measure the mass of the sludge. Each test had a different duration (the shortest was 21 minutes at power output of 6 kW) and lasted until the target DS of 85 % was achieved.

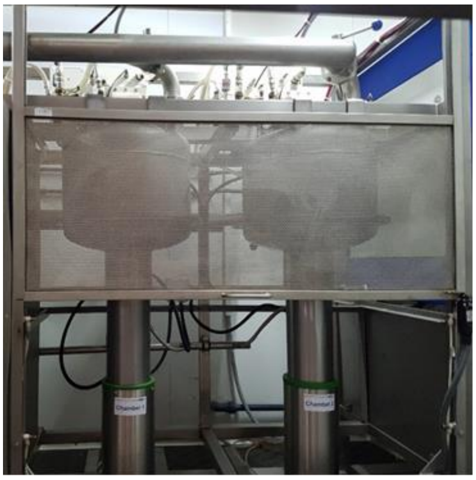

Figure 3.24. Left: an experimental microwave-based faecal sludge drying unit, and right: samples taken at different points in the process: a) filtrate from the sludge press, b) concentrated sludge from the sludge press, c) ultrafiltration concentrate, d) ultrafiltration permeate, e) reverse osmosis concentrate, f) reverse osmosis permeate, g) dry sludge, and h) condensate (photo: IHE Delft).

Interpreting the results

Figure 3.25 depicts the drying rate as a function of dry solids content at different power outputs of the microwave generators. As expected, the higher the power output, the higher the drying rate. At the start of the drying process the drying rate increased at all the evaluated power outputs until it reached a maximum and constant drying rate value. This constant drying phase was dominant and extended through almost the entire drying process; this is a positive characteristic of the microwave drying process and introduces a competitive advantage compared to thermal drying technologies where such constant drying phases are not commonly observed. Such a constant drying phase is associated with the removal of unbound (free) water from the surface of the sludge which demands much less energy to be evaporated than other types of water contained in the sludge (Figure 3.25).

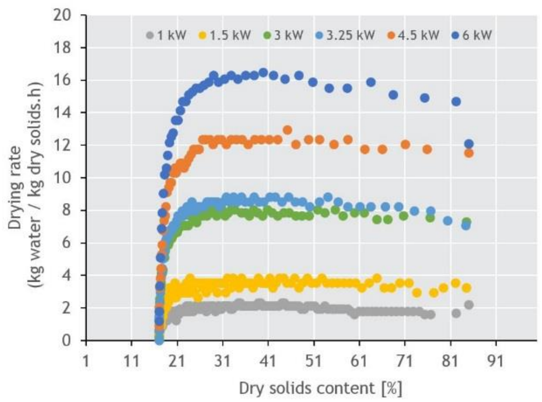

Figure 3.25. The sludge drying rate as a function of sludge dry-solids content at different power outputs of the microwave generators (Kocbek et al., 2020).

Figure 3.26 shows the SEC of the system during the period of drying sludge from 17 % to 85 % DS at the evaluated microwave generator power output range. It has been observed that increase in power output lowers the SEC. The lowest SEC of approximately 1 kWh/L of evaporated water was reported at power outputs higher than 3 kW. The observed changes in the SEC were due to the microwave radiation generation efficiency which was between 50 % (at power below 3 kW) and 70 % (at the highest power outputs).

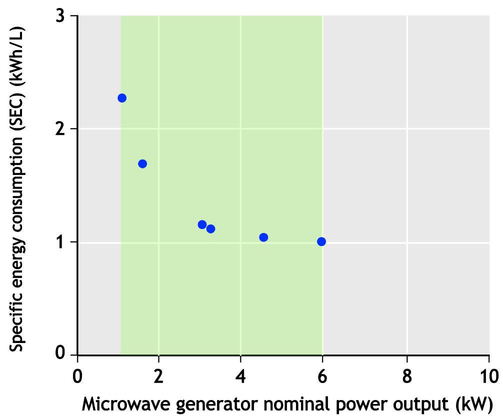

Figure 3.26. Effect of microwave generator power output on the specific energy consumption (SEC) (Kocbek et al., 2020).

Implications of scaling up

The SEC results obtained in this research provided the evidence that the modifications and innovations built in the Tehno Sanitizer mitigated the early development issues experienced with the Shit Killer, largely reducing the energy requirement resulting in achieving the target SEC of 1 kWh/L. Such results bring Tehno Sanitizer into the mix with conventional thermal drying (convective and conductive) technologies

(Bennamoun et al., 2013). Given the fact that in commercial-scale applications a more efficient microwave generator will be used (with an efficiency rate of up to 90 %), the SEC is expected to decrease by an additional 10 to 20 %. Furthermore, the energy recovery unit in the Tehno Sanitizer in this study was not turned on. With the additional heat becoming available from co-incineration of dry sludge (energetic value of obtained dry sludge was 20 MJ/kg or 5.6 kWh/kg) for pre-heating of the incoming sludge, and when the system starts to be continuously used, the calculated SEC will further decrease. If less stringent requirements for water treatment are applicable, an SEC of below 0.8 kWh/L can be achieved. Such results are promising and make this new technology a viable alternative for faecal sludge management.

3.7 Outlook

Faecal sludge management is a rapidly evolving sector. The information described in this chapter is important for developing new technologies, scaling up and transferring technologies, and optimizing established technologies. Experimentation is an iterative process, and research will need to be conducted back and forth between laboratory- and pilot-scale before technologies are ready for full-scale implementation. Projects that incorporate well thought-out experimentation ensure that an appropriate, context-specific treatment solution is selected, instead of assuming that a standard solution will fit. The inherent uncertainties in working with faecal sludge, and with innovative and transferring technologies, make risk management an essential focus in the development and scaling-up of any treatment technology. Risks can be mitigated through dedication to quality experimental design and execution, and through partnerships between municipalities and research institutions, which can help guide experimentation from the start of a project to the optimization and monitoring of a full-scale FSTP.

Future research needs for scaling-up dewatering and drying technologies will be driven by requirements to optimize treatment technologies that work for faecal sludge. The next advances in dewatering research will include establishing how to more rapidly and cost effectively monitor faecal sludge such that optimal conditioner dosing can be achieved. Another step will be acquiring a fundamental understanding of the processes occurring during stabilization that affect dewaterability. Future focuses in thermal drying research will address the need for a more holistic understanding of the drying process of faecal sludge, for example morphological changes that occur such as stickiness, and a better understanding of how moisture is bound to faecal sludge.

Chapter 4

Predictive models using “cheap and easy” field measurements: Can they fill a gap in planning, monitoring, and implementing fecal sludge management solutions?

This chapter was published in Water Research as:

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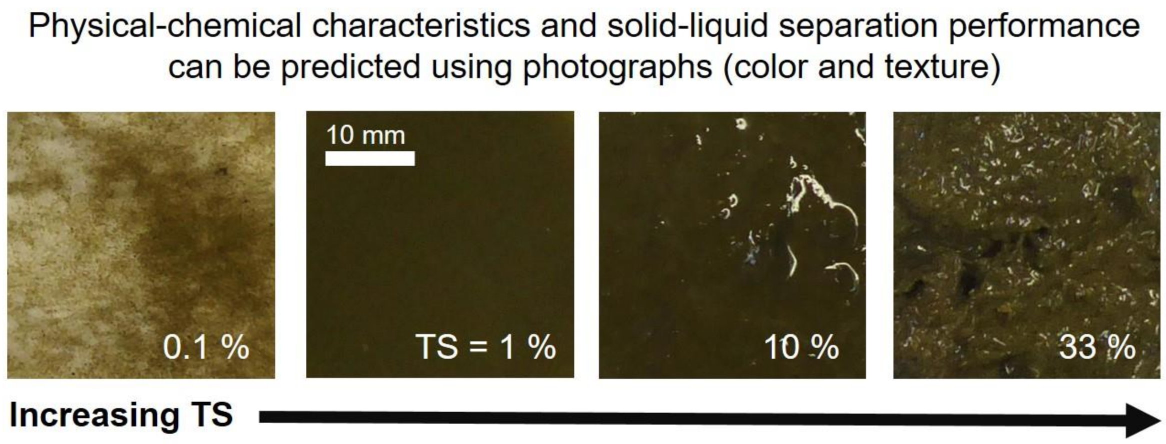
Graphical abstract



Highlights

- New method for image analysis of fecal sludge photos including color and texture
- Solid-liquid separation performance was predicted using image analysis of photos
- Simple decision tree models appear promising for citywide planning
- Machine learning predictions may be sufficient for real-time process control

Abstract

The characteristics of fecal sludge delivered to treatment plants are highly variable. Adapting treatment process operations accordingly is challenging due to a lack of analytical capacity for characterization and monitoring at many treatment plants. Cost-efficient and simple field measurements such as photographs and probe readings could be proxies for process control parameters that normally require laboratory analysis. To investigate this, we evaluated questionnaire data, expert assessments, and simple analytical measurements for fecal sludge collected from 421 onsite containments. This data served as inputs to models of varying complexity. Random forest and linear regression models were able to predict physical-chemical characteristics including total solids (TS) and ammonium (NH_4^+-N) concentrations, and solid-liquid separation performance including settling efficiency and filtration time (R^2 from 0.51-0.66) based on image analysis of photographs (sludge color, supernatant color, and texture) and probe readings (conductivity (EC) and pH). Supernatant color was the best predictor of settling efficiency and filtration time, EC was the best predictor of NH_4^+-N, and texture was the best predictor of TS. Predictive models have the potential to be applied for real-time monitoring and process control if a database of measurements is developed and models are validated in other cities. Simple decision tree models based on the single classifier of containment type can also be used to make predictions about citywide planning, where a lower degree of accuracy is required.

4.1 Introduction

The sanitation needs of 1/3 of the world’s population are met by non-sewered sanitation technologies, which can only provide protection of human and environmental health if the accumulated fecal sludge is adequately managed (WHO and UNICEF 2017, WHO 2018). To achieve this, characterization of process control parameters at treatment plants is required to ensure safe and efficient treatment (Bassan and Robbins 2014, von Sperling et al. 2020a). This includes projections for quantities and qualities of influent when designing a treatment plant, and routine monitoring for process control and compliance of effluent standards during operation. Although there has been considerable research focused on improving characterization and monitoring for centralized, sewer-based wastewater treatment in high-income countries, this is still lacking for fecal sludge treatment (Corominas et al. 2018, Klinger et al. 2019, WHO and UNICEF 2017, Yoo et al. 2008). A key obstacle is the lack of accessible analytical capacity, as there are few fecal sludge treatment plants (FSTPs) with onsite laboratories, and supply chains for procurement of chemicals are often complex and unreliable (Bassan et al. 2015, Bousek et al. 2018, Klinger et al. 2019).

Solid-liquid separation is a key step in fecal sludge treatment, and FSTPs are designed for settling and dewatering which typically precede subsequent treatment of liquid and solid fractions, and is most commonly achieved by settling-thickening tanks and/or drying beds, but could also entail mechanical dewatering, geo textiles, or new innovative technologies. The characteristics of fecal sludge are highly variable, resulting in correspondingly inconsistent settling and dewatering performance (Cofie et al. 2006, Gold et al. 2016). Without methods to predict the variable characteristics of sludge arriving for treatment, adjustments cannot be made to plant operations. The results are clogged drying beds or filter membranes, wasted space and decreased treatment capacity (Klinger et al. 2019). To address these problems, real-time monitoring for adaptive process control is required (Ward et al. 2021b). Parameters relevant for citywide planning and optimized process control at FSTPs include physical-chemical characteristics of influent (e.g., total solids (TS), chemical oxygen demand (COD), and ammonium nitrogen (NH_4^+-N) (Bassan and Robbins 2014).

Research is being conducted into alternatives to laboratorybased analysis for fecal sludge, but these alternatives are not yet well established and little quantitative research has been done (Bousek et al. 2018). However, several interesting correlations between field measurements and laboratory-based measurements have been reported. For example, electrical conductivity (EC) and pH have been correlated with settling performance and dewatering time of fecal sludge, suggesting that these might be possible field measurements to use as indicators of expected solid-liquid separation performance (Gold et al. 2018a, Junglen et al. 2020, Ward et al. 2019). Data gathered by questionnaire, including containment type (i.e., pit latrine or septic tank), presence of household water connection, source (i.e., household or commercial), and household income have been linked to physical-chemical characteristics such as TS and COD in several cities. However, these correlations are empirical and may differ between cities (Englund et al. 2020, Strande et al. 2021b, Strande et al. 2018, Tembo 2019). Expert assessments of color and odor are informally used as indicators of level of sludge stabilization, and it is generally believed that perceptible differences in color and odor are linked to different physical-chemical properties and dewatering performance (Hartenstein 1981, Schoebitz et al. 2016b). However, the capacity of color and odor to monitor fecal sludge characteristics has, to our knowledge, never before been quantified and is not well documented in the literature. Quantifying relationships between possible field measurements and laboratory-based measurements is the first step in establishing alternative field-based methods, but in order get the most utility out of field-based measurements, predictive models can be employed.

Based on experience using software sensors in the wastewater treatment sector, it should be possible to develop predictive models for fecal sludge using the types of field measurements previously discussed (Dürrenmatt and Gujer 2012, Tyralis et al. 2019). Currently, operators at FSTPs do not use predictive models, but may use expert knowledge or data collected from emptiers to make decisions about operation, process control, and maintenance. For example, mixing sludge from households and public toilets in a pre-determined ratio to achieve more consistent settling behavior (Cofie et al. 2006) or varying the dose of polymer flocculant for pit latrine sludge and septic tank sludge, based on observations of the differences in their solids contents (Ward et al. 2021b). Very little research has been published on predicting fecal sludge characteristics, however it has been proposed that questionnaire data can be used to model estimated loadings for planning new FSTPs (Strande et al. 2021b). A combination of data-driven and mechanistic models based on questionnaire data have demonstrated the ability to predict TS in fecal sludge with the goal of improved citywide planning (Englund et al. 2020). In order for predictive models to reduce dependence on laboratory-based characterization of fecal sludge, they must be accurate enough for routine monitoring and/or process control at FSTPs. However, so far, no such models have been reported in the literature. In this study, we use a large dataset that incorporates a combination of analytical field measurements, questionnaire data, and expert assessments to assess the suitability of predictive models for optimizing fecal sludge treatment.

The objective of this study was to investigate to what extent predictive models using field measurements can be used as a proxy for laboratory-based methods, based on a dataset of samples taken from 421 onsite containments in Lusaka, Zambia. We present results of field and laboratory data collection, and the predictive performance of the models, and discuss which models and field measurements would be appropriate for characterizing and monitoring fecal sludge in different scenarios including citywide planning, routine FSTP operations, and real-time process control of treatment technologies.

4.2 Materials and Methods

4.2.1 Sample collection

465 fecal sludge samples were collected in situ from 421 onsite containments (septic tanks and pit latrines) throughout the city of Lusaka, Zambia from September to December of 2019. Sampling replicates were taken at 8% of containments. A sampling plan was developed according to the method described in (Strande et al., 2021b). Household sampling sites were selected based on a modified version of stratified random sampling (including equal proportions of strata based on geological and demographic characteristics) in the non-sewered part of the city using ArcMap software (version 10.6). Non-household sampling sites (public toilets, offices, schools, and malls) were selected throughout the city based on local expert knowledge. Questionnaires were administered to the owner, tenant, or person in charge of operating and maintaining the system at each sampling site. Questionnaires included questions about designation of containment type, toilet type, water connection, and a number of other factors (questions available in SI). Answers were recorded with the KoBo Toolbox mobile phone app.

Because of the range of sludge consistencies present in Lusaka, different sampling devices were used to sample from pit latrines and septic tanks. A conical metal pit sampler, adapted from the design developed by James Tembo, presented in (Koottatep et al. 2021), was used to sample pit latrines. A composite sample was produced using three 1 L samples collected from the bottom (or the maximum reach of the sampler, 3 m), middle, and top of the pit. After homogenizing the composite sample in a bucket, a 0.9 L sample was taken for analysis. For septic tanks, a 3 m long core sampler adapted from the design from CDD Society, presented in Kootatep et al. (2021), was used. A composite sample was produced by emptying the contents of the core sampler into a bucket, homogenizing the contents, and taking a 0.9 L sample for analysis. Samples were transferred to a cooler for transport to the laboratory, where they were stored at 4°C until analysis. Detailed information about the sampling process, along with photographs of the sampling devices is provided in Kootatep et al. (2021).

4.2.2 Sample analysis

Sample processing

Incoming samples were homogenized thoroughly by shaking/stirring, and were divided into two portions – one to be blended before physical-chemical characterization, and the other to be analyzed for solid-liquid separation performance (avoiding blending in this case, so as to not disrupt any flocs or structure of the sludge which would influence settling and dewatering behavior).

Physical-chemical characterization

Samples were homogenized in a blender (3 minutes, medium setting). Foam height was measured immediately after blending using a ruler held to the wall of the blender. pH, EC, TS and volatile solids (VS) were analyzed according to the standard methods (APHA 2017). Density was measured by determining the mass of 25 mL of sample. COD was measured using the closed reflux titrimetric method (APHA 2017). Samples that could not be analyzed immediately were preserved by acidifying to pH ≤ 2 using H_2SO_4. NH_4^+-N was measured using the phenate method, following swirling with activated charcoal and filtration to remove filtrate color and residual turbidity (APHA 2017). Total organic carbon (TOC_{solids}) and total Kjeldahl nitrogen (TKN_{solids}) were measured on the dried solids as indicators of the potential for resource recovery of the solids, e.g., as a feedstock for composting (Al-Muyeed et al. 2017) or larvae production (Gold et al. 2020). Blended sludge was dried in a 105°C oven for 48 hours and dried solids were analyzed by an external laboratory (UNZA Department of Soil Science). TOCsolids was measured using the Walkely-Black procedure, with the endpoint determined by titrimetric method (Sparks et al. 2020). TKN_{solids} was measured using standard methods (Cottenie et al. 1982). A certified reference material, ISE 952 clay was used as a quality control standard.

Qualitative assessment of odor and color

Odor was assessed during sample processing after stirring, before blending. Samples were allowed to reach room temperature, at which point the lid of the sample container was removed, and the laboratory technician used standard chemical wafting technique (NRC 2010). Odors were classified into three categories: “fresh” (smells like fresh excreta, urine, or feces), “stabilized” (smells like compost, soil, or well biodegraded anaerobic digester sludge), or “middle” (sample odor falls somewhere between these categories). To keep the expert classifications as consistent as possible, the same laboratory technician was responsible for all odor classifications. Color was grouped into three qualitative categories: “light brown”, “black”, and “medium brown” (when sample falls somewhere between these categories). One laboratory assistant was responsible for all color classifications for consistency.

Solid-liquid separation performance

Supernatant turbidity was quantified as a metric of settling efficiency. Following centrifugation at 3,300 × g for 20 minutes, supernatant was decanted and turbidity was measured using a Hach 2100N turbidity meter and reported in NTU, adapted from the method described in (Mikkelsen and Keiding 2002). This was intended to represent the maximum possible reduction in suspended solids due to prolonged settling or mechanical solid-liquid separation. Capillary suction time (CST) was quantified as a metric of filtration time. CST was measured using a Triton 319 MultiCST apparatus with 18 mm funnel, according to Method 2710 G (APHA 2017), as adapted in (Velkushanova et al. 2021a). CST values are reported in seconds, and have been standardized by subtracting the CST of deionized water. TS in the dewatered sludge cake was quantified as a metric of maximum solids content achievable by dewatering, and was defined as the dry solids content in the dewatered sludge cake after dewatering via centrifugation at 3,300 × g for 20 minutes (Gold et al. 2018a). Supernatant was decanted for turbidity measurement and centrifuge tubes were left standing upside down on an absorbent cloth for 5 minutes to completely remove any free water before transferring solids out of the centrifuge tube for TS characterization.

Samples were screened for consistency, and only liquid and slurry samples (225 samples) were analyzed for solid-liquid separation performance, according to designations of sludge consistency presented in (Velkushanova and Strande 2021).

Color and texture quantification

Color and texture measurements were taken by analyzing photographs of 10 mL aliquots of sludge. To our knowledge, this is a novel method, developed as an alternative to the standard colorimetric/photometric method of color detection. It is intended to be used as a field measurement. Shown in Figure 4.1, 10 mL samples of unblended sludge and supernatant after centrifugation (when available) were poured into 10 mL petri dishes and placed on a white background next to a color checker chart (Datacolor Spydercheckr 24) (see Figures S4.1 and S4.2 in Appendix B for additional photographs of the setup). Samples were photographed in a designated location with consistent LED lighting (3 × 6W LED lamps, 3000 K (warm white color)) using a Panasonic DMC-TZ70 camera in JPEG format. JPEG format was selected over RAW in order to enable the broader application of color and texture assessment to photographs collected with smartphone cameras in the future. Photographs were post-processed for color correction using the color checker chart's standard RGB values and assuming CIE Standard Illuminant D65 (Schanda 2007). The color correction was performed using the Python Colour library (Version 0.1.5) using the transformation presented in Cheung et al. (2004). Sludge and supernatant color swatches were manually isolated from the color corrected photographs, and the average RGB color was calculated by taking the independent averages of the R, G, and B values in the swatch. The average sludge and supernatant color values were then converted to hue, saturation, and value (HSV).

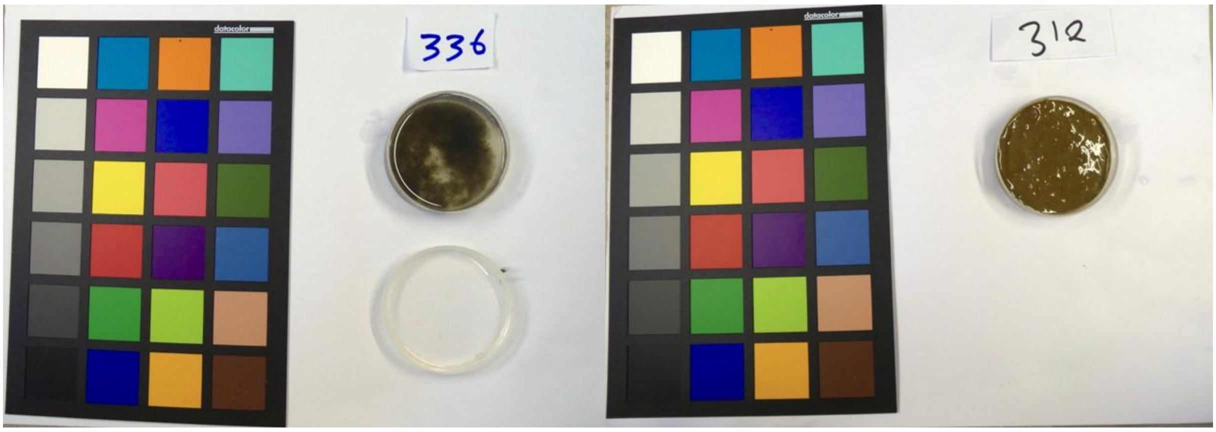


Figure 4.1. Examples of setup for sludge color and texture quantification. The color checker chart shown on the left side of the photographs was used to perform standardized color correction. Color and texture information were extracted from thumbnails of the samples isolated from the color corrected images. Left: Top petri dish contains bulk sludge, bottom petri dish contains supernatant after centrifugation. Right: Petri dish contains bulk sludge.

Texture analysis was performed using texture measures derived from a Grey Level Co-occurrence Matrix (GLCM) with the Python scikit-image library (version 0.17.2) (Van der Walt et al. 2014). Six common texture measures were calculated for each photograph: contrast, dissimilarity, homogeneity, correlation, angular second momentum (ASM), and energy. The mathematical descriptions of the texture measures are described in Haralick (1979) and HallBeyer (2017). All code related to color correction and texture analysis is provided along with the open dataset for this paper (link in SI).

QA/QC

According to QA/QC procedures, triplicate laboratory analyses were made for every 10th measurement for COD, NH_4^+-N, and supernatant turbidity, and every 5th measurement for TS, VS, and TS in the dewatered cake. For TOC_{solids} and TKN_{solids}, duplicate measurements were made every 7th measurement. Every CST measurement was replicated four times. Relative standard error on the replicates was (mean, 90th percentile): COD (8%, 16%), NH_4^+-N (4%, 9%), TS (5%, 15%), VS (4%, 8%), TOC_{solids} (8%, 13%), TKN_{solids} (18%, 34%), CST (5%, 11%), turbidity (4%, 9%), and TS in the dewatered cake (10%, 24%).

4.2.3 Data interpretation

Median values within a category were considered to be significantly different from one another if the 95% confidence intervals around the medians did not overlap (Chambers et al. 1983). Confidence intervals (CI) were calculated using the formula

$$CI = \text{median} \pm 1.57 \times IQR \sqrt{n}$$

Where IQR is the interquartile range and n is the number of data points in the category (Chambers et al. 1983). Confidence intervals around the median are represented as notches on boxplots (Figures S4.3 and S4.4 in SI).

4.2.4 Predictive models

Cost-efficient and simple to execute field measurements were evaluated in this study to determine whether they could be used to predict more costly laboratory-based analytical measurements. Field measurements evaluated as inputs to predictive models are detailed in Table 4.1. Questionnaire data is categorical and quantitative. Expert assessments are categorical, qualitative, and were assigned by trained laboratory technicians. Simple analytical measurements are quantitative, continuous-valued measurements collected using a probe, a ruler, or a camera. Although simple analytical measurements were performed in a laboratory as part of this study, they are all able to be measured in the field without laboratory equipment, and thus are labeled field measurements. The laboratory-based measurements that are desired outputs of the predictive models (target parameters) are outlined in Table 4.2.

Table 4.1. Cost-efficient and simple field measurements that were evaluated as predictors for the more costly laboratory-based analytical methods presented in Table 4.2.

Field measurements (model inputs)	Details
Questionnaire	Categorical, collected via questionnaire
containment type	septic tank, pit latrine
toilet type	wet flush, dry toilet
water connection on premises	yes, no
source	household, non-household
Expert assessments	Categorical, collected by laboratory assistant
odor	stabilized, middle, fresh
color (qualitative)	black, medium brown, light brown
Simple analytical	Continuous, collected by the following equipment (units):
EC	probe (mS/cm)
pH	probe
foam height	ruler (mm)
color (quantitative)	camera (hue, saturation, value)
texture	camera (contrast, dissimilarity, homogeneity, correlation, angular second moment (ASM), energy)
supernatant color	camera (hue, saturation, value)

Table 4.2. Laboratory-based measurements.

Laboratory-based measurements (target parameters)	Details
Solid-liquid separation performance:	
supernatant turbidity following centrifugation	Measure of settling efficiency (NTU)
CST	Measure of filtration time (seconds)
TS in dewatered cake following centrifugation	Measure of maximum water removal (% dry solids)
Physical-chemical characteristics:	
Complete sample	
COD	(g/L)
NH_4^+-N	(g/L)
TS	(% dry solids)
VS	(% of total solids)
Dried solids	
TOC_{solids}	(% of total solids)
TKN_{solids}	(% of total solids)

Data analysis, modelling, and visualization was performed using Python 3 (Van Rossum and Drake 2009). Models were implemented in scikit-learn (version 0.23.2) and statsmodels (version 0.12.1) packages in Python 3. Models, along with the complete dataset are available open source with this publication: https://doi.org/10.25678/00037X.

Predictive models using “cheap and easy” field measurements

Three types of models were evaluated in this study: i) a simple decision tree model, ii) a linear regression model, and iii) a random forest model. These models were chosen as they represent a range of complexities and unique advantages and disadvantages. Simple decision tree models are based on the decision tree presented in Strande et al. (2021b), which uses the median value of the target parameter in a category (e.g. median TS in septic tanks) to predict future values in that category. Simple decision tree models are reflective of the way that operational decisions may currently be made at treatment plants (Cofie et al. 2006, Ward et al. 2021b). They have the advantage of easy interpretation and visualization, but predictive capabilities are often poor. Linear regression models can also be easily interpreted, and are good descriptions of systems with linear behavior. In contrast, random forest models are a widely applied black-box machine learning algorithm that can deal with non-linearities and interactions, but cannot be interpreted directly (Hastie et al. 2009). A thorough description of the specifics for each model type is provided in the SI.

Model evaluation took part in two steps: a) the identification of relevant field measurement inputs for predicting each target laboratory-based measurement using a reduced dataset, and b) the final assessment of the performance of the three model types for each target, using expanded datasets.

Model inputs were evaluated using a reduced dataset (n = 244) comprising all inputs included in Table 2.1, with samples with missing input data removed. Supernatant color was included as an input for models predicting solid-liquid separation performance, but not for models predicting physical-chemical characteristics, as supernatant color data was collected only for the subset of samples that were evaluated for solid-liquid separation performance. For each target parameter, models were generated for every possible combination of maximal four inputs, and the performance of each model was evaluated using cross-validated R^2 and root mean squared error (RMSE) (5-fold cross validation, repeated 20 times). The input combination with the highest cross-validated R^2 was considered the best. Preference was given to models with fewer inputs, so if inputs could be removed from the model without a relevant decrease in R^2 (at two decimal places), those inputs were not included in the best model. Relative importance of inputs was evaluated by comparing the R^2 of models built with and without the input. The relative strengths of the inputs included in the best models were evaluated by comparing the R^2 of single-input models. The input with the largest R^2 was labeled the ‘strongest predictor’ if the R^2 of that model was at least 75% of the R^2 of the best multi-input model. Supporting predictors were defined as inputs that are included in the best model and increase the model R^2 when included as model inputs along with the strongest predictor. Detailed information about model performance and input importance are included in Appendix B (Tables S4.3-S4.9).

Input selection was also dictated by model type. The simple decision tree model (as defined in Strande et al. (2021b)) was designed to use only categorical data, so only questionnaire data and expert assessments were used as inputs to this model. The linear regression model was evaluated for all inputs (questionnaire data, expert assessments, and simple analytical measurements), except the texture parameters dissimilarity and ASM. These were removed after pre-screening for collinearity with other inputs (Pearson coefficient > 0.85), as they were strongly correlated with other texture parameters. The random forest model was evaluated for all inputs.

After the relevant inputs for predicting each target had been identified, final model performance was determined for each target. The dataset for the final evaluation included only the relevant inputs that were used in the best models (of each model type) for predicting a specific target, in order to maximize the number of data points used to train and evaluate each model. This allows the performance of the three model types in predicting a specific target to be compared. Cross-validated R^2 and RMSE were used in the final performance evaluation.

4.3 Results and Discussion

Data collected with cost-efficient and simple to execute field measurements were used to develop predictive models of the more expensive, laboratory-based analysis (i.e. physical-chemical characteristics and solid-liquid separation performance), in order to evaluate whether the less expensive methods could be used as proxies or partial replacements in data collection. The results and discussion are presented in the order of: 1) presentation of the collected field- and laboratory-based data; 2) a comparison of the developed models and best predictors; and 3) implications for use in characterization of fecal sludge for different fecal sludge management scenarios, including citywide planning, routine FSTP operations, and real-time process control of treatment technologies.

4.3.1 Overview of characteristics and trends in collected data

Results of the characterization of fecal sludge from septic tanks and pit latrines are presented in Table 4.3 and 4.4. Overall, the values were highly variable, and did not follow a normal distribution, which is consistent with other studies (Strande et al. 2021b). Median and mean field measurements, physical-chemical characteristics and solid-liquid separation performance metrics measured in this study are within the expected range based on reported median and mean values in the literature for sludge from septic tanks and pit latrines in Lusaka (Tembo 2019, Tembo et al. 2019), Kampala, Uganda. (Englund et al. 2020, Gold et al. 2018a, Semiyaga et al. 2017), Dakar, Senegal (Gold et al. 2016, Ward et al. 2019), Dar es Salaam, Tanzania (Ward et al. 2019), Durban, South Africa (Radford et al. 2015), Ouagadougou, Burkina Faso (Bassan et al. 2013), Accra, Ghana (Heinss et al. 1999), Bangkok, Thailand

(Heinss et al. 1999), Manila, Philippines (Heinss et al. 1999), and Hanoi, Vietnam (Englund et al. 2020, Gold et al. 2018a).

Table 4.3. Results of simple analytical field measurements grouped by septic tank and pit latrine. Literature values for comparison are ranges of reported mean or median values. *(a) Gold et al. 2018a, (b) Gold et al. 2016, (c) Ward et al. 2019, (d) Englund et al. 2020, (e) Heinss et al. 1999, (f) Bassan et al. 2013, (g) Tembo 2019, (h) Semiyaga et al. 2017, (i) Tembo et al. 2019*

	Field measurements														
	EC	pH	foam height	color			supernatant color			texture					
	(mS/cm)		(mm)	H	S	V	H	S	V	cont.	dissim.	homog.	ASM	energy	corr.
SEPTIC TANKS															
mean	2.6	7.64	5.62	53	28	21	54	17	87	2.69	0.44	0.85	0.35	0.56	0.63
std	2.5	0.45	6.05	15	19	20	24	15	10	7.87	0.55	0.09	0.20	0.17	0.22
median	**1.8**	**7.66**	**4.00**	**50**	**23**	**13**	**51**	**11**	**87**	**0.35**	**0.28**	**0.86**	**0.32**	**0.57**	**0.66**
25%	1.4	7.49	0.00	45	14	10	49	7	83	0.25	0.20	0.83	0.20	0.45	0.45
75%	2.7	7.83	10.00	60	39	19	56	23	93	0.77	0.36	0.91	0.44	0.66	0.81
n	202	202	202	200	200	200	181	181	181	197	197	197	197	197	197
literature values:															
reported means	2.3-15.4[a,b,c,d]	6.9-7.9[a,b,c,d,e]													
reported medians	2.0-13.5[a,c,d]	7.4-7.8[a,c,d]													
PIT LATRINES															
mean	14.2	7.59	2.71	50	46	21	41	69	66	9.33	1.10	0.74	0.23	0.45	0.50
std	5.3	0.53	6.63	9	20	9	10	18	16	9.50	0.73	0.11	0.16	0.15	0.15
median	**14.5**	**7.73**	**0.00**	**48**	**50**	**20**	**42**	**74**	**68**	**6.36**	**1.02**	**0.72**	**0.18**	**0.42**	**0.49**
25%	11.2	7.39	0.00	45	30	15	35	67	56	1.50	0.41	0.66	0.12	0.34	0.39
75%	17.2	7.96	2.00	54	61	26	48	82	78	14.81	1.61	0.82	0.29	0.54	0.59
n	207	207	207	203	203	203	46	46	46	198	198	198	198	198	198
literature values:															
reported means	12.1-14.6[a]	7.1-8.2[a,d,g]													
reported medians	12.0-13.6[a]	7.1-8.2[a,d,g]													

Table 4.4. Results of laboratory-based analysis grouped by septic tank and pit latrine. Literature values for comparison are ranges of reported mean or median values. *(a) Gold et al. 2018a, (b) Gold et al. 2016, (c) Ward et al. 2019, (d) Englund et al. 2020, (e) Heinss et al. 1999, (f) Bassan et al. 2013, (g) Tembo 2019, (h) Semiyaga et al. 2017, (i) Tembo et al. 2019*

	Laboratory-based measurements								
	Supernatant turbidity *(NTU)*	CST *(s)*	TS in dewatered cake *(% ds)*	COD *(g/L)*	NH4-N *(g/L)*	TS *(% ds)*	VS *(% of TS)*	TOC_{solids} *(% of TS)*	TKN_{solids} *(% of TS)*
SEPTIC TANKS									
mean	180	86	22.3	72.1	0.5	4.8	53.5	10.9	2.2
std	230	132	22.2	56.9	0.7	6.6	20.2	2.7	1.2
median	**100**	**42**	**14.8**	**53.3**	**0.3**	**2.0**	**51.8**	**11.0**	**2.1**
25%	40	13	7.4	32.0	0.1	0.6	41.8	9.1	1.4
75%	240	114	26.6	93.9	0.5	6.8	64.9	13.0	3.0
n	179	172	157	165	202	189	181	142	142
literature values:									
reported means		97.6[c]	11-18[a,c]	7.6-43.0[a,b,d,e,f]	0.18-0.6[a,b,d]	0.8-7.2[a,b,c,d,e,f]	48.3-76[a,b,c,d,e,f]		
reported medians		57.8[c]	12-14[a,c]	7.5-35[a,d]	0.24-0.63[a,d]	0.6-2.6[a,c,d]	52.7-75.5[a,c,d]		
PIT LATRINES									
mean	850	707	22.0	122.6	3.0	14.7	56.4	10.9	2.3
std	800	788	16.6	65.5	1.5	8.2	16.9	3.1	1.2
median	**650**	**468**	**18.2**	**121.1**	**3.0**	**14.8**	**59.0**	**9.9**	**2.2**
25%	390	304	9.4	82.1	2.1	9.8	44.3	9.1	1.4
75%	1000	834	31.8	156.5	3.7	18.9	69.2	13.6	2.8
n	46	45	43	154	206	195	193	124	124
literature values:									
reported means		179-1485[c,h]	18-31.8[a,c,h]	10.9-129[a,d,f,g]	1.4-3.2[a,d]	0.9-19[a,d,f,g,i]	43.2-63[a,d,f,g]		
reported medians		179[c]	17-30[a,c]	9.8-127.2[a,d,g]	1.3-3.1[a,d]	1.1-17[a,d,g]	52-64[a,d,g]		

As a first step in developing the predictive models, correlations and trends within the data were investigated. Values for EC, foam height, color (saturation), supernatant color (saturation), texture (contrast, dissimilarity, homogeneity, ASM, energy, and correlation), supernatant turbidity, CST, COD, NH_4^+-N, TS, and TOC_{solids} were significantly different for pit latrine and septic tank sludge, whereas pH, TKN_{solids}, and TS in the dewatered cake were not significantly different (based on 95% confidence intervals around the medians).

Dependencies between questionnaire categories of containment type, toilet type, water connection, and source exist. Illustrated in Figure 4.2, there is a dependency between containment type (i.e. pit latrine or septic tank) and toilet type (i.e. dry or wet), as water is used to convey excreta from toilets to septic tanks, whereas water is not required for conveyance to pit latrines. In a similar fashion, sites that had water connections were also more likely to have septic tanks. 55% of households had pit latrines and 45% septic tanks, whereas non-household establishments had a majority of septic tanks (80%). It was surprising that 2.5% of septic tanks were associated with dry toilets, and 12% with no water connection. This calls into question what people actually mean when they report ‘septic tank’ and brings attention to the apparent disparity between common assumed definitions of septic tanks and how they are actually defined in the field. In the future, instead of septic tank and pit latrine, descriptors of the actual containment technology (for example: lined, unlined, baffled, presence of overflow) are likely to provide more accurate and globally comparable descriptions (Johnston and Slaymaker 2020).

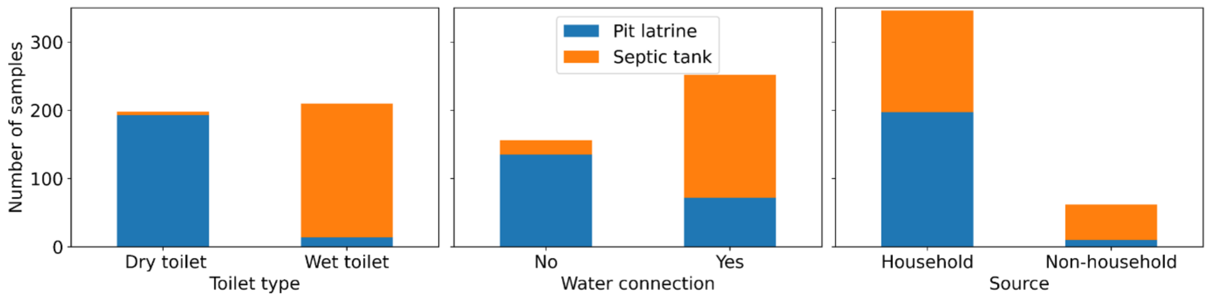


Figure 4.2. a) Distribution of pit latrines and septic tanks connected to dry and wet toilets, b) distribution of pit latrines and septic tanks at locations with and without onsite water connections, and c) distribution of pit latrines and septic tanks at households and non-household sites.

Questionnaire results for categories of containment type, toilet type, water connection, and source were related to laboratorybased measurements. Pit latrines, dry toilets, and sites with no water connection yielded correspondingly less diluted sludge (significantly higher median COD, NH_4^+-N, and TS) with poorer settling and filtration performance (higher supernatant turbidity and CST). Septic tanks, wet toilets (either pour-flush or cistern flush), and sites with a water connection yielded correspondingly more diluted sludge (significantly lower median COD, NH_4^+-N, and TS) with better settling and filtration performance (lower supernatant turbidity and CST). Because of the strong correlations between containment type and toilet type, only containment type is further included as a potential model input. Sludge from households was less diluted (higher median COD, NH_4^+-N, and TS), and had poorer settling and filtration performance (higher supernatant turbidity and CST) compared with sludge from non-household sources (Figures S4.2 and S4.3). These results contrast with observations in earlier studies in Dakar and Dar es Salaam, where sludge from public toilets (non-household) had poorer settling and dewatering performance than sludge from households (Ward et al. 2019), but agree with observations in Kampala that there was a difference between physical-chemical properties (COD, TS) of sludge from household and non-household sources (Strande et al. 2018). This suggests that source (i.e. household or non-household) may sometimes be a useful predictor of sludge characteristics, but the predictive relationships are different in different cities.

The expert assessment categories of color and odor show dependencies with each other, and with questionnaire categories. 98% of ‘fresh’ smelling samples were light or medium brown, while 74% of ‘stabilized’ smelling samples were black (see Figure S4.5). The majority of pit latrine sludge had a ‘fresh’ or ‘middle’ odor (79%) and was medium or light brown (74%), while the majority of septic tank sludge had a ‘stabilized’ odor (75%) and was black (74%) (see Figure. S4.6).

The expert assessment categories color and odor were associated with laboratory-based measurements. There were significant differences in the median supernatant turbidity, CST, NH_4^+-N, TS, COD, VS, and TOC_{solids} between ‘fresh’/light brown and ‘stabilized’/black samples (Figures S4.2 and S4.3). Interestingly, our results support common practitioner wisdom that has, to our knowledge, not been quantified in the literature: fecal sludge with higher organic matter content (COD, TOC, VS) was associated by odor and color as “fresh” sludge, whereas sludge labeled “stabilized” had correspondingly significantly lower organic matter. ‘Fresh’ sludge also took significantly longer to dewater and had poorer settling performance. This corroborates practitioner observations from the field that fresh sludge dewaters and settles more poorly than more stabilized sludge (Cofie et al. 2006, Heinss et al. 1999, Ward et al. 2019). This finding is interesting, because it offers insight into the transformation of organic matter in sludge as it stabilizes. The specific shift from the “fresh” feces odor to the more stabilized “barnyard” or “manure” odor during stabilization is a result of bacterial metabolism of organic acids, which produce the smells associated with fresh feces, leaving behind phenolic and sulfur-containing compounds, which are associated with the odor of stabilized feces (Lin et al. 2013, Starkenmann 2017). These results can direct future research into the transformation of organic matter in the sludge during stabilization and its effect on fundamental mechanisms controlling dewatering.

Relationships to simple analytical measurements are discussed in detail in the 'Suitability of field predictors' section.

4.3.2 Comparison of model input combinations and performance

The collected data were evaluated to see if field measurements (Table 4.1) could be used to predict laboratory-based measurements (Table 4.2). A range of field measurements were evaluated as inputs in the predictive models, including questionnaire data such as containment type (i.e. pit latrine or septic tank), source (i.e. household or non-household)), expert assessments such as odor and color (qualitative), and simple analytical measurements such as EC, pH, foam height, color (quantitative), texture, and supernatant color. Table 4.5 summarizes the combinations of field measurements that were selected as relevant inputs in every model type for each target parameter.

Table 4.5. Field measurements identified as relevant model inputs to predict laboratory-based measurements, and corresponding R^2 and RMSE of models run with selected inputs. Strongest predictors (accounting for at least 75% of R^2) shown in black, supporting predictors shown in grey, inputs to poorly performing models (R2<0.[2]) shown in light grey, inputs not used in model shown in white, and inputs not evaluated shown with grey dots. Model performance metrics, R^2 and RMSE for each target parameter by type of model are displayed on the right.

strongest predictor (accounts for at least 75% of R^2)
supporting predictor
input to a poorly performing model ($R^2 < 0.2$)
not a predictor
not evaluated

Field measurements (model inputs)

Laboratory-based measurements (target parameters)			Questionnaire + expert assessments: Containment type	Water connection	Source	Odor	Color (qualitative)	Simple analytical measurements: EC	pH	Foam height	Color (quantitative)	Texture	Supernatant color	R^2	RMSE
Solid-liquid separation performance	Supernatant turbidity	decision tree	strongest predictor					not evaluated	not evaluated	not evaluated	not evaluated	not evaluated	not evaluated	0.21	280 NTU
		linear			supporting predictor				supporting predictor				strongest predictor	0.62	190 NTU
		random forest							supporting predictor			supporting predictor	strongest predictor	0.66	180 NTU
	CST	decision tree	strongest predictor					not evaluated	not evaluated	not evaluated	not evaluated	not evaluated	not evaluated	0.37	200 s
		linear								supporting predictor			strongest predictor	0.51	170 s
		random forest										supporting predictor	strongest predictor	0.64	140 s
	TS in dewatered cake	decision tree		input to a poorly performing model				not evaluated	not evaluated	not evaluated	not evaluated	not evaluated	not evaluated	-0.06	13 %ds
		linear								input to a poorly performing model		input to a poorly performing model		0.00	12 %ds
		random forest						input to a poorly performing model		input to a poorly performing model		input to a poorly performing model	input to a poorly performing model	0.01	12 %ds
Physical-chemical characteristics	COD	decision tree	input to a poorly performing model					not evaluated	not evaluated	not evaluated	not evaluated	not evaluated	not evaluated	0.09	62 g/L
		linear	supporting predictor							supporting predictor			not evaluated	0.25	56 g/L
		random forest	supporting predictor							supporting predictor			not evaluated	0.27	55 g/L
	NH4-N	decision tree	strongest predictor					not evaluated	not evaluated	not evaluated	not evaluated	not evaluated	not evaluated	0.51	1.2 g/L
		linear						strongest predictor	supporting predictor	supporting predictor		supporting predictor	not evaluated	0.66	1.0 g/L
		random forest						strongest predictor	supporting predictor			supporting predictor	not evaluated	0.65	1.0 g/L
	TS	decision tree	strongest predictor					not evaluated	not evaluated	not evaluated	not evaluated	not evaluated	not evaluated	0.26	7.7 %ds
		linear	supporting predictor								supporting predictor	strongest predictor	not evaluated	0.55	6.0 %ds
		random forest								supporting predictor	supporting predictor	strongest predictor	not evaluated	0.55	6.0 %ds
	VS	decision tree					input to a poorly performing model	not evaluated	not evaluated	not evaluated	not evaluated	not evaluated	not evaluated	0.07	17 % of TS
		linear			input to a poorly performing model				input to a poorly performing model		input to a poorly performing model		not evaluated	0.12	17 % of TS
		random forest			input to a poorly performing model		input to a poorly performing model						not evaluated	0.06	17 % of TS
	TOC_{solids}	decision tree				input to a poorly performing model		not evaluated	not evaluated	not evaluated	not evaluated	not evaluated	not evaluated	0.01	2.8 % of TS
		linear		input to a poorly performing model		input to a poorly performing model					input to a poorly performing model		not evaluated	0.13	2.6 % of TS
		random forest							input to a poorly performing model	input to a poorly performing model	input to a poorly performing model	input to a poorly performing model	not evaluated	0.13	2.6% of TS
	TKN_{solids}	decision tree				input to a poorly performing model		not evaluated	not evaluated	not evaluated	not evaluated	not evaluated	not evaluated	-0.02	1.2 % of TS
		linear				input to a poorly performing model			input to a poorly performing model				not evaluated	0.01	1.2 % of TS
		random forest				input to a poorly performing model							not evaluated	-0.01	1.2 % of TS

Predictive models using “cheap and easy” field measurements

The strongest predictor for each model is shown in black, and supporting predictors, which increase the model performance when included as model inputs, are shown in grey. Model inputs included in poorly performing models ($R^2 < 0.2$) are shown in light grey. The model performance metrics on the right of the figure are cross-validated R^2 and RMSE for models using the highlighted field measurements as inputs. Since simple decision tree models are based on single inputs, the input associated with the highest performing model for each target was designated as the strongest predictor. Tables showing model prediction accuracy and error depending on the inclusion of different predictors are included in Appendix B (Tables S4.3-S4.9)

Model performance

As seen in Table 4.5, in predicting a given target, simple decision tree models did not perform as well, while linear regression models and random forest models generally improved model fit and reduced error. Decision tree models were able to account for 21-51% of the variance in supernatant turbidity, CST, NH_4^+-N, and TS. Linear models improved on the predictions made by decision tree models for supernatant turbidity, CST, NH_4^+-N, TS, and COD, with 195%, 38%, 178%, 29%, and 112% increases in model fit (R^2) and 32%, 15%, 10%, 17%, and 22% reductions in prediction error (RMSE) respectively. Random forest models improved on the predictions made by decision tree models for supernatant turbidity, CST, NH_4^+-N, TS, and COD, with 214%, 73%, 200%, 27%, and 112% increases in model fit (R^2), respectively, and 36%, 30%, 11%, 17%, and 22% reductions in prediction error (RMSE), respectively. Random forest models outperformed linear models only in predicting the solid-liquid separation performance metrics supernatant turbidity and CST. Random forest models improved on the predictions made by linear models for supernatant turbidity and CST with 6% and 25% increases in R^2 and 5% and 18% reductions in RMSE, respectively. It was unexpected that random forest models did not provide higher accuracy predictions than linear models for most parameters, as random forest models are able to capture nonlinear interactions between predictors (Hastie et al. 2009), and nonlinear relationships have been observed in models of nutrients in wastewater effluent (Castrillo and García 2020). It is possible that given access to a larger training dataset, random forest model performance could exceed that of linear models.

Something to consider when selecting a predictive model is the trade-off between increased predictive power and interpretability of the model. Simple decision tree models and linear regression models tend to be less flexible and provide less robust predictions for complex nonlinear datasets, but it is relatively easy to understand the relationships between a field measurement and the target parameter. In contrast, nonlinear methods such as random forest models are more flexible and may produce better predictions, but at the expense of model interpretability (Hastie et al. 2009). In cases where linear models and random forest models provide comparable predictions, linear models may be preferred for their relative simplicity. In cases where data will be collected routinely, and a dataset will continue to grow in size and complexity, random forest models may be preferred for their ability to extract nonlinear relationships from larger datasets.

Suitability of field predictors

Although questionnaire data and expert assessments were useful inputs for the decision tree models, they were only incrementally helpful to include in the linear and random forest models. Containment type was the strongest predictor for decision tree models, explaining 21-51% of the variance in supernatant turbidity, CST, NH_4^+-N, and TS. For all linear and random forest regression models with R^2 higher than 0.5, a single field measurement (the strongest predictor) was responsible for at least 75% of the prediction accuracy (black in Table 4.5). Although the inclusion of questionnaire data (containment type and source) did contribute to increased fit for some linear and random forest models, the simple analytical parameters were always the strongest predictors for these types of models. This indicates that the predictive detail represented by the separation into septic tank and pit latrine or household and non-household is largely captured by the differences in the physical-chemical compositions of sludge from septic tanks and pit latrines or household and non-household sources. The simple analytical field measurements may be better model inputs than questionnaire data or expert assessments because they are continuous instead of categorical and are thus able to provide higher resolution information.

Supernatant turbidity and CST were predicted primarily by supernatant color. Supernatant color contributed 86% and 87% of the linear and random forest model fits, respectively, in predictions of supernatant turbidity, and 75% and 90% of linear and random forest model fits, respectively, for predictions of CST. Including texture as an input further improved random forest model predictions of both supernatant turbidity and CST. Our results support and quantify previous observations of a relationship between qualitatively measured supernatant color and settling and filtration performance in fecal sludge from Dakar and Dar es Salaam (Ward et al. 2019). This relationship is hypothesized to be a result of high concentrations of suspended and soluble organic matter, which also contribute to high supernatant turbidity and filter clogging during dewatering (Ward et al. 2019). Field measurements can be selected to suit available technical and financial resources. Although supernatant color was the strongest predictor of supernatant turbidity and CST, it may not be an ideal field predictor in every case, as it requires settling prior to measurement. However, settling tests are also simple analytical tests that can be performed in the field with low-cost equipment in less than an hour (e.g. Imhoff cones or settling columns) (Junglen et al. 2020). It is not surprising that supernatant turbidity is strongly related to supernatant color. Now that this relationship has been quantified for fecal sludge in Lusaka, it seems promising for estimations of effluent turbidity using color in photographs as a proxy, and could replace turbidity measurements where spectrophotometers are not available.

NH_4^+-N was predicted primarily by EC, which contributed 91% and 82% of the linear and random forest model fits, respectively. (The linear model is simple: EC (mS/cm) $*$ 0.2 = NH_4^+-N (g/L), see Figure S4.7). The inclusion of texture and pH in the models further improved predictions. There is precedent in the literature for the use of EC as a predictor of ammonia in manure and fecal sludge. Onsite measurement of EC has been suggested as a good proxy for predicting nutrient concentrations, including ammonia nitrogen (R^2 = 0.91) in swine manure (Suresh et al. 2009). For fecal sludge from pit latrines and septic tanks in Uganda, Vietnam, and Japan, Gold et al. (2018a) observed a linear correlation (R^2 = 0.6) between NH_4^+-N and EC. Based on these results, onsite measurement of EC using a conductivity probe is a promising option for cheaper and less resource-intensive monitoring of NH_4^+-N.

TS was predicted primarily by texture, which contributed 81% and 86% of the linear and random forest model fits, respectively. The inclusion of color (quantitative) further improved both model types, and including containment type additionally improved linear model predictions. These results make sense based on scientific knowledge that sludge surfaces gain texture as they dry, changing from a smooth liquid to a lumpy slurry, to a rough semi-solid or solid. The physical transformation of sludge as it dries following dewatering has been well characterized for sediment sludges on drying beds, and TS has been shown to be predictable to a high level of accuracy (R^2 > 0.92) using machine learning based on texture analysis (Bodun et al. 2000). Our results indicate that using texture measurements extracted from photographs for prediction of TS could be useful in eliminating the time- and equipment intensive laboratory analysis of TS for fecal sludge.

The target parameters VS, TOC_{solids}, TKN_{solids}, and TS in the dewatered cake were not able to be predicted using any of the models. These parameters, along with COD (for which models were only able to account for ~25% of the variation in the data), are all associated with the sludge organic matter. It appears that the field parameters evaluated in this study are not sufficient to fully characterize and predict the organic components in fecal sludge. This is consistent with our previous observations that it was difficult to predict variation in dewatered cake solids and VS in fecal sludge from Dakar and Dar es Salaam (Ward et al. 2019). This was hypothesized to be partly due to the influence of soil and solid waste on measured VS and sludge dewatering behavior. A rapid field measurement to predict silica content as a proxy for soil could be a possible solution. Miller et al. (2012) proposed the use of field portable infrared spectrometers to predict silica content in coal dust. Another possibility could be monitoring organic matter composition using in situ fluorescent sensors. This has been demonstrated in the field, where COD and soluble COD in the effluent of decentralized wastewater treatment systems were able to be monitored using fluorescence as a proxy measurement (Mladenov et al. 2018). Fluorescent sensing may also be able to provide better predictions of the composition and transformation of organic matter in sludge during stabilization; fluorescent peaks have been associated with the concentrations of high molecular weight humic substances and other organic matter, and thus may be a promising method for monitoring level of stabilization (Mladenov et al. 2018, Yao et al. 2016). Further fundamental understanding of the mechanisms controlling fecal sludge stabilization and dewatering will likely be instructional in identifying possible field predictors.

Models can also be adapted to incorporate new field measurements, especially in cases when practitioners have identified them to be operationally relevant. For example, we found that including density as an input can significantly improve the prediction accuracy of the TS model (random forest model prediction improves to R^2 = 0.70, RMSE = 4.8 %ds). For practitioners who have access to a penetrometer for field density measurements as described in Radford and Sugden (2014), this could be a valuable field predictor to incorporate.

4.3.3 Implications for the field

In this study, we evaluated three types of models with increasing levels of complexity. The selection of the best model for different applications will depend on the required level of accuracy and on the available resources. Here, we explore where and how each model type could be relevant, considering several situations where predictive models could be useful for characterization and monitoring, as summarized in Figure 4.3.

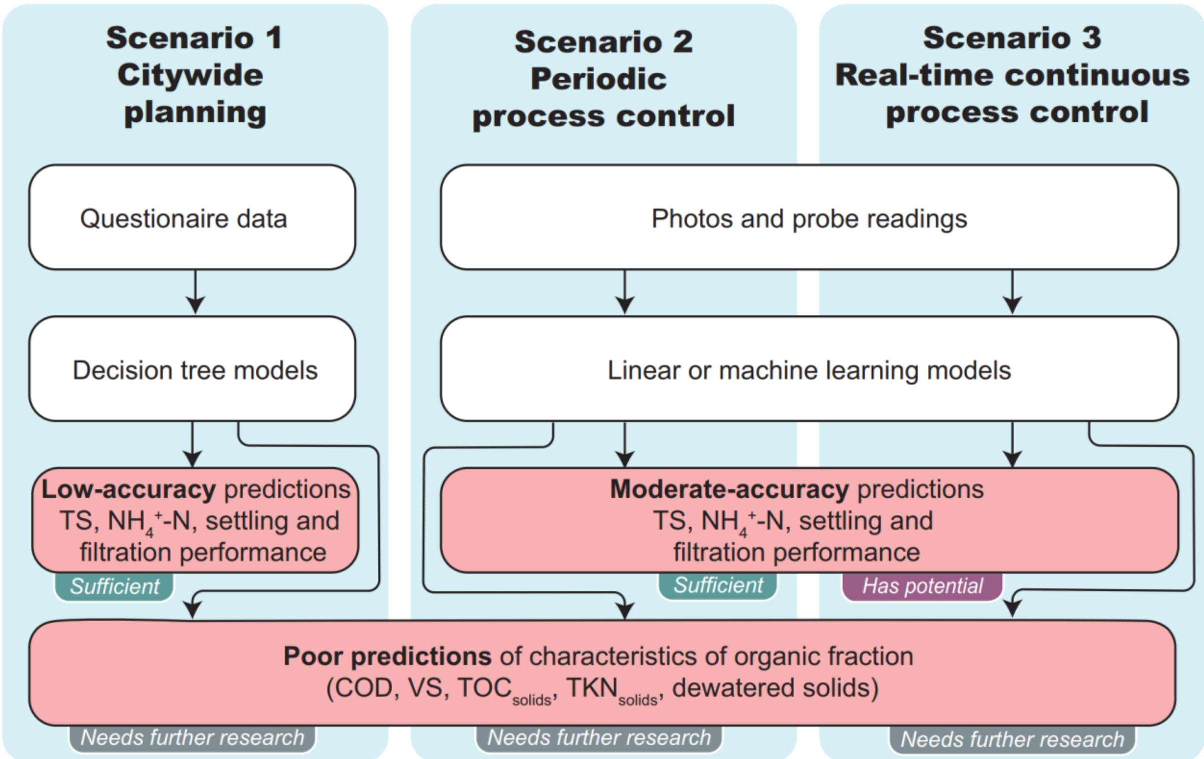


Figure 4.3. This flowchart summarizes three types of planning and operation scenarios where cost efficient and simple field measurements and models could be employed. Appropriate inputs and model types for each scenario are shown as white boxes, and prediction accuracies of different outputs are shown as pink boxes. The tabs at the base of each pink box indicate whether the prediction accuracy is likely to be sufficient, has potential with further data collection, or requires additional research prior to implementation.

Scenario 1: Community to citywide planning

Information provided by simple decision tree models may be adequate for the level of detail required for community to citywide planning. Incorporating questionnaire data such as containment type into simple decision tree models can already provide a moderate amount of information about sludge characteristics and solid-liquid separation performance. Siting decisions and designs for new FSTPs are often based on citywide averages of loadings and volumes (Klinger et al. 2019), but it has recently been suggested that incorporating decision tree models to estimate quantities and qualities of fecal sludge accumulating on a neighborhood level would greatly improve projections of loadings for planning of FSTPs, transfer stations, and regularly scheduled empting programs (Strande et al. 2021b). Our study confirms that such an approach would provide an improvement over citywide averages: a simple decision tree model based on containment type accounts for 26% of the variance in TS. Because this application does not require a high-resolution prediction, decision tree models providing low-accuracy predictions (R^2 of 0.26 for TS) maybe be adequate, as they are easy to understand conceptually and are in many cases based on information that city planners might already have access to. This can include information collected in a shit flow diagram (SFD), for example, prevalence of pit latrines and septic tanks by neighborhood, location of water taps, or data on housing density or residential/commercial land use (Peal et al. 2020).

Scenario 2: Periodic process control of fecal sludge treatment technologies

Operating a fecal sludge treatment plant requires more refined predictions than citywide planning. Combining cost-efficient and simple field measurements with linear or machine learning models contributes a further improvement in prediction accuracy, providing moderate-accuracy predictions of TS, NH_4^+-N, and settling and filtration performance (R^2 of 0.51-0.66). Operators routinely make decisions about how much sludge to load on drying beds, and when to remove dewatered/dried sludge. Improvements in consistency of solids loading and shorter, more consistent residence times could substantially increase treatment plant capacity and performance (Klinger et al. 2019, Seck et al. 2015). Currently, the drying beds at most FSTPs are operated at constant hydraulic loadings and residence times, with no monitoring of influent sludge characteristics (Klinger et al. 2019). As a result of not being able to monitor influent sludge, common operational problems due to the variability in influent TS arise. These problems include overloaded drying beds clogging and dewatering too slowly, and underloaded beds wasting space and decreasing treatment capacity (Klinger et al. 2019). Being able to adjust the loading of drying beds based on TS concentrations and dewatering time, together with monitoring of TS on the drying beds, would increase treatment performance and treatment plant capacity. This could be done by incorporation of linear or machine learning models into a smartphone app, so that pictures taken with the smartphone would provide estimates of TS concentrations and dewatering time. A more simple application could be employed with printed cards with example colors and textures for comparison and decision-making (von Sperling et al. 2020b).

Scenario 3: Real-time continuous process control of fecal sludge treatment technologies

The capacity to make rapid predictions based on photos and probe measurements could be a serious game-changer for processes requiring real-time monitoring. Online conditioner dosing for advanced settling and dewatering is one example (Ward et al. 2021b). The current state of the art in FSM is to adjust polymer dosing flowrates based on containment type: pit latrine sludge is dosed at a higher flowrate than septic tank sludge to account for the differences in average TS of sludge in each category (Ward et al. 2021b). This method provides insufficient resolution for predictions, and it is very difficult to avoid over- and under-dosing conditioner due to the high variability in sludge characteristics. In this case, a smartphone app with photo and probe inputs (obtained at the treatment facility) could be used to monitor TS. If the TS value of the influent sludge is 2% ds, the random forest model should predict within a range of 0.7 – 3.3 % ds. This may be sufficiently high resolution, depending on the propensity of the selected conditioner to overdosing. For example, using a 2000 L mixing tank at a transfer station (Rhodes-Dicker et al. 2020) and a target polymer conditioner dose of 2 mL/g TS, an actual dose between 0.7 and 3.3 mL/g TS could be achieved (see Appendix B for calculation). This is comparable to the observed optimal dosing window for the conditioner (1-3 mL/g TS). These models are a proof of concept; they will also need to be refined depending on the technology and validated prior to use in other cities. Currently, there is a significant lack of information on fecal sludge characterization to drive the development of predictive models. The development of a global database of fecal sludge field measurements uploaded together with laboratory results using standardized methods (Velkushanova et al. 2021b) will allow for the continuous improvement of models and global applications of these predictions. Such an approach has significant potential to provide reliable characterization data and enable real-time monitoring and process control for fecal sludge treatment in cities all over the world.

4.4 Conclusions

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

• Cost-efficient and simple field measurements from photos (color, texture) and probes (EC, pH) can be used as predictors of fecal sludge characteristics and solid-liquid separation performance (TS, NH_4^+-N, settling efficiency, and filtration time) when combined with linear regression and machine learning models.

Predictive models using “cheap and easy” field measurements

• Containment type is a good predictor of fecal sludge characteristics and solid-liquid separation performance (TS, NH_4^+-N, settling efficiency and filtration time) and can be especially helpful for making low-resolution predictions when combined with simple decision tree models, e.g. for projecting loadings for FSTP design.

• Laboratory-based measurements associated with the organic matter in the sludge solids (COD, VS, TOC_{solids}, TKN_{solids}, TS in the dewatered cake) could not be predicted using the methods we evaluated. A better understanding of the organic matter in fecal sludge and how it relates to solid-liquid separation performance is needed to identify good predictors.

• Based on this proof of concept, which indicates that predictions of characteristics using photographs and probe measurements are possible, focus should be placed on validating this approach in other cities. The collection of worldwide datasets would allow for global implementation and continuously improving machine learning models. Our ongoing research includes development of an app for field practitioners that can predict fecal sludge characteristics based on pictures taken with a smartphone.

4.5 Acknowledgements

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4.6 Supporting information and additional data

Supplementary information is included in Appendix B at the end of this document. The complete dataset is published at https://doi.org/10.25678/00037X.

Chapter 5

Sludge Snap: A machine learning approach to fecal sludge characterization in the field

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Ward, B.J., Allen, J., Escamilla, A., Sivick, D., Sun, B., Yu, K., Dahlberg, R., Niu, R., Ward, B.C., Strande, L. (2021) Sludge Snap: A machine learning approach to fecal sludge characterization in the field. 42nd WEDC International Conference, Loughborough University. Conference contribution. https://hdl.handle.net/2134/16866394.v1

5.1 Background

Fecal sludge characterization is necessary to ensure adequate fecal sludge management and protection of public and environmental health. However, characterization is currently very difficult to achieve in many places relying on non-sewered sanitation due to limited access to analytical laboratories and limited supply chains for chemicals and laboratory equipment. To address this, we developed predictive models that use photographs, probe measurements, and field observations to estimate fecal sludge characteristics without the need for a lab (Ward et al. 2021a). This study was a proof of concept, based on 465 samples collected from pit latrines and septic tanks throughout Lusaka, Zambia. We found that physical-chemical characteristics of fecal sludge, including total solids (TS) and ammonium-nitrogen (NH_4^+-N), and dewatering performance metrics (supernatant turbidity and filtration time) can be predicted using machine learning and linear models based on sludge photographs and probe readings (pH, conductivity).

These models were able to predict the results of lab characterization with moderate accuracy (R^2 of 0.55-0.66). For example, if the TS of an influent sludge sample is 20 g/L, the random forest model should predict within a range of 7-33 g/L. While estimating TS to this level of accuracy could be sufficient for better calculating loadings on drying beds or conditioner dosing, we are hopeful that estimates can be improved in the future as more data are gathered and models are refined (Ward et al. 2021a). In addition, before these models can be useful in practice, they need to be validated and refined in other cities. To make validation data collection easier, and for these models to ultimately be useful to practitioners and researchers, we developed an app that can intake field data and generate real-time predictions in the field.

5.2 Sludge Snap app

The Sludge Snap app was developed by a team of computer science and engineering students at Virginia Commonwealth University in collaboration with Eawag. The app is intended to be used for lab-independent sludge characterization by operators of fecal sludge treatment facilities or researchers performing community- or city-wide studies of sludge quantities and qualities. The user interface is designed to be simple to use for field data collection. The app prompts the user to take a photograph of a sludge sample with their smartphone, then to add additional field measurements or observations (for example, pH, conductivity, containment type). Once all of the field data have been entered, the app generates predictions of sludge characteristics (for example, TS, NH_4^+-N, or dewatering performance). If the user has created an account and is logged in, recent entries are saved and accessible for the user to review in the app, and results can be exported by emailing a report to the user.

5.2.1 App structure and server

The first iteration of the Sludge Snap app is designed as a web app to ensure that it will function on all smartphones and will not be limited to a specific mobile platform. The application is made up of two parts: a user interface (front-end) on the smart phone built using React (JavaScript), and back-end, which does most of the sophisticated processing of the data, on a remote server accessible by the front-end over the internet, built using Vercel combined with Hasura. The user interface application on the mobile device captures images, performs some simple processing to correct the image, and forwards it to the server back-end for futher analysis. If the internet is not available, the Sludge Snap app on the mobile device will collect the sample image and preliminary information and store them on the mobile device until internet service is available. The app will then forward the collected information to the server back-end for a full analysis. The server back-end provides the Sludge Snap app with in-depth analysis of samples. The server stores the sample images and results of analysis for future reference. The software for the app is modular to allow for new features to be easily added and removed over time.

5.2.2 In-app photo processing module

Once a photo is uploaded to Sludge Snap, the image is pre-processed and color and texture data are extracted. First, the photo is color corrected using a standardized color checker chart included in the image. Then, a computer vision library is used to identify petri dishes containing sludge samples in the color corrected image. Isolated thumbnails of the sludge samples are then processed to extract average sludge color (in HSV) and texture measures. Extracted color and texture data are then fed as inputs to the machine learning model, which predicts the desired outputs.

5.2.3 Machine learning model module

When the input data are submitted, the app runs a pre-defined empirical model and displays the outputs. The machine learning model runs on the back-end server and returns its predicted results to the user on the front-end. The first iteration of the Sludge Snap app runs the random forest models generated from 465 samples in Lusaka (Ward et al. 2021a). However, the app is designed to allow for different models to be easily defined and uploaded, to accommodate use of the app in different cities where different empirical relationships likely exist.

5.3 Next steps

Sludge Snap could be used to predict characteristics in Lusaka, since that is where the data were collected, but a percentage of the samples should still be analyzed in the laboratory for quality assurance purposes. To be able to start using Sludge Snap in other cities, practitioners will need to start populating a global database with pictures and lab results from their cities to see if the same or different trends hold true. If the same trends are seen, it means that some relationships may be globally relevant for fecal sludge, and we may be able to use existing models based on data from many cities. If different trends are seen in different cities, then the models could be adapted to fit each application. Data collection campaigns in seven new cities will begin this summer, to help us begin to validate these models and the Sludge Snap app in new locations and start building a global database of fecal sludge characteristics.

A video of the presentation given at the WEDC conference, which includes a demonstration of the Sludge Snap app is viewable at this link:

https://repository.lboro.ac.uk/articles/conference_contribution/Sludge_Snap_a_machine_learning_approach_to_fecal_sludge_characterization_in_the_field/16866394

Chapter 6

Particle size as a driver of dewatering performance and its relationship to stabilization in fecal sludge

Submitted manuscript

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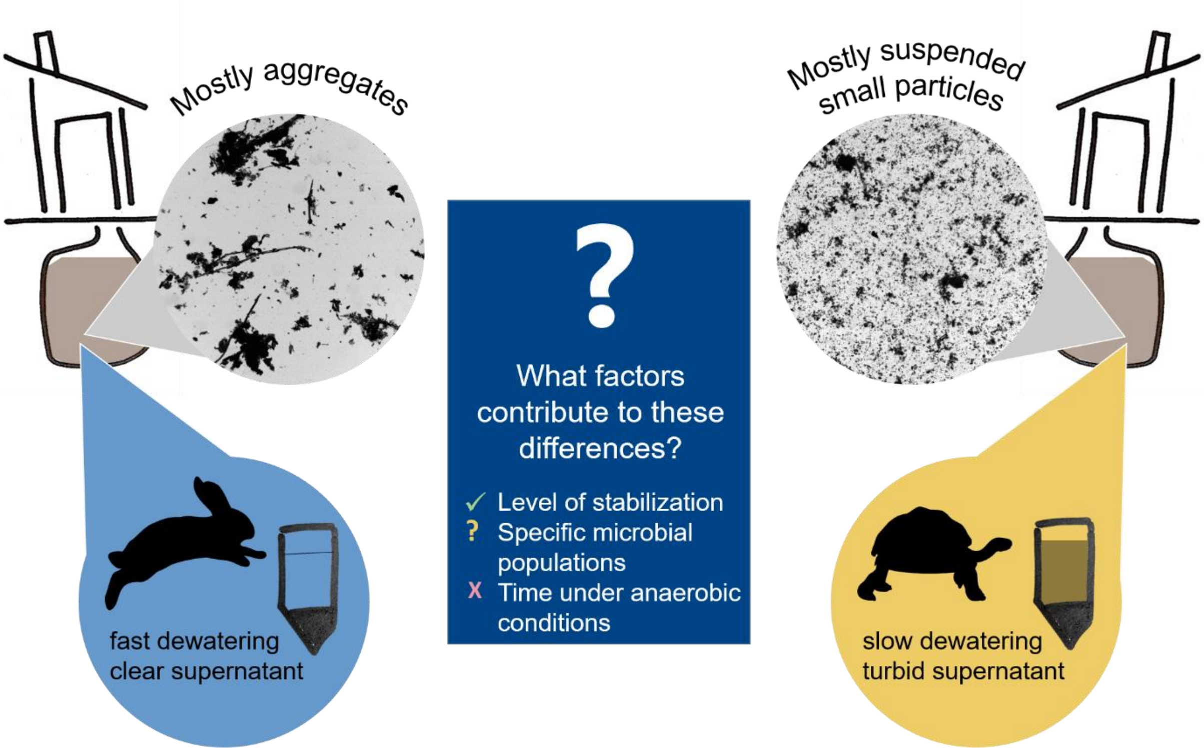
Graphical abstract



Highlights

- Particles <10 µm areresponsible for poor dewatering in fecal sludge
- Stabilization indicators correspond to dewatering performance
- Storage time is not a predictor of stabilization or dewatering in fecal sludge
- Aggregate size is associated with specific microbial communities
- Treatment should focus on removing small particles

Abstract

Poor and unpredictable dewatering performance of fecal sludge is a major barrier to sanitation provision in cities not served by sewers. Each batch of influent fecal sludge arriving at a treatment facility can have 1-2 orders of magnitude different characteristics and dewatering performance. Based on observations by practitioners, it is thought that fecal sludge stabilization increases with time in onsite storage, and that dewatering performance increases with stabilization. It is also known that particle size distribution is a key driver in the dewatering of wastewater sludges. However, a relationship between particle size distribution and dewatering has not been established in fecal sludge, and it is not understood how changes during storage in containment are affecting dewatering. We validate that particle size distribution is a driver of dewatering performance in fecal sludge, and is associated with level of stabilization. We found that the concentration of particles smaller than 10 µm and the median aggregate size (D50) are correlated with capillary suction time (CST) and supernatant turbidity in field samples from Naivasha, Kenya and Kampala, Uganda and in controlled anaerobic storage experiments. While dewatering performance was associated with stabilization indicators (pCOD/COD, VSS/TSS and C/N), stabilization and dewatering were not dependent on time in storage. Samples with larger aggregates had higher abundance of the Gammaproteobacteria Pseudomonas, and samples with smaller aggregates were characterized by the Bacteroidetes families Vadin HA17 and Rikenellaceae. Our results suggest that within the typical timescales between emptying events of onsite containments in urban areas, the stabilization process is not time dependent but is more likely associated with specific microbial populations and the in-situ environmental conditions (e.g. redox conditions, limiting nutrients, qualities of organic matter) which promote or discourage their growth.

6.1 Introduction

Over half of the world's urban population lacks access to safely managed sanitation, a majority of whom rely on onsite sanitation systems like septic tanks and pit latrines (WHO and UNICEF 2017) (Peal et al. 2020). Solid-liquid separation of fecal sludge that accumulates in these systems is one of the greatest technical barriers to achieving better sanitation coverage in non-sewered urban areas. A major issue is inconsistent dewatering performance, which is associated with high variability in the physical-chemical characteristics of influent fecal sludge (Gold et al. 2016). Experienced fecal sludge management practitioners often use descriptive information about influent (e.g. neighborhood of origin, pit latrine or septic tank, public or private facility, color, or odor) to predict dewatering performance and adjust process controls accordingly (Cofie et al. 2006, Ward et al. 2021b). Many practitioners have made connections between the observed level of stabilization of incoming sludge and how well it dewaters, noting that more stabilized sludge is easier to dewater, while fresher sludge is more difficult (Cofie et al. 2006, Heinss et al. 1999). Ward et al. (2019 and 2021a) corroborated practitioner experience by assessing field samples of fecal sludge in Dakar, Senegal; Dar es Salaam, Tanzania; and Lusaka, Zambia, and noted that characteristics associated with fresh sludge, e.g. light brown color and odor of fresh excreta, corresponded to poorer dewatering performance. There are few published studies about the stabilization processes occurring during storage in onsite containments, but stabilization is thought to be primarily anaerobic (Shaw and Dorea 2021, van Eekert et al. 2019), and it is commonly accepted that the degree of stabilization increases with longer time intervals between emptying events, as recalcitrant organic matter builds up in the solids (Lopez Vazquez et al. 2021). In addition, it is not currently understood how and why the stabilization processes in onsite containments affects dewatering.

The factors limiting dewatering in various organic sludges and slurries have been extensively studied. It is well accepted within the wastewater sludge literature that poor dewatering performance (for example, with sludge drying beds, membranes, or mechanical filter presses) is caused by small particles clogging pores and interstitial spaces in the sludge cake (Christensen et al. 2015, Karr and Keinath 1978). Karr and Keinath (1978) saw that the concentration of small particles in the supracolloidal (1 – 100 μm) range was the key driver of differences in filtration performance in wastewater sludges. While we would expect fecal sludge to adhere to the same physical rules governing the dewatering of other sludges, a link between particle size and dewatering performance has not yet been demonstrated. Semiyaga et al. (2017) and Gold et al. (2018a) characterized particle size distribution in fecal sludge field samples using sequential sieving, but did not report a relationship between small particles and filtration performance. Sam et al. (submitted) measured particle size distribution of sludge from one pit latrine over the course of anaerobic storage using static light scattering, but did not observe an association between supracolloidal particles and dewaterability. However, in practice, the addition of polymer conditioners to fecal sludge can greatly improve filtration and settling performance, which leads us to believe that suspended small particles are in fact key determinants of dewatering performance in fecal sludge (Gold et al. 2016, Rhodes-Dicker et al. 2020, Shaw et al. submitted).

Flocculation and coagulation processes, either facilitated by the addition of conditioners or by microbial populations promoting bioflocculation from produced extracellular polymeric substances (EPS), can reduce the concentration of small particles by allowing them to agglomerate into larger flocs, thereby improving filtration (Wei et al. 2018, Wilén et al. 2003). Anaerobic stabilization can either improve or worsen dewatering performance in wastewater sludge. Much of the literature focuses on what happens to activated sludge during anaerobic digestion or storage under anaerobic conditions. The general consensus is that activated sludge dewaterability (measured by filtration and supernatant turbidity) typically worsens after anaerobic stabilization, as small particles that were previously bound within flocs are released into the bulk (Christensen et al. 2015). However, the dewatering of primary wastewater sludge does not respond consistently to anaerobic digestion, and studies have reported either improved or worsened dewatering performance with time under anaerobic conditions (Lawler et al. 1986, Mahmoud et al. 2006, Miron et al. 2000). Since fecal sludge has much lower concentrations of EPS than activated sludge, and comparable or lower concentrations than primary sludge (Ward et al 2019, Sam et al submitted), it is expected that it may respond differently to anaerobic stabilization. It is not known whether aggregates, defined here as clusters of smaller individual particles (e.g. bacterial cells, organic fibers, inorganic particles), are present in fecal sludge. We distinguish aggregates in fecal sludge from flocs in biological activated wastewater sludge, to avoid implying that they have similar properties. If aggregates are present in fecal sludge, we would expect them to contain much less EPS, and not be as large as flocs present in activated sludge based on anaerobic conditions in containments. Because the literature suggests a connection between more stabilized fecal sludge and dewatering performance in field samples, we hypothesize that fecal sludge dewatering will be improved through anaerobic stabilization due to the degradation of small particles, which will reduce pore clogging and supernatant turbidity.

Based on current understandings reported by practitioners and the literature for fecal sludge management and municipal wastewater treatment, we designed this study to evaluate the following questions (1) does stabilization of fecal sludge increase with storage time in onsite containment, (2) is the level of stabilization related to particle size distribution, and (3) are changes in particle size distribution responsible for changes in dewaterability. The overall objective of this study was to evaluate whether particle size distribution is a determining factor in the dewatering performance of fecal sludge, and to investigate to what extent differences in particle size and dewatering performance are driven by stabilization, using field samples and controlled anaerobic storage.

6.2 Materials and Methods

6.2.1 Field sample collection

Twenty field samples were collected from onsite containments, 10 from Kampala, Uganda, and 10 from Naivasha, Kenya. Sampling locations were selected based on a diverse range of operating and storage conditions in order to obtain a diverse range of physical-chemical characteristics. Samples were collected from 11 households, 3 public toilets, 3 offices, 1 hotel, 1 tourist campsite, and 1 place of worship. For analysis, the type of establishment was grouped into “household”, “public toilet”, and “commercial”, where the “commercial” category contained the offices, hotel, campsite, and place of worship. The dataset included fully-lined containments (14) and partially-lined containments (4), with (8) and without (12) baffles, and with (11) and without (9) outflows. Toilets leading into the containments were a mix of cistern-flush (7), pour-flush (11), and dry (2) toilets.

Composite samples were collected from each containment. In Kampala, samples were collected directly from the containment using a multi-stage sampling device developed by Water for People Uganda (described in Semiyaga et al. (2017)). Equal volumes were taken from the bottom, middle, and top of the containment depth and thoroughly mixed in a bucket to create a composite sample, Figure S6.1. Five liters of the composite sample was collected in a plastic bottle and stored in a cooler with ice, and shipped directly to Eawag in Dübendorf, Switzerland in refrigerated storage. In Naivasha, each containment was fully emptied by a vacuum truck, and samples were collected from the vacuum truck during discharge at Naivasha Water and Sanitation Company's wastewater treatment plant. At the start of discharge, ¼ of the sample volume was collected, followed by ½ at the mid-point, and ¼ at the end, following the protocol described in Bassan et al. (2013), Figure S6.2. Samples were combined in a bucket and thoroughly mixed to create a composite sample. Five liters of the composite sample was collected in plastic bottles, and refrigerated at 4°C after collection. Samples were stored at 4°C until shipping to Switzerland in coolers with dry ice. In this text, field samples are labeled with their original sample IDs from the study, with "N" indicating that it was collected in Naivasha, Kenya, and "K" indicating that it was collected in Kampala, Uganda. Original sample IDs were included to facilitate easy reference to the published data package.

Questionnaires were administered on site during sample collection in Kampala, and during vacuum truck emptying in Naivasha. Respondent answers were logged using the KoBo Toolbox mobile application. Questionnaires included demographic information about the users, the type of establishment, types of waste streams entering the containment, technological details about the containment and the toilet, and the time since the containment was last emptied. The full questionnaire is included in Appendix C and raw response data are included in the published data package, which can be accessed at the following link: https://polybox.ethz.ch/index.php/s/rHfZJEgL7UAEjED (Link to data package will be replaced with a DOI linking to the open data repository before manuscript is published).

6.2.2 Sample analysis

Sample processing

Upon arrival at Eawag, samples were well mixed by vigorous shaking and stirring, and divided into 500 mL bottles for further analysis. For total solids (TS), volatile solids (VS), chemical oxygen demand (COD), soluble COD (sCOD), total organic carbon (TOC), total nitrogen (TN), ammonium nitrogen (NH_4^+-N), pH, and electrical conductivity (EC), samples were homogenized with an IKA Ultra-Turrax homogenizer at 25,000 rpm for 90 seconds. For analysis of particle and agglomerate characteristics, including total suspended solids (TSS), volatile suspended solids (VSS), particle size distribution, microscopic imaging, capillary suction time (CST), and supernatant turbidity after centrifugation, samples were homogenized by shaking and stirring, as to not disrupt the integrity of agglomerates and particles in the sample.

Sample characterization

Physical-chemical characteristics TS, VS, TSS, VSS, pH, EC, COD, sCOD, NH_4^+-N, TN, and TOC were performed according to standard methods for fecal sludge analysis (Velkushanova et al. 2021a). Samples were filtered through 0.45 µm syringe filters prior to analysis of NH_4^+-N and sCOD, and 0.45 µm glass fiber filters were used for TSS and VSS analysis. Particulate COD (pCOD) was calculated by subtracting sCOD from total COD. Stabilization indicators VSS/TSS, C/N, and pCOD/COD were used to quantify the level of stabilization in this study.

Particle size distribution was measured by static light scattering following Method 2560 D (APHA 2017) using a Beckman Coulter LS 13 320 Laser Diffraction Particle Size Analyzer with Universal Liquid Module. When dilutions were necessary to meet the range of the analyzer, they were performed immediately before adding the sample to the instrument and mixed gently to avoid disruption of flocs. Presented particle size distributions are on a volume-basis. Characteristics values were derived from volume based size distributions. These include the percentage of solids above or below specific particle diameters, and D10, D50, and D90, which are percentiles describing the particle size distribution: D50 is the median particle diameter, 10% of the sample is smaller than D10, and 90% of the sample is smaller than D90. D50 is referred to as the “median aggregate size” in the text.

Confocal laser scanning microscopy was used to image aggregates and particles. Samples were gently shaken and 3-5 drops were put on a glass slide using a Pasteur pipette. 9 – 11 images of each sample were captured at x5 magnification, bright field, and 2048 x 2048 pi resolution. Images were analyzed with Fiji (Schindelin et al. 2012). Image analysis was performed following the protocol outlined in Ferreira and Rasb (2012). Average particle area was defined as the area of the image occupied by particles (in μm^2) divided by the particle count. Major axis length was defined as the primary axis of the best fitting ellipse. Reported mean values were calculated for all images of the same sample. Determination of aggregate presence or absence was qualitative, using visual inspection of images. Aggregates were assessed by shape, i.e. they appeared to be aggregated smaller particles instead of larger single particles of sand or organic fibers, and by size, i.e. they were larger than ~50 µm. Samples with no discernable aggregates were also secondarily determined by particle size distribution as having more than 50% of solids composed of particles less than 50 µm in diameter.

Zeta potential was characterized to gain insight about mechanisms impacting aggregate formation and disintegration in fecal sludge. A Malvern Nano-ZS zetasizer was used. However, due to measurement errors generated when characterizing high conductivity samples, we were only able to obtain meaningful measurements for samples with EC < 3 mS/cm, which excluded about half of the field samples. Zeta potential results are not discussed further here, but are included in the data package.

Capillary suction time (CST) and supernatant turbidity following centrifugation were characterized as metrics of dewatering performance. CST was measured using a Triton 310 Multi-CST apparatus and an 18 mm funnel, following Method 2710 G (APHA 2017), as adapted in Velkushanova et al. (2021b). Measured CST values were standardized by subtracting the CST of deionized water. In this study, CST values are presented in seconds when comparing changes in dewatering performance for one sample, while CST values are normalized by TSS when comparing performance between field samples with different solids concentrations, as is recommended in the standard methods since CST is known to increase with increasing TSS (APHA 2017). Supernatant turbidity was measured following centrifugation at 3300 *x g* for 20 minutes, using HACH 2100N turbidity meter (Ward et al. 2021a).

Particle size manipulation

Field samples were manipulated to produce three fecal sludge samples with the same TSS and solution characteristics, but with different particle size distributions. Three field samples (N9, N5, and K4) were sieved sequentially through 2 mm and 100 µm sieves. Solids remaining on the 2 mm sieve were discarded. Solids remaining on the 100 µm sieve, and the sample passing through the 100 µm sieve, were collected separately. A portion of the sample that passed through the 100 µm sieve was filtered through a 0.45 µm syringe filter, and the filtrate was used to dilute the particles remaining on the 100 µm sieve and to dilute the original sample to the same TSS as the sieved sample. For verification, particle size distribution was quantified, and dewatering performance measured as CST.

Microbial community characterization

Samples were preserved for microbial analysis by centrifuging 2 mL aliquots at 6000 *x g* for 10 minutes, discarding the supernatant, resuspending the pellet in 1 mL RNAlater and storing at -20°C prior to DNA extraction. DNA was extracted using a modification of the method of Griffiths et al. (2000) described in (Sam et al. submitted).16S rRNA amplicon sequencing, data processing, OTU clustering and taxonomic annotation and further analysis of alpha and beta diversity was performed by Novogene.Co, Ltd. Analysis of similarities (Anosim) was used to evaluate whether variation in microbial community composition between groups was significantly larger than variation within groups (at 95% confidence level). Different methods for characterizing differential abundance between groups can produce quite disparate results, and it has been recommended as best practice to compare several methods and use a consensus approach (Nearing et al. 2022) To assess which taxa significantly differed between groups, three methods of differential abundance were used and the results compared: t-test, Metastat, and LefSe (linear discriminant analysis (LDA) effect size). Taxa are only presented as associated with group differences if all three methods agreed on the result.

6.2.3 Controlled anaerobic storage

Five field samples (N9, N10, K4, K5, K9) were used as inocula for controlled anaerobic storage experiments to evaluate changes with stabilization, following the setup developed and described in (Sam et al. submitted). Fresh excreta (feces and urine collected in urine diverting dry toilets at Eawag, combined with a feces to urine ratio of 2.5 by wet weight and diluted with tap water to 48 gTSS/L) was mixed with inoculum at a ratio of 1:4 (gVSS/mL basis). The excreta-inocula mixtures were added to four 200 mL serum bottles for each inoculum and incubated at 37°C in a shaking incubator for 7 weeks. Gas was released daily for the first 3 weeks using a needle inserted into the rubber septum, after gas production slowed in week 4, bottles were degassed every 2-3 days. Gas production had ceased for all bottles by 6 weeks, and tests were continued for a subsequent week with no additional gas production. Samples were taken weekly for physical-chemical characterization, particle size, microscopy, and dewatering performance. Microbial community was characterized for samples taken at time 0, week 3, and week 7. After sampling, bottles were bubbled with N_2 for 30 seconds to ensure anaerobic conditions were maintained.

6.3 Results and Discussion

6.3.1 Characteristics of field samples

Table 6.1 summarizes physical-chemical characteristics of field samples collected in Kampala, Uganda and Naivasha, Kenya, grouped by the type of establishment from which they were collected and compared with literature values for fecal sludge and wastewater sludges. TS and TSS concentrations were comparable between cities and establishment types, except for samples from public toilets, which had an order of magnitude higher solids concentration. The samples in this study were on the dilute end of the range of reported literature values for fecal sludge. This corresponds to COD concentrations on the low end of published values as well, although EC and NH_4^+-N were within previously reported ranges. EC and NH_4^+-N were highest in public toilet sludge, which was also observed by (Ward et al. 2019), most likely due to high urine fraction. pCOD/COD was comparable with measured values for fecal sludge, but generally lower than typical values for wastewater sludges. Public toilet sludge and commercial sludge from Naivasha had especially high sCOD fractions. C/N is consistent with reported values for fecal sludge, but very low compared to values for wastewater sludges. The particle size distribution in fecal sludge samples was much broader than in reported values for wastewater sludges, with higher proportions of very large (>300 µm) and very small (<10 µm) particles. Samples from Naivasha had larger median aggregate sizes (D50) comparable to activated sludge, and samples from Kampala had smaller median aggregate sizes similar to primary or anaerobically digested sludges and blackwater (Hocaoglu and Orhon 2013).

Table 6.1. Metrics of dewatering performance and physical-chemical characteristics of field samples, categorized by city of origin and type of establishment. D10, D50, and D90 are percentiles describing the particle size distribution on a volume basis: D50 is the median particle diameter, 10% of the sample is smaller than D10, and 90% of the sample is smaller than D90. Literature values are a range of values for published fecal sludge and wastewater sludge characteristics. Literature values are medians unless marked otherwise. [a](Ward et al. 2021d), [b] (Gold et al. 2018a), [c] (Ward et al. 2019), [d] (Sam et al. submitted), [e] (Sakaveli et al. 2021), [f] (Houghton and Stephenson 2002), [g] (Turovskiy and Mathai 2006), [h] (Houghton et al. 2001), [i] (Ahnert et al. 2021), [j] (Tchobanoglous et al. 2014), [k] (Siddiqui et al. 2011), [l] (Jin et al. 2004), [m] (Rasmussen et al. 1994), [n] (Zhang et al. 2016), [o] (Shao et al. 2010), [p] (Sanin and Vesilind 1994), [q] (Jørgensen et al. 2017), [r] (Yang et al. 2019), [s] (Neyens et al. 2004).

				Metrics of dewatering performance			Physical-chemical characteristics												
				CST	normalized CST	Supernatant turbidity	TS	TSS	VSS/TSS	EC	pH	NH_4^+-N	COD	pCOD/COD	C/N	D10	D50	D90	Time since last emptied
City	Establishment	n		(s)	(sL/gTSS)	(NTU)	(g/L)	(g/L)		(mS/cm)		(mg/L)	(g/L)			(µm)	(µm)	(µm)	(weeks)
Kampala, Uganda	Household	4	*mean*	114	17	160	6.3	4.0	0.79	7.0	7.4	267	10.8	0.94	1.2	6	41	419	42
			median	**44**	**16**	**130**	**4.2**	**2.2**	**0.82**	**5.9**	**7.4**	**190**	**9.1**	**0.93**	**1.1**	**5**	**40**	**441**	**4**
			std	171	13	160	6.3	5.1	0.08	5.0	0.2	209	4.5	0.04	0.7	2	17	170	76
	Public toilet	3	*mean*	205	10	270	25.4	24.8	0.78	9.1	7.3	566	30.7	0.76	3.2	8	50	596	6
			median	**136**	**8**	**230**	**19.3**	**19.9**	**0.78**	**9.0**	**7.3**	**582**	**23.0**	**0.78**	**3.0**	**9**	**50**	**602**	**6**
			std	144	8	140	15.1	14.6	0.08	0.6	0.2	189	18.5	0.12	1.2	3	12	12	4
	Commercial	3	*mean*	9	8	29	3.1	1.7	0.78	4.9	7.4	344	4.1	0.94	1.1	10	63	677	100
			median	**10**	**5**	**37**	**2.8**	**2.2**	**0.83**	**2.9**	**7.3**	**217**	**4.9**	**0.95**	**0.9**	**10**	**64**	**509**	**32**
			std	6	5	16	2.0	1.3	0.15	3.4	0.4	245	1.6	0.03	0.8	0	8	333	139
Naivasha, Kenya	Household	7	*mean*	11	5	60	10.8	9.6	0.65	5.2	7.8	216	9.5	0.86	3.7	13	90	503	14
			median	**6**	**2**	**14**	**3.3**	**2.2**	**0.64**	**2.6**	**7.8**	**132**	**3.5**	**0.91**	**2.0**	**13**	**86**	**538**	**12**
			std	12	5	80	11.6	11.5	0.10	4.3	0.3	269	10.2	0.16	4.1	4	38	259	18
	Commercial	3	*mean*	4	4	30	3.6	2.0	0.74	3.9	7.1	299	2.5	0.75	10.4	16	146	725	5
			median	**4**	**5**	**28**	**4.2**	**1.0**	**0.76**	**1.4**	**7.2**	**43**	**3.0**	**0.71**	**1.0**	**14**	**184**	**886**	**1**
			std	1	2	26	1.9	2.2	0.18	5.2	1.1	478	1.5	0.11	16.5	5	73	383	6
Literature values																			
Fecal sludge				42-468 [a]	9-56 [b+]	100-650 [a]	11 - 148 [b]	4.9-8.5 [c]	0.62-0.85 [c]	1.7-14.5 [a,b]	7.1-7.8 [b]	178-3110 [b]	12.6-127.2 [b]	0.67-0.94* [d]	0.8-26* [a]	3* [d]	51* [d]	549* [d]	-
Wastewater primary sludge				61* [e]	2.1* [f]	-	20-70* [g]	14-46* [h]	0.76 [i]	2.6-3.1* [b]	5-8* [j]	132* [e]	48.6* [e]	0.95 [i]	11.6-13.6* [e,k]	7* [f]	27* [f]	233* [f]	-
Wastewater activated sludge				13-20* [l]	1.6 -2.3* [n,o]	60* [p]	4-15* [g]	1.54-13.7* [h]	0.59-0.84* [l]	0.6-0.9* [m]	6.5-8* [j]	0.3-53.9* [q]	0.165-11.3* [q]	0.98 [k]	5.4 - 10* [k,r]	33* [s]	107* [s]	169* [s]	-
Wastewater anaerobically digested sludge				26* [e]	7.1-11.1* [f,o]	-	30-60* [g]	15.6-34.1* [h]	0.68 [i]	1.6-10* [e,m]	7.9* [e]	490* [e]	20.1* [e]	0.97* [i]	10.0* [e]	10* [d]	50* [d]	145* [d]	-

* mean values
[+] units in sL/gTS

Samples from Naivasha generally had lower CST (s) and supernatant turbidity compared with samples from Kampala. CST and supernatant turbidity from public toilets in Kampala was significantly higher than for any other group, which was also observed in Dakar, Senegal (Ward et al. 2019). This appears to be explained in Kampala by the order of magnitude higher TSS in public toilet samples, as normalizing CST by TSS (which is conventional in comparing between field samples (APHA 2017)) effectively canceled out differences between public toilets and other establishments. Normalized CST (sL/g TSS) measured in this study was on the low end for fecal sludge reported in the literature. Samples from Naivasha had lower normalized CST than reported in other studies – comparable to primary or activated wastewater sludges, while normalized CST values from Kampala samples were higher, and comparable to fecal sludge from other cities and anaerobically digested wastewater sludge.

Figure 6.1 summarizes the dewatering performance (normalized CST) of fecal sludge based on city, type of establishment, time since last emptied, and containment type (baffles, outflow, lined/unlined). When CST was normalized by TSS concentration, differences in dewatering performance between containment type and establishment type were not detectable. However, dewatering performance was still better in Naivasha compared with Kampala. It is notable that dewatering performance did not improve with increased time between emptying events, which was unexpected based on existing literature and practitioner experience. Longer intervals between emptying events are thought to correspond with improved stabilization of the sludge, yielding more complete particle hydrolysis and an improvement in dewatering (Lopez Vazquez et al. 2021). The observations in this study could be associated with the relatively short timescales between emptying events. The majority of containments (60%) were emptied every 3 months or more often, and 50% were emptied every 2 months or more often. 90% were emptied more than once per year. The frequent emptying and inconsistent technical features observed in this study do not correspond with expectations based on design guidelines for septic tanks and pit latrines, which are intended to be emptied every few years ((Lopez Vazquez et al. 2021). Hence, containment type is described here using technical details instead of technology name (e.g. “septic tank”, “pit latrine”), as the majority of containments did not conform to common technical descriptions (e.g. 29% of “septic tanks” had no outflow, 50% had no baffles, 17% of “pit latrines” had an outflow, baffles, and were fully lined, and 67% were connected to cistern flush toilets). This supports the push within the global sanitation sector to move away from classifying containments using uninformative designations of “septic tank” and “pit latrine” (Isunju et al. 2013, Strande et al. 2021a). Our findings are consistent with reported technical characteristics and emptying frequencies from other cities in sub-Saharan Africa and Asia, and we posit that our observations are a reflection of the actual sanitation situation in many cities (Koottatep et al. 2012, Nakagiri et al. 2015, Strande et al. 2018, Ward et al. 2021a).

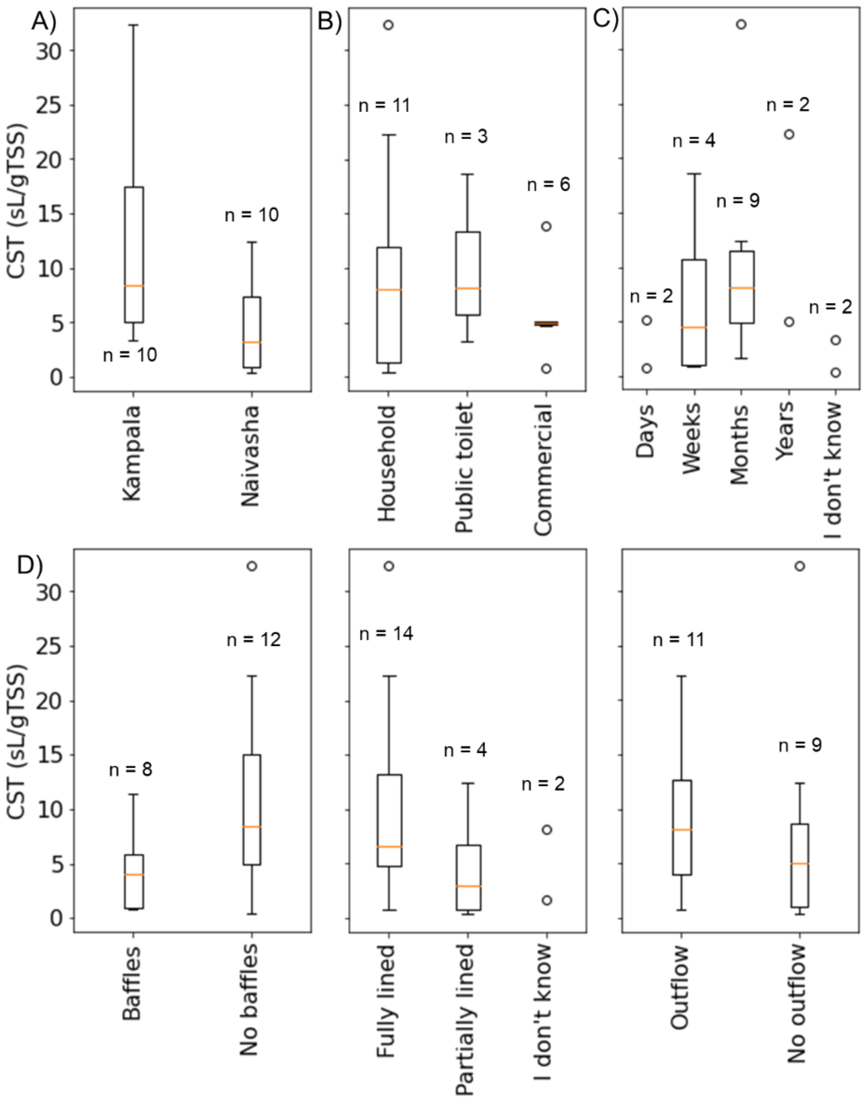


Figure 6.1. Dewatering performance characterized by CST normalized by TSS, grouped by (A) city, (B) establishment type, (C) time interval since last emptied, and D) containment technology. For categories where n=2, data points are provided instead of boxplots.

Illustrated in Figure 6.2, the dewatering performance metrics considered in this study, CST and supernatant turbidity, were linearly related. A linear relationship between these metrics was also observed in fecal sludge from Lusaka, Zambia (Ward et al. 2021a). Because of the strong correlation, dewatering performance is discussed in terms of CST and normalized CST in this paper. Supernatant turbidity is reported in the open data package.

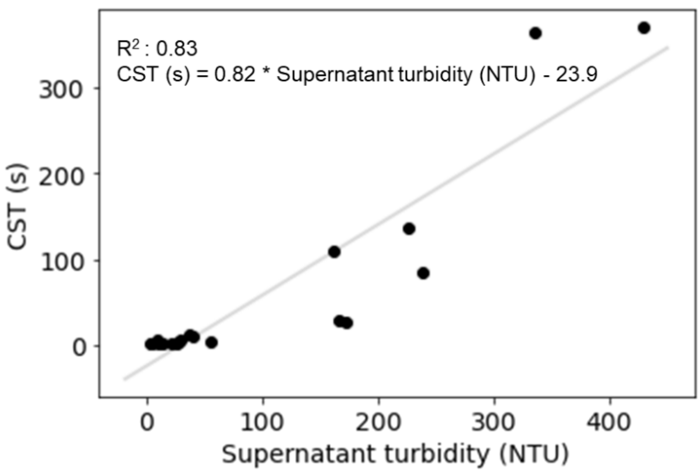


Figure 6.2. Scatterplot showing relationship between metrics of dewatering performance CST (s) and supernatant turbidity (NTU).

6.3.2 Particle size and dewatering performance

Field Samples

Illustrated in Figure 6.3, the percent of solids composed of macrocolloidal particles (1-10 µm), true colloidal particles (0.45-1 µm), and settleable particles (>100 µm) were all associated with dewatering performance in field samples. Fecal sludge with higher fractions of small particles < 10 µm had correspondingly reduced dewatering performance (higher normalized CST), whereas sludge with higher fractions of aggregates > 100 µm dewatered better. These observations are consistent with wastewater literature. Rasmussen et al. (1994) observed that the particle fraction from 0.45 – 10 µm was most associated with poor filtration and high supernatant turbidity after anaerobic storage of activated sludge. Karr and Keinath (1978) saw a relationship between dewatering performance and supracolloidal particle fraction (1-100 µm) in primary, activated, and anaerobically digested wastewater sludges, but did not measure macrocolloidal particles separately. The absence of clear relationship between the particle fraction 10-100 µm and dewatering performance (shown in Figure 3A) indicates that the smallest particles (macrocolloidal and true colloidal, <10 µm) are most influential in governing the dewatering performance of the fecal sludge field samples. This could explain why previous studies of fecal sludge did not observe strong relationships between particle size and dewatering, as they were specifically looking for associations between CST and the supracolloidal particle fraction (1-100 µm) (Gold et al. 2018a, Sam et al. submitted). The importance of macrocolloidal and true colloidal particles explains the trend illustrated in Figure 3B, where samples with the largest aggregates (indicated by D50) have the best dewatering performance, and Figure 3A where samples with the highest fraction of aggregates >100 µm have the best dewatering performance. It is logical to expect that samples with a high fraction of particles consolidated within large aggregates would have fewer suspended small particles to contribute to clogging.

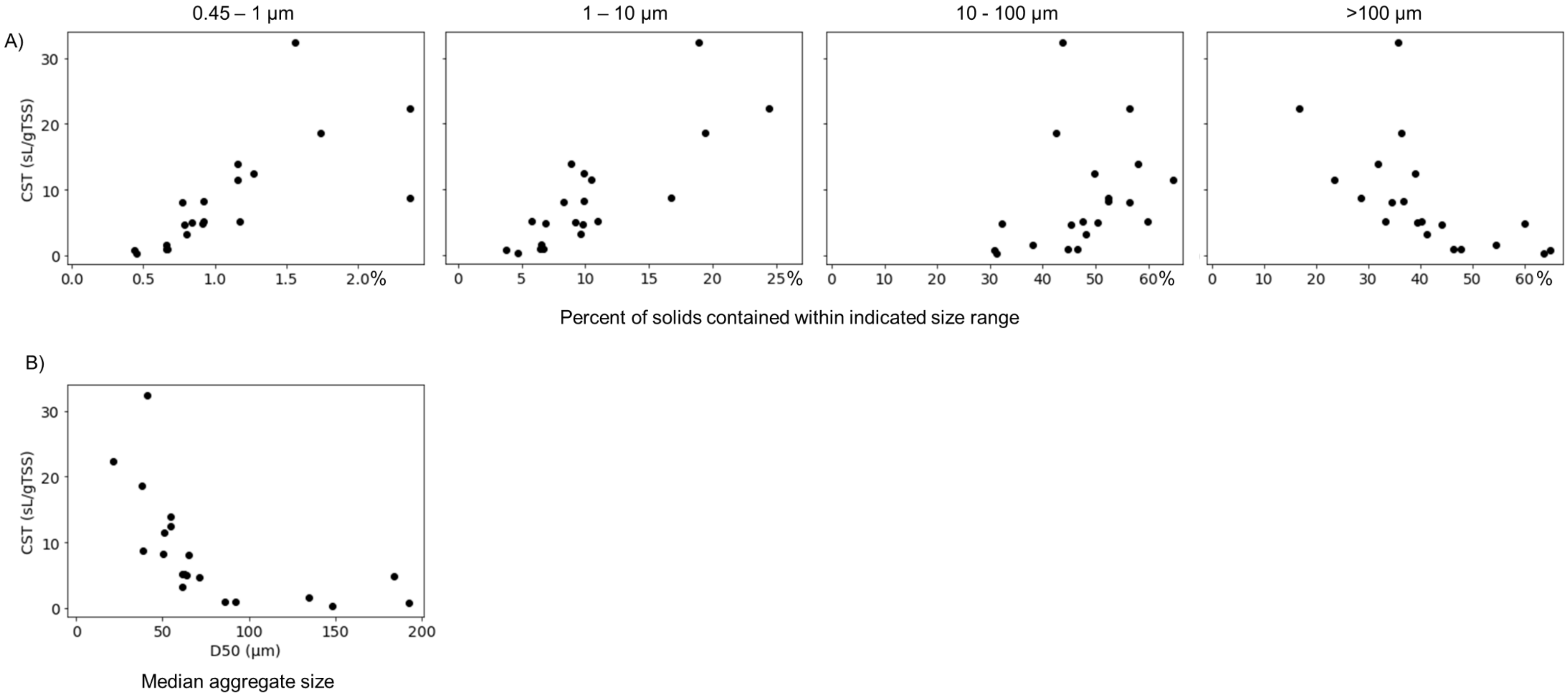


Figure 6.3. (A) Scatterplots showing relationships between normalized CST and characteristics values derived from volume based size distributions. The x-axis of each figure represents the percent of solids contained within the indicated size range (0.45-1 µm, 1-10 µm, 10-100 µm, and >100 µm). (B) Scatterplot showing the relationship between normalized CST and median aggregate size (D50)

Particle size manipulation

As illustrated in Figure 6.4, manipulating the particle size distribution of field samples verified that particle size distribution plays a role in governing fecal sludge dewatering performance. Three field samples representing a range of dewatering performances: “high” normalized CST (27 sL/gTSS), “moderate” (5 sL/gTSS), and “low” (2 sL/gTSS), were used to create a series of samples with distinct particle size distributions, but the same TSS, as described in the Methods Section. For every field sample, the measured dewatering performance was directly associated with the proportion of small particles. The “small” distribution samples had a correspondingly worse dewatering performance (higher normalized CST) than the original, while the “large” distribution samples dewatered better. The only exception to this was the low CST sample (bottom of Figure 6.4), where the large fraction dewatering performance is not significantly different from the original sample. This can be seen on the corresponding particle size distribution figure and is explained by the similar particle size distributions in the two samples.

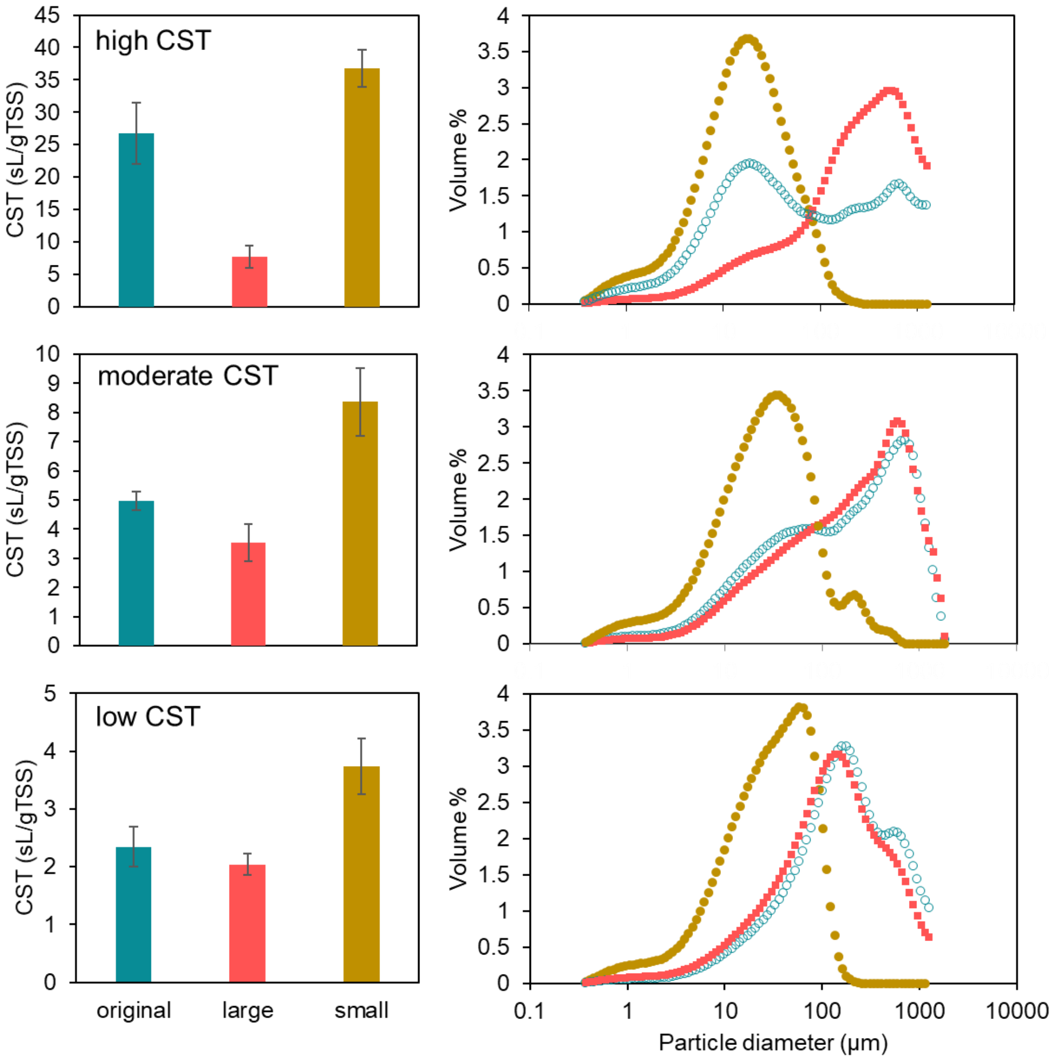


Figure 6.4. Left: Normalized CST measurements for three field samples (high CST, moderate CST, low CST) and for the same samples with manipulated particle size distributions. Right: Measured particle size distribution (volume basis). Original field samples shown in green (open circles), “large” samples composed of particles remaining on a 100 µm sieve shown in red (squares), “small” samples composed of particles that passed through a 100 µm sieve shown in gold (filled circles).

6.3.3 Aggregates and small particles in fecal sludge

In situ processes during onsite storage could influence the concentration of small particles in fecal sludge. These include incorporation of small particles into larger aggregates, degradation of aggregates or large primary particles into small particles, and destruction of small particles as part of the stabilization process. In field samples, we can see evidence of aggregates and particle destruction, and look at how that relates to reported residence time in containment.

Aggregates were visible in stereomicroscopic images of a majority of field samples (examples in Figure 6.5A, complete set of images in open data package). Illustrated in Figure 6.5B, samples with and without microscopically evident aggregation showed different relationships between CST and TSS. It makes sense that dewatering performance would be worse at lower TSS concentrations if a sample contained mostly suspended small particles, compared with mostly aggregated particles. Unexpectedly, the presence or absence of aggregates was not associated with time since the containment was emptied. Samples with no aggregates were taken from containments emptied every few days to every few years (Figure S6.3).

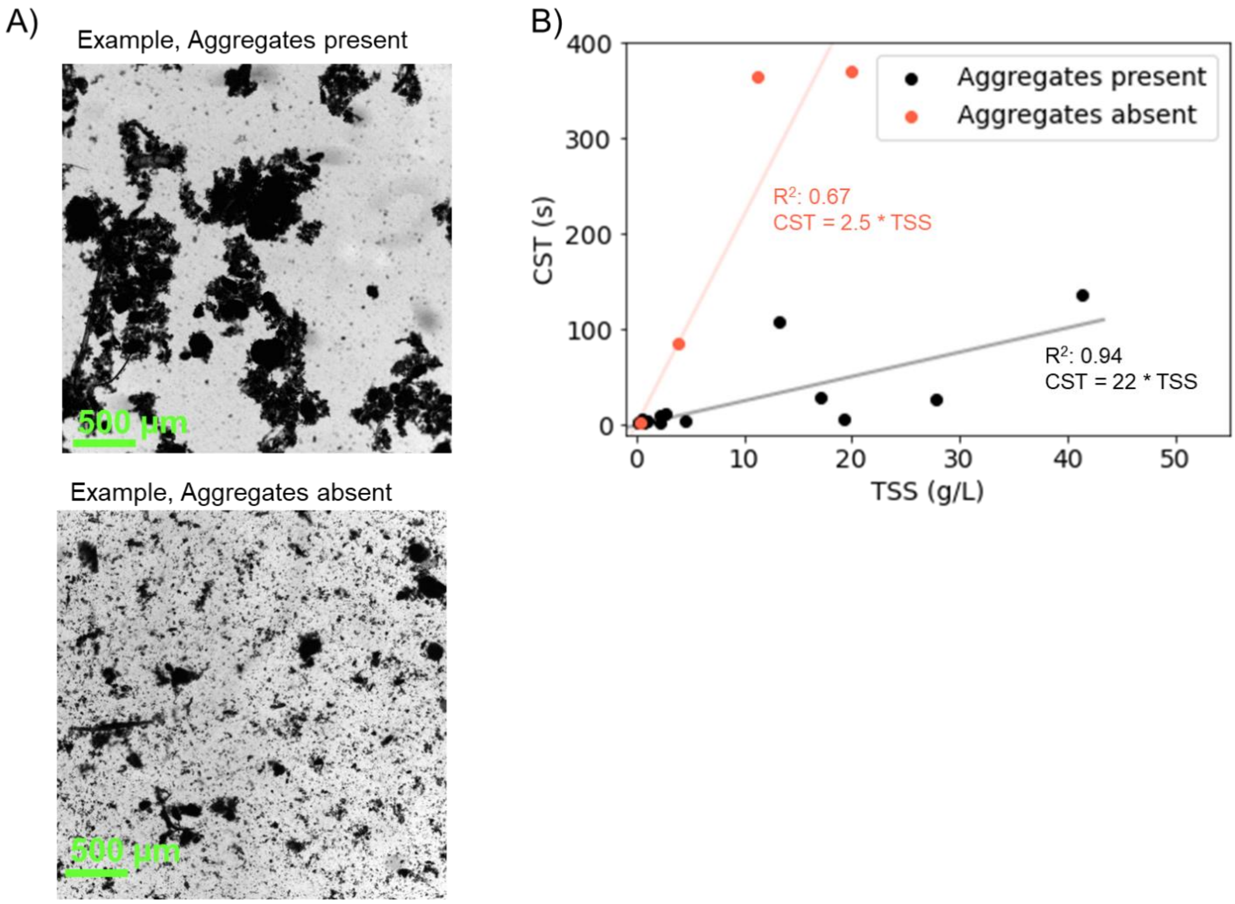


Figure 6.5. (A) Example stereomicroscope images of field samples with aggregates (top) and without aggregates (bottom). (B) Scatterplot of CST (s) vs TSS (g/L) for field samples with evident particle aggregation (black) and samples where aggregation is not evident in microscopic images (red).

Microbial community analysis of field samples may provide insight into possible links between aggregate properties and microorganisms in fecal sludge. Figure 6.6 shows phylum and genus-level community compositions for every field sample in this study, ordered from highest to lowest aggregate size. Additionally, we divided field samples into groups by aggregate size: largest aggregates (top 25% of measured D50), medium aggregates (middle 50%), and smallest aggregates (bottom 25%). There was a higher variation between the largest and smallest aggregate groups than within the groups, based on an analysis of similarities (anosim) evaluation (Table S6.1). Samples with the largest aggregates had higher relative abundance of the Gammaproteobacteria genus Pseudomonas, and samples with the smallest aggregates had higher relative abundance of Bacteroidetes genus Vadin HA17 and family Rikenellaceae (Figures S6.4-S6.6). These specific genera have been associated previously with differences in floc properties and solid-liquid separation in fecal sludge and wastewater. The genus Pseudomonas has been associated with faster dewatering and less turbid supernatant in fecal sludge field samples from Dakar, Senegal and Dar es Salaam, Tanzania (Ward et al., 2019). In high strength wastewater, species of Pseudomonas have been shown to promote bioflocculation, and to form especially strong aggregates under high NH_4^+-N conditions (Yang et al. 2021). Conversely, Vadin HA17 was associated with increased incidences of biofouling in anaerobic membrane bioreactors (Liu et al. 2020). Vadin HA17 genus members are fermentative proteolytic amino acid degraders (Mei et al. 2020), so if aggregates in fecal sludge are bound together by extracellular polymers, it is possible that protein hydrolysis is driving aggregate disintegration in these samples. Also noteworthy is the high abundance of Pseudomonas, a facultative aerobe, and Lactobacillus, a microaerophilic fermenter, in the field samples with the largest aggregate sizes. This suggests that microaerobic conditions could be present in some onsite containments. Based on microbial community differences between field samples, it is possible that specific onsite storage conditions, (e.g., redox conditions), may promote microbial populations that facilitate improved aggregate formation or degrade aggregates through hydrolysis of biopolymers, impacting dewatering performance when fecal sludge is delivered to treatment facilities.

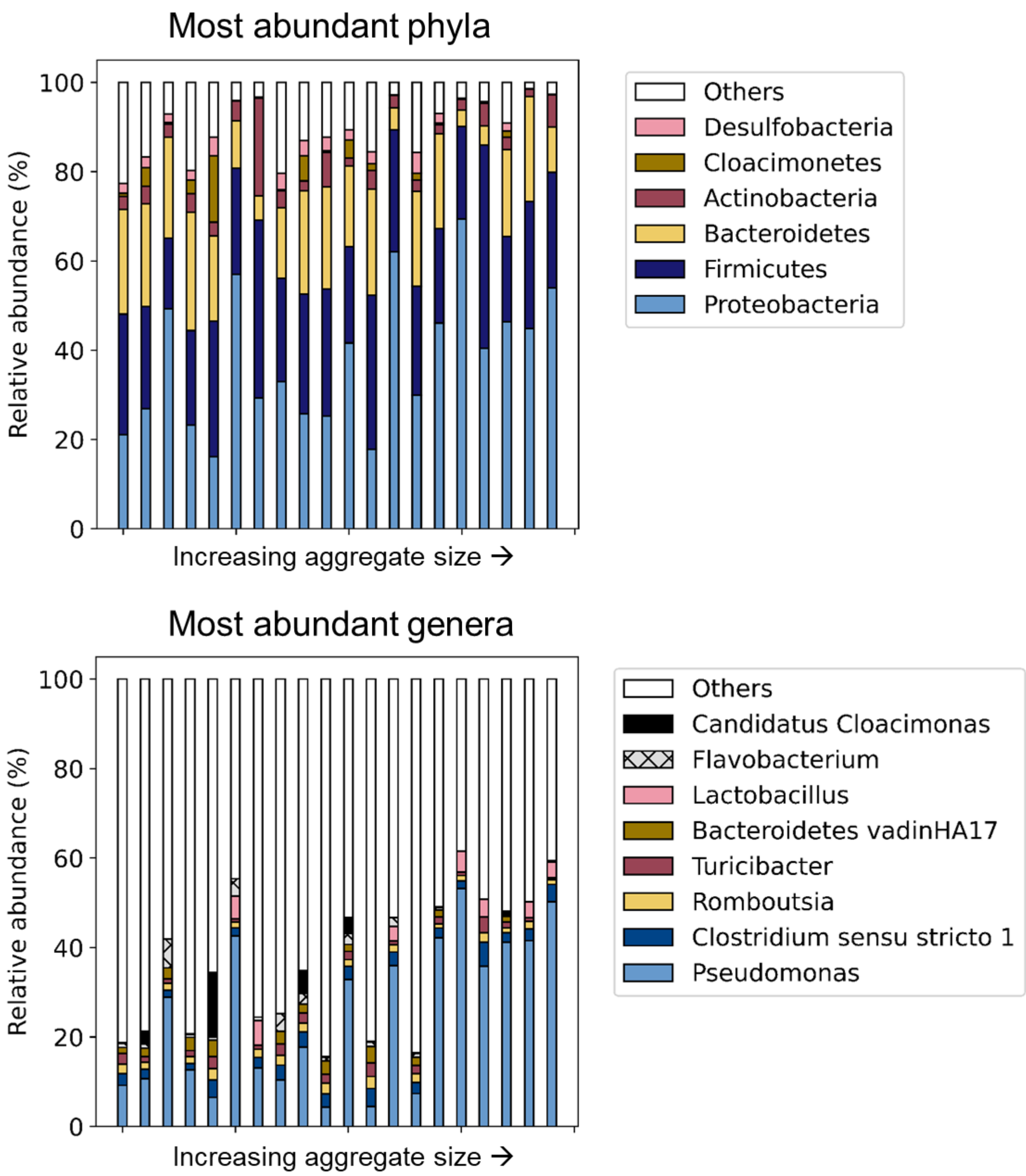


Figure 6.6. Relative abundance of most abundant phyla (top) and genera (bottom) for every fecal sludge field sample, ordered by increasing aggregate size (D50).

Indicators of stabilization (pCOD/COD, C/N, VSS/TSS) in the field samples indicate varying degrees of breakdown of organics. Illustrated in Figure 6.7, samples with the worst dewatering performance (highest normalized CST) and highest proportion of particles smaller than 10 µm have high pCOD/COD (>0.85), low C/N (<5), and high VSS/TSS (>0.8). This is interesting, because we expected high pCOD/COD to be indicative of more stabilized sludge, as it is a measure of whether soluble organics have been digested, while low C/N and high VSS/TSS are characteristic of less stabilized sludge (Tchobanoglous et al. 2014). Aggregate size (D50) does not show consistent relationships with indicators of stabilization. Based on our original hypothesis, we expected to see a higher proportion of small particles present in samples that had not been fully stabilized (i.e. where particle hydrolysis was still ongoing). We would have expected a higher proportion of small particles to correspond not only with high VSS/TSS and low C/N (which we observed), but also with low pCOD/COD (which we did not observe).

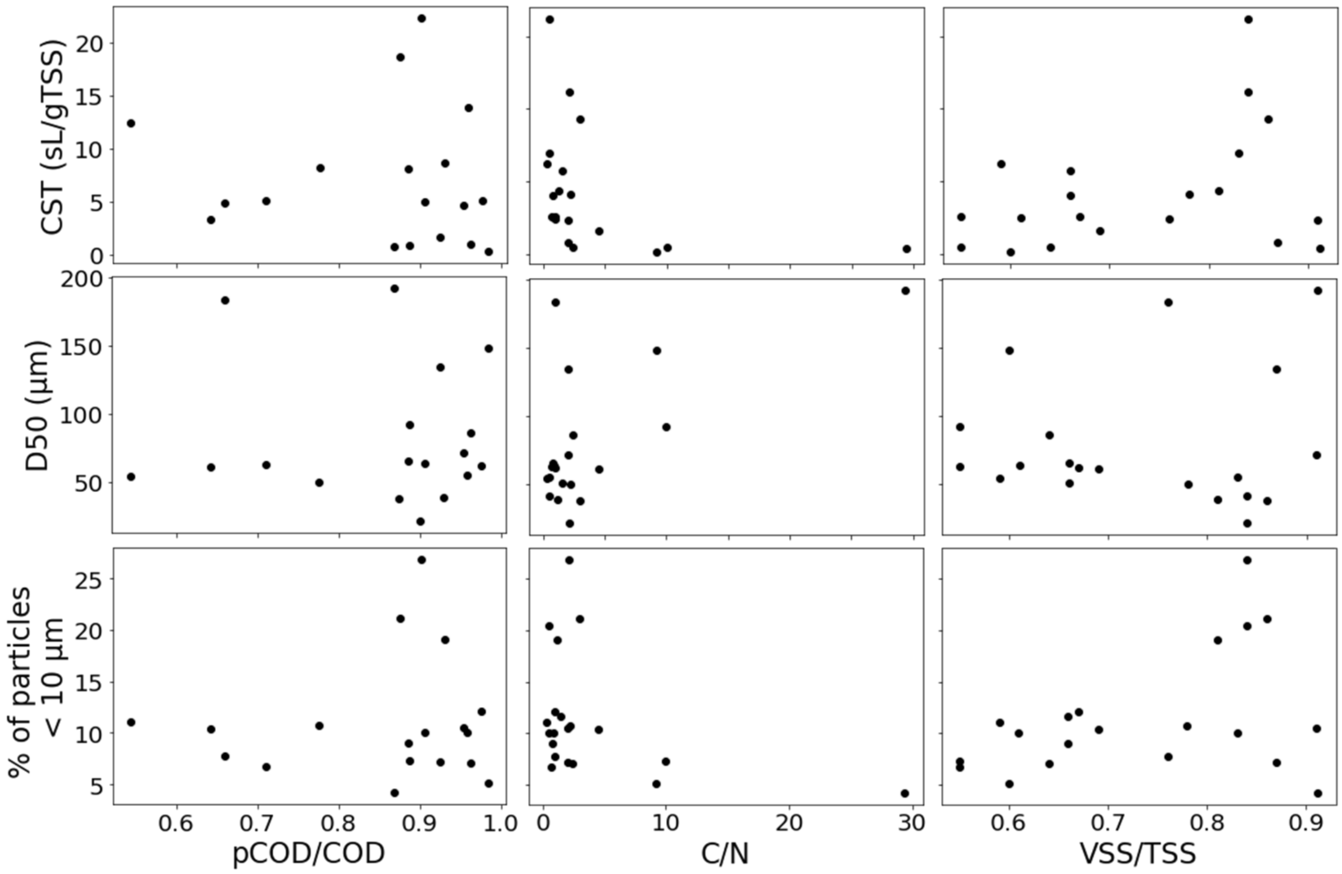


Figure 6.7. Scatterplots illustrating the relationships between normalized CST, median aggregate size (D50), the percent of particles below 10 µm, and stabilization indicators pCOD/COD, C/N, and VSS/TSS in fecal sludge field samples.

6.3.4 Particle size, stabilization, and time in containment

We saw an association between metrics of stabilization and particle properties that affect dewatering (Figure 6.7), but did not observe a corresponding improvement in dewatering performance based on time between emptying events (Figure 6.1). Figure 6.8A shows particle and aggregate characteristics broken down by time since last emptied, and offers an explanation of CST results. There were not observable differences in aggregate size or the proportion of small particles based on time since last emptied, and there were also no differences in stabilization indicators C:N (Figure 6.8B), pCOD/COD or VSS/TSS (Figure S6.13) based on time. These observations are supported by results from (Ward et al. 2021d), who also saw no relationship between time between emptying and C/N for fecal sludge in

Lusaka, Zambia. These results suggest that time in containment is not a relevant factor in determining stabilization and dewatering performance in fecal sludge.

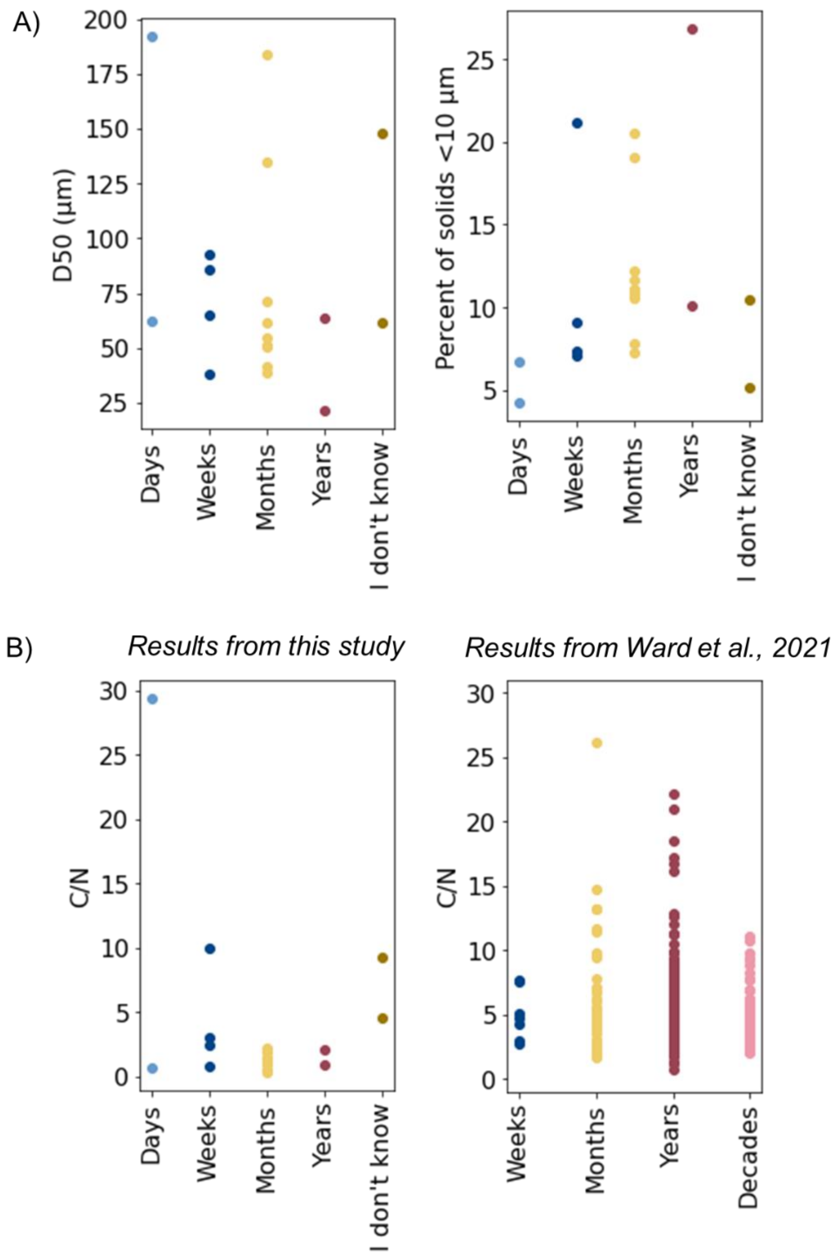


Figure 6.8. (A) Plots showing the distribution of particle size characteristics on a volume basis (D50 and percent of particles less than 10 µm) broken down by the time since the containment was last emptied. (B) Plots showing stabilization metric C/N for this study broken down by time since last emptied, and compared against data from samples taken from containments in Lusaka, Zambia ((Ward et al. 2021d)).

6.3.5 Controlled anaerobic storage experiments

In order to further investigate the relationship between storage time, stabilization, and particle size distribution, we conducted controlled anaerobic storage experiments in the laboratory. Five field samples (K4, K5, K9, N9, N10) were mixed with fresh excreta and stored under anaerobic conditions for 7 weeks.

Stabilization indicators and dewatering performance did not change as we had hypothesized over the course of controlled anaerobic storage. We were not able to replicate the characteristics of “stabilized” field samples, and there was very little or no consistent change in stabilization indicators from time 0 to week 7 (Figure S6.14). VSS/TSS reduced by an average of 14% (ranged from 7-23% reduction), which is lower than expected based on anaerobic digestion of wastewater sludge (50-65% VS reduction) (Tchobanoglous et al. 2014), but comparable to results achieved with fecal sludge in an identical experimental setup (20% VS reduction) (Sam et al. submitted). C/N ranged from 40% increase to 50% decrease. pCOD/COD essentially did not change after 7 weeks, except for K5, which increased by 13%. CST, supernatant turbidity, and particle size distribution changed over the 7 week storage period (Figure 6.9). In reactors, as with field samples, we saw indications that particle size distribution (especially higher fractions of small particles <10 µm) is driving changes in dewatering performance. CST did not decrease consistently over the 7 week residence time as we expected, instead it decreased, then increased again around week 4, corresponding to an increase in supernatant turbidity. These changes in dewatering performance were mirrored by differences in aggregate and particle properties, with the fraction of small particles (<10 µm) reducing for the first 3 or 4 weeks, then increasing.

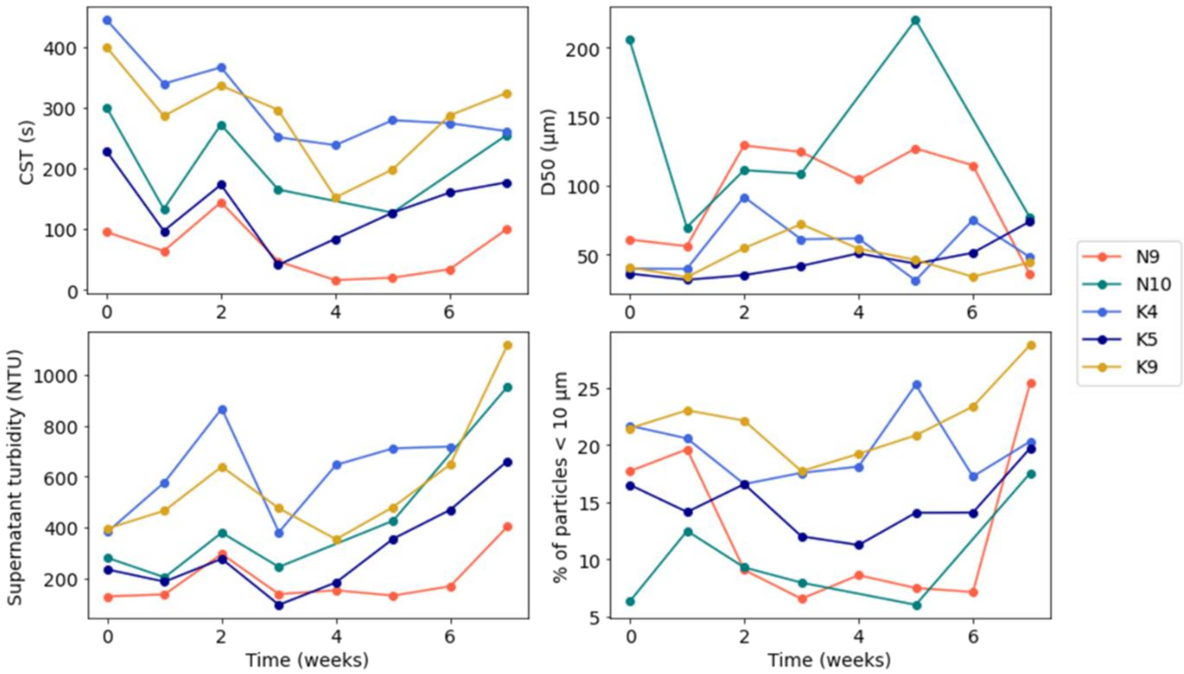


Figure 6.9. Changes in solid-liquid separation performance (CST (s), supernatant turbidity) and particle size distribution (median particle diameter (D50), percent of particles less than 10 µm) over the course of 7 weeks of anaerobic storage.

Illustrated in Figure 6.10, quantitative analysis of microscopy images indicated that the average size of aggregates (on a number basis) is decreasing over the course of anaerobic storage. In some samples (e.g. K4, K9), the aggregate size appears to stay the same or slightly increase between time 0 and week 3, which suggests along with the results presented in Figure 10, that small particles are being destroyed during the first 3 weeks of anaerobic storage, but in the following weeks, larger aggregates are deteriorating and releasing small particles into suspension. There was not any evidence of aggregate formation during controlled anaerobic stabilization experiments. The initial reduction and subsequent release of small particles into the bulk during anaerobic digestion and the corresponding changes in dewatering performance have been reported in literature for primary wastewater sludge (Mahmoud et al. 2006) and a mix of primary and activated sludge (Lawler et al. 1986), and was attributed to the presence of stressful or inhibitory conditions, which kept the released small particles from being further degraded, resulting in the accumulation of small particles and poor dewatering. It is possible that nutrient limitations (Colón et al. 2015), high antibiotic contamination (Bischel et al. 2015), high concentrations of recalcitrant organic matter (Krueger et al. 2021), or other inhibiting factors could be impeding the complete anaerobic stabilization of fecal sludge.

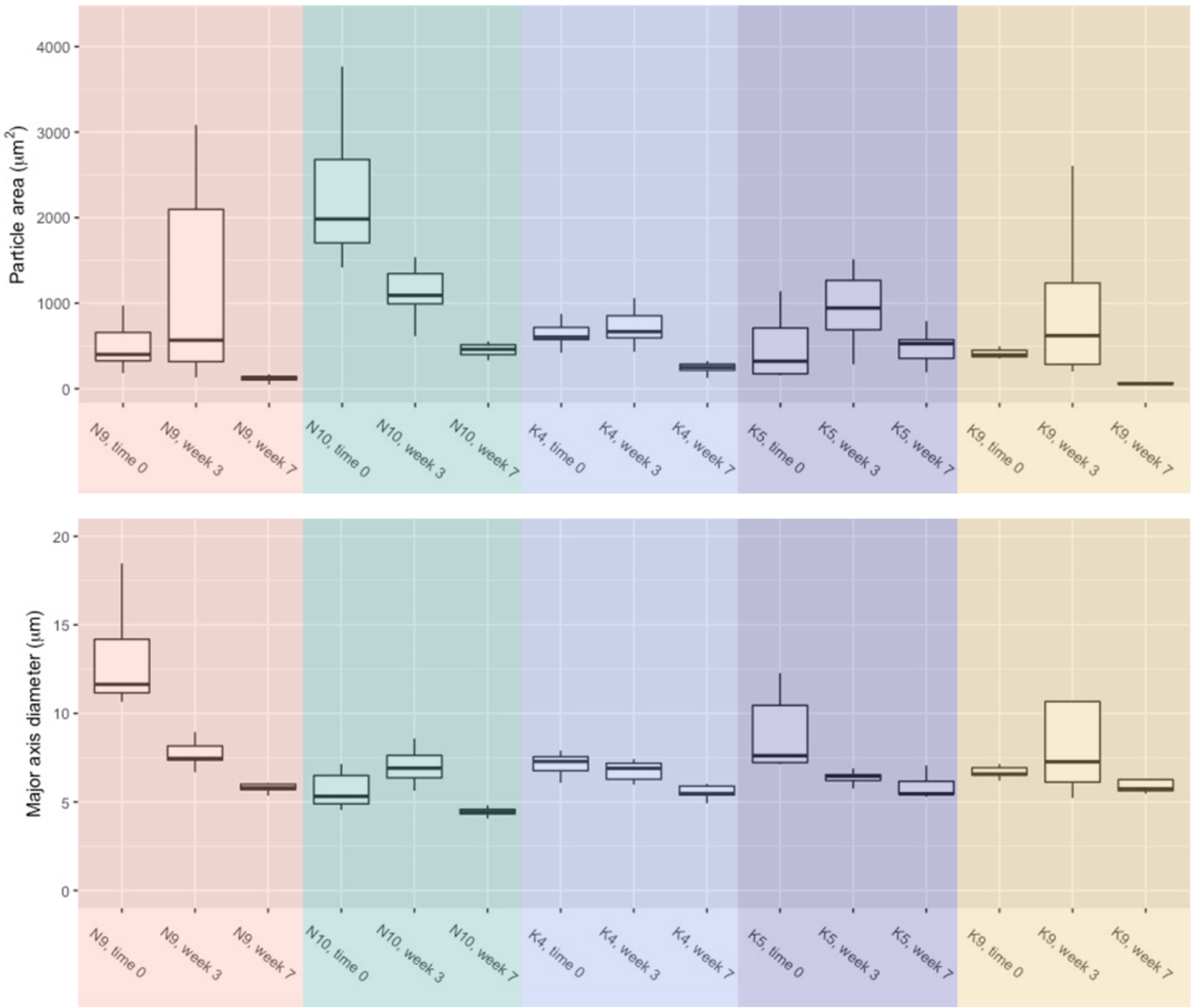


Figure 6.10. Boxplots illustrating results of quantitative analysis of microscopy images of samples taken during controlled stabilization. Particle area is the average area of a single particle or aggregate (total area taken up by particles/particle count), and major axis diameter is the average length of the primary axis of the best fitting ellipse. Colors correspond to each inoculum.

Although the extent of change in stabilization was less than expected, and particle size distribution did not change in the way we hypothesized, anaerobic digestion processes were taking place during this period. Gas was produced by all samples during the first month, and eventually stopped at or before week 6. Microbial community shifted over the course of anaerobic storage. Firmicutes displaced Proteobacteria in every reactor (Figure S6.16), which is a commonly observed transition in anaerobic digesters (Alalawy et al. 2021, Díaz et al. 2018). Differential abundance analysis between time 0 and week 7 showed higher abundances of strictly anaerobic fermenter genus Clostridium sensu stricto after 7 weeks of anaerobic storage (Table S6.1 and Figures S6.10-S6.12).

Notably, although changes with time in anaerobic storage were not what we expected, indicators of stabilization were still related to metrics of dewatering performance and particle size distribution. Shown in Figure 6.11, for samples taken once a week over the course of anaerobic storage, the relationships we had hypothesized were observed, with dewatering performance generally improving with level of stabilization (as indicated by high pCOD/COD, high C/N, and low VSS/TSS). Relationships between stabilization indicators and particle size distribution are in Figure S6.15. These results support the currently accepted understanding that fecal sludge dewatering and particle size distribution are associated with level of stabilization, but gaining a deeper understand the how small particles and aggregates are destroyed or created during the stabilization process will likely also require an understanding of the interactions and transformations of soluble and particulate organic matter (e.g. EPS, recalcitrant organic matter from excreta, and surfactants from greywater (Cecconet et al. 2019, Krueger et al. 2021, Sam et al. submitted)).

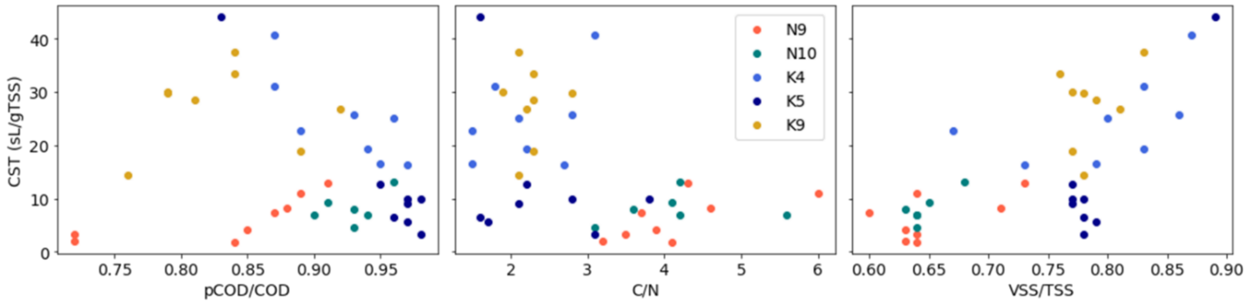


Figure 6.11. Scatterplots illustrating the relationships between stabilization indicators and normalized CST (sL/gTSS) for reactors inoculated with five field samples (N9, N10, K4, K5, K9) over the entire course of anaerobic storage.

6.3.6 Implications for researchers and practitioners

In field samples, and in controlled anaerobic storage experiments, dewatering performance and particle properties were linked to stabilization, but not to time in storage. These results question currently accepted understandings within the fecal sludge management community that stabilization and dewaterability continually improve with longer residence times in containment. This could be expected, as conditions in onsite containments are not analogous to highly controlled anaerobic digestion occurring at wastewater treatment facilities, and containments are continuously fed with fresh inputs (e.g. excreta). Therefore, practitioners should reconsider the common perception that fecal sludge undergoes steady anaerobic degradation in containment becoming fully stabilized after some amount of time, and in the design of treatment technologies consider that fecal sludge will still contain readily biodegradable organic matter even following longer-term storage.

Particle size as a driver of dewatering performance

The links between aggregate size and microaerophilic bacteria in field samples could indicate that the conditions producing aggregated fecal sludge during storage may not be entirely anaerobic. In order to understand what conditions facilitate or inhibit in situ stabilization processes in onsite containments research should focus on characterizing substrate availability, redox conditions, availability of macro- and micronutrients, and inhibitors (e.g. high concentrations of ammonium). This could lead to a better understanding of subsequent dewatering performance at treatment facilities

Usage practices that influence particle size distribution in fecal sludge could also have an impact on dewaterability at treatment facilities. Possibilities include greywater containing surfactants going into onsite containments (which can reduce membrane clogging through micelle formation (Cecconet et al. 2019)) or food waste (which is high in pCOD (Guven et al. 2019)). Likewise, emptying and transport practices that apply high shear to the fecal sludge could disrupt aggregates and decrease dewatering performance (Christensen et al. 2015). However, samples collected from vacuum trucks in Naivasha, Kenya in this study had overall larger aggregates compared to samples collected in situ from onsite containments in Kampala, Uganda, which would not have been expected if emptying with vacuum trucks was destroying aggregates. Based on our results, the concentration of small particles plays an important role in filtration and settling performance, indicating the most important consideration for improved dewatering of fecal sludge at treatment facilities is the elimination of small particles. This could be achieved for example through the addition of conditioners (Shaw et al. submitted), which reduce the concentration of small particles by trapping them inside of larger aggregates. This translates to better dewatering through reduced clogging with passive filtration (e.g. sand drying beds, geotextile bags), and improved removal of suspended particles by settling, and allow for the use of mechanical dewatering. However, these technologies are not yet established for fecal sludge and will need to be adapted with input from bench- and pilot-scale research and cannot be directly transferred from wastewater treatment.

6.4 Conclusions:

To the best of our knowledge, this is the first study examining stabilization with storage time in onsite containment with regards to dewatering performance at treatment facilities, and with a focus on understanding the role of particle size distribution in relation to stabilization and dewatering of fecal sludge. Based on the findings from this study, the key conclusions are:

- The common perception that stabilization and dewatering performance of fecal sludge are linked to storage time in onsite containments does not hold up to scientific investigation. However, although time is not a predictor, particle size distribution and dewatering performance are related to level of stabilization.
- Particle and aggregate size distribution, especially the concentration of small particles <10 µm, is a driver of dewatering performance, indicating that improved dewatering of

fecal sludge at treatment facilities can be achieved through the removal of small particles. With the current state of knowledge, this cannot be reliably achieved during storage in containment, but could be achieved with treatment options that promote flocculation (e.g. conditioners), or destruction of small particles (e.g. hydrolysis pretreatment followed by controlled anaerobic digestion).

- Microbial communities appear to be associated with aggregate size, and therefore dewatering performance of fecal sludge. With a deeper understanding of in-situ stabilization mechanisms and microbial degradation pathways, better predictions of characteristics of influent fecal sludge arriving for treatment and improved recommendations for optimal onsite storage conditions could be made.

Acknowledgements

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Statement of co-author contributions

BJ Ward designed the study, coordinated sample analysis, analyzed data, and took the lead in writing the manuscript. Thu Nguyen characterized samples for physical chemical characteristics and dewatering performance, collected and developed data analysis techniques microscopic images, and contributed to data interpretation and writing. Stanley Sam was responsible for preservation and DNA extraction of samples for microbial community analysis, contributed to study design and writing. Naomi Korir supervised sample collection in Naivasha, Kenya, and provided context and insight during project conception and writing. Charles Niwagaba supervised sample collection in Kampala, Uganda, and provided context and insight during project conception and writing. Eberhard Morgenroth contributed to study design, and provided insight on data interpretation and writing. Linda Strande conceived of the project idea, contributed to study design, data interpretation, and writing, and supervised the project.

Supporting information and additional data

Supplementary information is included in Appendix C at the end of this document. The complete dataset is published at https://10.25678/000702.

Chapter 7

Conclusions, Future Research, and Practical Implications

7.1 Conclusions

The necessity for low-footprint efficient dewatering of fecal sludge will continue to grow as urban populations expand. Immediate solutions to treat fecal sludge are essential. To achieve this, we need to understand the factors that determine settling and dewatering performance in fecal sludge, how they differ from municipal wastewater, and how they are impacted by stabilization processes during storage, in order to make recommendations for how to optimize design and manage fecal sludge treatment. The objective of this thesis was to advance understanding of fundamental drivers of solid-liquid separation in fecal sludge and develop methods for rapid field characterization, in order to implement robust and reliable treatment solutions that can be readily adapted to work with highly variable influent. In this section, the key findings of this thesis research are presented in the green textboxes with supporting bullets below.

This research identified that dewatering and settling in fecal sludge delivered to treatment facilities are driven by the concentration of ***suspended small particles*** and ***colloidal/soluble EPS***. Fecal sludge has different solid-liquid separation behavior from wastewater sludges and the ***existing conceptual model of dewatering needs to be modified and expanded*** to incorporate properties of fecal sludge. The mechanisms leading to aggregate formation and disintegration in fecal sludge and the role of EPS in aggregate structure require further research.

- Fecal sludge has a broader distribution of particle sizes than municipal wastewater influent or wastewater sludges, but the way that particle size distribution impacts dewatering fits with the accepted conceptual understanding of physical solid-liquid separation processes. Fecal sludge samples with smaller aggregates and higher fractions of particles smaller than 10 µm exhibited comparable or worse dewatering to anaerobically digested wastewater sludges, and fecal sludge samples with larger aggregate sizes and fewer small particles dewatered comparably to primary or activated wastewater sludges.
- Fecal sludge generally has much lower EPS concentrations and higher fractions of humic acids compared to wastewater sludges. Contrary to the role that EPS plays in wastewater activated sludge, EPS in fecal sludge did not measurably contribute to binding water in the sludge cake. In fecal sludge, EPS appears to be most relevantly characterized as soluble and colloidal organic matter which contribute to clogging and turbid supernatant.
- Aggregates are present in some, but not all, fecal sludge field samples, and their size is associated with abundant populations of bacteria that have been observed to contribute to bioflocculation.

The common perception that stabilization and solid-liquid separation of fecal sludge arriving at treatment facilities are linked to storage time in onsite containment does ***not hold up to scientific investigation***. However, although ***time is not a predictor***, ***particle size distribution and dewatering performance are related to level of stabilization***. In order to move forward in understanding how the transformation of organic matter during storage in onsite containment affects treatment performance, we need to understand the environmental and microbial drivers and inhibitors of fecal sludge stabilization.

- Fresh fecal sludge has more turbid supernatant and takes more time to dewater than stabilized fecal sludge, based on qualitative observations of color and odor and on analytical indicators of stabilization.
- Dewatering performance and stabilization did not improve with longer time intervals between emptying events, based on field samples collected in five different cities.
- In controlled anaerobic storage experiments, neither a consistent relationship between dewatering and storage time nor between metrics of stabilization and storage time was observed. However, more stabilized samples with lower VSS/TSS, higher C/N, and higher pCOD/COD had good dewatering performance compared to less stabilized samples.

Dewatering performance and influent fecal sludge characteristics can be ***predicted with reasonable accuracy*** using inexpensive and simple ***field measurements*** and ***black box models***. However, developed models are based on samples collected in one city, and will need to be expanded and validated with sampling campaigns in additional cities in order to be more widely applicable.

- Machine learning and linear regression models can use cost-efficient and easy to implement field measurements, including photographs (color, texture) and probe readings (EC, pH) to predict fecal sludge characteristics and dewatering performance (TS, NH4+-N, supernatant turbidity, and dewatering time). Prediction accuracy is an improvement on existing qualitative methods and could be useful for dosing conditioners in a small-footprint transfer station or mobile operation.
- Simple decision tree models using containment type or other environmental, technical, or demographic factors in different cities (e.g. establishment type as observed in Dakar and Kampala) can be helpful for making low-resolution predictions for city-scale planning or projecting loadings for the design of fecal sludge treatment facilities.

- An app was developed to automate color and texture data collection and make predictions of fecal sludge characteristics and dewatering performance based on field measurements. This provided a proof of concept with models developed from data collected in Lusaka, Zambia.

7.2 Future Research

The outcomes of the work presented in this thesis include ideas for targeted follow-up fundamental research to understand the ***formation and disintegration of aggregates*** in fecal sludge and the ***factors that facilitate or hinder in situ stabilization*** in onsite containment.

We saw aggregates in field samples, but it is still not known whether aggregates actually form in containments, or whether they are residual pieces of feces that have not been broken down. Future research should investigate ***what aggregates are composed of***, and evaluate how they are different from activated sludge flocs. Our research found that EPS is present in much smaller concentrations than in wastewater sludges, and we expect fecal sludge aggregates are not only composed of EPS and microbial biomass. Future research should zoom out to incorporate a wider pool of organics including remnants of undigested foods, soil organic matter, detergents, municipal solid waste, and cleansing materials. Qualitative analysis of components and structure of fecal sludge aggregates could be accomplished using confocal microscopy in combination with staining to identify specific macromolecules and FISH to stain for specific microorganisms. This could be quantitatively supported by isolating aggregates and single suspended particles by sequential sieving and filtration, and separately characterizing the organic matter in the different size fractions.

Future research also needs to focus on understanding ***what forces are keeping aggregates together*** or allowing them to disintegrate. This should include evaluation of surface charge, which can give us information about whether fecal sludge aggregation is governed by electrostatic interactions. For such measurements to be possible, the method for characterizing surface charge by measuring zeta potential in fecal sludge needs to be adapted for higher conductivity samples. Additionally, hydrophobicity of aggregates would be informative to characterize, as this could be a driver of aggregation and has been shown to change as wastewater sludge organics are preferentially hydrolyzed and transformed during the course of anaerobic digestion. It will be important to characterize surface charge and hydrophobicity in tandem with the concentrations and fractionations of macromolecules and concentrations of monovalent and divalent cations associated with aggregates and with the bulk. Finally, in addition to characterizing CST and supernatant turbidity for dewatering performance, it makes sense to also evaluate aggregate resistance to shear. This is practically important for dewatering performance, as aggregates that easily break up, for example when subjected to shear during emptying or mechanical dewatering, will not dewater the same as stronger aggregates that do not disintegrate.

To advance understanding of how the transformation of organic matter during storage in onsite containment affects treatment performance, we need to understand the environmental and microbial ***drivers and inhibitors of fecal sludge stabilization*** processes. Future research should focus on identifying in situ conditions that facilitate and inhibit the breakdown of organic matter, including availability of nutrients, presence of inhibitors, bioavailability of organic matter, and redox conditions. This could for example, be broken into a field work component and a controlled digestion component. First, field samples could be collected from various depths of onsite containments and characterized for macro- and micro-nutrients, inhibitors (e.g. H_2S, NH_4^+-N, antibiotics), speciation and recalcitrance of organic matter, and electron acceptors (e.g. Fe (II), Fe (III), sulfate), along with indicators of stabilization. These characteristics can then be related back to differences in level of stabilization, with identified key nutrients, inhibitors, substrates, and electron acceptors. Next, controlled degradation experiments in reactors can be performed, adjusting the identified inhibiting or facilitating factors accordingly to validate the observations made in the field and identify microbial stabilization pathways and relevant in situ conditions for promoting stabilization and changes in dewatering performance.

7.3 Implications for practice

The research presented in this thesis led to applied recommendations for fecal sludge treatment practitioners for how to ***better predict influent fecal sludge characteristics*** and ***optimize treatment performance***.

Characterization of fecal sludge is imperative for designing and operating sustainable treatment solutions. In many cases, ***estimates of characteristics*** are accurate enough and could supplement expensive laboratory analysis. For example, for designing a treatment facility based on estimated sludge characteristics in the neighborhoods the facility will serve, or adjusting drying bed loading rates or conditioner dosing based on estimated characteristics of each batch of influent fecal sludge. The level of accuracy of estimates ***only needs to be good enough*** for the application: lower accuracy models are sufficient for estimating the quantities and qualities of fecal sludge for citywide planning, while greater accuracy will be necessary for designing treatment facilities and informing process control. The predictive models we developed are immediately applicable for Lusaka, as long as they are combined with periodic quality control checks using laboratory analytical methods. However, in order for predictions to be relevant to additional locations, datasets need to be collected in other cities in collaboration with local partners to build a global database of fecal sludge characteristics. The database will allow for the models to be updated and validated outside of Lusaka, to determine which relationships are city/region specific and which could be applicable globally. At the same time, prediction uncertainties will need to be quantified and improved as models are updated with larger datasets, to aid in practical decision making at fecal sludge treatment facilities. The Sludge Snap app is a promising proof of concept, but will also need to be refined with attention to user experience and data processing efficiency, to make it faster and simpler to use in the field. Based on the feedback of prospective users, the next iteration of the Sludge Snap app should be updated to make process control recommendations for specific treatment technologies based on field measurements (e.g. a suggested conditioner dose based on total solids).

Based on these findings, removing small particles from suspension is the key to achieving better solid-liquid separation in fecal sludge. With the current state of knowledge, doing this with anaerobic digestion during storage is infeasible and unreliable. Treatment options that ***promote flocculation*** (e.g. conditioners), or consistently and thoroughly ***destroy small particles*** (e.g. hydrolysis pretreatment followed by controlled anaerobic digestion) should be explored to optimize dewatering performance. However, these technologies are not yet established for use with fecal sludge, and ***applied research will be needed*** to adapt them to work for highly variable influent. Technology transfer will require multiple iterations between lab-, pilot-, and full-scale testing, and critical evaluation of performance to achieve well-functioning robust treatment technologies. Partnerships between municipalities and research institutes and universities can help to guide optimization of fecal sludge treatment technologies to address the ever growing urban sanitation need.

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Appendix A

Supplemental Information

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

Barbara J. Ward[1,2,*], Jacqueline Traber[1], Amadou Gueye[3], Bécaye Diop[3], Eberhard Morgenroth[1,2], Linda Strande[1]

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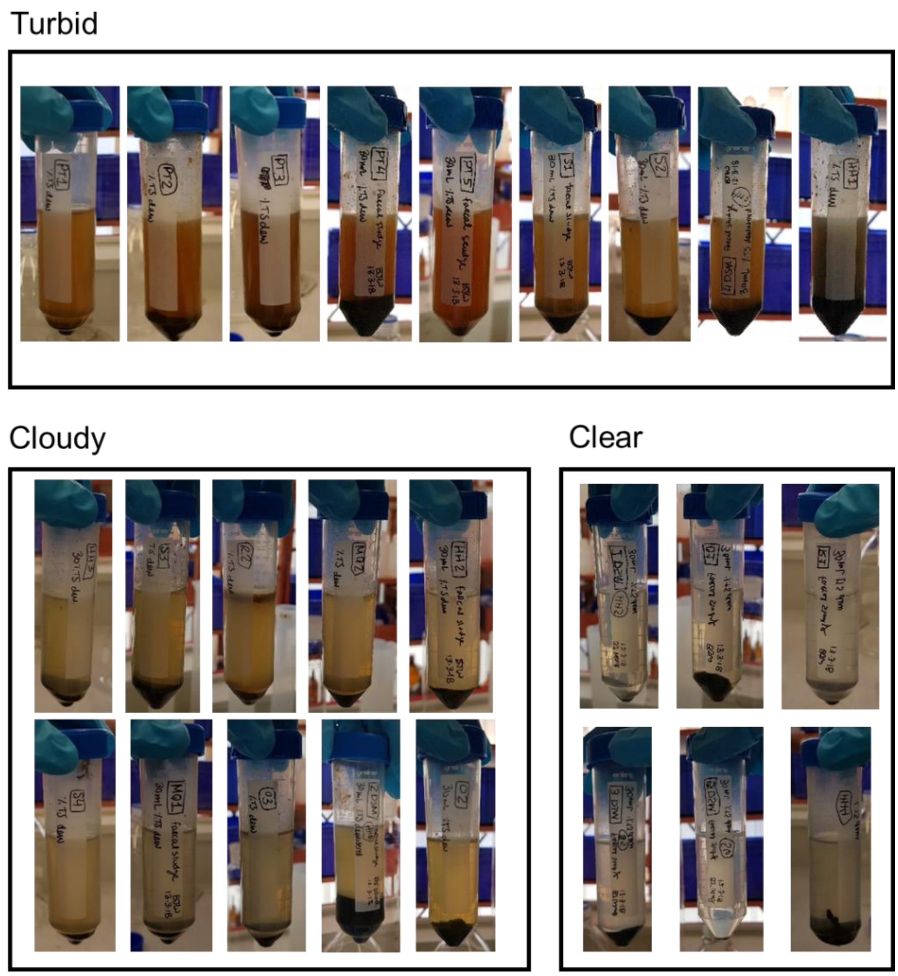


Figure S2.1. Photographs of samples used for ranking of supernatant turbidity after prolonged settling. Samples were grouped into three categories: clear, cloudy, or turbid.

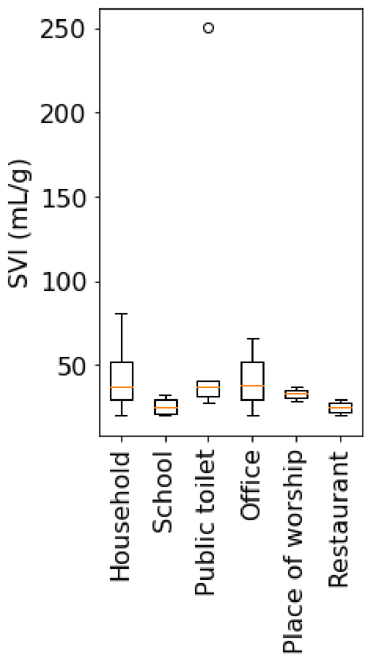


Figure S2.2. Boxplot illustrating distribution of sludge volume index (SVI) by faecal sludge source.

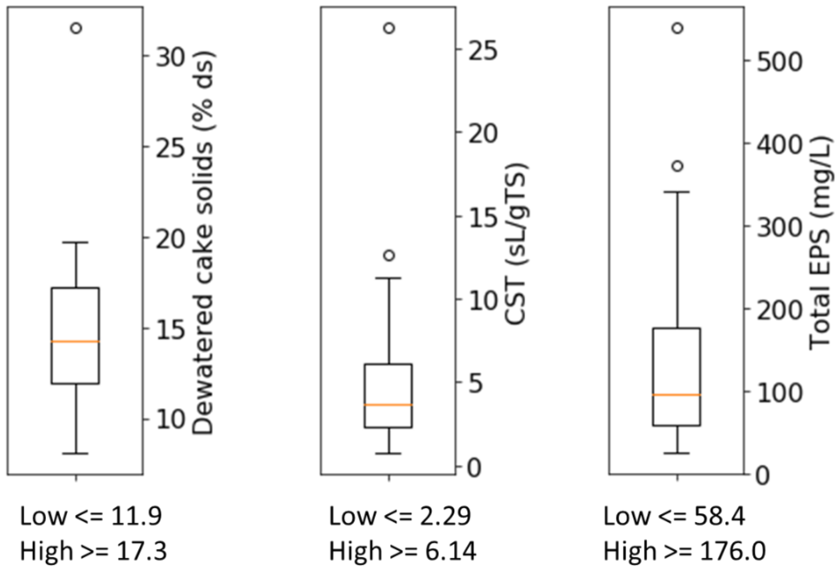


Figure S2.3. Boxplots illustrating distribution of dewatered cake solids (left), CST (middle), and total EPS concentration (right). “High” values are defined as data points falling within the top 25%, and “Low” values are points falling within the bottom 25%. “High” and “Low” designations were used to assign categories for differential abundance analysis.

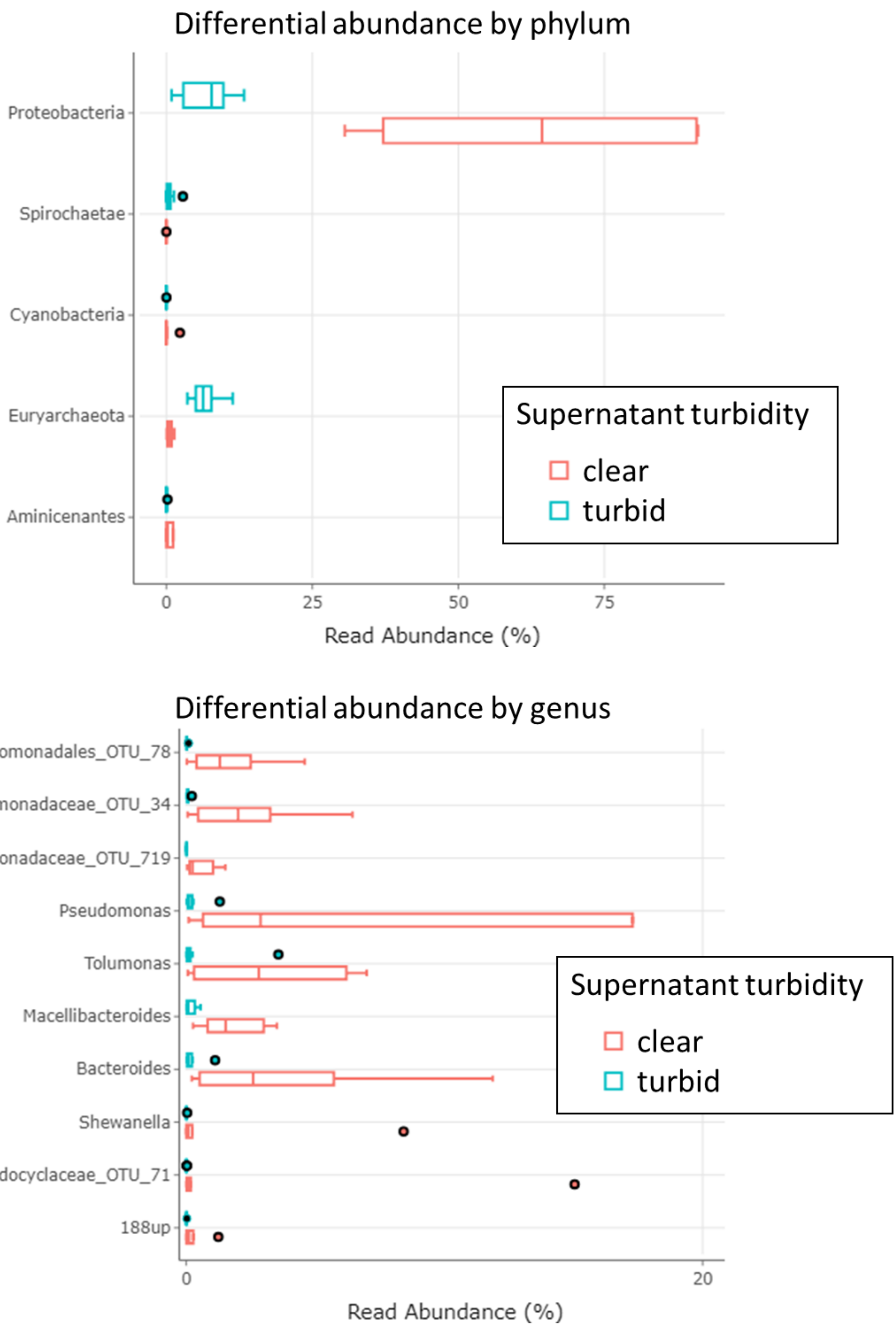


Figure S2.4. Boxplots showing differential abundance of specific phyla (top) and genera (bottom) between samples with clear supernatant after settling (red) and turbid supernatant after settling (blue). Plots produced using shinyapps.io data analysis platform provided by DNASense.

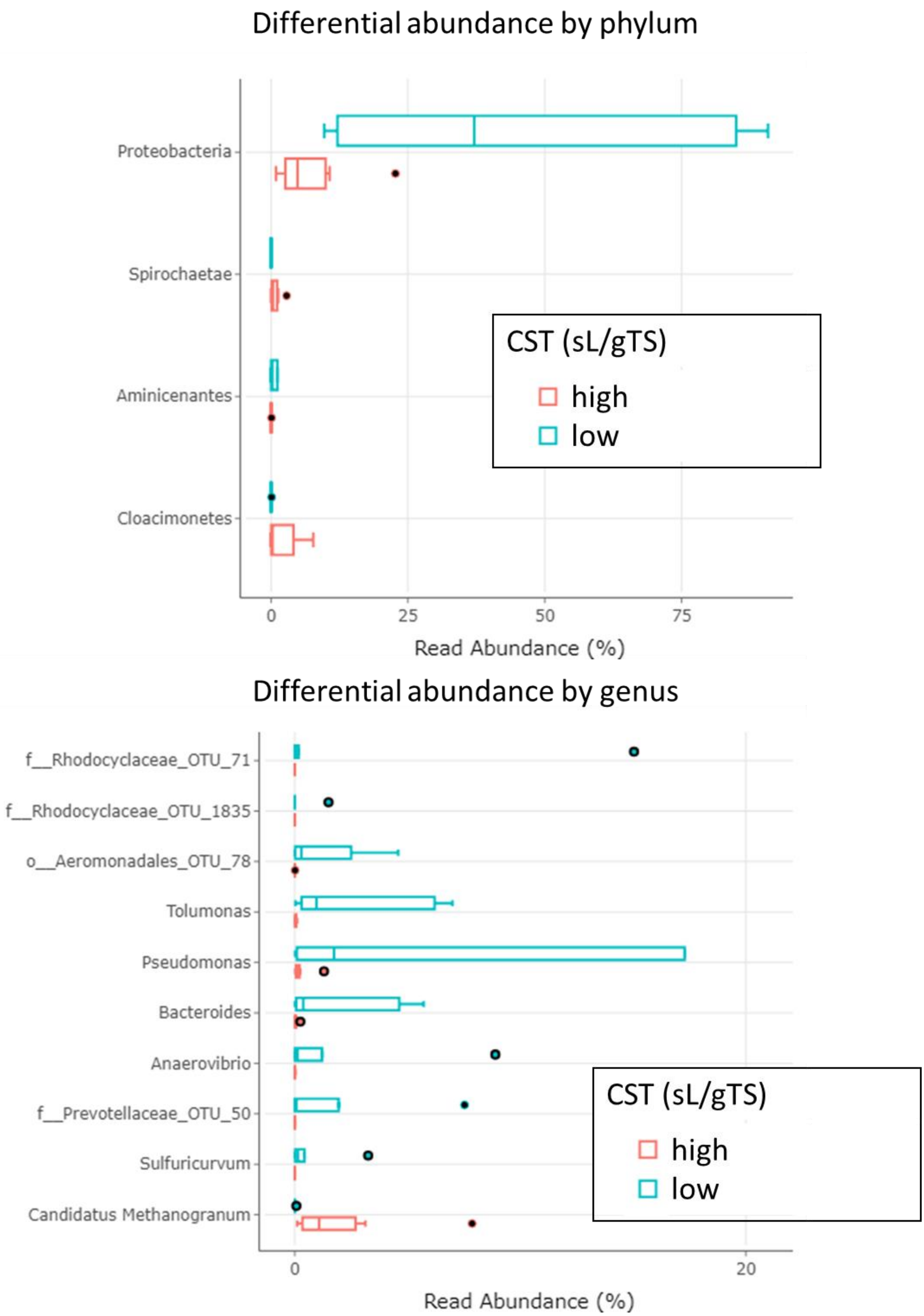
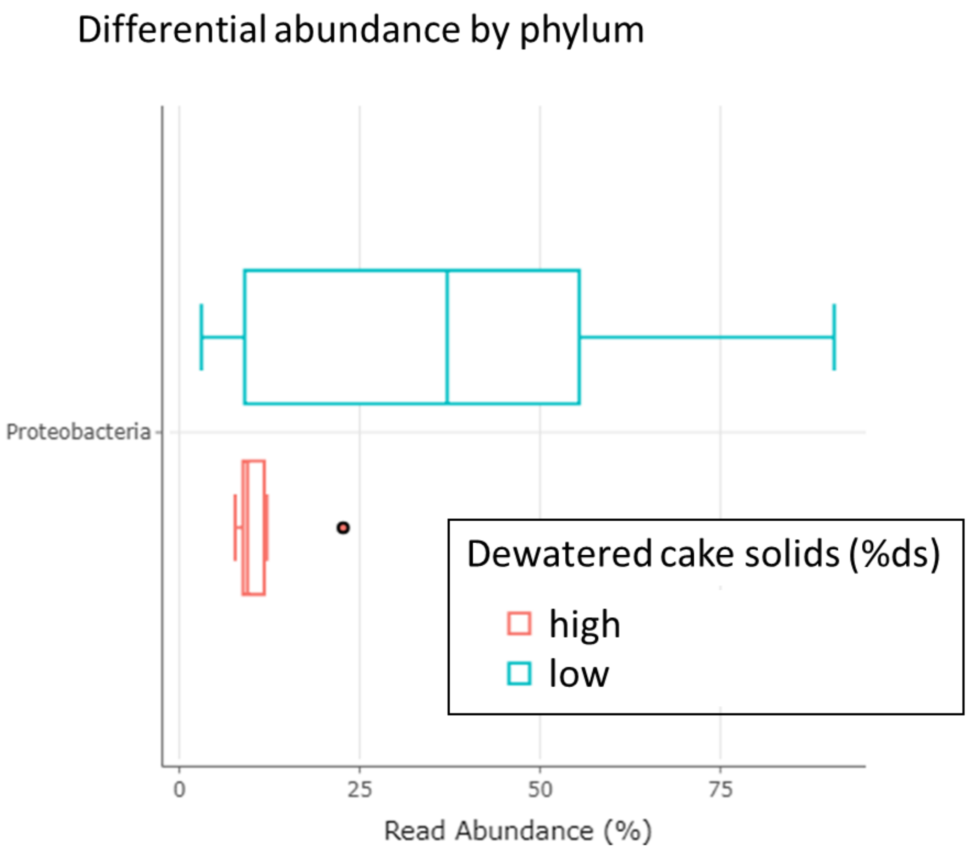


Figure S2.5. Boxplots showing differential abundance of specific phyla (top) and genera (bottom) between samples with high CST (≥6.14 sL/gTS) (red) and low CST (≤2.29 sL/gTS) (blue). Plots produced using shinyapps.io data analysis platform provided by DNASense.



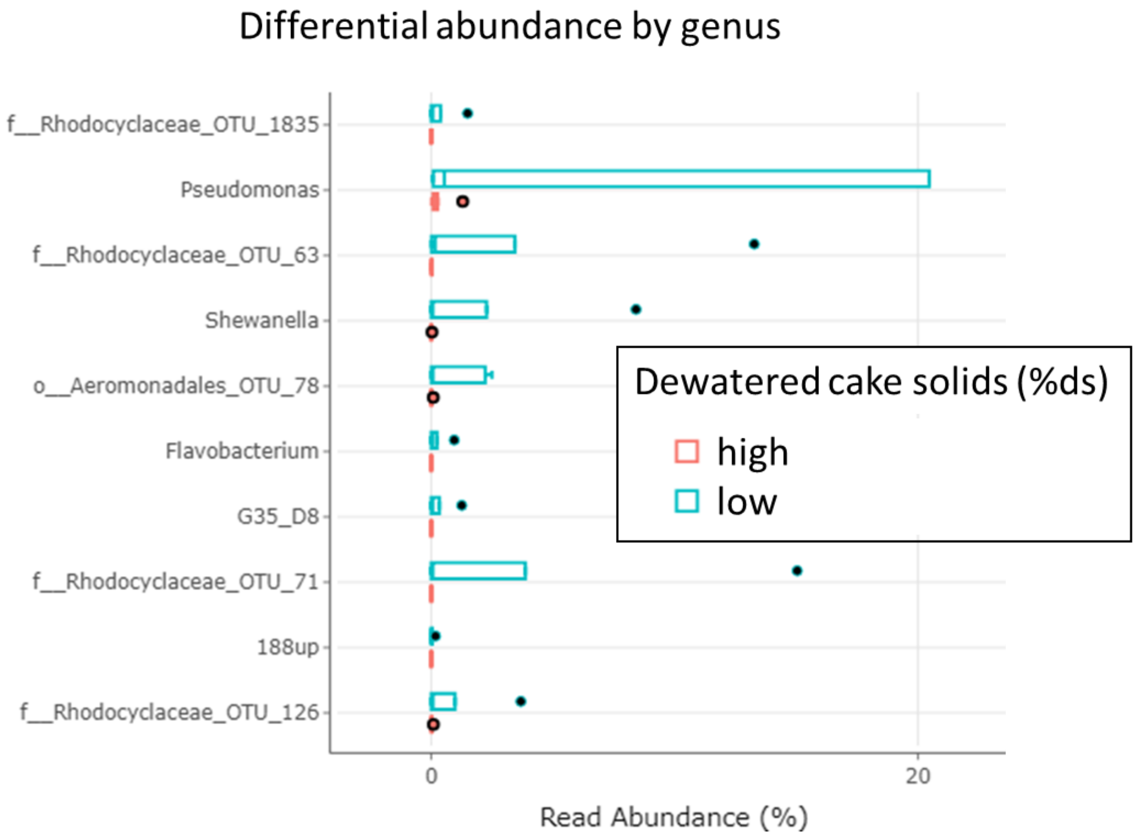


Figure S2.6. Boxplots showing differential abundance of specific phyla (top) and genera (bottom) between samples with high dewatered cake solids (≥17.3 %) (red) and low dewatered cake solids (≤11.9 %) (blue). Plots produced using shinyapps.io data analysis platform provided by DNASense.

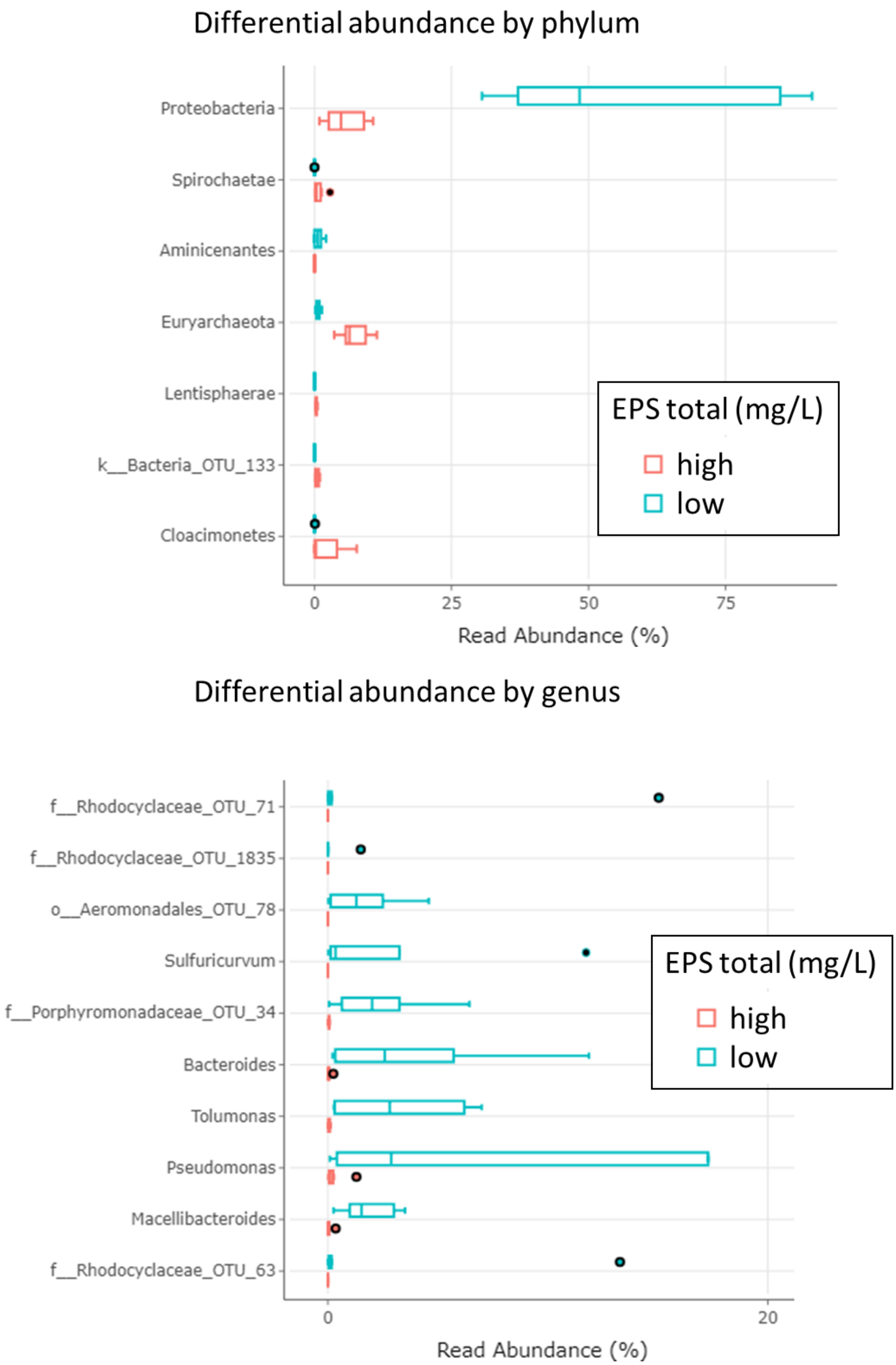


Figure S2.7. Boxplots showing differential abundance of specific phyla (top) and genera (bottom) between samples with high EPS concentrations (≥176.0 mg/L) (red) and low EPS concentrations (≤58.4 mg/L) (blue). Plots produced using shinyapps.io data analysis platform provided by DNASense.

Appendix B

Supplemental Information

Predictive models using "cheap and easy" field measurements: Can they fill a gap in planning, monitoring, and implementing fecal sludge management solutions?

Barbara J. Ward[a,b,*], Nienke Andriessen[a], James M. Tembo[c], Joel Kabika[c], Matt Grau[d], Andreas Scheidegger[a], Eberhard Morgenroth[a,b], Linda Strande[a]

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Links to open dataset and code

The complete dataset and code used for this study are accessible at this DOI:

https://doi.org/10.25678/00037X

Questionnaire questions

Is the establishment accessible for sampling?
If no, why not?
Sample identification number (###-Team-YYYYMMDD)
Local area name
Type of establishment
Optional additional information to type of establishment
How many residents live in the household?
Building type
Optional additional information to building type
Roof type
Optional additional information to roof type
Primary occupation of the head of the household
Highest educational qualification
Monthly income level
Type of containment
Does this septic tank have one or more baffles?
Is there an outflow?
If yes, where does the outflow go to?
What type of toilet(s) feed into this containment system?
Number of users
Type of anal cleansing material
Is there solid waste in the containment?
If yes, what type(s) of solid waste?
Do you add anything to the containment (for example ash, bio additives, enzymes)?
Type of wastewater entering the containment
Age of the system
Do you notice a change in the sludge level in your system between the wet and dry seasons?
Is there a water connection on the premises?
When was the system last emptied?
Was it fully emptied at that time?
GPS location
Take a picture of the toilet superstructure
Take a photo of the building

Example photographs from color and texture analysis

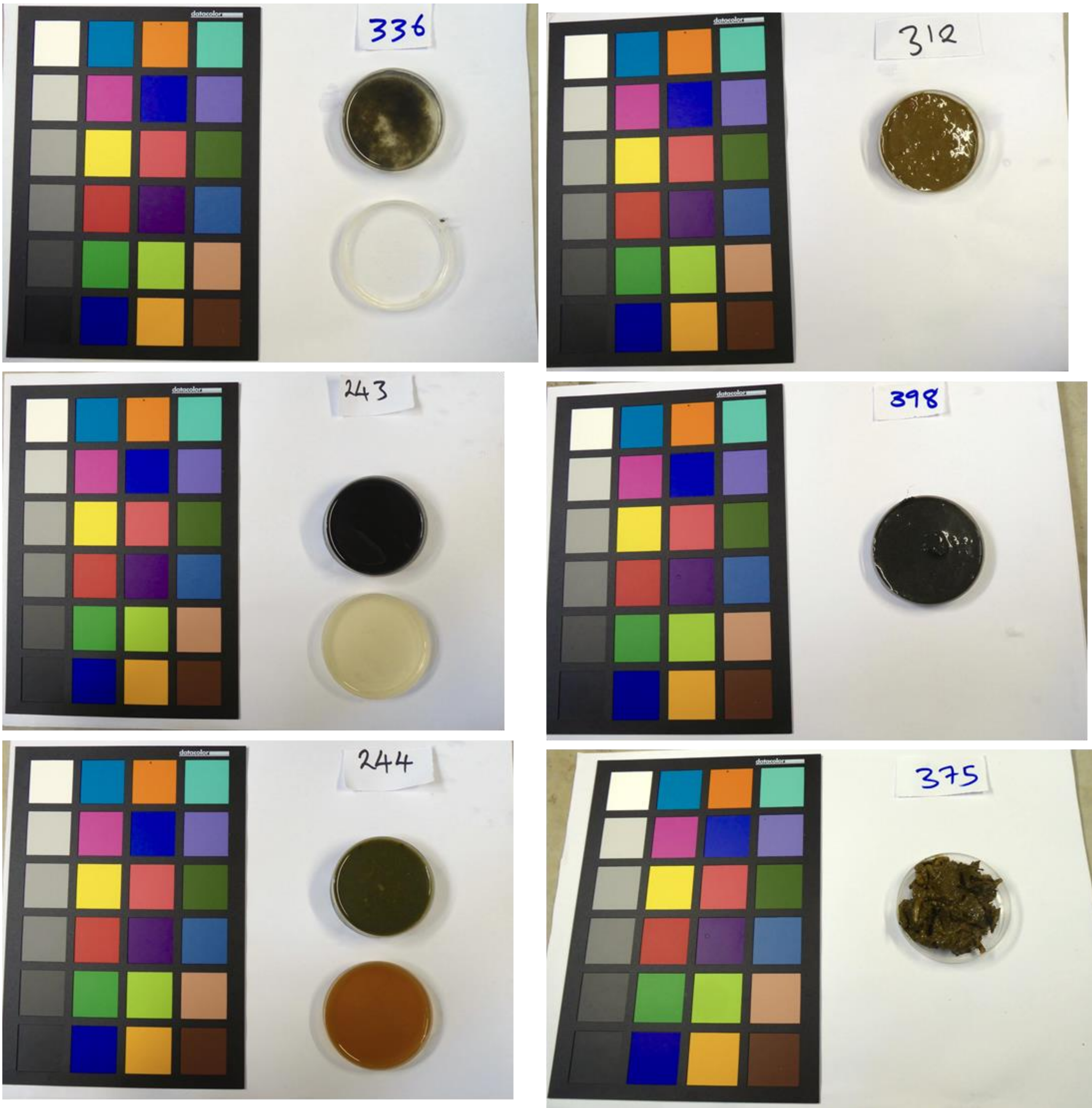

Figure S4.1 Example photographs of fecal sludge samples (including supernatant samples in photographs on left) with color checker charts.

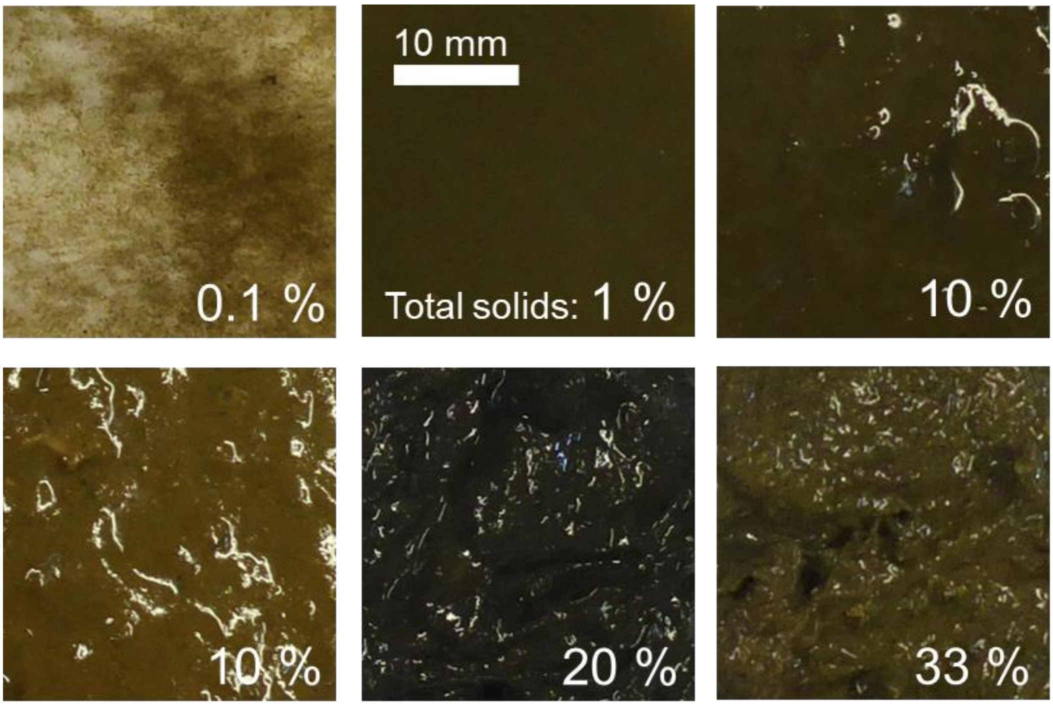

Figure S4.2. Example swatches used to extract texture and color information.

Descriptive statistics for all samples

Table S4.1. Summary statistics for field measurements of all fecal sludge samples.

ALL SAMPLES

	Field measurements														
	EC	pH	foam height	color			supernatant color			texture					
	(mS/cm)		(mm)	H	S	V	H	S	V	cont.	dissim.	homog.	ASM	energy	corr.
mean	8.4	7.61	4.21	51	37	21	51	27	83	5.85	0.76	0.79	0.29	0.51	0.57
std	7.0	0.48	6.52	12	21	15	21	26	14	9.09	0.71	0.11	0.19	0.17	0.19
median	**5.9**	**7.69**	**2.00**	**49**	**36**	**16**	**51**	**14**	**85**	**0.84**	**0.38**	**0.83**	**0.25**	**0.50**	**0.54**
25%	1.8	7.44	0.00	45	17	11	46	8	79	0.30	0.25	0.70	0.13	0.37	0.41
75%	14.4	7.89	5.00	60	56	25	55	38	91	7.96	1.23	0.88	0.39	0.63	0.72
n	465	465	465	456	456	456	253	253	253	448	448	448	448	448	448

Table S4.2. Summary statistics for laboratory measurements of all fecal sludge samples.

ALL SAMPLES

	Laboratory-based measurements									
	Supernatant turbidity	CST	TS in dewatered cake	COD	NH4-N	TS	VS	TOC_{solids}	TKN_{solids}	Density
	(NTU)	*(s)*	*(% ds)*	*(g/L)*	*(g/L)*	*(% ds)*	*(% of TS)*	*(% of TS)*	*(% of TS)*	*(g/mL)*
mean	311	208	21.7	97.3	1.7	9.9	55.2	11.0	2.2	1.10
std	472	437	20.9	66.8	1.7	8.8	18.3	2.8	1.2	0.07
median	**144**	**64**	**15.1**	**86.4**	**1.0**	**8.5**	**54.1**	**10.5**	**2.1**	**1.08**
25%	53	16	7.7	39.8	0.3	1.5	43.1	9.2	1.4	1.06
75%	373	193	26.4	141.6	3.0	16.0	68.5	13.0	2.8	1.11
n	253	237	223	362	464	439	428	301	301	298

Appendix B

Boxplots

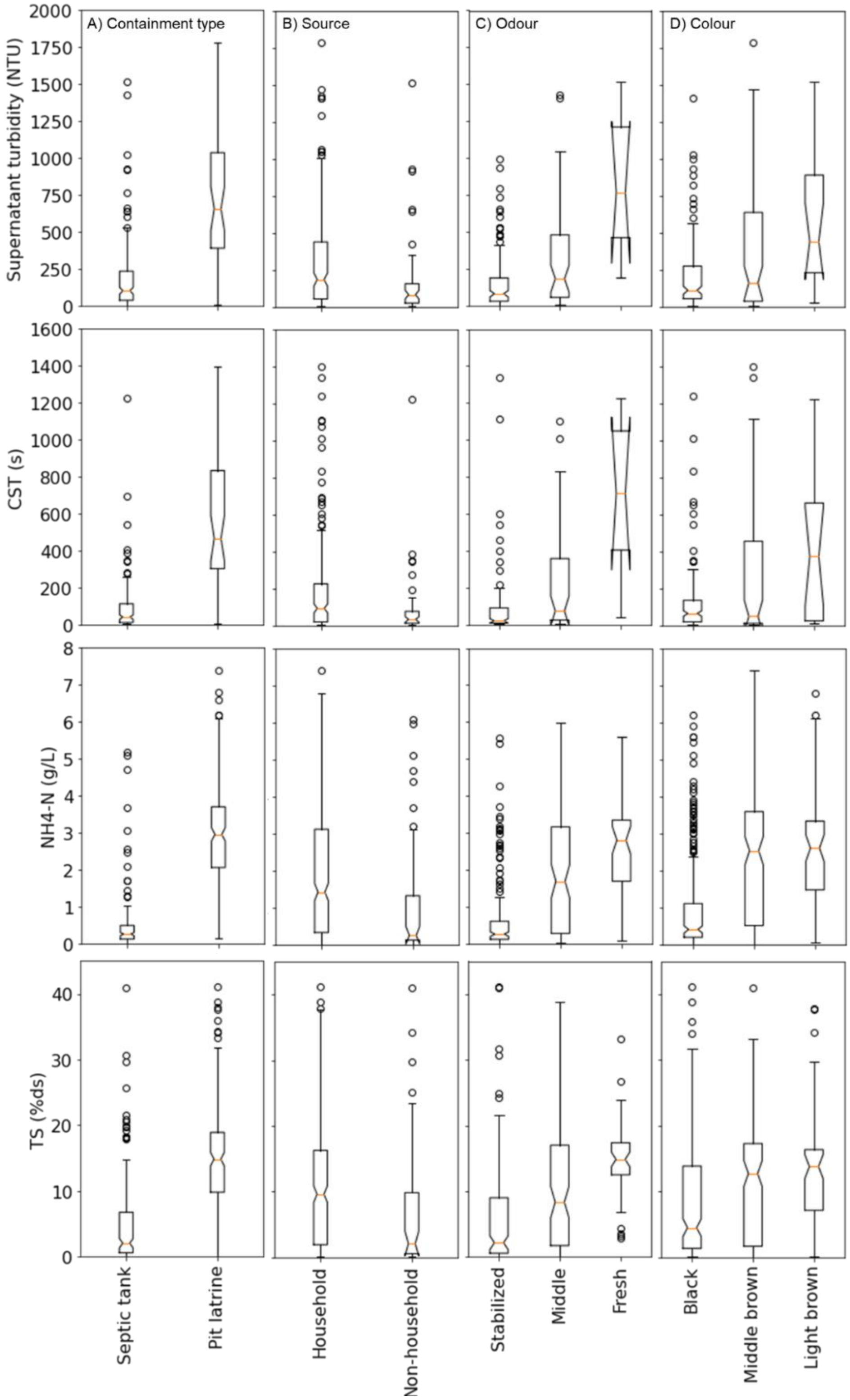

Figure S4.1. Boxplots showing relationship between questionnaire data, expert knowledge, and target parameters

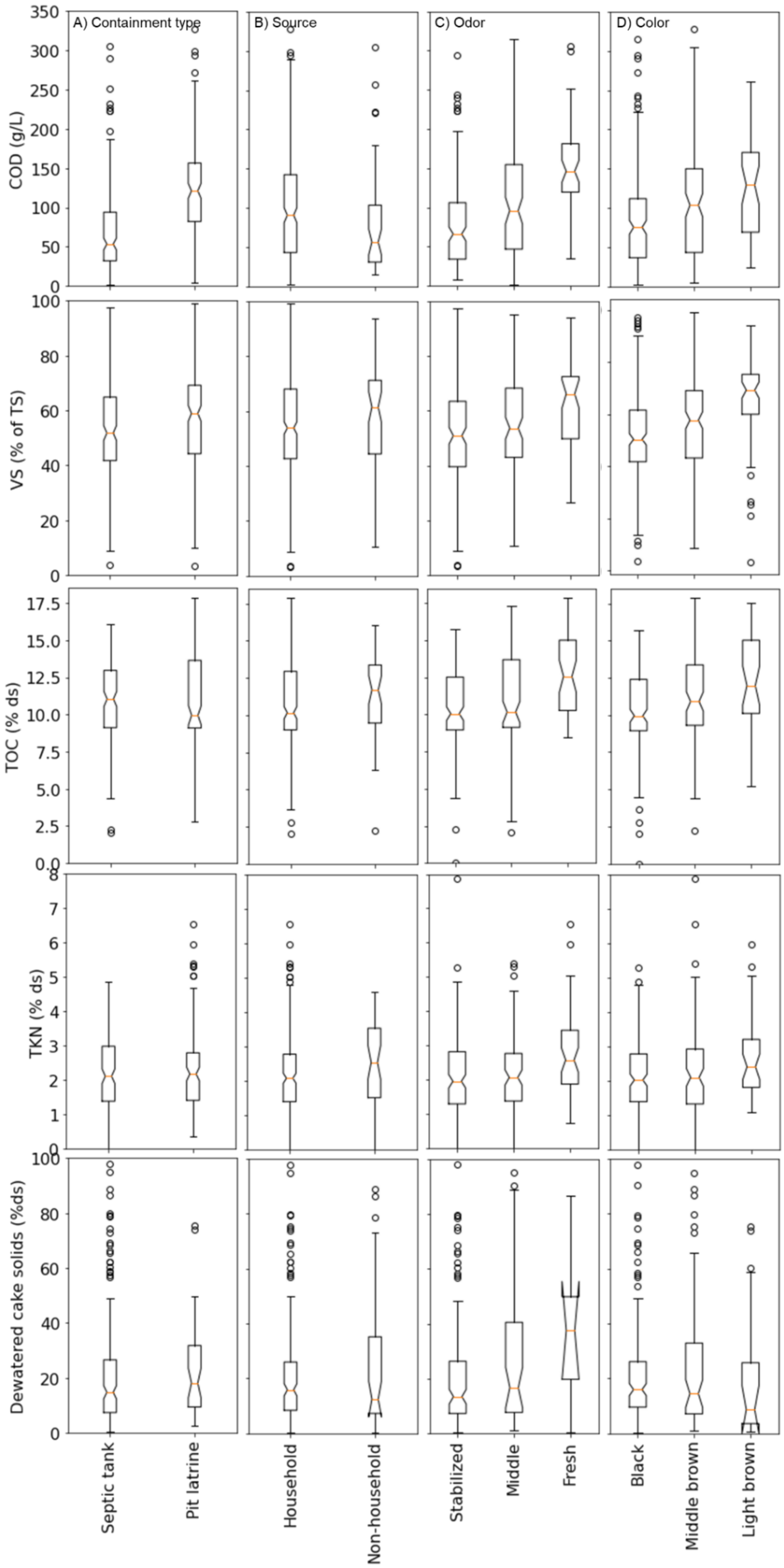

Figure S4.4. Boxplots showing relationship between questionnaire data, expert knowledge, and target parameters

Heatmaps

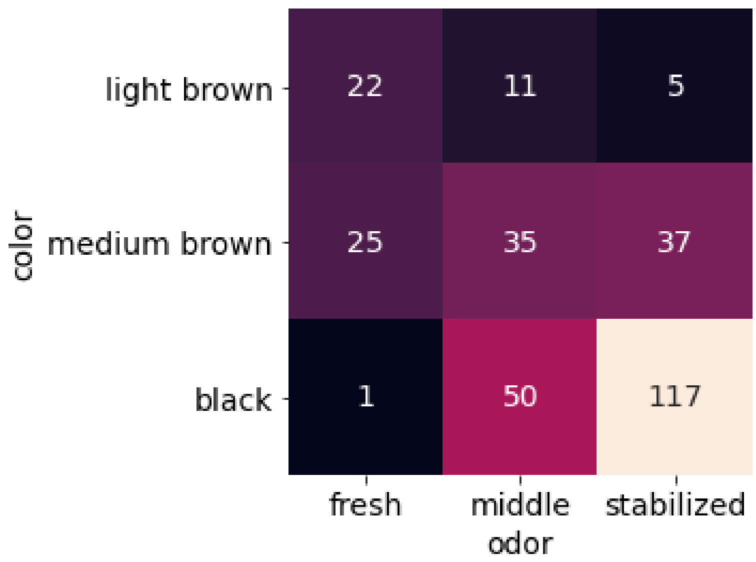

Figure S4.5. Heatmap of samples designated by expert assessments of color and odor.

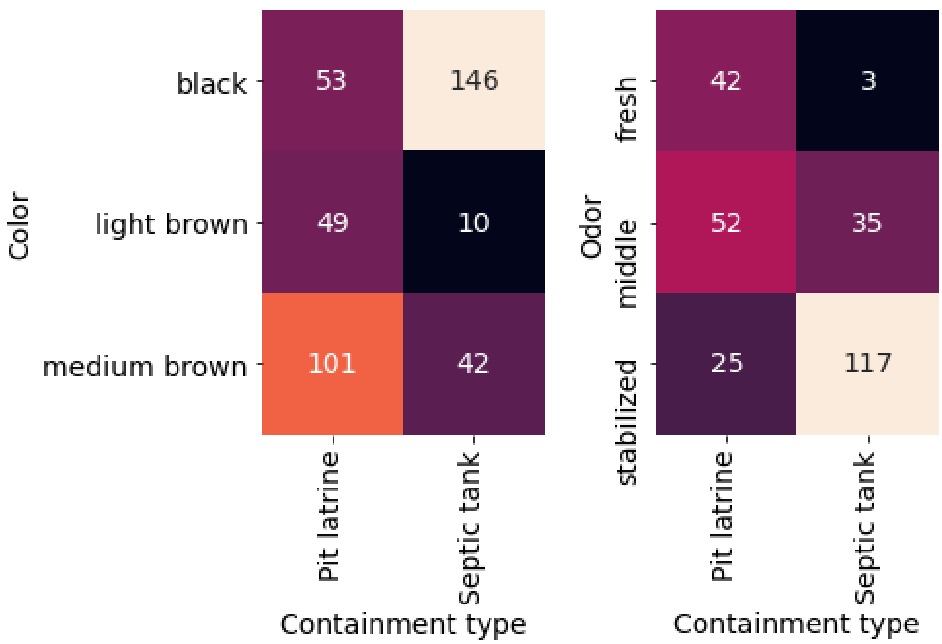

Figure S4.6. Heatmaps of samples designated by containment type and expert assessments of color and odor.

Linear relationship between NH_4^+-N and EC

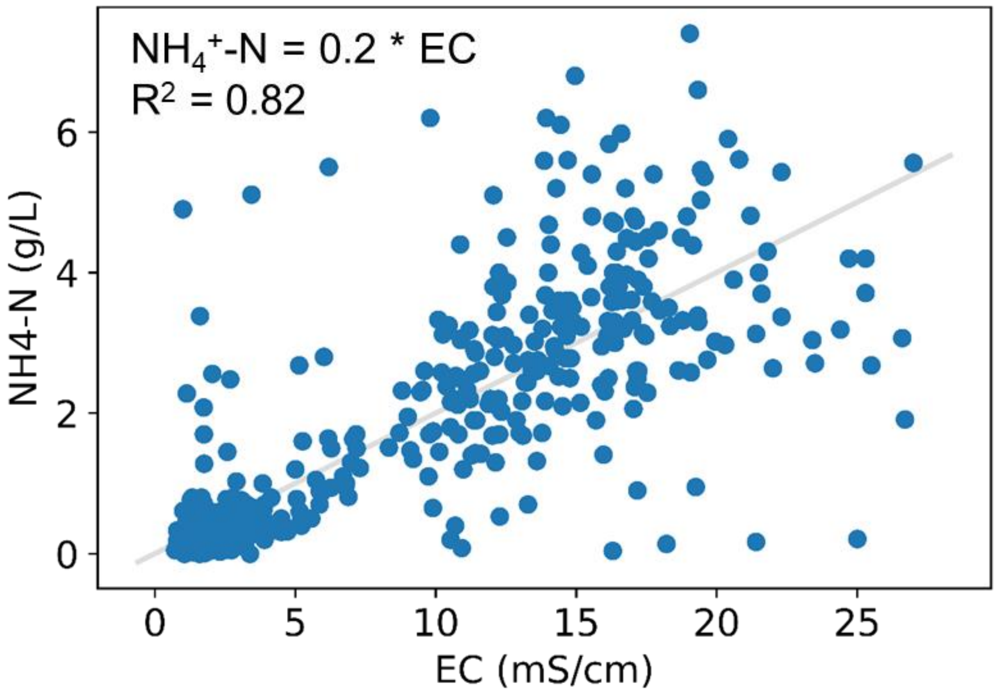

Figure S4.7. Scatterplot showing relationship between NH_4^+-N and EC.

Descriptions of models

Random forest model description

- Random forest built with 300 trees
- Default hyperparameters were used: Number of estimators = 100; Max features = 'auto'; Max depth = 'None'; Min samples per split = 2; Min samples per leaf = 1; Bootstrap = 'True'.

Linear model description

- Linear ordinary least squares model

Simple decision tree model description

- The simple decision tree model uses the median value of the target parameter in a category (e.g., median TS in septic tanks) to predict future values in that category. Inputs to this model must be categorical data. For example, if the training dataset has median TS in septic tanks at 3%, and median TS in pit latrines at 10%, the simple decision tree will predict 3% TS for future septic tank samples and 10% TS for pit latrine samples.

Selection of strongest predictors and supporting predictors

For each target parameter, models were generated for every possible combination of maximal four inputs, and the performance of each model was evaluated using cross-validated R^2 and root mean squared error (RMSE) (5-fold cross validation, repeated 20 times). Normalized RMSE (nRMSE) was also calculated, by dividing RMSE by the mean of the observed data. The input combination with the highest cross-validated R^2 was considered the best. Preference was given to models with fewer inputs, so if inputs could be removed from the model without a relevant decrease in R^2 (at two decimal places), those inputs were not included in the best model. Relative importance of inputs was evaluated by comparing the R^2 of models built with and without the input. The relative strengths of the inputs included in the best models were evaluated by comparing the R^2 of single-input models. The input with the largest R^2 was labeled the 'strongest predictor' if the R^2 of that model was at least 75% of the R^2 of the best multi-input model. Supporting predictors were defined as inputs that are included in the best model and increase the model R^2 when included as model inputs along with the strongest predictor.

A detailed example of the selection of best inputs for the random forest model for TS follows. Table S4.3 shows an example of cross-validated model performance results ordered by R^2 for models produced with different input combinations.

Table S4.3. Example of model performance outputs generated by the random forest model predicting TS. Models are ranked by R^2. The highest R^2 model containing ≤ 4 inputs is the model circled in red. Inputs included are foam height; color (quantitative) which comprises H, S, and V color values of the bulk sludge; texture, which comprises the texture measures contrast, dissimilarity, homogeneity, ASM, energy, and correlation; and containment type.

inputs	N	R2	RMSE	nRMSE
['Foam_height', 'bH', 'bS', 'bV', 'contrast', 'dissim', 'homog', 'ASM', 'energy', 'corr', 'containment_type', 'source', 'color_qualitative']	244	0.572277	5.776583	62.64996
['pH', 'Foam_height', 'bH', 'bS', 'bV', 'contrast', 'dissim', 'homog', 'ASM', 'energy', 'corr', 'containment_type', 'color_qualitative']	244	0.571779	5.784853	62.73965
['Foam_height', 'bH', 'bS', 'bV', 'contrast', 'dissim', 'homog', 'ASM', 'energy', 'corr', 'containment_type', 'color_qualitative']	244	0.571613	5.778912	62.67522
['Foam_height', 'bH', 'bS', 'bV', 'contrast', 'dissim', 'homog', 'ASM', 'energy', 'corr', 'containment_type', 'odor', 'color_qualitative']	244	0.571196	5.785699	62.74883
['Foam_height', 'bH', 'bS', 'bV', 'contrast', 'dissim', 'homog', 'ASM', 'energy', 'corr', 'containment_type']	244	0.571158	5.78222	62.7111
['Foam_height', 'bH', 'bS', 'bV', 'contrast', 'dissim', 'homog', 'ASM', 'energy', 'corr', 'containment_type', 'source', 'odor', 'color_qualitative']	244	0.571105	5.787302	62.76621
['pH', 'Foam_height', 'bH', 'bS', 'bV', 'contrast', 'dissim', 'homog', 'ASM', 'energy', 'corr', 'containment_type', 'source', 'color_qualitative']	244	0.570985	5.789234	62.78716
['pH', 'Foam_height', 'bH', 'bS', 'bV', 'contrast', 'dissim', 'homog', 'ASM', 'energy', 'corr', 'containment_type', 'source']	244	0.570923	5.790242	62.7981
['Foam_height', 'bH', 'bS', 'bV', 'contrast', 'dissim', 'homog', 'ASM', 'energy', 'corr', 'containment_type', 'source']	244	0.57075	5.784717	62.73818
['pH', 'Foam_height', 'bH', 'bS', 'bV', 'contrast', 'dissim', 'homog', 'ASM', 'energy', 'corr', 'containment_type', 'odor']	244	0.57056	5.794112	62.84008
['Foam_height', 'bH', 'bS', 'bV', 'contrast', 'dissim', 'homog', 'ASM', 'energy', 'corr', 'containment_type', 'water_connection', 'source', 'color_qualitative']	244	0.570492	5.788736	62.78176

Highest R^2 model with 4 inputs

Once the best performing 4-input model was identified, model performance of every possible combination of the 4 inputs in that model were tabulated and ranked by R^2 (Table S4). Input importance is determined using this table. The R^2 value of the model including texture as a single input is higher than the R^2 of the other single-input models (and is in fact higher than those of many of the multi-input models). The R^2 of the texture single-input model is 0.49, which is 86% as high as the R^2 for the 4-input model. Thus, texture fulfils the criteria to be labeled the strongest predictor of TS.

Table S4.4. All combinations of inputs included in best 4-input random forest model for TS, ranked by R^2. Black rectangles indicate the inclusion of an input in the model.

Total solids

Cross-validated model performance			Features included			
R^2	RMSE (s)	nRMSE (%)	texture	color (quantitative)	foam height	containment type
0.57	5.78	62.71	■	■	■	■
0.57	5.81	62.97	■	■	■	
0.55	5.92	64.22	■	■		■
0.54	5.98	64.82	■	■		
0.50	6.27	67.98	■		■	■
0.50	6.28	68.16	■		■	
0.50	6.31	68.39	■			■
0.49	6.33	68.64	■			
0.44	6.63	71.94		■	■	■
0.43	6.71	72.78			■	■
0.36	7.07	76.64		■	■	
0.33	7.28	78.95			■	
0.27	7.62	82.70				■
0.20	7.90	85.67		■		■
-0.10	9.36	101.48		■		

To assess the other inputs for their role as supporting predictors, the best 2-input and 3-input models are identified using R^2, and are included in a condensed summary table (Table S4.5). Because the 4-input model has the same R^2 (to 2 decimal places) as the 3-input model, the 3-input model is selected as the overall best random forest model for TS. Texture is designated as the strongest predictor, and color (quantitative) and foam height are designated as supporting predictors. Containment type is not designated as a supporting predictor, because its inclusion in the best model does not yield an improvement in R^2 (Table S4.6).

Table S4.5. Condensed summary table of best inputs, ranked by R^2. Black rectangles indicate the inclusion of an input in the model.

Total solids

Cross-validated model performance			Features included			
R^2	RMSE (s)	nRMSE (%)	texture	color (quantitative)	foam height	containment type
0.57	5.78	62.71	■	■	■	■
0.57	5.81	62.97	■	■	■	
0.54	5.98	64.82	■	■		
0.49	6.33	68.64	■			

Table S4.6. Final summary table of strongest and supporting predictors, ranked by R^2. Black rectangles indicate the inclusion of an input in the model.

Total solids

Cross-validated model performance			Features included		
R^2	RMSE (s)	nRMSE (%)	texture	color (quantitative)	foam height
0.57	5.81	62.97	■	■	■
0.54	5.98	64.82	■	■	
0.49	6.33	68.64	■		

This analysis was performed for every model type and every predictor. Final summary tables are presented for every model in Tables S4.7 (random forest), S4.8 (linear regression), and S4.9 (decision tree).

Table S4.7. Random forest model performance tables for comparison of model inputs, based on reduced dataset (R^2 do not match best models developed with expanded dataset in every case). Black rectangles indicate the inclusion of an input in the model.

Supernatant turbidity (settling efficiency)

Cross-validated model performance			Features included		
R^2	RMSE (NTU)	nRMSE (%)	supernat. color	texture	pH
0.57	183.21	80.33	■	■	■
0.53	190.81	83.67	■	■	
0.49	195.37	85.67	■		

CST (filtration time)

Cross-validated model performance			Features included	
R^2	RMSE (s)	nRMSE (%)	supernat. color	texture
0.56	145.87	96.83	■	■
0.42	165.61	109.93	■	

Total solids in dewatered cake

Cross-validated model performance			Features included			
R^2	RMSE (% ds)	nRMSE (%)	EC	foam height	texture	supernat. color
-0.03	12.61	78.54	■	■	■	■

COD

Cross-validated model performance			Features included	
R^2	RMSE (g/L)	nRMSE (%)	foam height	containment type
0.31	52.61	53.02	■	■
0.22	56.10	56.54	■	

NH_4^+-N

Cross-validated model performance			Features included		
R^2	RMSE (g/L)	nRMSE (%)	EC	texture	pH
0.79	0.70	47.45	■	■	■
0.78	0.71	48.03	■	■	
0.65	0.90	61.01	■		

Total solids

Cross-validated model performance			Features included		
R^2	RMSE (s)	nRMSE (%)	texture	color (quantitative)	foam height
0.57	5.81	62.97	■	■	■
0.54	5.98	64.82	■	■	
0.49	6.33	68.64	■		

Volatile solids

Cross-validated model performance			Features included	
R^2	RMSE (% of TS)	nRMSE (%)	color (qualitative)	odor
0.07	17.44	32.12	■	■
0.07	17.48	32.21	■	

TOCsolids

Cross-validated model performance			Features included			
R^2	RMSE (% of TS)	nRMSE (%)	pH	color (quantitative)	foam height	texture
0.13	2.63	24.38	■	■	■	■
0.10	2.67	24.74	■	■	■	
0.07	2.72	25.18	■	■		

TKNsolids

Cross-validated model performance			Features included
R^2	RMSE (% of TS)	nRMSE (%)	odor
0.01	1.21	52.53	■

Table S4.8. Linear model performance tables for comparison of model inputs, based on reduced dataset (R^2 do not match best models developed with expanded dataset in every case). Black rectangles indicate the inclusion of an input in the model.

Settling efficiency (supernatant turbidity)

Cross-validated model performance			Features included		
R^2	RMSE (NTU)	nRMSE (%)	supernatant color	pH	source
0.54	187.18	82.08	■	■	■
0.47	199.63	87.53	■		

CST (filtration time)

Cross-validated model performance			Features included	
R^2	RMSE (s)	nRMSE (%)	supernatant color	foam height
0.42	165.25	109.69	■	■
0.38	169.85	112.74	■	

Total solids in dewatered cake

Cross-validated model performance			Features included	
R^2	RMSE (NTU)	nRMSE (%)	foam height	texture
-0.05	12.94	80.59	■	■

COD

Cross-validated model performance			Features included	
R^2	RMSE (NTU)	nRMSE (%)	containment type	foam height
0.33	52.02	52.43	■	■
0.19	57.09	57.54	■	

NH_4^+-N

Cross-validated model performance			Features included			
R^2	RMSE (NTU)	nRMSE (%)	EC	texture	pH	foam height
0.80	0.68	45.96	■	■	■	■
0.79	0.69	46.72	■	■	■	
0.77	0.73	48.93	■	■		
0.73	0.79	53.54	■			

TS

Cross-validated model performance			Features included		
R^2	RMSE (% of TS)	nRMSE (%)	texture	containment type	color (quantitative)

0.57	5.81	63.03	■	■	■
0.55	5.95	64.53	■	■	
0.46	6.52	70.74	■		

VS

Cross-validated model performance			Features included		
R^2	RMSE (% of TS)	nRMSE (%)	pH	source	color (quantitative)
0.12	16.92	31.17	■	■	■
0.10	17.19	31.68	■	■	

TOCsolids

Cross-validated model performance			Features included		
R^2	RMSE (NTU)	nRMSE (%)	color (quantitative)	odor	water connection
0.13	2.64	24.44	■	■	■
0.10	2.67	24.76	■	■	

TKNsolids

Cross-validated model performance			Features included	
R^2	RMSE (NTU)	nRMSE (%)	pH	odor
0.04	1.19	51.71	■	■

Table S4.9. Simple decision tree model performance tables. Rows highlighted in yellow are models with $R^2 > 0.2$. Rows highlighted in grey are the highest performing models for a target parameter that have $R^2 \leq 0.2$.

Supernatant turbidity (settling efficiency)

Simple decision tree performance			n	Predictor
R^2	RMSE (NTU)	nRMSE (%)		
0.22	282.42	104.00	202	Containment type
0.15	294.33	108.78	203	Toilet type
-0.03	326.05	120.51	203	Water connection
-0.10	338.34	125.05	203	Source
0.00	286.98	124.86	168	Odor
-0.07	322.58	121.89	222	Color (qualitative)

Appendix B

CST (filtration time)

Simple decision tree performance			n	Predictor
R^2	RMSE (s)	nRMSE (%)		
0.40	195.73	112.13	196	Containment type
0.36	204.40	117.59	197	Toilet type
-0.06	264.21	152.00	197	Water connection
-0.10	270.48	155.61	197	Source
0.01	248.52	165.58	148	Odor
-0.15	272.48	157.81	212	Color (qualitative)

TS in dewatered cake

Simple decision tree performance			n	Predictor
R^2	RMSE (% ds)	nRMSE (%)		
-0.08	13.00	77.71	179	Containment type
-0.06	12.97	77.52	180	Toilet type
-0.06	12.95	77.39	180	Water connection
-0.09	13.12	78.40	180	Source
-0.11	13.80	85.10	158	Odor
-0.08	13.02	78.09	197	Color (qualitative)

COD

Simple decision tree performance			n	Predictor
R^2	RMSE (g/L)	nRMSE (%)		
0.09	61.65	62.46	298	Containment type
0.12	60.36	61.47	297	Toilet type
0.01	64.25	65.43	297	Water connection
-0.03	65.55	66.42	299	Source
0.09	61.99	61.41	288	Odor
0.01	65.79	65.76	331	Color (qualitative)

NH4-N

Simple decision tree performance			n	Predictor
R^2	RMSE (g/L)	nRMSE (%)		
0.51	1.18	66.70	385	Containment type
0.47	1.22	69.43	386	Toilet type
0.02	1.67	94.73	386	Water connection
-0.07	1.74	98.88	387	Source
0.18	1.37	94.68	287	Odor
0.12	1.59	89.48	429	Color (qualitative)

TS

Simple decision tree performance			n	Predictor
R^2	RMSE (% ds)	nRMSE (%)		
0.24	7.81	76.66	362	Containment type
0.20	7.98	78.56	363	Toilet type
-0.06	9.24	90.92	363	Water connection
-0.04	9.21	90.23	364	Source
0.00	8.87	94.73	287	Odor
-0.04	9.05	88.15	405	Color (qualitative)

VS

Simple decision tree performance			n	Predictor
R^2	RMSE (% of TS)	nRMSE (%)		
-0.03	18.31	33.21	352	Containment type
-0.02	18.22	33.04	353	Toilet type
-0.03	18.29	33.16	353	Water connection
-0.04	18.37	33.34	354	Source
-0.01	17.97	33.23	284	Odor
0.06	17.20	30.95	397	Color (qualitative)

TOC_{solids}

Simple decision tree performance			n	Predictor
R^2	RMSE (% of TS)	nRMSE (%)		
-0.09	2.96	27.39	247	Containment type
-0.09	2.96	27.36	247	Toilet type
-0.07	2.94	27.12	247	Water connection
-0.09	2.97	27.44	249	Source
-0.01	2.83	25.85	273	Odor
-0.07	2.83	26.05	273	Color (qualitative)

TKN_{solids}

Simple decision tree performance			n	Predictor
R^2	RMSE (% of TS)	nRMSE (%)		
-0.04	1.24	54.71	247	Containment type
-0.04	1.24	54.88	247	Toilet type
-0.06	1.25	55.16	247	Water connection
-0.06	1.24	55.10	249	Source
-0.02	1.24	54.83	273	Odor
-0.02	1.24	55.00	273	Color (qualitative)

Table S4.10. Normalized mean squared error (nRMSE) of the best performing models generated with expanded datasets. nRMSE is calculated by dividing RMSE by the mean of the observed data.

Laboratory-based measurements (target parameters)			nRMSE (%)
	Supernatant turbidity	decision tree	104
		linear	82
		random forest	80
	CST	decision tree	112
		linear	109
		random forest	97
	TS in dewatered cake	decision tree	78
		linear	81
		random forest	79
	COD	decision tree	62
		linear	52
		random forest	53
	NH4-N	decision tree	67
		linear	46
		random forest	47
	TS	decision tree	77
		linear	63
		random forest	63
	VS	decision tree	31
		linear	31
		random forest	32
	TOC_{solids}	decision tree	26
		linear	24
		random forest	24
	TKN_{solids}	decision tree	55
		linear	52
		random forest	53

TS prediction for conditioner dosing

Prediction of accuracy of batch conditioner dosing operation, based on the setup of the Sanergy transfer station in Nairobi, Kenya.
Assumptions:

- Dosage of polymer flocculant is based on TS of influent sludge
- Influent sludge has TS of 2% dry solids (20 g/L)
- Influent volume of 2000 L of fecal sludge per batch
- Target flocculant dose is 2 mL/g TS
- Density of this fecal sludge is same as water (1 g/L)

Prediction error of best predictive model for TS is ± 63%, based on nRMSE (Table S4.6).

Calculations:

Low bound on TS prediction = 2% ds - (0.63 * 2% ds) = 0.74% ds, or 7.4 g/L TS
High bound on TS prediction = 2% ds + (0.63 * 2% ds) = 3.26% ds or 32.6 g/L TS

Low bound on total mass TS in batch = 7.4 g/L TS * 2000 L = 14,800 g TS
High bound on total mass TS in batch = 32.6 g/L TS * 2000 L = 65,200 g TS

Low volume of flocculant added = 2 mL floc/g TS * 14,800 g TS = 29,600 mL floc
High volume of flocculant added = 2 mL floc/g TS * 65,200 g TS = 130,400 mL floc

Actual dose range:
Lowest dose = 29,600 mL floc / (20 g/L TS * 2000 L TS) = **0.74 mL/g TS**
Highest dose = 130,400 mL floc / (20 g/L TS * 2000 L TS) = **3.26 mL/g TS**

Appendix C

Supplemental Information

Particle size as a driver of dewatering performance and its relationship to stabilization in fecal sludge

Submitted

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Link to open dataset

The complete dataset used for this study is accessible at this link:

https://10.25678/000702

Questionnaire

Sample ID (##-city-DDMMYYYY)

*A unique number for each sampling location. Format: Sample number-first letter of city-Date (for example, 09-K-31012021).

____

GPS location: ____

Questions for observer

Location

1. What is the neighborhood/local area name?

____

Type of establishment

2. What type of establishment does the containment serve?
    - ☐ Household (single or multiple)
    - ☐ School
    - ☐ Commercial (includes hotel, restaurant, mall, shop)
    - ☐ Factory/industry
    - ☐ Office building (includes municipal offices)
    - ☐ Public toilet (includes market, ablution block)
    - ☐ House of worship
    - ☐ Other (specify): ____

User information

3. Describe toilet users (select appropriate questions for each type of establishment)
    3.1. Household
    - Is this toilet shared between multiple households?
        - ☐ Yes
        - ☐ No

- What is the number of users? This is the average total number of users per day who use the containment. (So if multiple toilets go into one containment, add up the number of users of each of these toilets).________________________________
- Do people pay to use this toilet?
  - ☐ Yes
  - ☐ No
- Primary occupation of the head of the household ____________________________________
- Highest educational qualification of the head of household
  - ☐ None
  - ☐ Primary education
  - ☐ Secondary education
  - ☐ Tertiary education
  - ☐ Postgraduate education or higher
  - ☐ I don't want to say
  - ☐ Other (specify) ________________________________
- Do you rent or own your home?
  - ☐ Rent
  - ☐ Own
- How many rooms in your home? ________________________________
- How many occupants in your household? _________________________
- Which of the following items are owned by at least one member of the household? (Select all that apply)
  - ☐ Automobile
  - ☐ Motor bike
  - ☐ Bicycle
  - ☐ Television
  - ☐ Refrigerator
  - ☐ Mobile phone

3.2. School
- How many students and staff? ______________
- How many days per week is the school open? _________

3.3. Factory/industry
- How many employees work here per day? ________________
- How many days per week is the factory/industry open? ___________

3.4. Office building
- How many employees work here per day? ___________
- How many days per week is the office open? __________

3.5. Commercial (restaurant, hotel, mall, or shop)
- How many employees work here per day? ________
- How many guests per week? ________
- Does number of guests change seasonally?
  - ☐ Yes
  - ☐ No

3.6. House of worship
- How many people attend services per week? ________
- How many people use the building for other events per week? _______________

3.7. Public toilet
- How many people use the toilets per week? ____________
- How many days per week is the facility open? _____________
- What are the hours of operation? __________
- What is the cost per use? ___________
- What is the weekly revenue? ___________

- Are there showers?
  - ☐ Yes
  - ☐ No
- Is this a primary toilet for any users? If so, for what percent of users? ____________

3.8. Other (specify): _________

- How many people use the toilet per week?__________

Describe the onsite containment:

4. How would you classify this type of containment?
   - ☐ Pit latrine
   - ☐ Septic tank
   - ☐ Holding tank
   - ☐ Cesspit / leach pit
   - ☐ Other (specify): ____
5. Describe the containment's lining:
   - ☐ Fully lined (watertight)
   - ☐ Partially lined (water permeable)
   - ☐ Unlined
   - ☐ I don't know

   5.1. If fully or partially lined, what material is the lining made of?
   - ☐ Concrete
   - ☐ Fiberglass
   - ☐ PVC or plastic
   - ☐ Brick
   - ☐ Stone
   - ☐ Cinderblocks
   - ☐ Wood
   - ☐ Other (specify):_________________________________
6. Do you notice a change in the liquid level inside the containment between rainy and dry seasons?
   - ☐ Yes
   - ☐ No
   - ☐ I don't know

   6.1. If Yes, what is the difference?
   - ☐ Liquid level is higher during the rainy season
   - ☐ Liquid level is lower during the rainy season
   - ☐ Other (specify): ____________________
7. Does the containment have one or more baffles?
   - ☐ Yes
   - ☐ No
   - ☐ I don't know
8. Does the system have an outflow?
   - ☐ Yes
   - ☐ No
   - ☐ I don't know

   8.1. If Yes, where does the tank drain to?
   - ☐ Open drain
   - ☐ Sewer
   - ☐ Soak pit
   - ☐ I don't know
   - ☐ Other (specify): _________________________________
9. Provide any additional information about the containment:

Appendix C

Description of toilet technology

10. What type of toilet(s) feed into this containment system? (select all that apply)
    - ☐ Cistern flush toilet
    - ☐ Pour-flush toilet
    - ☐ Dry toilet
    - ☐ Urine-diverting toilet

Wastewater and solid waste

11. What type of anal cleansing material is used? (select all that apply)
    - ☐ Toilet paper
    - ☐ Water
    - ☐ Other (specify): ___________

    11.1. Is toilet paper or other anal cleansing material disposed of into the toilet?
    - ☐ Yes
    - ☐ No
    - ☐ I don’t know
12. Is anything added to the containment to help with smell or improved degradation?
    - ☐ Yes
    - ☐ No
    - ☐ I don’t know

    12.1. If Yes, what is added?
    - ☐ Bio additives/enzymes
    - ☐ Lime
    - ☐ Ash
    - ☐ Other (specify):_________________

    12.2. Specify brand name if possible: _________________________

    12.3. How often is it added, and how much is added every time? ____
13. Are cleaning chemicals disposed of into the toilet?
    - ☐ Yes
    - ☐ No
    - ☐ I don’t know

    13.1. If Yes, what chemicals? Specify brand names if possible: ____
14. Does solid waste enter the faecal sludge containment system?
    - ☐ Yes
    - ☐ No
    - ☐ I don’t know

    14.1. If Yes, what type(s) of solid waste? ________________________________
15. What types of wastewater enter the containment system? (select all that apply)
    - ☐ Toilet
    - ☐ Bathing
    - ☐ Laundry
    - ☐ Kitchen
    - ☐ I don’t know

Water accessibility

16. Is there a water connection on the premesis?
    - ☐ Yes
    - ☐ No
    - ☐ I don’t know

17. Where do you get your water?
    - ☐ Tap inside building
    - ☐ Standpipe outside building
    - ☐ Other (specify): _________________________
    - ☐ I don't know
18. If you get your water outside of the building, what is the walking distance to the standpipe or water source? *(specify distance in meters)*: ____________________

Quality of construction

19. How old is the containment?
    - ☐ Less than 1 year old
    - ☐ 1-5 years old
    - ☐ 6-10 years old
    - ☐ More than 10 years old
    - ☐ I don't know
20. Who constructed your containment system?
    - ☐ Professional engineer
    - ☐ Technician/mason
    - ☐ Myself, or family/friends
    - ☐ I don't know

Accumulation rate

21. What is the volume of the containment system? (*specify L or m3*) ___________
22. When was the system last emptied? (You can write either the month+year when it was last emptied, or the time since last emptying event.) ________________________
23. Was it fully emptied at that time?
    - ☐ Yes
    - ☐ No
    - ☐ I don't know
    - ☐ System has never been emptied
24. What is the typical emptying interval of your containment system?
    - ☐ Days
    - ☐ Weeks
    - ☐ Months
    - ☐ Every year
    - ☐ Every ______ years (specify)
    - ☐ I don't know

Photographs of sampling campaign

Figure S6.5. Sample collection in Kampala, Uganda

Figure S6.6. Sample collection in Naivasha, Kenya

Presence/Absence of aggregates in relation to time between emptying

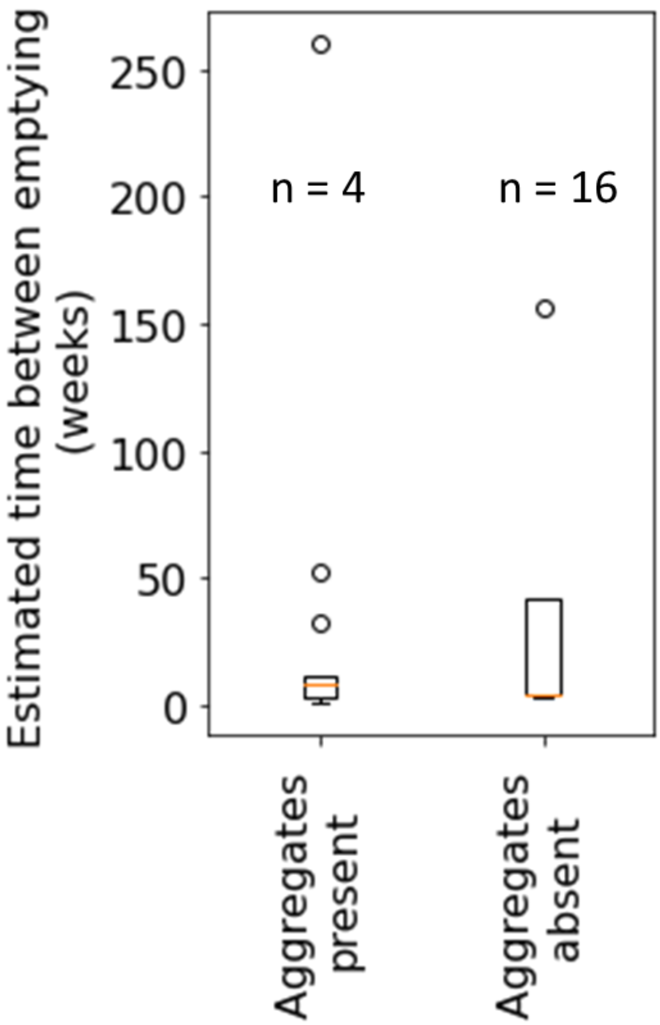

Figure S6.7. Boxplot illustrating the estimated time between emptying for containments where aggregates were present or absent in the fecal sludge.

Appendix C

Microbial community analysis: Beta Diversity

We divided field samples into groups based on aggregate size (D50) and percent of particles smaller than 10 μm. We also looked at community differences between weeks of anaerobic digestion. Groups were broken down as follows:

Aggregate size: The largest aggregates: “High.D50” (top 25% of measured D50), medium aggregates: “Mid.D50” (middle 50% of D50), and smallest aggregates: “Low.D50” (smallest 25% of D50).

Percent of small particles: The most small particles: “Many.small” (top 25% of solids <10 μm), medium amount of small particles: “Moderate” (middle 50% of solids <10 μm), the fewest small particles: “Few.small” (bottom 25% of solids <10 μm).

Time under anaerobic storage: Undigested samples are “WK.0”, samples taken after 3 weeks of digestion are “WK.3”, and samples taken after 7 weeks of digestion are “WK.7”.

Statistical tests of beta diversity were performed by Novogene. Figures included here were generated by Novogene.

Analysis of similarity (Anosim)

Anosim is a nonparametric test to determine whether differences between groups are significantly larger than differences within groups. R-values range from -1 to 1. Positive R-value means the variation within a group is smaller than the variation between groups. The P-value represents the degree of confidence. When the P-value is less than 0.05, it suggests that the result is statistically significant to a 95% confidence value.

Table S6.10. Anosim analysis results.

Groups	R-value	P-value
High.D50-Mid.D50	0.2956	0.022
Low.D50-Mid.D50	-0.06909	0.645
Low.D50-High.D50	0.684	0.011
WK.7-WK.0	0.68	0.015
WK.7-WK.3	0.34	0.038
WK.3-WK.0	-0.116	0.87
Many.small-Few.small	0.424	0.011
Moderate-Few.small	-0.05382	0.662
Moderate-Many.small	-0.01709	0.396

Differential abundance analysis

Three metrics of differential abundance analysis were selected to evaluate which specific microbial OTUs, genera, families, phyla, etc. were differently abundant between groups: t-test, Metastat, and linear discriminant analysis (LDA) Effect Size (LefSe). We used the results of differential abundance analysis to identify species with significant variation between groups, at a confidence value of 95%. When all three methods identified the same species/genus/family, etc., we presented these in the article as possibly characteristic of differences between groups, and thus potentially associated with physical differences between groups.

Aggregate size

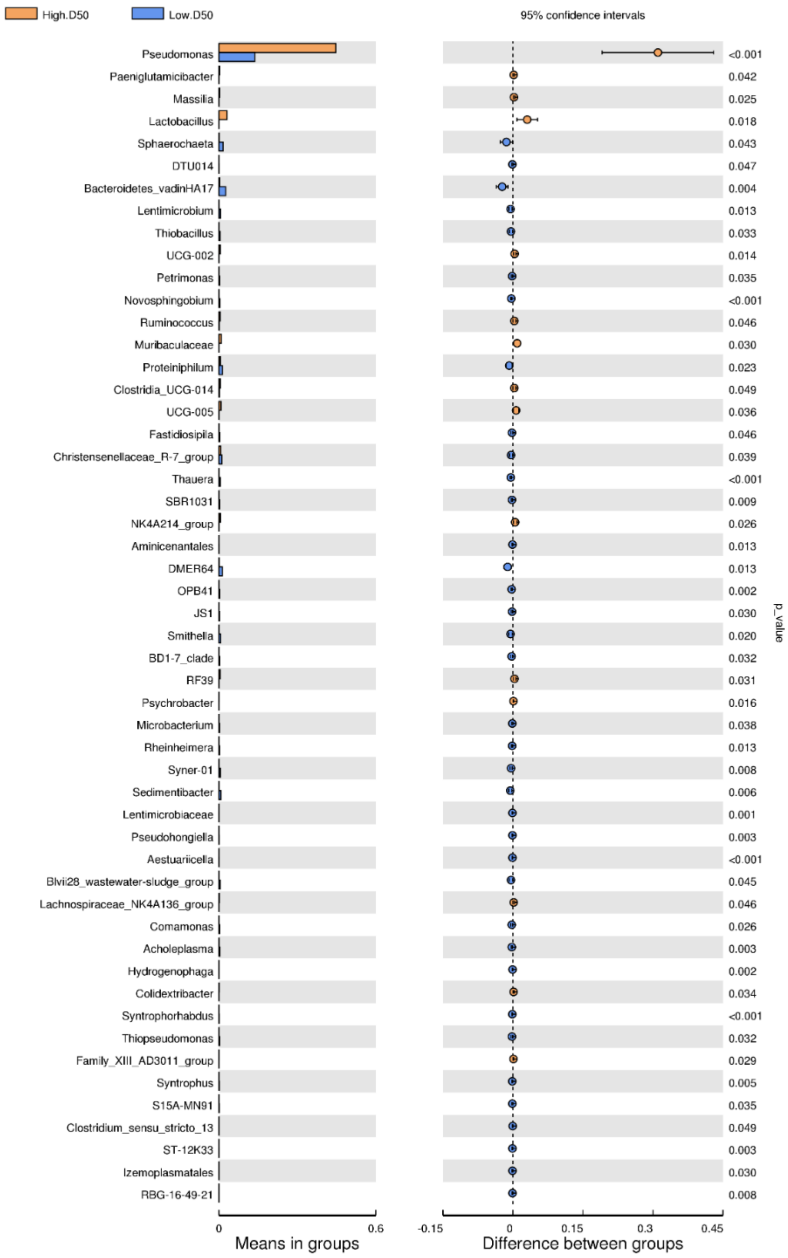

Figure S6.8. T-test results for samples with the largest (High.D50) and smallest (Low.D50) aggregates, at the genus-level.

Appendix C

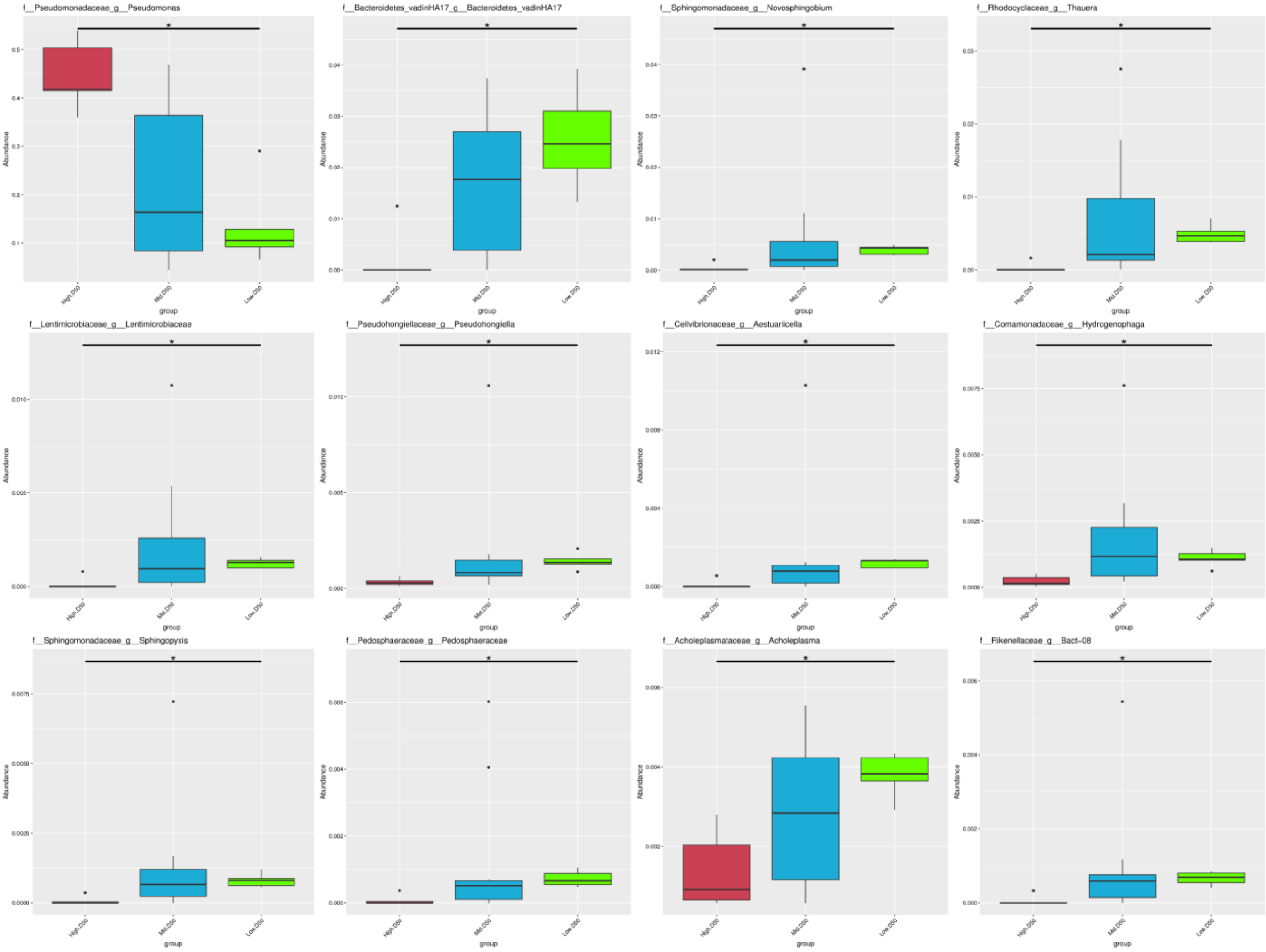

Figure S6.9. Metastat results for samples with the largest (High.D50), median (Mid.D50), and smallest (Low.D50) aggregates, at the genus-level.

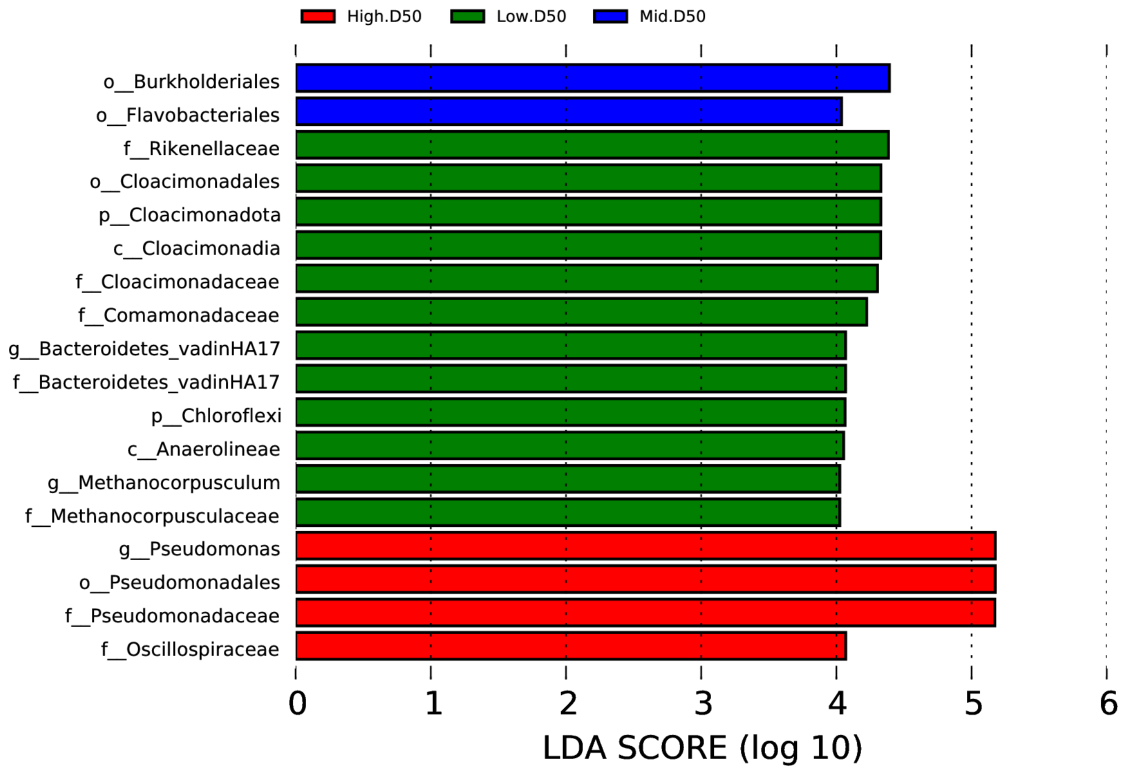

Figure S6.10. LefSe results for samples with the largest (High.D50), median (Mid.D50), and smallest (Low.D50) aggregates.

Appendix C

Percent of small particles

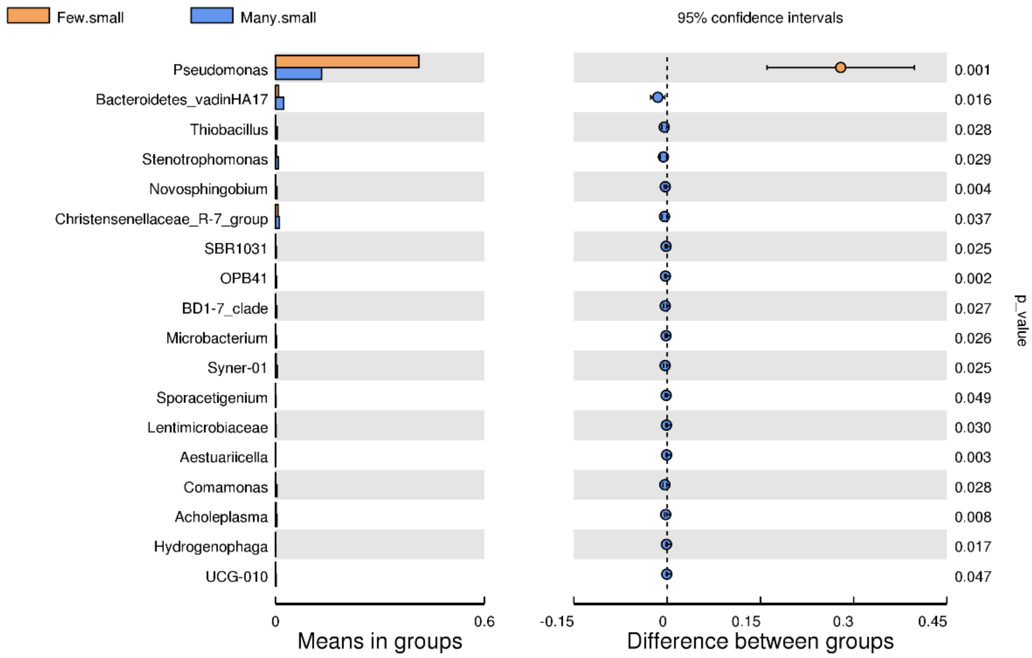

Figure S6.11. T-test results for samples with the highest proportion of small particles <10 μm (Many.small), and smallest proportion (Few.small), at the genus-level.

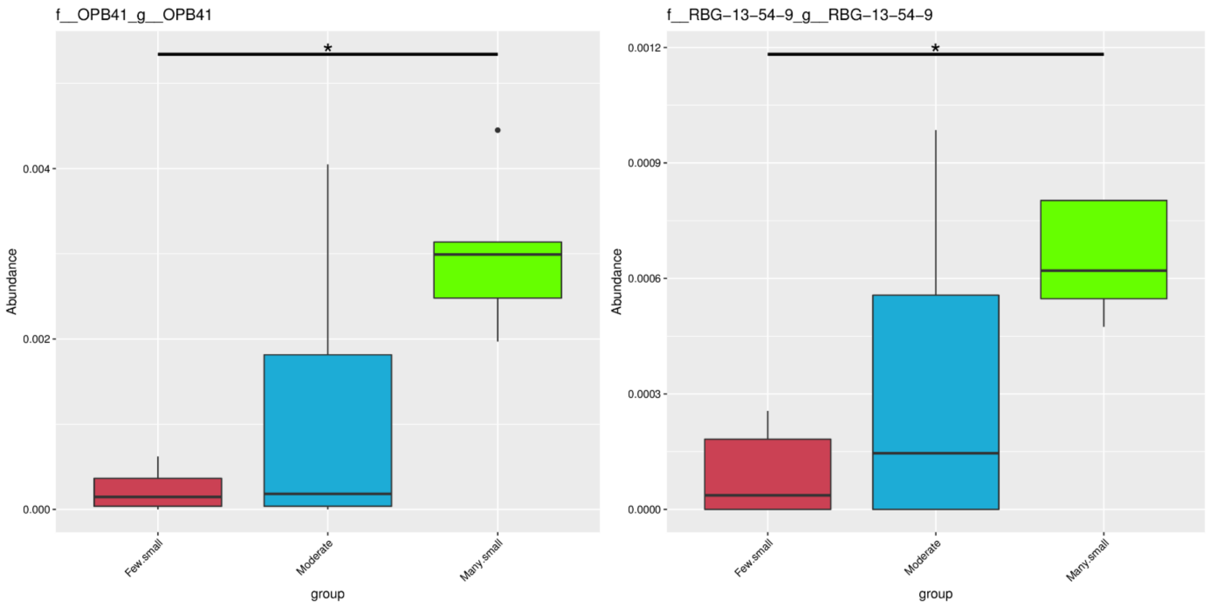

Figure S6.12. Metastat results for samples with the highest proportion of small particles <10 μm (Many.small), median proportion (Moderate), and smallest proportion (Few.small), at the genus-level.

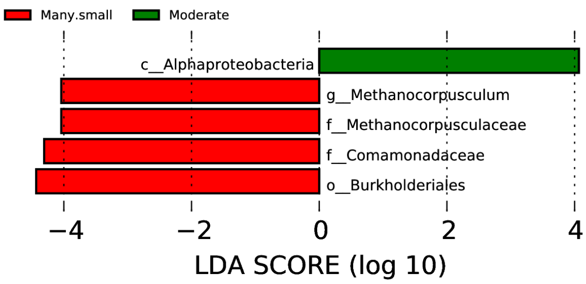

Figure S6.13. LefSe results for samples with the highest proportion of small particles <10 µm (Many.small), median proportion (Moderate), and smallest proportion (Few.small), at the genus-level.

Time under anaerobic storage

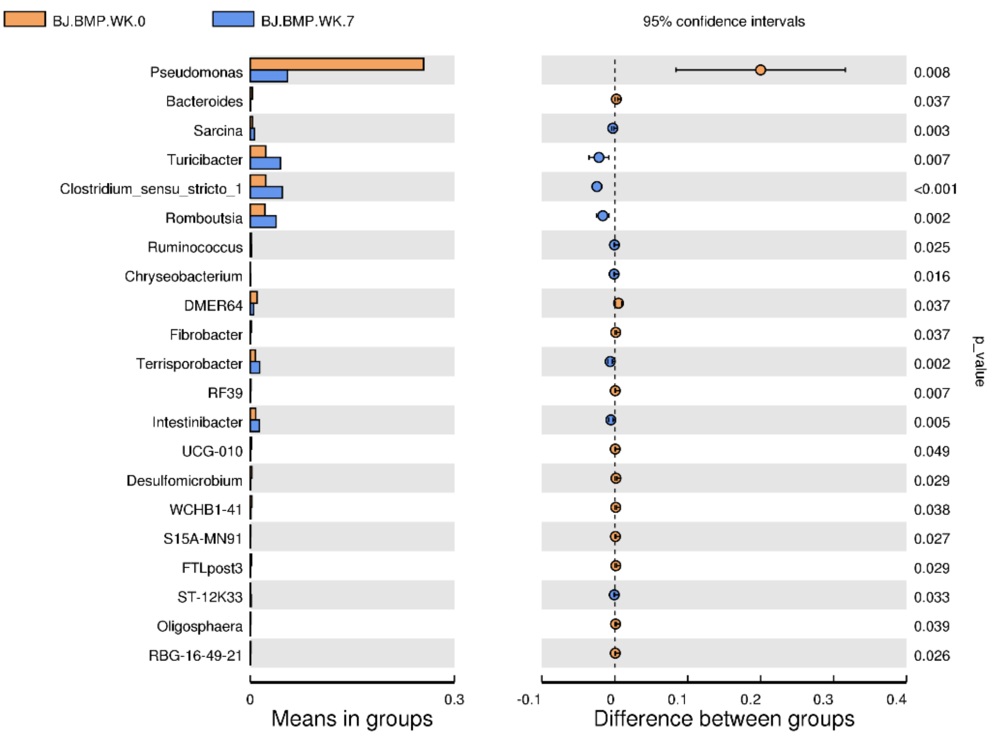

Figure S6.14. T-test results for undigested samples (BJ.BMP.WK.0) and samples digested for 7 weeks under anaerobic conditions (BJ.BMP.WK.7), at the genus-level.

Appendix C

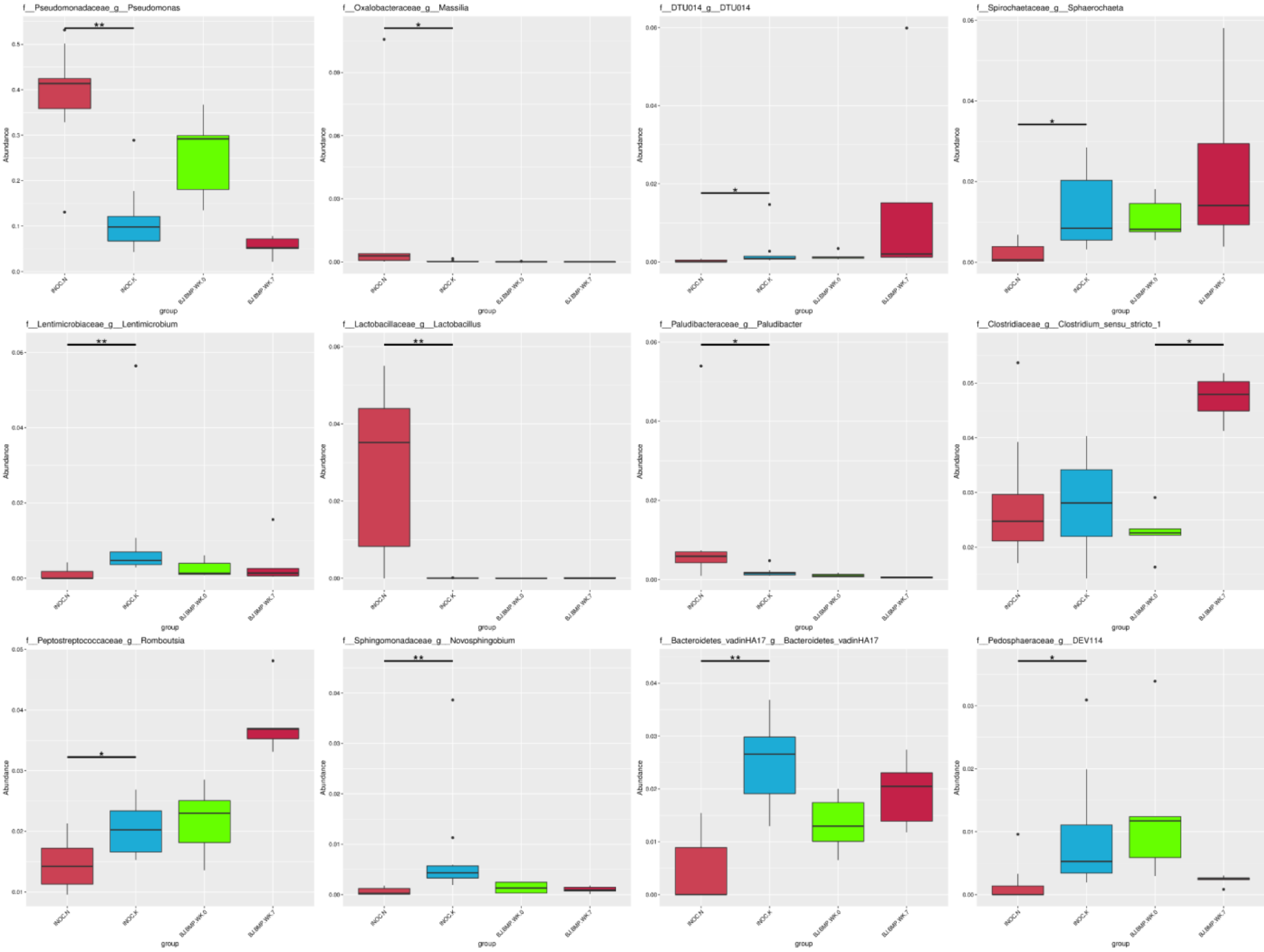

Figure S6.15. Metastat results for undigested samples (BJ.BMP.WK.0) and samples digested for 7 weeks under anaerobic conditions (BJ.BMP.WK.7), at the genus-level. Differences between sludge from Naivasha (INOC.K) and Kampala (INOC.K) are also presented.

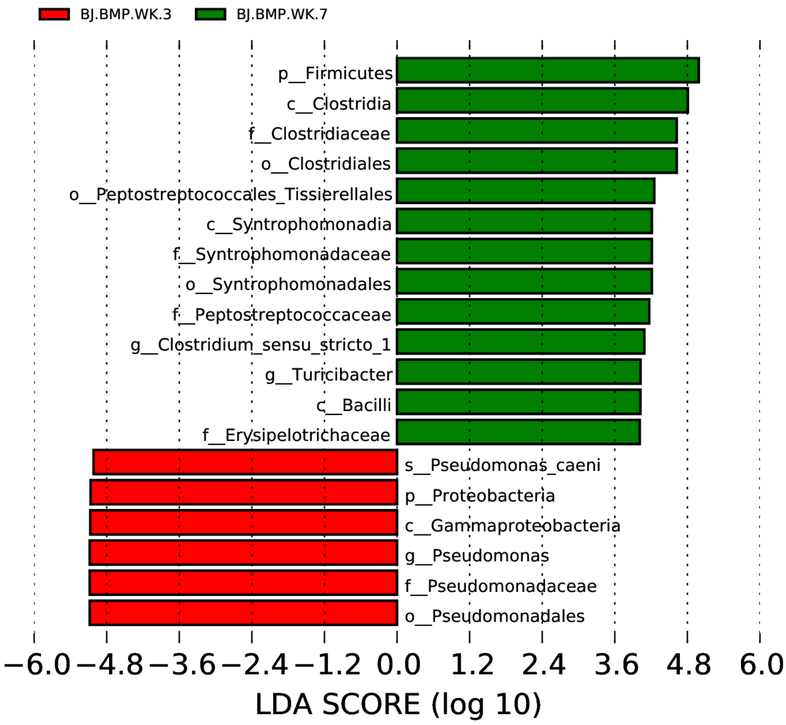

Figure S6.16. LefSe results for undigested samples (BJ.BMP.WK.0), samples digested for 3 weeks under anaerobic conditions (BJ.BMP.WK.3) and samples digested for 7 weeks under anaerobic conditions (BJ.BMP.WK.7).

Stabilization metrics and time since last emptied

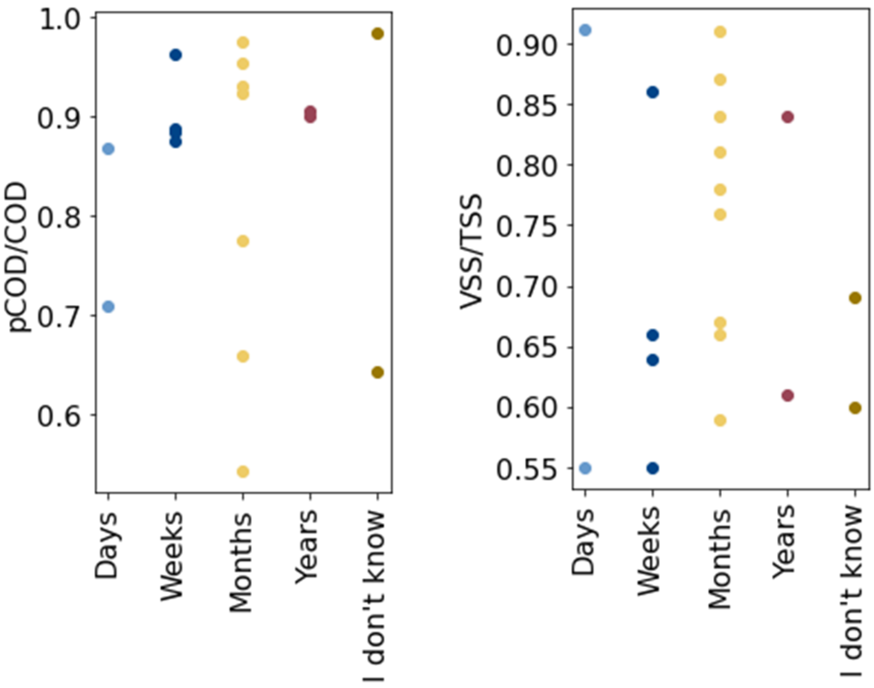

Figure S6.17. Plots showing the stabilization metrics pCOD/COD and VSS/TSS broken down by time since containment was last emptied.

Changes in stabilization indicators during controlled anaerobic storage

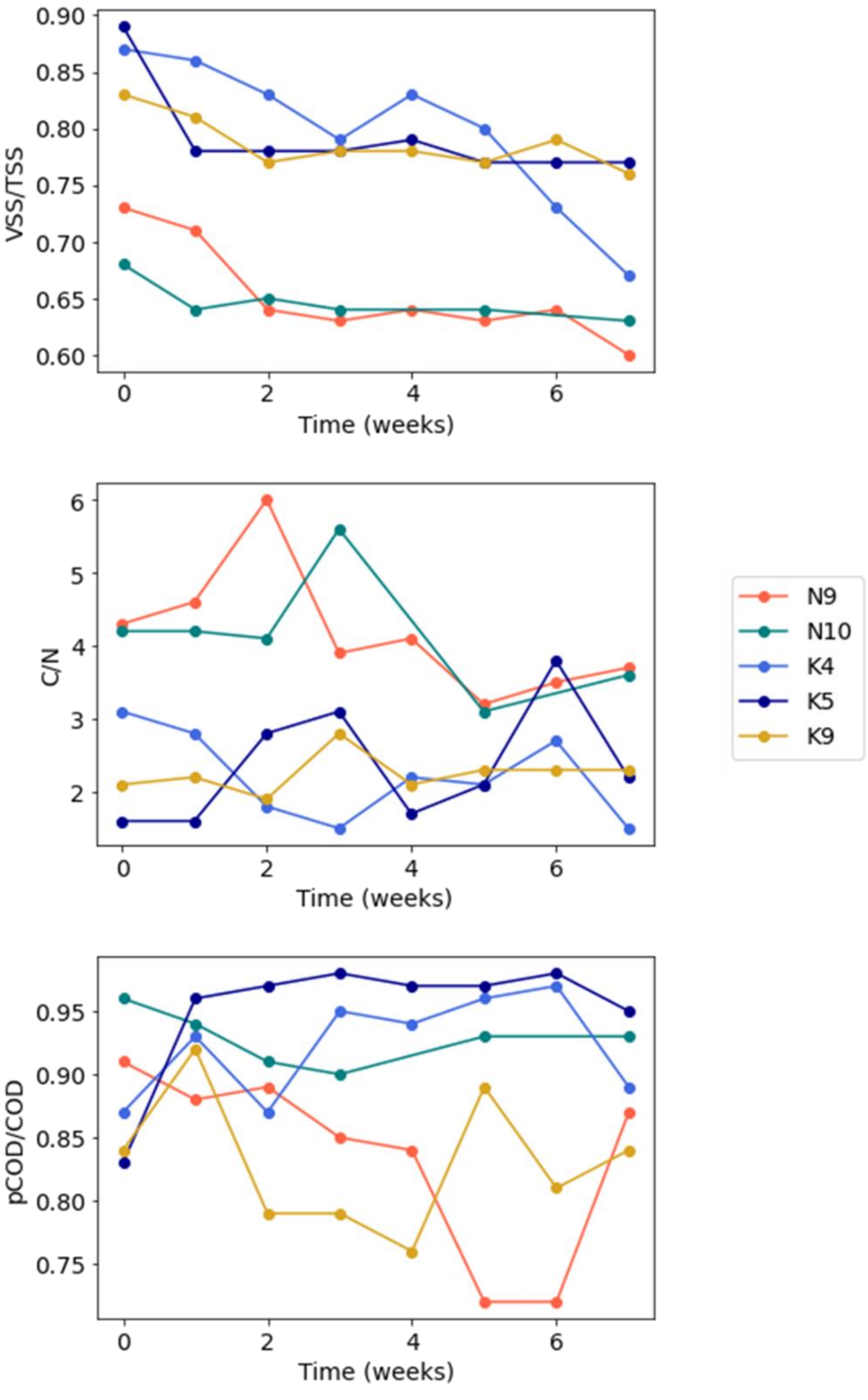

Figure S6.18. Weekly values of stabilization indicators as fecal sludge field samples are incubated under controlled anaerobic conditions.

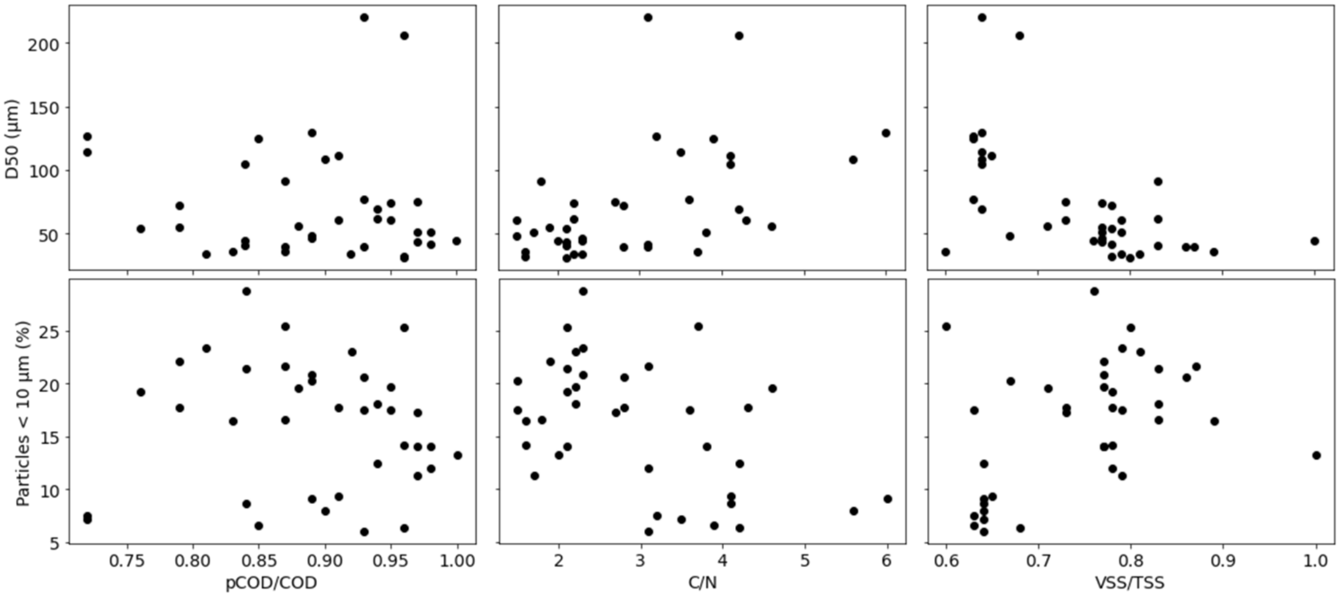

Figure S6.19. Scatterplots illustrating the relationships between stabilization indicators and median aggregate size (D50) and the percent of particles smaller than 10 µm for samples over the entire course of anaerobic storage.

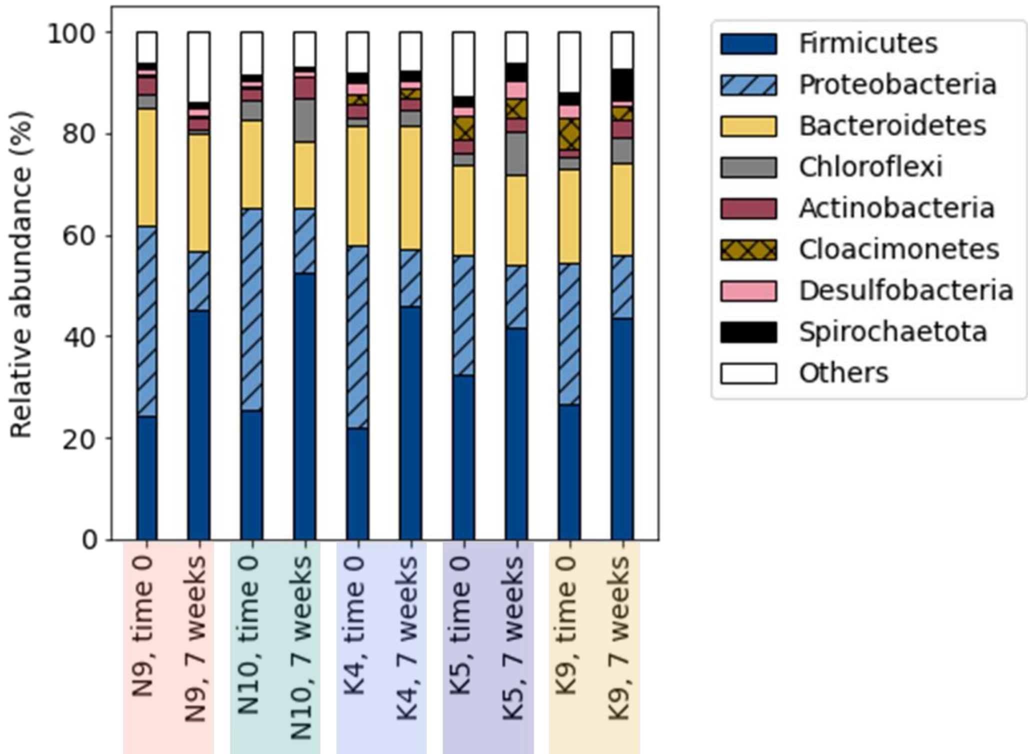

Figure S6.20. Relative abundance of the top 8 phyla present in controlled anaerobic storage reactors inoculated with five field samples (N9, N10, K4, K5, and K9) at time 0 (before incubation), and after 7 weeks in a shaking incubator at 37°C under anaerobic conditions.

Curriculum Vitae

Barbara J Ward

Environmental Engineer, M.S.

PhD Candidate

Nationality: USA

EDUCATION

Dec 2017 – June 2022	**PhD Candidate Environmental Engineering** **ETH Zurich**, Zurich, Switzerland Thesis: Settling and Dewatering of Faecal Sludge: Building Fundamental Knowledge for Improved Global Sanitation Advisor: Eberhard Morgenroth Supervisor: Linda Strande
2013	**MSE Environmental Engineering** **Graduate Certificate, Engineering for Developing Communities** **University of Colorado Boulder**, Boulder, Colorado, USA Thesis: Carbonized feces from the Sol-Char Toilet for use as a solid fuel Adviser: Lupita Montoya
2009	**BS Chemical Engineering** **North Carolina State University**, Raleigh, North Carolina, USA

PROFESSIONAL EXPERIENCE

Dec 2017 - present	**PhD Researcher** studying faecal sludge dewatering and characterization Management of Excreta, Wastewater, and Sludge Research Group Sandec – Department of Sanitation, Water, and Solid Waste for Development Eawag – Swiss Federal Institute of Aquatic Science and Technology Dübendorf, Switzerland
April 2016 – Nov 2017	**Project Manager** for Water Hub @ NEST blackwater dewatering for resource recovery Management of Excreta, Wastewater, and Sludge Research Group Sandec – Department of Sanitation, Water, and Solid Waste for Development Eawag – Swiss Federal Institute of Aquatic Science and Technology Dübendorf, Switzerland
Oct 2014 – Feb 2016	**Project Manager** for Renewable Hydrogen Solarthermal Redox Water Splitting project Department of Chemical and Biological Engineering University of Colorado Boulder Boulder, Colorado, USA
Jan 2014 – Oct 2014	**Professional Research Assistant** for BMGF Reinvent the Toilet Sol-Char Toilet project Department of Environmental Engineering University of Colorado Boulder Boulder, Colorado, USA

Aug 2013 – Dec 2013 **Research Assistant** for Faecal Sludge to Biodiesel project
Pivot Works, Ltd. (formerly Waste Enterprisers)
Kumasi, Ghana

SCIENTIFIC PUBLICATIONS

Authored Book Chapters

1. ***Ward, B.J.***, Septien, S., Ronteltap, M., Strande, L. (2021) Experimental design for the development, transfer, scale-up, and optimization of treatment technologies: Case studies of dewatering and drying. Velkushanova, Strande, Ronteltap, Kootatep, Brdjanovic, D. & Buckley, C. (Eds.), Methods for Faecal Sludge Analysis, 85-114. IWA, ISBN: 9781780409115.
2. Penn, R., ***Ward, B.J.***, Strande, L., Maurer, M. (2021) Faecal sludge simulants: review of synthetic human faeces and faecal sludge for sanitation and wastewater research. Velkushanova, Strande, Ronteltap, Kootatep, Brdjanovic, D. & Buckley, C. (Eds.), Methods for Faecal Sludge Analysis, 195-234. IWA, ISBN: 9781780409115.
3. Velkushanova, K., Reddy, M., Zikalala, T., Gumbi B., Archer, C., ***Ward, B.J.***, Andriessen, N., Sam, S., Strande, L. (2021) Laboratory procedures and methods for characterisation of faecal sludge. Velkushanova, Strande, Ronteltap, Kootatep, Brdjanovic, D. & Buckley, C. (Eds.), Methods for Faecal Sludge Analysis, 235-395. IWA, ISBN: 9781780409115.
4. ***Ward, B.J.***, Strande, L. (2019) Chapter 5.2 Conditioning. Englund & Strande (Eds.), Faecal Sludge Management: Highlights and Exercises, 151-162. Dübendorf, Switzerland: Eawag. ISBN: 978-3-906484-70-9.

Refereed Journal Articles

1. ***Ward, B.J.***, Nguyen, M.T., Sam, S.B., Korir, N., Niwagaba, C.B., Morgenroth, E., Strande, L., Particle size as a driver of dewatering performance and its relationship to stabilization in fecal sludge. *Submitted to Water Research.*
2. Sam, S.B., ***Ward, B.J.***, Niederdorfer, R., Morgenroth, E., Strande, L., Elucidating the role of extracellular polymeric substances (EPS) in dewaterability of fecal sludge, and changes during anaerobic storage in onsite sanitation systems. *Submitted to Water Research.*
3. ***Ward, B.J.***, Andriessen, N., Tembo, J.M., Kabika, J., Grau, M., Scheidegger, A., Morgenroth, E., Strande, L. (2021) Predictive models using “cheap and easy” field measurements: Can they fill a gap in planning, monitoring, and implementing fecal sludge management solutions? *Water Research*. https://doi.org/10.1016/j.watres.2021.116997
4. Fisher, R.P., Lewandowski, A., Yacob, T.W., ***Ward, B.J.***, Hafford, L.M., Mahoney, R.B., Oversby, C.J., Mejic, D., Huschutz, D.H., Summers, R.S., Linden, K.G., Weimer, A.W. (2021) Solar Thermal Processing to Disinfect Human Waste. *Sustainability,* 13(9), 4935.
5. Rhodes-Dicker, L., ***Ward, B.J.***, Mwalugongo, W., Stradley, L. (2021) Permeable membrane dewatering of faecal sludge from pit latrines at a transfer station in Nairobi, Kenya. *Environmental Technology.* 1-28. https://doi.org/10.1080/09593330.2020.1870573.
6. Junglen, K., Rhodes-Dicker, L., ***Ward, B.J.***, Gitau, E., Mwalugongo, W., Stradley, L., Thomas, E. (2020) Characterization and prediction of fecal sludge parameters and settling behavior in informal settlements in Nairobi, Kenya. *Sustainability*.
7. ***Ward, B.J.***, Traber, J., Gueye, A., Diop, B., Morgenroth, E., Strande, L. (2019) Evaluation of conceptual model and predictors of faecal sludge dewatering performance in Senegal and Tanzania. *Water Research*, 167, 115101.
8. Andriessen, N., ***Ward, B.J.***, Strande, L. (2019). To char or not to char? Review of technologies to produce solid fuels for resource recovery from faecal sludge. *Journal of Water, Sanitation and Hygiene for Development*, 9 (2), 210-224.
9. Penn, R., ***Ward, B.J.***, Strande, L., Maurer, M. (2018). Review of synthetic human faeces and faecal sludge for sanitation and wastewater research. *Water Research*, 132, 222-240.
10. Hafford, L.M., ***Ward, B.J.***, Weimer, A.W., Linden, K. (2018). Fecal sludge as a fuel: characterization, cofire limits, and evaluation of quality improvement measures. *Water Science and Technology*, *78*(12), 2437-2448.
11. Strande, L., Schoebitz, L., Bischoff, F., Ddiba, D., Okello, F., Englund, M., ***Ward, B.J.***, Niwagaba, C. (2018). Methods to reliably estimate faecal sludge quantities and qualities for the design of treatment technologies and management solutions. *Journal of Environmental Management*, *223*, 898-907.
12. ***Ward, B.J.***, Yacob, T.W., Montoya, L.D. (2014). Evaluation of solid fuel char briquettes from human waste. *Environmental science & technology*, *48*(16), 9852-9858.

Refereed Conference Papers

1. ***Ward, B.J.***, Allen, J., Escamilla, A., Sivick, D., Sun, B., Yu, K., Dahlberg, R., Niu, R., Ward, B.C., Strande, L. Sludge Snap: A machine learning approach to fecal sludge characterization in the field. 42nd WEDC International Conference, Online, Extended abstract and podium presentation, 2021.
2. ***Ward, B.J.***, Gold, M., Turyasiima, D., Studer, F., Getkate, W., Maiteki, J.M., Niwagaba, C.B., Strande, L. SEEK (Sludge to Energy Enterprises in Kampala): Co-processing faecal sludge for fuel production. *40th WEDC International Conference,* Loughborough, UK, Paper and Podium Presentation, 2017.
3. Hafford, L.M., ***Ward, B.J.***, Weimer, A.W., Linden, K. Fecal sludge as a fuel: Characterization, co-fire limits, and two methods to improve its quality. *IWA Specialist Conference on Sludge Management: SludgeTech,* London, UK, Paper and Podium Presentation, 2017.
4. Etter, B., Wittmer, A., ***Ward, B.J.*** Udert, K.M., Strande, L., T.A. Larsen, E. Morgenroth. Water Hub @ NEST: A Living Lab to Test Innovative Wastewater Treatment Solutions. *IWA Small Water and Wastewater Systems & Specialized Conference on Resources-Oriented Sanitation*, Athens, Greece, Paper and Podium presentation, 2016.

CONFERENCE PRESENTATIONS AND PROCEEDINGS

Invited Talks

1. ***Ward, B.J.***, Strande, L. Determining predictors of faecal sludge quantities, qualities, and dewatering performance. *Sanergy Speaker Series*. Nairobi, Kenya. 8 July, 2019.
2. ***Ward, B.J.***, Strande, L. FSM Field Characterization – Experiences from Africa. *Workshop on Sustainable Sanitation – Global & Local Partnerships, Experiences and Way Ahead*. Cape Town, South Africa. 18 February, 2019.
3. ***Ward, B.J.***, Strande, L. End use and resource recovery from faecal sludge – focusing on solid fuels. *Beyond development aid: Sanitation financing & revenue models in reuse (human) waste.* Organized by Institute of Fiscal Studies (IFS), UK & WASTE, Netherlands. The Hague, Netherlands, 15 May, 2017.

Presentations and Proceedings

1. ***Ward, B.J.***, Integrated approach to faecal sludge dewatering. *SuSanA 30th Global Meeting*, Presentation, 2020.
2. ***Ward, B.J.***, Andriessen, N., Strande, L., Estimating quantities and qualities (Q&Q) of faecal sludge. *SuSanA 30th Global Meeting*, Presentation, 2020.
3. **Ward, B.J.**, Sam S.B., Morgenroth, E., Strande, L. Progress in Faecal Sludge Dewatering: Evaluating a conceptual model and predictors of dewatering performance. *Faecal Sludge Management – FSM5*, Cape Town, South Africa. Podium Presentation, 2019.
4. **Ward, B.J.**, Sam, S.B., Morgenroth, E., Strande, L. Water Hub @ NEST: Faecal Sludge Dewatering for Resource Recovery. *Eawag Symposium*, Duebendorf, Switzerland, Podium Presentation, 2018. First place, best presentation.
5. ***Ward, B.J.***, Morgenroth, E., Strande, L. Water Hub @ NEST: Faecal Sludge Dewatering for Resource Recovery. *Eawag Symposium,* Duebendorf, Switzerland, Poster and Flash Presentation, 2017. First place, best presentation.
6. ***Ward, B.J.***, Gold, M., Morgenroth, E., Kimwaga, R., Strande, L. Faecal Sludge Dewatering: New Research Facilities for a Multi Directional Approach. *Faecal Sludge Management – FSM4,* Chennai, India. Podium Presentation, 2017.
7. Fisher, R., Yacob, T.W., Mahoney, R.B., ***Ward, B.J.***, Linden, K.G., Weimer, A.W. Distributed Activation Energy Model for Solar-Driven Pyrolysis of Untreated Human Feces, *American Institute of Chemical Engineers (AIChE) 14th Annual Meeting*, Atlanta, Georgia USA. Podium Presentation, 2014.
8. ***Ward, B.J.***, Montoya, L.D. Investigating energy sources from solar-biochar toilet, *Association of Environmental Engineering & Science Professors 50th Anniversary Conference*, Golden, Colorado USA. Podium Presentation, 2013.

Science Writing and Magazine Articles

1. ***Ward, B.J.***, Andriessen, N., Tembo, J.M., Kabika, J., Scheidegger, A., Morgenroth, E., Strande, L. Smartphone app and predictive models for field characterization of faecal sludge, *Sandec News,* 2021.
2. ***Ward, B.J.***, Sam, S., Kapanda, K., Tembo, J.M., Morgenroth, E., Strande, L. Research strategy for overcoming the faecal sludge dewatering bottleneck, *Sandec News,* 2020.
3. Sam, S., Kapanda, K., ***Ward, B.J.***, Tembo, J., Morgenroth, E., Strande, L. The trials of developing a biomethane potential method for faecal sludge, *Sandec News,* 2020.
4. ***Ward, B.J.***, Sam, S., Gueye, A., Diop, B., Morgenroth, E., Strande, L. Predicting faecal sludge dewatering performance for improved treatment, *Sandec News,* 2019.
5. ***Ward, B.J.***, Sam, S., Andriessen, N., Morgenroth, E., Strande, L. Research Priorities and Progress in Faecal Sludge Dewatering, *Sandec News,* 2018.

6. ***Ward, B.J.***, Morgenroth, E., Strande, L. Dewatering Characterisation of Synthetic Faecal Sludge, *Sandec News,* 2017.

Teaching and Knowledge Dissemination

1. Guest lecturer for Water supply, sanitation and waste infrastructure and services in developing countries ETHZ course, April 2021 (remote classroom virtual lecture via Zoom).
2. Guest lecturer for Water supply, sanitation and waste infrastructure and services in developing countries ETHZ course, April 2020 (remote classroom virtual lecture via Zoom).
3. Expert panelist. IWA Non-Sewered Sanitation Webinar, Understanding the behavior of faecal sludge during thermal drying, November 2019.
4. Workshop co-facilitator and trainer. Faecal sludge Q&Q Kickoff Workshop, Climate-Friendly Sanitation Lusaka Project. Lusaka, Zambia, July 2019.
5. Workshop facilitator and presenter. Moisture content: Solving the Achilles heel of faecal sludge treatment and processing options. Cape Town, South Africa, February 2019.
6. Workshop presenter. Workshop on sustainable sanitation – Global & local partnerships, experiences and way ahead. Cape Town, South Africa, February 2019.
7. Workshop presenter. Characterization and quantification of faecal sludge. Cape Town, South Africa, February 2019.
8. Guest lecturer for Water supply, sanitation and waste infrastructure and services in developing countries ETHZ course, April 2018.
9. Guest lecturer for Sanitary engineering in developing countries EPFL course Env 402, October 2017.
10. Presentations and workshop. City-wide Sanitation Advocacy and Planning: Tools for FSM Diagnostics and Project Planning. Loughborough, UK, July 2017.
11. Presentations and workshop. Citywide Inclusive Sanitation for Bank Operations – Strategic Knowledge and Learning Event. Durban, South Africa, December 2016.
12. Presentations and workshop. Introduction to Faecal Sludge Management workshop. Dar es Salaam, Tanzania, November 2016.
13. Guest lecturer for Sanitary engineering in developing countries EPFL course Env 402, December 2016.
14. Guest lecturer for Water, Sanitation and Hygiene, University of Colorado Boulder course CVEN 5969, September 2015.
15. Guest lecturer for Water, Sanitation and Hygiene, University of Colorado Boulder course CVEN 5969, September 2014.

AWARDS & HONORS

ETH Zurich Engineering for Development (E4D) Doctoral Fellowship, 175,000 CHF	2018 - 2021
Best Student Presentation 2018 Eawag Symposium	2018
Outstanding Reviewer for Environmental Science: Water Research & Technology	2017
Best Student Presentation 2017 Eawag Symposium	2017