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WASH R&D CENTRE

Water Sanitation & Hygiene Research & Development Centre

**Characterisation of faecal material waste from rural schools’
onsite sanitation systems for its safe disposal and optimal
valorisation**

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PREFACE

The experimental work described in this dissertation was conducted at the Water Sanitation & Hygiene, Research & Development Centre, School of Engineering University of KwaZulu-Natal, Durban, South Africa, From February 2022 to February 2023, Under the supervision of Dr Santiago Septien Stringel.

This work has not been submitted in any form for any degree or diploma to any tertiary institution, where use has been made of the work of others, it is duly acknowledged in the text.

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As the candidate's supervisor I agree to the submission of this thesis

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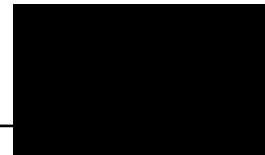
DECLARATION

I Yuri Ramruthan declare that:

1. The research reported in this dissertation, except where otherwise indicated, is my original work.
2. This thesis has not been submitted for any degree or examination at any other university.
3. This data does not contain other person's data, graphs, pictures, or other information, unless specifically acknowledged as being sourced from other persons.
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Y. Ramruthan

Signature



PUBLICATIONS AND PRESENTATIONS

Oral presentation

Ramruthan, Y., Mercer, E., and Septiens, S. Characterisation of faecal material waste from rural schools' Onsite sanitation systems, for its optimal valorisation and safe disposal; IWA-Non Sewered Sanitation, 15th- 18th October, Johannesburg South Africa.

Oral presentation

Ramruthan, Y., Mercer, E., and Septiens, S. Characterisation of faecal material waste from rural schools' Onsite sanitation systems, for its optimal valorisation and safe disposal; IWA young water professionals, 1st-3rd June 2023, Vancouver, British Columbia Canada.

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- First in the SENG Oral category for the 2022 College of Agriculture, Engineering and Science Postgraduate Research and Innovation Symposium (PRIS).
- Won the most Impactful research in the SENG Oral category for the 2022 College of Agriculture, Engineering and Science Postgraduate Research and Innovation Symposium (PRIS).

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DEDICATION

Lord Jesus this song is forever yours; a thousand hallelujahs and a thousand more.

This dissertation is dedicated to my mother. Mom, without your unfailing love, countless sacrifices, and unlimited support, I would not be where I am today. I love you and appreciate everything that you have done for me. My gratitude is eternal.

“You are my reason”.

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ABSTRACT

Poor sanitation facilities in South African rural schools are barricading youth from their education, due to an unsafe learning environment. Inadequate sanitation facilities in South African rural schools not only pose severe health risks but also hinder progress towards achieving the 4th and 6th Sustainable Development Goals (SDGs). This study aims to assess the optimal valorisation of faecal material derived from rural school sanitation systems, including ventilated improved pit latrines (VIPs), mobile toilets (MTs), and septic tanks (STs) in Durban, South Africa. Faecal material, defined as the combination of solid waste, urine, and other gastrointestinal excretions, underwent a thorough analysis of its physico-chemical, thermal, thermodynamic, and mechanical properties. Analytical tests for characterizing faecal material were conducted in line with standard methods utilized by the WASH R&D Centre and the Methods for Faecal Sludge Analysis by Velkushanova et al. (2021). The findings were used to formulate decision matrices for determining the most efficient treatment and emptying strategies based on the characteristics of the faecal material. Notably, all containment systems exhibited comparable energy and nutrient potential suitable for the production of fuels and fertilizers, respectively. Particularly, fresh faeces from MTs exhibited the highest solids concentration (~18% TS), organic fraction (~78.37% Volatile Solids), and calorific value (23.24 MJ/kg dry solids). The opposite was observed for VIPs (~45.47% VS, ~8.5% TS) with a calorific value of (18.4 MJ/kg dry solids). MTs illustrated the highest mean nutrient composition and improved treatability - a lower shear yield stress at the same solid's concentration to the more degraded faecal sludge from VIPs and STs. This inadvertently improved flowability and reduced the pump head requirement for emptying technologies and treatment processes.. Ultimately faeces from MTs were considered to have high strength. VIPs demonstrated the highest TSS, TDS and lowest SVI at 3,6 g/L, 1422 mg/L and 75 ml/g respectively. Together faecal material from all On Site Sanitation systems, exhibited a water activity of ~1, suggesting good dewaterability potential, with results signifying the moisture in the sludge is unbound, so less difficult to remove. Particle size for school toilets ranged between 0,7- 2046,7 μm 0,6 -1202,3 μm and 0,7-1492,5 μm for VIPs, MTs, and STs, respectively. Thermal conductivity for each OSS (MTs, VIPs, STs) represented a narrow range for faecal sludge and faeces ranging between 0,48-0,59 (W/K/m), which were in close approximation to the thermal conductivity of water.

Overall, this study guides engineers and regulators in establishing safe, equitable and sustainable rural school sanitation systems, by promoting resource recovery options such as biochar, fertilizers, and biogas. High strength faecal material from MTs offered the greatest potential for resource recovery, however, requires multiple treatment stages to ensure safe environmental discharge standards are met. They also act as a temporary containment solution, with limited maintenance and negative user perception. The large presence of trash, high TS% and denser faecal material within MTs rendered it most applicable to manual emptying methods (MAPET and manual diaphragm). Consequently, the preferable direction for advancing rural school sanitation, is the adoption of water-borne systems where possible. A transition towards these systems will minimize the spread of diseases associated with non-flush systems (MTs and VIPs). Additionally, it will enhance the overall comfort of female students using school toilets, subsequently reducing the fatalities that have occurred from children drowning in pit latrines. Water-borne sanitation systems, such as STs better facilitate the use mechanical emptying methods, providing a wider array of mechanical emptying options. Furthermore, STs demonstrated viable treatment options owing to its better stabilized waste, like composting and vermicomposting. Moreover, progressive options for areas with no water access, include, contained-based sanitation or re-invented toilets such as composting toilets. This will allow for faeces to be valorised relatively fresh, obtaining direct resource recovery, for high quality end-use products like biogas, and biochar for a fuel source.

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List of abbreviations

AD	Anaerobic Digestion
C: N	Carbon: Nitrogen
COD	Chemical oxygen Demand
CV	Calorific Value
EC	Electrical Conductivity
FS	Faecal Sludge
FSM	Faecal Sludge Management
MC	Moisture Content
MTs	Mobile Toilets
OSS	Onsite Sanitation
PDB	Planted Drying Beds
PSD	Particle Size Distribution
SA	South Africa
SD	Solar Drying
STs	Septic Tank systems
SVI	Sludge Volume Index
TDS	Total Dissolved Solids
TS	Total Solids
TSS	Total Suspended Solids
UPDBs	Unplanted Drying beds
VIPs	Ventilated Improved Pit latrines
VS	Volatile Solids

Chapter 1: Introduction

1.1. Background

Sanitation is the hygienic disposal or recycling of faeces, household wastewater, and industrial waste i.e., the prevention of pathogenic contact between humans (UNESCO, 2003). In recent years, the World Health Organization (WHO) published statistics that stated 1,7 billion of the world's population have inadequate sanitation facilities, such as private toilets or basic pit latrines; of this, 494 million people defecate in the open (behind trees, open water bodies, and in the streets) (WHO, 2022). This leads to a longstanding public health crisis with high transmission of diarrhoeal diseases, especially within developing countries that demonstrate rapid population growth, industrialisation, and urbanisation.

Safe sanitation globally lowers morbidity rates and reduces disease severity by improving the quality of life (Esrey et al., 1991, Merchant et al., 2003). Pruss-Ustun and Organization (2008) correlated poverty, infancy and 10% of disease burden with poor sanitation. The World Health Organization further explained that 80% of developing urban populations in Africa, Asia and Latin America bare diseases linked with inadequate sanitation, hygiene, or water conditions (WHO, 1998)

Studies currently show faecal sludge generation and wastewater to vary from 100-1000 L per capita per year and 20-150 L capita⁻¹ day⁻¹. The WHO projects this to triple in the upcoming 30 years, resulting in 2,8 billion people lacking safely managed sanitation (UN-Habitat, 2004). Poor sanitation impacts human wellbeing and contributes to environmental challenges. Hence, the urgency to address the 6th Sustainable development goal (SDG), which targets to guarantee the availability and sustainable management of water and sanitation for all (United Nations, 2022). By protecting our water sources, through the improvement of sanitation facilities, treatment technologies, and safe discharge methods, optimal valorisation solutions for faecal material will be established.

While progress has been made in recent years within the water and sanitation department in South Africa, rural areas still fall short with limited access to safe drinking water and a lack of adequate sanitation facilities (Stats, 2020). This is primarily due to high financial costs linked with skilled labor, maintenance needed for centralized sanitation, and proper construction of wastewater and faecal sludge treatment plants (large scale piped water, and sewer networks), that offers a safer discharge into the environment (Nikiema and Cofie, 2014, Bahri et al., 2008, Hutton et al., 2007).

Contrastingly, high costs of centralised sanitation stimulates the use of decentralised sanitation facilities such as pit latrines, which reduces capital costs up to 84% because of minimal sewer networks (Jung et al., 2018). However, these systems only provide limited solutions and store faecal matter for a short

duration, eventually requiring emptying. Improper emptying methods practiced in rural areas contribute to eutrophication in the presence of shallow aquifers. This, in turn, leads to pathogenic contamination in drinking water, resulting in the subsequent transmission of diseases.(Templeton et al., 2015).

Adequate sanitation is essential due to the harsh contaminants present in faecal material. Faecal sludge and faeces, for example, may contain over 100 types of bacteria, viruses, protozoa, and helminths (Cave and Kolsky, 1999). According to the Institute of Municipal Engineers of Southern Africa, poor sanitation, and personal hygiene lead to the spreading of helminth and cholera infections, which are dispersed through food or water that has been contaminated by human waste, diarrhoeal diseases, malaria infections, typhus fever, and typhoid trachoma (De Souza et al., 2006)

The most vulnerable group of individuals affected by helminths and other pathogenic contaminant's found in faecal matter are children (Bartlett, 2005). According to the World Health Organization (2005) these effects are associated with children's nutritional deficiencies, impaired physical and mental development. Hence, inadequate sanitation facilities in rural schools combined with poor sanitation practices such as, open defecation, and improper disposal of faecal material, have resulted in an upsurge of school incidents causing severe illnesses. Which attribute to the presence of faecal matter pathogens and maintenance challenges (Nansereko, 2010). For girls, effects of improper sanitation are exacerbated as they are forced to abstain from regular schooling activities because of inadequate menstrual hygiene facilities.

The National Education Infrastructure Management System (NEIMS), Water Research Commission (WRC) and Department of Basic Education (DBE), conveyed that in 2011 approximately 50,5% of normal South African schools had non-existent or inadequate sanitation facilities, This included 11 450 schools with unimproved pit latrines and 155 chemical toilets. The remaining schools had moderately acceptable sanitation systems, which included approximately 35% onsite sanitation, in the form of VIPs (20%), septic tanks (10%) and Enviro-Loo systems (5%) (DBE, 2011).

In over a decade, only limited technical improvements have been made to advance sanitation infrastructure in rural schools. However, the increase of pupils drowning and subsequently dying from falling into pit latrines persists (Rother et al., 2019). Despite the severe health problems echoed from unsafe school toilets, prohibiting the rural youth from completing their education, data on unsafe school toilets continue to emerge. Effective sanitation in addition to proper education are vital for enhancing child survival, human and environmental well-being, empowering women, promoting economic growth, combatting poverty, and establishing a basis of peace (UNICEF, 1998).

Nansereko (2010) outlines the most problematic sanitation issues faced by rural schools include non-existent, insufficient, or unsafe water supply, toilets not adapted to the basic needs of children, as well as broken, dirty, and unsafe hand washing facilities. These persisting problems derail from the 4th and 6th Sustainable Development Goal (SDG), which aims to guarantee inclusive and equitable quality education, promoting lifelong learning opportunities; and ensure the sustainability of clean water and safe sanitation for all, respectively.

Wolf et al. (2018) showed that improved sanitation can lower the effects of diarrhoea by 25%. Pathogen reduction (Faecal coliforms, E. coli, and Helminth eggs) depends on the disposal options, and end use of treated faecal sludge. Thus, for effective and sustainable faecal sludge management (FSM) in schools and globally, sanitation needs to be attended at all stages of the service chain (Figure1).

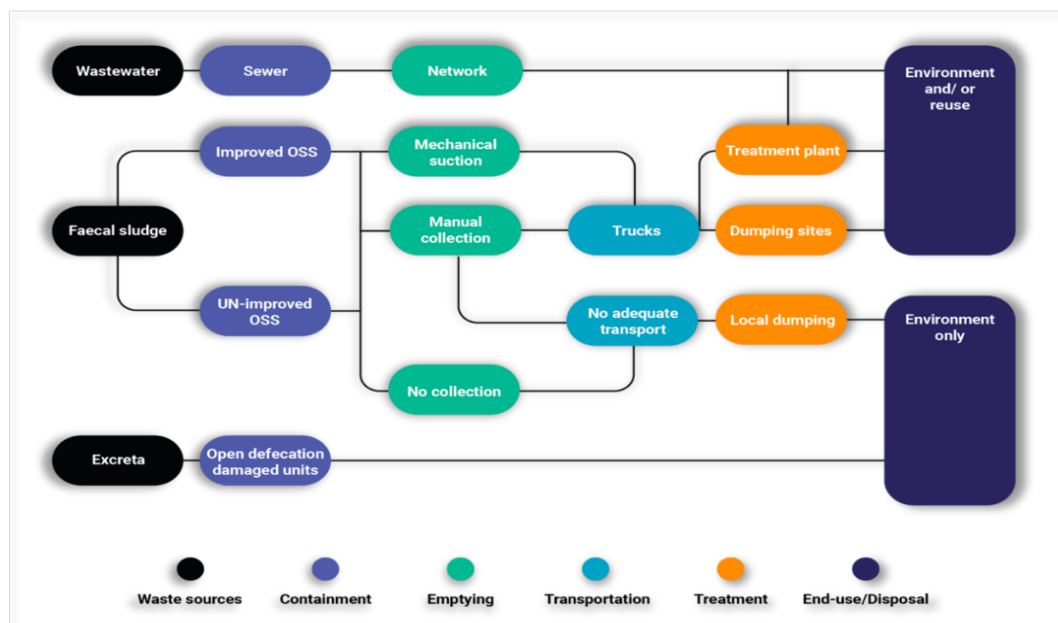


Figure 1: Faecal sludge, excreta, and wastewater management along the sanitation value chain. Source: Nikimem et al. (2015)

1.2 Problem statement

In over a decade, only limited technical improvements have been made to improve sanitation infrastructure in rural schools. Despite the increases in reports of pupils drowning and [or] subsequently dying from falling into pit latrines. Moreover, the inadequate disposal (open dumping) of faecal waste compounds this problem by causing environmental contamination, which can otherwise be used as a sustainable resource. This poses a critical challenge to the health, wellbeing, and educational prospects of rural youth.

This study intends to characterise faecal material (faecal sludge and faeces) from rural schools' onsite sanitation systems (OSS). Comprehending the chemical, physical, thermal, mechanical, and thermodynamic properties of faecal material is crucial in aiding design engineers and technological developers in creating better sanitation facilities, as well as innovative emptying, and treatment technologies. Based on the characteristics of faecal matter, data will be established that determines optimal valorisation solutions, producing organic and inorganic products, such as biochar, biogas and fertilizers. This will promote a healthier and more sustained environment for future generations. It will guide regulators and decision-makers to enforce policies around FSM especially in rural areas where poverty is high. It will improve the associated behaviour around rural sanitation facilities. In turn, ensuring the safety of the environment and health risks posed on youth, through introducing efficiently sustained school toilets. Ultimately providing a solution to the lack of sanitation management within the rural schooling sector, persuading learners to attend school and fully engage in their education within a safe environment, assisting in the move towards the 4th and 6th SDG.

1.3 Aims and Objectives

This study aims to investigate faecal material characteristics from rural school toilets, for its safe disposal and optimal valorisation.

To reach the aim, the following objectives were set:

- To characterise faecal sludge and faeces from rural school onsite sanitation systems.
- To analyse and interpret the collected data which will assist in developing a decision-making tool for the optimal valorisation and treatment of faecal material.
- To propose safe methods of emptying sanitation systems.

1.4 Research Questions

- What are the physico-chemical, thermal, mechanical, and thermodynamic properties of faecal material from rural school toilets?
- How can faecal material from rural school toilets be treated and re-used for its optimal valorisation?
- What are safe emptying methods for faecal material in rural schools' toilets?

1.5 Scope of the study

This research is focused on characterising faecal material from rural school sanitation systems in Durban, South Africa. Fresh faeces were obtained from mobile toilets (MTs) and faecal sludge (FS) from ventilated improved pit latrines (VIPs) and septic tanks (STs). Laboratory analysis was conducted to analyse the physical, chemical, thermal, mechanical, and thermodynamic properties. Samples were analysed according to IWA published book about Methods for Faecal Sludge Analysis Velkushanova et al. (2021) and standard operating procedures produced by the WASH R&D centre. Results of this study is used to develop a decision matrix which provides further clarity on improving school toilet designs, treatment technologies, options for faecal sludge end-use and emptying solutions for different types of sanitation system waste.

1.6 Thesis Structure

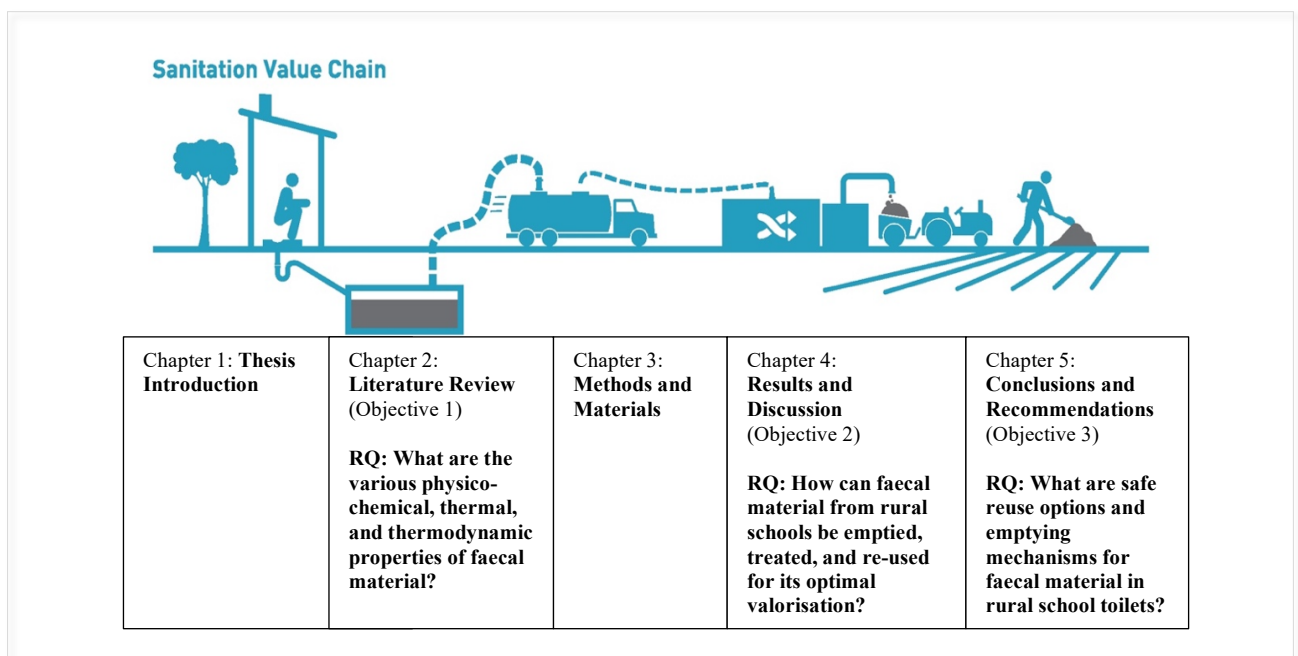


Figure 2: Outline of thesis.

Figure 2 outlines the thesis structure and explores research questions addressed in each chapter. Chapter one serves as the introduction of this dissertation. It explores the general context by framing the current situation on water, sanitation, and educational impacts resulting from the prevalence of unsanitary toilets in South Africa. It further comprises the research aims, objectives, problem statement and scope of the study. Chapter two encompasses the literature review, with a focus on the diverse characteristics of faecal material from different sanitation systems. Chapter three focuses on the research methodology, experimental work undertaken and data analysis to achieve various results and relationships. Chapter four presents and discusses the research findings, elaborates on the synopsis of research findings and provides a decision-making tool for the selection of appropriate treatment and emptying methods. Lastly, chapter five provides a synopsis of the research objectives, findings and addresses recommendations for future studies on rural school sanitation.

Chapter 2: Literature Review

This chapter provides an overview of existing sanitation practices in South African (SA) rural schools, it outlines various characteristics and treatment processes of faecal material and evaluates the significance of characterising faecal matter. Lastly, it examines the potential of faecal material reuse, and safe emptying strategies.

2.1 The role of the government in providing sustainable sanitation facilities for schools.

Although sanitation statistics on a macro level in SA have depicted improvement, rural and low-income areas still demonstrate a high percentage of limited access to safe sanitation (Mirugi-Mukundi, 2014). According to statistics South Africa (2021), 64,8% of South Africans had access to a flush toilet connected to a public sewerage system, while 32,7% of the population relies on decentralised pit latrines systems and 1,1 % make use of the bucket system (StatsSA,2021).

The Water Service Act 108 of 1997, stated “every person has the right to basic water supply and sanitation” (RSA, 1997:12). Hence, the Governments role in providing essential services like education, health care, and safe sanitation facilities is mandatory. It is the responsibility of the local government to guarantee basic sanitation and water services for South African citizens, while the role of the National/Provincial government is to strengthen the local government’s capacity and supervise the construction of water and sanitation infrastructural development (Naidoo et al., 2008). Thus, it is integral that the government’s legal framework for providing safe sanitation is inspected.

2.1.1 School Sanitation Systems in South Africa

In SA 3297 schools still use pit latrines (Table 1), affecting 37 858 teachers and 1 039 117 learners (South African human rights commission, 2021). Provinces KwaZulu-Natal and Eastern Cape exhibited the highest percentage of schools using pit toilets (Table 1 and 2). Pit toilets can lead to educational disadvantages such as absence from class owing to sanitation-related illness or unwillingness to attend school, if not properly managed (Ross and Mirowsky, 2001)

Table 1: Number of schools using pit latrines in South Africa

Province	Number of schools using pit latrines	The number of teachers affected	The number of learners affected
Eastern Cape	2236	24 705	653 516
KwaZulu-Natal	983	12 978	349 826
Mpumalanga	59	885	30 270
Northwest	19	175	5 505
Total	3297	37 858	1 039 117

Source: (Odeku, 2022)

According to UNICEF (2018), approximately 570 million children around the world had limited or no access to basic water, hygiene, and sanitation services, which is a basic requirement for any beneficial educational environment (Jasper et al., 2012). The National Education Infrastructure Management (2011) indicated that 50% of South Africa's government schools had inadequate or non-existent sanitation facilities. Table 2 gives an overview of the types of school sanitation facilities used in South Africa and demonstrates the percentage of children still using pit toilets. The department of education identified KwaZulu-Natal as one of the provinces with the lowest number of flush toilets (Table 2) (StatsSA, 2021).

In most cases where there are insufficient toilet facilities for the total number of learners, queuing for extensive time periods causes learners to miss class, and detracts from their educational attainment. Decentralised sanitation systems such as pit latrines are unsafe, and convey severe health hazards, by contributing to the spread of diseases, particularly when shared amongst many individuals (Figure 3). Still et al. (2012a) also highlighted the many children who have fallen in pits toilets, drowned, and subsequently died. Louton et al. (2015) explained the unpleasant smell, high disease transmission, and difficulty in emptying pit latrines leads to excreta accumulation. Pit emptying further renders a major concern for rural schools, as pits overflow, faecal material thickens, and trash accumulates prohibiting typical emptying methods (Louton et al., 2015).

Table 2: Types of school sanitation facilities

Provinces	Enviro Loo	Municipal Flush Toilet	None	Pit Toilets	Ventilated Improved Toilets	Chemical Toilets	Septic Tanks	Composting toilets	Municipal reticulation Network
Eastern Cape	2%	20%	4%	44%	30%				
Free State	1%	77%	1%		6%		15%		
Gauteng						11%	89%		
KwaZulu-Natal				42%	41%		1%	16%	
Mpumalanga	17%			4%	1%		32%		51%
Northern Cape	8%	55%		4%			33%		

Source: SAHRC (2021)

The accumulation of trash leaves several schools with inefficiently working toilets or no toilets at all (Louton et al., 2015). Owing to the majority of complications experienced in rural schools relating to inadequate sanitation and infrastructure, the government authorities, policymakers, school officials and design engineers need to be guided in establishing improved sanitation facilities (Louton et al., 2015). These facilities should exhibit an all-encompassing solution, allowing for the health, environment, and dignity of learners to be protected (Louton et al., 2015).



Figure 3: Quality of rural school toilets in South Africa, Source: Neethling and Still (2020).

2.1.2 Impact of poor sanitation and hygiene in schools

The lack of adequate sanitation facilities in public schools impacts the hygiene and health of school learners (Mbele, 2011). The importance of well-maintained toilet facilities within educational institutions remains a global initiative towards improved cognitive development, well-being of individuals, economic and social development (Devnarain and Matthias, 2011). Outbreaks of diseases like gastroenteritis, caused by enteric pathogens, are directly associated with chronic illnesses, preventing school attendance and undermining the achievement of the 4th SDG. (Yolken et al., 1982)

However, in certain schools, particularly those in rural areas, the absence of running water poses a significant concern for the effective functioning of centralized sanitation facilities. Similarly, in locations where decentralized facilities are in place, these systems are unable to support the threshold of individuals using them (Bartram et al., 2009). For women and girls, appropriate water, hygiene, and sanitation services is integral for their wellbeing (WHO, 2017). The challenges faced by girls are significantly higher than those faces by boys, especially in maintaining good hygiene during menstruation at school. This dissuades them from education due to schools lacking adequate hygiene facilities. (Kirk and Sommer, 2006).

UNESCO (2014) estimates one in ten girls within Sub-Saharan Africa either misses fundamental schooling classes or discontinues her education entirely due to menstrual related issues.

Moreover, for females, the lack of privacy forces them to wait until a suitable toilet is found when urinating or defecating in school toilets, exposing them to bladder and bowel dysfunctions. (Kajbafzadeh et al., 2006). Alternatively, the child avoids using the toilet by abstaining from food and liquids increasing their susceptibility to health risks (Bartram et al., 2009). Hence, improved sanitation is vital to maintain the wellbeing and create opportunities for girls, allowing them to develop a

foundational education without being interrupted or forced to leave school, in search of an appropriate toilet.

Collectively for all children the exposure to faecal material becomes a vector for the transmission of numerous parasites due to its biohazardous nature. The primary issues affiliated with rural school toilets is owed to the absence of basic maintenance, such as regular cleaning and replacement of mandatory hygiene utilities i.e., toilet paper, soap and handwashing facilities (Hutton and Chase, 2016). Data from WHO (2004) outlined that if every person had access to regulated piped water and sewage supply, 1863 million days of school attendance would be gained due to the reduced diarrheal disease rate.

Poorly maintained toilets in rural schools, contribute to the build-up of excreta (containing solid and liquid waste) within the toilet bowl or urinal. The accumulation of excreta in the bowls of various toilets progress to high transmission of pathogens and severe odour in school toilets. These pathogens can be transmitted by inhalation and contaminated surfaces. Studies by Palmer et al. (1981) and Rajaratnam et al. (1992) expressed that the highest outbreaks of infectious diseases have stemmed from unhygienic public toilet facilities. Pathogenic transmission is heightened in cases of diarrhoea where splashing on and under toilet bowl rims are common as seen in figure 3 (Barker and Bloomfield, 2000).

A vast number of enteric pathogens have been established in high volumes within stool, these pathogens are summarised in table 3. Studies by Lennox et al. (1954) and (Newsom, 1972), showed an infected individual can cast off, up to 10^{11} colony forming units of *Salmonella* and *Shigella* per stool. A concern to school toilets within KwaZulu-Natal is Helminth infections. *Ascaris lumbricoides* is one of the most hazardous pathogens in human faeces (BARTRAM, 2004). Important factors leading to *Ascaris lumbricoides* in poor sanitation facilities include, environmental health, population size per one toilet, socio-economic status and failure of deworming children, all of which are directly linked with faecal contamination (BARTRAM, 2004).

Table 3: Summary of various pathogens excreted in stool.

Microorganism	Reference	Microorganism	Reference
Coliforms Faecal al.	Haas et al. (2014)	Coxiella burnetii	Sinclair et al. (2008)
Faecal coliforms	Haas et al. (2014)	Viral encephalitis viruses	Sinclair et al. (2008)
Escherichia. Coli	Haas et al. (2014)	Nipah virus	Sinclair et al. (2008)
Salmonella	Haas et al. (2014)	Rabies virus	Sinclair et al. (2008)
Shigella	Haas et al. (2014)	Smallpox virus	Sinclair et al. (2008)
Enterovirus	Pepper et al. (2014)	Cytomegalovirus	Paduch (2007)
Hepatitis A	Pepper et al. (2014)	SARS-CoV	Xu et al. (2005)
Rotavirus	Pepper et al. (2014)		
Norovirus	Pepper et al. (2014)	Adenovirus	Gresser and Katz (1960); Paduch (2007)
Adenovirus	Haas et al. (2014)	Measles (rubella)	Gresser and Katz (1960), Paduch (2007)
SARS-CoV-2	Xiao et al. (2020)	Salmonella typh	Abney et al. (2021)
Cryptosporidium	Pepper et al. (2014)	Salmonella paratyphi	Abney et al. (2021)
Giardia	Magana-Arachchi and Wanigatunge (2020)	Leptospira interorgans	
Coxiella burnetii	Haas et al. (2014)		

2.2 Onsite sanitation

Fourie and Van Ryneveld (1995), Torondel (2010) and WHO/UNICEF (2006) outlined that onsite sanitation relates to the situation whereby excreta is stored/collected on the area it is generated. These systems experience degradation of faecal material within its respective containment (Lawrence et al., 2001). Eventually the build-up of faecal matter within the containment requires emptying, which can be accomplished periodically or through the development of a new containment after closing the full original one (Fourie and Van Ryneveld, 1995). Thereafter the emptied waste can be reused or disposed (Franceys et al., 1992). FSM has always constituted a major challenge for rural areas, where sewer pipelines are expensive, consequently impacting the surrounding public health. Due to limited attention, FS collection services for OSS are provided by the informal sector, which lack technology, regulations and quality control (Jayathilake et al., 2019). To ensure proper valorisation and disposal of FS, it is crucial to assess how FS is managed in conjunction with the design and maintenance of OSS.

In South Africa the most utilised OSS includes septic tanks (STs), shallow pit latrines, ventilated improved pit (VIP) latrines, Mobile toilets (MTs), aqua-privy's (also known as low flush onsite sanitation systems) and urine diversion toilets (UDDTs) (Fourie and Van Ryneveld, 1995; Franceys et al., 1992). Tilley. (2014), Lawrence et al. (2001), and Brikké et al. (2003) reported that dry and water-based sanitation are subclasses of onsite sanitation. STs are classed as wet systems, as it receives "flush" and grey water, while VIPs, MTs, UDDTs and aqua privies are classed as dry systems, owing to the lack of "flush water" entering these systems (Fourie and Van Ryneveld, 1995).

For the purpose of this research, faecal material was obtained from the most prevailing school sanitation systems in Durban namely, VIPs, STs and MTs. Sanitation systems were also selected based on its faecal material properties, for example fresher faeces (< 1 week old) are found in MTs, while STs hold FS comprised of urine, faeces and flush water i.e., more diluted contents, and VIPs contains thicker and drier FS owing to minimal “flush water” entering the system (Fidjeland, 2015).

2.2.1 Ventilated improved pit latrine (VIPs)

Mara. (1996) outlined that VIPs are constructed for the storage of faecal matter, and systems differ from traditional pit latrines as they include a tall ventilation pipe (structure showed in Figure 4). Mara (1984) explained that the pipe serves as a medium to control flies and odour. Foxon & Buckley (2008) outlined that for pit latrines to be sustainable and serve a long-life, the rate at which material is degraded or leached should align with the rate at which the pit is filled.

Typical contaminants within VIPs include urine, faeces, sand, toilet paper and a large proportion of trash (non-degradable material) which subsequently increases the rate at which the pit is filled. Buckley et al. (2008), Still et al. (2010) and Cairncross and Feachem (2018) similarly outlined the main problems with the operation of VIPs is related to the volume of individuals utilizing the toilet, and their habits of disposing trash within these systems. The management of full pit latrines poses challenges on health and surrounding environments. Gudda et al. (2019) outlined a draft guideline with options for full pit latrines (DWAF, 2007). It showed that rural communities exhibiting full pit latrines demonstrated no difference to those with no sanitation with options including, (1) abandoning old latrines and constructing new ones, (2) addition of water to extend life of the VIP, or (3) use methods to reduce non-degradable contents. Alternatively, users opt for VIP disinfectant additives, which are often ineffective and stipulate negative environmental impacts, such as pathogenic infiltration and pollution of groundwater (Graham., 2013).

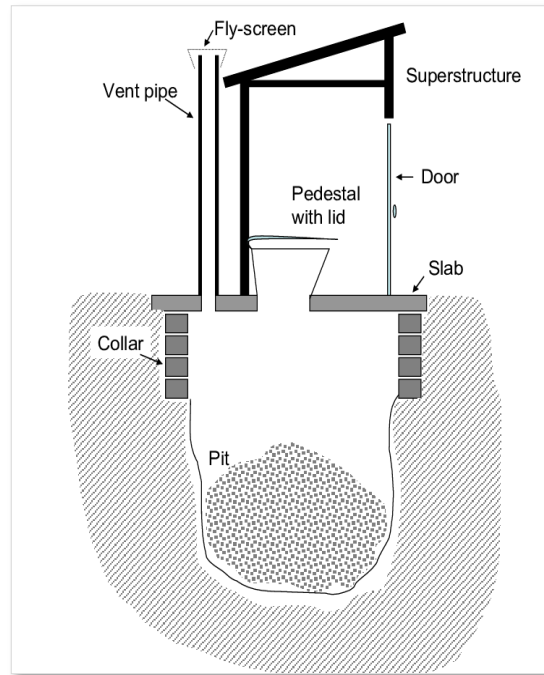


Figure 4: Basic Structure of ventilated improved pit latrine Source: Foxon & Buckley, (2008).

2.2.2 Septic Tank Systems (STs)

Depending on the STs locality, it receives blackwater from a flush toilet and eventually greywater (Luostarinen et al., 2007). As seen in Figure 5, these streams flow into a tank with heavy particles sinking to the bottom, and scum (consists of fats and oils) floating on top. Over time sludge thickens as solids settle to the bottom of the tank and degrade anaerobically. Effluent from STs is dispersed by a soak pit, leach field, or transported to a treatment facility. Tilley et al. (2008) and Taweesan et al. (2015) explained that moderate treatment of solids and organic concentration reduction are demonstrated from settling and anaerobic process within STs.

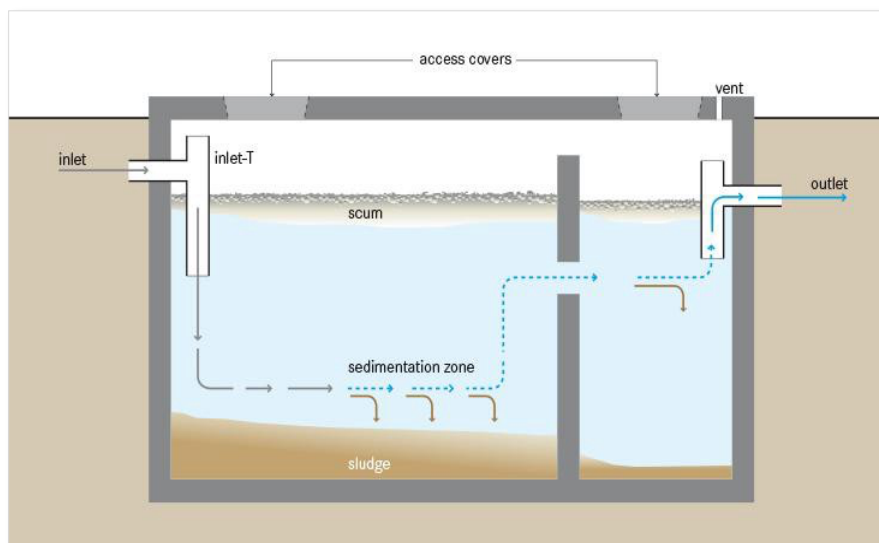


Figure 5: Basic structure of septic tank system Source: Tilley et al., (2008).

2.2.3 Mobile toilets (MTs)

Mobile toilets are lightweight and transportable (not connected to the ground) as seen in Figure 6. Faeces stored within these toilets (in a containment, below the seat) should be disposed in accordance with environmental regulations (Bonifazi et al., 2022, Widodo and RohanaNasution, 2019). In comparison to VIPs and STs, MTs are emptied more frequently, and therefore contain relatively fresher faeces. For sanitization and to suppress the odour from MTs, chemical additives are employed within the system (Bonifazi et al., 2022). These additives were studied to inhibit microbiological activity (Cairncross & Feachem, 2018). Bonifazi et al. (2022) showed that the chemicals are carcinogenic, posing threats to human health. MTs are often placed in areas as a temporary solution such as, construction sites, event venues and public spaces, and should not be utilized as a permanent resolution (Bonifazi et al., 2022). In recent years, there has been increased scrutiny due to the hygiene of these systems and chemicals utilized within them (Bonifazi et al., 2022).

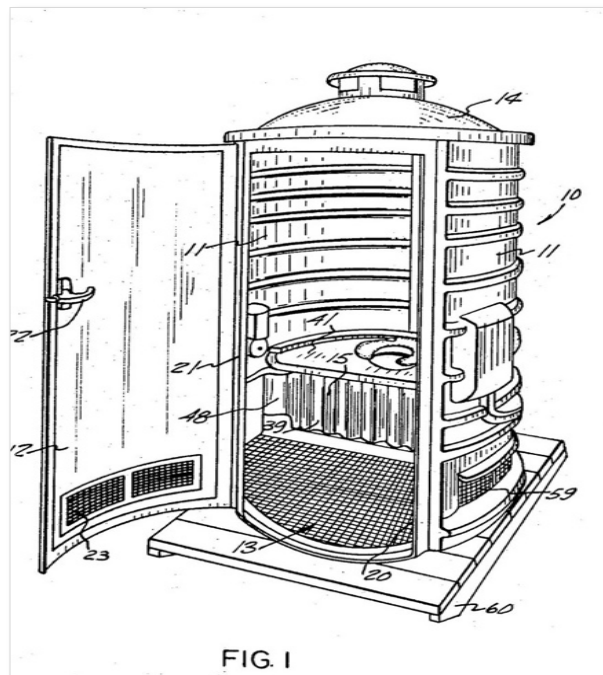


Figure 6: Basic structure of mobile toilet
(Source: Poly John).

2.3 Key characteristics of faecal sludge and faeces

Faeces and sludge are heterogenous and vary due to many factors (Bakare et al., 2012). These factors include (1) environmental such as, demographic and geographic location, the climate and ground water presence; (2) the type of onsite sanitation technology; (3) the age of faecal material within containment; (4) the frequency of toilet use; (5) type of usage of the toilet, (defined by the use of water in the system i.e., dry/flush and the culture of toilet users i.e., wipers or washers); (6) the addition of additives such

as trash or odour suppressants; and (7) the regularity of sludge collection (weekly, annually, or barely) (Semiyaga et al., 2015).

Typically, the variation of faecal material within different OSS is a result of organic matter digestion. For example, MTs are frequently occupied and regularly emptied, resulting in these systems containing less stabilized faecal material. This is compared to VIPs and STs which are rarely emptied and therefore contain more stabilized sludge (Strauss et al., 1997). The difference in stabilisation alters faecal matter treatment and valorisation options (Strauss et al., 1997). Jenkins et al. (2014) outlined that the presence of trash, stabilised and thick waste constituted a major handicap for emptying technologies. As it is prone to clogging, subsequently reducing flowability, and therefore needs to be carefully examined prior to emptying systems. Wanda et al. (2021) reported on the presence of non-faecal material established in VIPs versus STs in West Cameroon, 75% of individuals reported high presence of non-faecal material in VIPs and only 22% for STs.

FS include, Nutrients (P, K, N, C: N, NH₄ and NH₄-N), chemical oxygen demand (COD), pH, Electrical conductivity (EC), rheological shear stress, thermal conductivity, total solids (TS), volatile solids (VS), total dissolved solids (TDS), total suspended solids (TSS), water activity, sludge volume index (SVI), density, particle size distribution and calorific value. Essentially the above properties are grouped in five overarching groups namely, chemical, physical, mechanical, thermal and thermodynamic.

2.3.1 Chemical and physico-chemical properties of faecal material

An understanding of the physico-chemical properties is necessary to evaluate the potential reuse of faecal material in establishing solid and liquid fertilizers, energy sources such as biofuel and treatment for safe disposal (Velkushanova et al., 2021). This research will examine the following physico-chemical processes.

2.3.1.1 Chemical oxygen demand

Chemical oxygen demand (COD) corresponds to the oxygen needed to oxidise the organic fraction of a sample, of which is susceptible to oxidation by a strong chemical oxidant (Ramalho, 1977). COD in FS is indicative of the organic matter i.e., biodegradability and degree of stabilisation. Hence it is important for the optimisation of treatment, as it facilitates on the reuse and co-treatment of faecal matter (Strand et al., 2015; Rose et al., 2014). Additionally, COD is used as an indicator of exposure to the oxidation of inorganic materials existing within wastewater and faecal sludge (Cao and Xia, 2013).

Strande et al. (2014) explained that high-strength sludge or faeces (COD ~250,000 mg/L) from OSS were regularly emptied, poorly stabilized, and exhibited low dewaterability potential, responding better to digestion treatments. Whereas low strength sludge or faeces (COD ~10 000 mg/L) are more stabilised and exhibit higher dewaterability potential and respond well to treatment by drying beds (Strande et al., 2014). Montangero and Strauss (2002) summarised high and low strength faecal sludges from OSS according to table 4.

It is worth noting that COD concentrations and organic stability of FS and faeces are extremely varied. This variation is the result of anaerobic digestion occurring within the sanitation system (Montangero and Strauss, 2002). Bassen et al. (2013) showed that COD concentrations for ST sludge averaged around 7607 mg/L and VIP sludge around 12 437 mg/L, whereas Rose et al. (2015) reported a COD of 48 900 mg/L for fresh faeces, which is significantly higher than that of aged sludge (STs and VIPs). The difference in COD concentrations is largely owed to the storage time within its containment, for example the longer retention of FS experienced in STs, and VIP latrines leads to more biochemically stabilised sludge, compared to MTs which are frequently emptied, therefore illustrate stronger COD concentrations and less stabilised sludge.

Table 4: Strength of FS from different onsite sanitation systems

Item	Type “A” (High strength)	Type “B” (Low strength)	Sewage for comparison’s sake
Example	Public toilet or bucket latrine sludge	Septage	Tropical sewage
Characterisation	Highly concentrated, mostly fresh FS stored for weeks or days only.	FS of Low concentration; usually stored for several years; more stabilised than type “A”	
COD (mg/L)	20,000 - 50,000	< 15,000	500 - 2,500
COD/BOD	5: 1	10: 1	2: 1
NH ₄ -N (mg/L)	2, - 5,000	< 1,000	30 - 70
TS (mg/L)	≥ 3.5%	< 3 %	< 1 %
SS (mg/L)	≥ 30,000	< 7,000	200 - 700
Helm. Eggs, (no./l)	20, - 60,000	< 4,000	300 - 2,000

Source : Montangero and Strauss (2002)

2.3.1.2 pH

The measurement of pH informs on the water chemistry processes within the sludge. These processes cover acid-base chemistry, neutralisation, alkalinity, precipitation, coagulation, biological stabilisation corrosion control, and disinfection, (APHA AWWA, 2005). The pH of FS can be utilised as an indicator for the various treatment options and design (Mawioo et al., 2016). In terms of design, sludges with an

acidic pH degrade connecting pipes faster, resulting in toxic material seeping into the water supply, and causing groundwater and environmental contamination (Niwagaba et al., 2014a).

Niwagaba et al. (2014a) showed that the standard pH of FS ranged between 6,5 – 8. Torondel, (2010) outlined the pH of FS as primarily basic ranging from 7,1 – 9, this was supported by Kuffour et al. (2009), who observed pH values of 7,7 – 9. However, other studies showed wider variation ranging between 1.5 – 12.6 (Cofie et al., 2006, Al-Sa'ed and Hithnawi, 2006). A wide-ranging pH value is risky as, outside a pH range of 6-9 biological processes are disturbed, which constrains anaerobic digestion (AD) and the formation of methane. For optimal FS treatment, pH levels should be controlled ensuring that the required chemical or microbial reactions run competently (Ganta et al., 2022).

2.3.1.3 Electrical conductivity

The electric conductivity of a sample is associated with the dissolved salts, therefore used as an indicator salinity (Velkushanova et al., 2021). The concentration of salts within the sample is essential for determining dewatering and flocculation properties of FS, moreover excessively high EC measurements can inhabit biological processes (Velkushanova et al., 2021). Hence, this is useful for determining treatment processes and resource recovery options such as soil conditioners of faecal matter, as high concentrations of dissolved salts (depending on the nature of the solutes) can pose threats to crops (Velkushanova et al., 2021).

Junglen et al. (2020) found that VIP sludge exhibited an EC concentration of 20 mS/cm, and 15,65 mS/cm for ST sludge. Whereas Yadav et al. (2010) outlined an EC of 60 mS/cm for fresh faeces. Koné et al. (2007) explained that the maximum conductivity plants can tolerate is 3 mS/cm, thus respective treatment pathways should aim to lower this content prior to environmental discharge. Niwagaba et al. (2014) also noted that high EC values can be indicative of an increased ammonia concentrations within the sample.

2.3.1.4 Nutrients

Various studies reveal the high abundance of nutrient content within human excreta (Lu et al., 2017, Harder et al., 2019, Kelova et al., 2021), making nutrient recovery a major objective within FSM. However, poor management and insufficient control of nutrients can have an opposite effect on the environment leading to ground water contamination and eutrophication (Balemi and Negisho, 2012; Taliman et al., 2019). The nutrient concentration within faecal matter can largely enhance the production of fertilisers, which are otherwise dependent on finite resources, such as nitrogen and

phosphorus (Buzie-Fru, 2010). Rose et al. (2015) described that dried faecal solids comprise 13% carbon, 14-18% nitrogen (N), and 3,7% phosphorus (P) and 3,7 potassium (K). Whilst Yadav et al. (2010) showed faeces contained 11 mg/g, 28mg/g and 41 mg/g of phosphorus, potassium, and total nitrogen respectively.

Results emanating from the nitrogen (N), phosphorus (P), potassium (K), and carbon (C) tests will be used to determine the C:N ratio. Determining an optimum C:N ratio (20-30:1) is key for the re-use of faecal material as a fertiliser and maximising agricultural benefits by minimising potential nutrient imbalances or losses (Hema et al., 2022). Similarly, macronutrients (N, P, K) are vital for plant growth and required in large quantities to eventually impact improved food quality and plant growth, for a healthier environment and sustainable livelihood, particularly for rural farmers who cannot afford commercial fertilisers.

2.3.1.4.1 Nitrogen and ammonia

Nitrogen is an important component of amino acids which constitute the building blocks of proteins. These proteins are essential for plant functions such as enzyme activity, structural support and regulation of metabolic pathways. Nitrogen can be present in several forms which depend on the pH, presence of oxygen, type of FS, and the duration of sludge storage (Niwagaba et al., 2014b). Forms nitrogen include ammonium ($\text{NH}_4\text{-N}$), ammonia ($\text{NH}_3\text{-N}$), nitrate ($\text{NO}_3\text{-N}$), nitrite ($\text{NO}_2\text{-N}$), as well as organic forms of nitrogen (amino acids and amines). The measurement of nitrogen indicates the stability and availability of nutrients in treatment and resource recovery processes (Rose et al., 2015). However, nitrites in high concentrations (commonly seen in effluents) can pose adverse effects on aquatic life. In aerobic treatment processes an increase of nitrites can alert deficiencies within treatment processes (Pescod, 1992). Nitrates can become toxic, through the continuous indiscriminate dumping of FS, which eventually threatens public health when entering the food chain by ground water and surface contamination (Velkushanova et al., 2021).

Typical concentrations of nitrogen from VIP sludges range between 3000-5000 mg/l (Heinss et al., 1999, Doku, 2003, Chaggu, 2004, Coetzee et al., 2011, Bakare et al., 2012, Salisbury et al., 2018). However, this range is significantly smaller for septage FS at 190-2100 mg/L (Heinss et al., 1999; Koné and Strauss., 2004). For FS or faeces to be used as a source for nitrogen, the nitrogen application rate should not surpass the agronomic rate (~ 20% of nitrogen within fertilizers). Metcalf et al. (1991) outlined that treated FS contained 1-6% of nitrogen on a dry weight basis. Whereas Rose et al. (2015) described a much higher nitrogen concentration of dried solid faeces (14-18% N). For commercial fertilisers Lorenz and Maynard (1988) showed a nitrogen concentration of 28-32%.

The proportion in which nitrogen can be found i.e., organic, and plant-available inorganic forms is dependent on the way in which the sludge is processed. Organic forms of nitrogen need to undergo mineralisation by microorganisms to be in its inorganic form (Bardgett et al., 2003, Schimel and Bennett, 2004). Sludge type, carbon-nitrogen ratio, climate (high rainfall), soil type, the structure and composition of microbial communities all influence the rate of mineralization (Grzyb et al., 2020). Epstein et al. (1978) described that the rates of mineralisation of organic nitrogen in composted dry sludges are less than those of liquid sludges. Universities and Colleges (1973) explained that sludge undergone anaerobic digestion exhibited a higher microbial oxidation of the organic material, with a subsequent lower residual of organic nitrogen (e.g., sludge from STs and VIPs).

Ammonia can be present as free ammonia (NH₃) or within an ionic form of ammonium (NH₄⁺). Free ammonia is volatile i.e., converts to the gas phase easily (Bassan et al., 2014, Krueger et al., 2021b, Fakkaw, 2016). Typically, ammoniacal nitrogen comprises ~ 10% total N in aerobically digested sludge (degradation in the presence of oxygen) and ~30% in anaerobically digested sludge (Linden et al., 1983). Commonly in aqueous solutions there is a pH dependent equilibrium established amongst the two, which allows the ammonia to occur as free ammonia or within the form of ammonium (Velkushanova et al., 2021). Strande et al. (2014) outlines that free ammonia is reliable for the inactivation of microbes not ammonium ions. The pK_a of ammonia is 9,25, i.e., the pH where 50% is NH₃ and 50% is NH₄⁺. According to Strande et al. (2014), Equation 1 can be used to determine the percent of NH₃ concentration.

$$\text{NH}_3\% = \frac{100}{1 + [\text{H}^+]/\text{K}_a} \quad \text{Equation 1}$$

Ammonia is essential to consider as high concentrations can hinder biological processes and lead to effects such as eutrophication. However, while the harmful effects of NH₃ are broadly studied, Nordin (2010) explained NH₄⁺ ions (even at high concentrations) are more tolerated by majority of microorganisms. Contrastingly, increased ammonia concentrations can also decrease pathogen development. Studies by Englund and Strande (2019) depicted that during alkaline FS stabilisation processes, the introduction of lime rises the pH which subsequently leads to an increase of free ammonia rather than an ionic form of ammonium. Hill et al. (2013) and Fidjeland et al. (2013) described sanitisation by ammonia as one of the most fundamental methods for the inactivation of pathogens.

For FS and faeces, Strauss et al. (1997) reported NH₄-N concentrations ranged between 2 000-5 000 mg/L. This result was supported by Gudda et al. (2017), who further showed that the highest

concentration of ammonia within faecal matter was observed from samples collected from the top layers of OSS (less stabilised waste).

2.3.1.4.2 Phosphorus

Phosphorus is important for plant and organism growth. The role of phosphorus in a plants metabolism is necessary, photosynthesis, and root development (Ullah et al., 2015, Cockefair, 1931). Thus, to ensure on-going food production, while minimising environmental pollution, phosphorus from faeces and FS must be considered for agricultural reuse. Within FS, phosphorus is present as orthophosphates, polyphosphates, and organically bound phosphates (Strande and Brdjanovic, 2014). Table 5 and 6 illustrates the daily loadings and concentration of phosphorus within faeces and FS respectively. These values can be altered depending on the various FS treatments undertaken.

Typically, sewage sludge exhibits low concentrations of phosphorus (Torri et al., 2017). Whilst other studies outlined FS exhibited between 0,9-6,1% of phosphorus on a dry basis (Metcalf et al., 1991). However, Lorenz and Maynard (1988) described that commercial fertilisers comprised of approximately 8-24% of phosphorus. To promote optimal agricultural growth and environmental protection, it is significant to monitor the rates of phosphorus ensuring that it does not exceed the maximum levels needed for crop yields. This is done to prevent surface and water contamination.

Table 5: Daily loadings and concentrations of phosphorus in faeces

Value (g/cap/day)	Value on a wet basis (g/kg)	References
0,35	3,40	Vinneras et al. (2006)
0,5	0,5 -1,83	Czemiel (2000)
0,5	0,5 – 3,59	Vinneras (2002)
0,51	0,1-1,77	Goldblith and Wick (1961)
0,65-0,87	0,65-0,87	Calloway and Margen (1971) Meinzinger
0,5	7,76-8,92	and Oldenburg (2009)
0,69-2,5	0,5 – 3,8	chaggu (2004)
0,9-2,7	0,69-2,5	Wignarajah et al. (2003)
	4,80-9,86	

Source: (Rose et al., 2015).

Table 6: Concentration of phosphorus in faecal sludge

Parameter	Pit latrines	Septic tanks	Portable toilets
P (mg/L)	6 – 474	65 – 2,040	~396,2

Source: Schoebitz et al. (2016); Rose (2015).

2.3.1.4.3 Potassium

Potassium is essential for biological and physiological processes (Wishart, 2019). It promotes plant growth and improves its metabolism by regulating root and apex growth (Wang et al., 2013). Examining

the potassium content within faecal matter will enhance the knowledge on fertilizer development yielding a larger quantity of crops. Strand et al. (2014), Cofie et al. (2006), Koné et al. (2007, 2010), and Nartey (2013) reported that FS contained ~1,9-15,4 % (K) for faeces. Table 7 summarises the various daily loadings and concentrations of faeces on a wet basis.

Table 7: Daily loadings and concentrations of potassium in faeces.

Value (g/cap/day)	Value on a wet basis (g/kg)	References
0,2-0,24	1,78–2,14	Calloway and Margen (1971)
0,47	3,10	Goldblith and Wick (1961)
0,75-0,88		Wignarajah et al. (2003)
0,8	4,936	Eastwood et al. (1984)
0,8-1,0		Kujawa-Roeveld et al. (2006)
		(Meininger and Oldenburg, 2009)
0,7	3,3	Chaggu (2004)
0,8-2,1	2,712	Vinnerås et al. (2006)
1,48-2,52	7,16	

Rose et al., (2015).

2.3.1.5 Solids concentration

The concentration of total solids (TS) in faecal sludge is crucial for selecting appropriate separating technologies and utilizing faecal sludge as a fuel resource (Seck et al., 2015, Diener et al., 2014, Strauss et al., 2000). Solids are divided or grouped according to its organic content i.e., volatile, (VS) or fixed, or on its physical properties i.e., total suspended (TSS) or total dissolved solids (TDS) (Velkushanova et al., 2021). TDS are referred to solids that can pass through a filter with a pore size 2.0 µm or less, whereas the TSS are the solids which do not pass through the filter. TSS is a metric that examines particulate loading within the supernatant post-settling (Velkushanova et al., 2021).

Increasing concentration of solids is the aim of fundamental pre-treatment practices like dewatering and drying processes (Velkushanova et al., 2021). Dewatering is performed on samples exhibiting a high-water content. Through dewatering the volume of sludge is reduced which subsequently lowers transportation costs along the sanitation value chain (Strande et al., 2014). Dewatering methods currently practiced include, unplanted drying beds, planted drying beds, and mechanical dewatering (Semiya et al., 2017). Sludge drying is the process of extracting moisture from faecal sludge in the form of water vapor, accomplished through methods such as heating, as opposed to liquid removal through techniques like mechanical dewatering (Zhang et al., 2021). The FS is often dried to a TS > 65%, depending on the treatment. Thermal drying methods, unlike dewatering, enable a substantially greater reduction in moisture levels. This lower moisture content is ideal for efficient energy recovery

and the reuse of faecal sludge as a biofuel (Getahun et al., 2020). Hence, understanding the solids concentration remains vital for onsite containment technologies and safe disposal mechanisms.

The solids concentration of FS is based on temperature, humidity, air velocity, sludge source, and physical transformations of the material (Banga and Singh, 1994). The solids concentration in sludge can directly or indirectly effect biodegradability, viscosity, solid-liquid separation, dewaterability, flowability and drying potential (Velkushanova et al., 2021). Woolley et al. (2014) further explains that the concentration of solids within a faecal sample influences its shear stress. Fresh material with a high solids concentration exhibits high apparent viscosity. Indeed, at a given shear rate, fresh faeces with lower moisture content exhibits a higher viscosity (Woolley et al., 2014).

Typically, liquid FS exhibits a TS concentration of < 5%, usually obtained from wet containments such as STs and wet/leached VIPs (Velkushanova., 2021). Slurry FS is slightly thicker than liquid but still has liquid/muddy consistency, herein literature values shows that the TS concentration ranges between 5-15% (Velkushanova., 2021). These types of samples are often collected from improved/unimproved pit latrines or at the bottom of STs. Semi-solid FS has a soft paste like texture. This type of the sludge is not pumpable at higher ranges and should be gathered by spade (Velkushanova., 2021). Semi-solid sludge is often obtained from composting toilets, leech pits latrines, UDDTs, or dewatering treatment technologies (Velkushanova., 2021).

Koné and Strauss (2004) showed an average TS concentration of 19 000 mg/l for samples taken from STs. Similarly, samples from a case study in Dakar showed a variable TS concentration of 4500-14 000 mg/l (Vonwiller, 2007, Walker, 2008). The difference in TS concentration can be explained by user behaviour, or type of sanitation system i.e., if there is an inclusion of flush water. Moreover, Fry and Merrill (1973) showed that VS make up 92% of the TS content in faeces. Table 8 compares TS and TVS from STs and discharging trucks, while Table 9 outlines average TSS, TVS, and TS collected from VIPs and STs during dry and rainy seasons (December 2010 – September 2011) in Ouagadougou, Accra and Dakar.

Table 8: Comparison of TS, and TVS from the characterisation of FS in West Africa

Location	Sampling method	TS (mg/l)	TVS (%TS)
Ouagadougou.	From discharging trucks, at the beginning, middle, and end	11820	48
Ouagadougou ¹	From septic tanks	19000	47
Accra ²	From septic tanks	11900	59
Dakar (Rufisque) ³	From discharging trucks, every 30 second	14000	NA

Source: Koné and Strauss, 2004; Heinss, et al. 1999; and Walker, 2008.

Table 9: FS characterisation for dry and rainy seasons, from VIPs and STs in Burkina Faso

Parameter	FS sampled during dry season		FS sampled during rainy season		FS from pit latrines only		FS from septic tanks only	
	Mean	σ	Mean	σ	Mean	σ	Mean	σ
TS [mg/l]	10658	8264	12919	10989	13349	10755	8984	8926
TVS [% TS]	53	-	61	-	58	-	57	-
TSS [mg/l]	6826	5032	11084	10406	10982	10700	7077	7478

Source: Bassan et al. (2013).

2.3.2 Physical and Mechanical Properties

The physical properties of excreta aid the selection of emptying and treatment technologies for pumping, compressing, extruding, among other operations (Velkushanova et al., 2021; Strand et al., 2014).

2.3.2.1 Density

Density is determined by dividing mass by volume, it indicates how compact a material is. Penn et al. (2018) showed that the ideal density for healthy human faeces ranged between 1,06-1,09 g/mL. SCHERTENLEIB (1985) reported that the average density of FS was around 1,423 kg.m⁻³. AIT (2002) further outlined that wet sludge from the bottom of STs exhibited a density range of 1,092 - 1,159 kg.m⁻³. Niwagaba et al., (2014) reported the density from dry and wet pit latrines as 1,356 kg.m⁻³ and 1,443 kg.m⁻³ respectively. Essentially, density varies depending on type of storage containment, age of the sludge, and frequency of desludging. Seleman et al., (2021) reported a 0,999 mg/l mean density for STs and 0,992 mg/l for VIPs. The higher density observed in STs was likely due to the sludge age (Seleman et al., 2021).

2.3.2.2 Particle size distribution

The particle size distribution (PSD) provides an indication on the breakdown of faecal material or the particle agglomeration (Velkushanova et al., 2021). The varying particle sizes of faeces and FS is important for the design of FS treatment technologies, such as dewatering and filtration (Kopp and Dichtl, 2001, Spellman, 1997). PSD is greatly influenced by the dehydration process, which is owed to individual particles forming clumps post-dehydration, subsequently affecting flowability and the rotation of the particle, eventually increasing viscosity (Harboe and Modigell, 2012). Strande et al. (2014) showed that particle size influences the BOD reaction rates and solid-liquid separation

(filtration), which was owed to smaller and more soluble particles illustrating faster biochemical oxygen demand (BOD) reaction rate coefficients. Additionally, Metcalf et al. (1991) explained the importance of PSD for filtration (mechanism for solid-liquid separation in FSM), where smaller particle size distributions are not as effectively removed by filtration processes.

The average particle size of human faeces ranges between 0,49 and 624 μm (Chatema et al., 2023). However, FS particle size varies depending on the type of sanitation system (Semiyaga et al., 2017). This was also observed from Semiyaga et al. (2017) who showed FS from unlined pit latrines exhibited a lower particle size range ($<0,032$ mm) compared to lined pits.

2.3.2.3 Rheological properties

Rheological properties of faeces and sludge inform on the operation and design of FS emptying devices and treatment technologies, such as viscous heaters (FS technology that pasteurises the sludge by heating by viscous friction) (Belcher et al., 2015). The yield stress of sludge is defined as the minimum shear stress value to overcome the resistance of flow (Shafiq et al., 2020, Mercer et al., 2021). Hence, this study examines the shear yield stress on FS and faeces for school sanitation systems, to advise on minimum energy needed for faecal material to flow and assist in establishing optimal emptying techniques.

Cao et al. (2016), Thota Radhakrishnan et al. (2018), Woolley et al. (2014) and Mercer et al. (2021) showed that the moisture content is a central factor influencing the rheological properties of faecal matter, by proportionally inducing shear stress to yield flow. This was supported by Septien et al. (2018) who demonstrated a yield stress decrease at a higher moisture content for FS. Faecal material exhibits shear thinning (a decrease in viscosity by increasing shear rate) with thixotropic behaviour i.e., a progressive decrease in viscosity to 0,5% of the original value, after 20 s to a shear rate of 10 s^{-1} (Woolley et al., 2014).

Septien et al. (2018) reported that Ventilated Improved Pit latrines (VIPs) FS exhibited a yield stress between 500 – 1,000 Pa. De Loubens et al. (2020) explained, that between a solid content range of 10 - 70/100 g/g of faeces, there is a yield stress between 20 – 8000 Pa. For fresh faeces, Mercer et al. (2021) outlined that a lower shear stress is required to begin flow compared to aged VIP sludge or sewage sludge at the same solids' concentration. At 20% TS concentration, Mercer et al. (2021) reported that yield stress values are around 120 Pa for fresh faeces, whereas Septien et al. (2018) reports a yield stress between 500 - 1000 Pa for aged faecal sludge at 10 - 23% TS, as seen by rheological curve in figure 7. Figure 7 implies that FS, in comparison to fresh faeces, would require a greater pumping power at the

same flow rate. Indeed, factors such as solids content, aging and degradability degree alter rheological behaviour (Mercer et al.,2021).

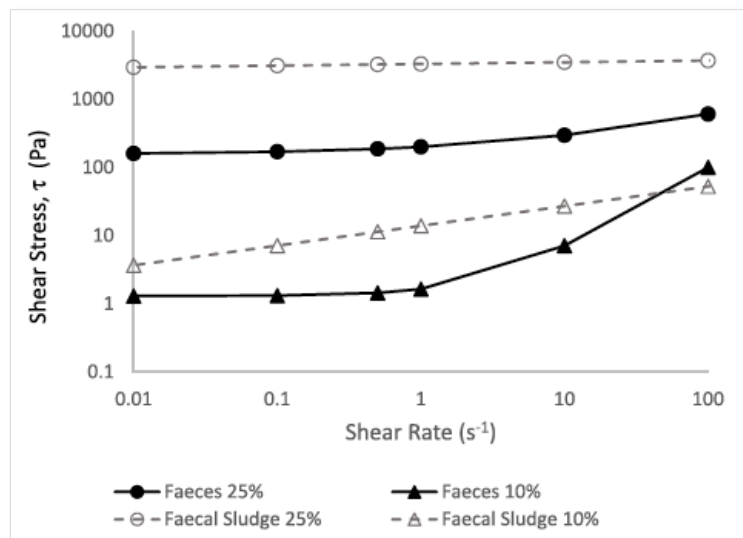


Figure 7: Rheological curve of fresh faeces (Mercer et al., 2021) and faecal sludges (Septiens et al., 2018).

2.3.3 Thermal and Thermodynamic Properties

Thermal and thermodynamic properties of FS and faeces are necessary for thermochemical treatment processes (Gold et al., 2017; Muspratt et al., 2014; Strande and Brdjanovic, 2014). Possible thermal treatment pathways include gasification/pyrolysis, aerobic fermentation, hydrothermal carbonization, and incineration; subsequently recovering resources like char, biodiesel, methanol, and energy (Rose et al., 2015; Ting and Lee, 2006). Figure 8 shows the current thermal and biological options for production of energy from faecal sludge.

Thermal treatments are most influenced by TS and the FS energy content (Rose et al., 2015). This is owed to the TS concentration determining the viability and feasibility of thermal processes i.e., whether it can be used as a suitable feedstock (Rose et al., 2015). The average TS of faeces is approximately 25%, similar to the range of dewatered sludge 22-36% (Tchobanoglous et al., 2003). This suggests that thermal treatment could be performed without dewatering processes, however, would still require drying owing to the high-water content in raw material. Hence for optimal combustion, dewatering and drying of FS are necessary. Additionally, it increases the efficiency of thermal processes, congruently stabilizing the material and inactivating the present pathogens (Rose et al., 2015; Naidoo et al., 2019).

Drying of faecal material is affected by the operating parameters such as temperature, airflow rate, humidity, pressure, and type of heating source, as well as feedstock characteristics, including type of sludge, size and geometry (Makununika, 2016). Getahun et al. (2018) further explained that bound

water in FS required a higher energy input to remove the bound moisture and break the interactions between the solid matrix and bound moisture. Getahun et al. (2018) also showed that FS required approximately 2 to 3 that of the latent heat vaporisation of water for drying. To test for optimal energy potential of faecal material thermal conductivity, calorific value and water activity are determined.

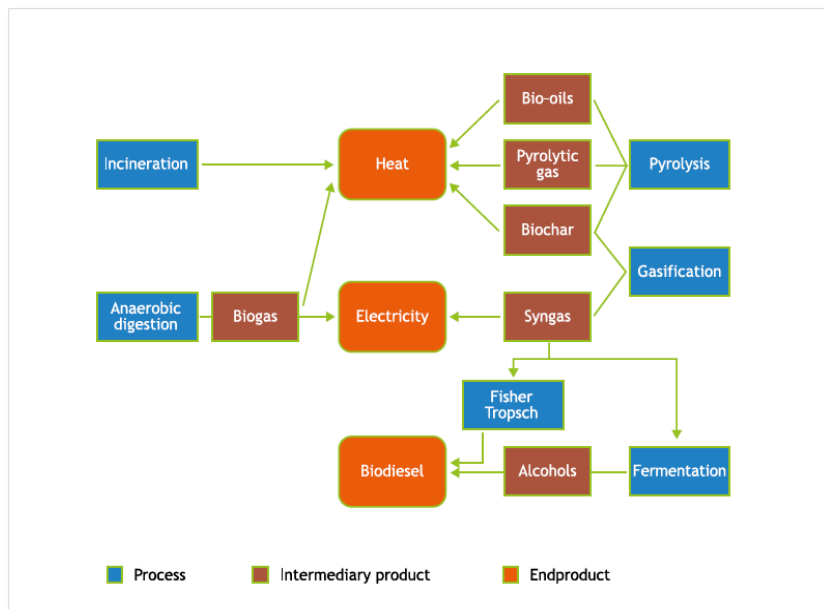


Figure 8: Thermal and biological options for energy production from FS . Source: (Niwagaba et al., 2014b).

2.3.3.1 Thermal Conductivity

Thermal conductivity refers to the capacity of a material, such as FS, to conduct heat (Velkushanova et al., 2021). Chatema. (2021) outlined that thermal conductivity generally decreased when water activity decreased. This was supported by Lewicki (2004), who explained the primary contributor to thermal conductivity was water activity, and Septien et al. (2020), who showed the decrease in thermal conductivity when moisture is removed. The relationship of moisture content to thermal conductivity for FS is illustrated in Figure 9.

For VIP FS with a moisture content between 0,69 and 0,87 g water/g wet mass, Zuma et al. (2015) reported that thermal conductivity was between 0,48 W/m.K. – 0,55 W/m.K, with no significant differences between various pit layers. These findings are supported by Niwagaba et al.(2014), who reported the thermal conductivity of 0,4W/m.K. and 0,55 W/m. K for dry and wet pit latrines respectively. This was supported by Makununika (2016) and Septien et al. (2020) who showed wet FS to be 0,55 W/m.K (comparable to pure water ~ 0,6 WK/m) and dried FS to be 0,044 W/m.K (in resemblance to air ~ 0,03 W/m.K). Septien et al. (2020) explained that at moisture contents < 40 % wet basis, thermal conductivity did not vary and achieved a constant value of 0,04 W/m/K. Hence, wetter

sanitation systems such as STs and wet VIPs exhibited FS with a higher thermal conductivity compared to drier systems. Hanson et al. (2000) and Bart-Plange et al. (2009) explained the decrease in thermal conductivity for dry FS is likely due to the large empty spaces within the solid, which are occupied by air post FS drying, leading to the lower conductivity of the faecal material.

Septien et al. (2020) further explained that thermal conductivity of FS was lower post drying. However, as the thermal diffusivity (ability of a body to conduct heat, relative to its ability to store thermal energy), is higher in a dried material, this allows for an improved conduction of heat which is positive for thermal processes i.e., making dried material more suitable for reuse as a biofuel.

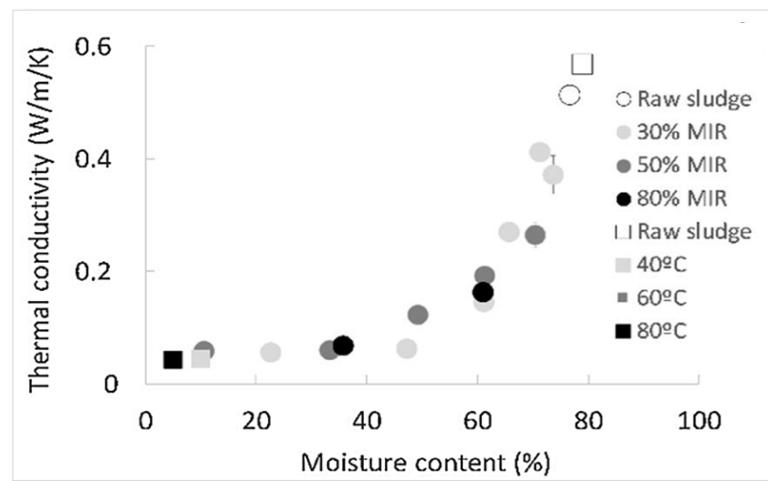


Figure 9: Thermal conductivity versus moisture content .
Source: Septien et al., (2020).

2.3.3.2 Calorific value

Calorific value is a measure of the energy released by the mass of a sample, when it undergoes combustion within an enclosure of fixed volume. It indicates the potential energy content within a sample. Calorimetric analysis is crucial for assessing the suitability of faeces and FS as biochar and pellets, hence for using material as a source of fuel. The calorific value of FS is influenced by several factors, including the type of OSS, level of stabilisation, moisture content, and available sand content within sample (Niwigaba et al., 2014a).

Zuma et al. (2015) and Muspratt et al. (2014), reported the calorific value of VIP FS ranged between 11- 18 MJ/kg dry solid. Onabanjo et al. (2016) and Muspratt et al. (2014) reported higher calorific values (~25MJ/kg) for fresh faeces, suggesting human faeces can be a viable option for energy recovery through combustion (Rose et al., 2015). Figure 10 depicts the various calorific values found in FS and

wastewater sludge and compares it to the calorific value in biomass fuels. The FS values compare well with traditional biomass fuels reported in (Muspratt et al., 2014, Alakangas, 2016).

Faecal sludge stored in its containment for a long time exhibits higher biological degradation compared to fresh faeces which causes the decrease in calorific value (Zuma et al., 2015; Ostrem et al., 2004). This was supported by Gold et al. (2017) and Andriessen et al. (2019), who explained that the decline in calorific value was associated with the depletion of readily available organic content. However, to optimize on the energy potential of FS, initially a degree of drying is needed. Drying is fundamental to decrease the moisture content and increase the calorific value for the possibility of fuel ignition (Onabanjo et al., 2016).

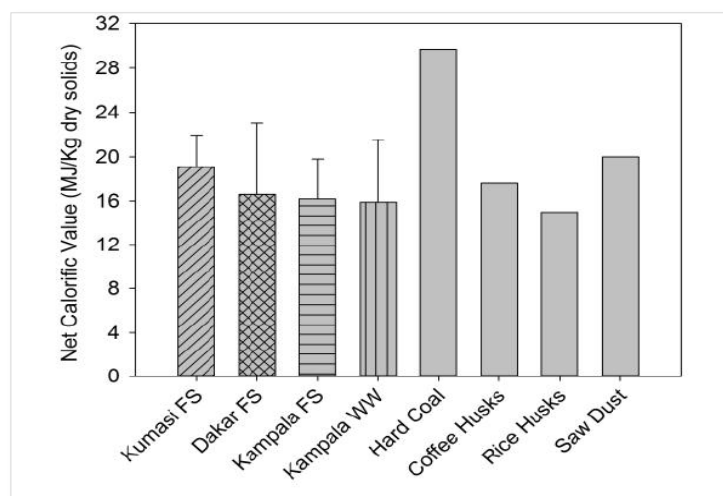


Figure 10: Calorific value for faecal sludge, wastewater sludge and biomass fuels. Source: (Muspratt et al., 2014, Alakangas, 2016).

2.3.3.3 Water Activity

Water activity is a thermodynamic parameter outlined as the vapour pressure of water within the sample, divided by the vapour pressure of pure water at the same temperature. It is indicative of the binding strength of moisture within the solid matrix of the faecal matter sample. Water activity (a_w) ranges from 0-1. A value of 1 signifies that the sample is unbound to solids and easier to remove, while a value of 0 indicates that all the water within the sample is strongly bound to solids and more difficult to remove. Understanding the boundness of moisture to the solid material is fundamental for engineering dewatering and drying methods (Tao et al., 2008). Water activity additionally suggests the microbial activity within a sample, as microorganisms are unable to survive within low water activity environments. Barbosa-Cánovas (2003) explained that at water activities lower than 0,62, there is usually no microbial growth.

Getahun et al. (2020) outlined that water activity is a function of moisture content, hence as moisture content decreases so does water activity, as seen in Figure 11. This was supported by Remington (2019), who showed that between moisture contents of 27-34%, water activity is approximately 0,85, while for moisture contents between 63-86%wt, water activity ranged between 0,93-1 a_w . Getahun et al. (2020) further showed a significant change of a_w when moisture content was below 30% (0.85 a_w), while above 60% moisture a_w was ~ 1 , and suggested that microbial growth is unlikely for samples with less than 10 % moisture (0,1 a_w).

Almeida et al. (1999), Zavala et al. (2002), Nwaneri et al. (2008), Woolley et al.(2014), Remington (2019) and Chatema et al. (2023) reported that fresh faeces illustrated moisture contents ranging between 77-82% wt, which corresponded with water activities of 0,99 a_w . Similarly Remington (2019) showed water activities of 0,93-1 a_w for moisture contents between 63-86 %wt. For faecal sludge Getahun et al. (2020) showed typical water activities of ~ 1 a_w until moisture content is $< 60\%$. Figure 11 outlines a sorption isotherm for FS (the relationship between equilibrium moisture content and water activity) in Durban, South Africa. It shows that water activity ~ 1 is above 35 %wt and there is a drastic decrease of water activity below 35 %wt (Mupinga, 2021).

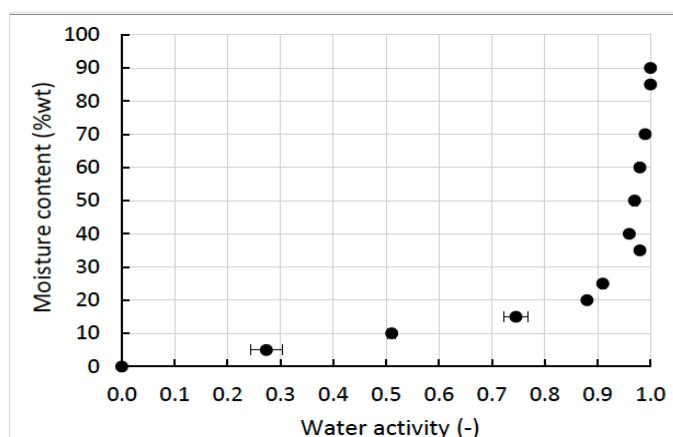


Figure 11: Relationship between water activity and moisture content. Source: Mupinga et al. (2021).

2.3.4 Settleability of faecal material

Sludge settleability is an indicator of how well faecal material can settle (Heins et al.,1994). The settleability of sludge is based on the volume of occupied sludge after a known settling period. Amongst all settling tests, sludge volume index (SVI) is the most widely practiced. Dick and Vesilind (1969) outlined that the settleability of sludge is influenced by TSS, rheological characteristics, interface velocity and temperature. SVI measures the sludge settling performance, based on the quantity of suspended solids which have settled in a specific time.

The time needed for sludge settling increases with the age of FS (Chandana and Rao, 2021). The SVI for septage sample ranges from 40-80 ml/g, which is significantly lower than that of wastewater, ranging between 75-100 ml/g (Heins et al., 1994). Results by Niwagaba et al. (2014) showed FS obtained from wet and dry pit latrines exhibited an SVI of 40 ml/g and 110 ml/g respectively. Hence, the SVI from wet pit latrines is similar to ST sludge. Moreover, Niwagaba et al. (2014b) suggested that an SVI lower than 100 ml/g achieved good solid-liquid separation within settling thickening tanks. This was supported by Heins et al. (1998) who outlined a good settling ability was achieved with an SVI of 30-80 ml/g and Bassen et al. (2013), who showed that an SVI of 26-29 mg/l exhibited excellent settling.

2.4 Emptying, transport, treatment and end-use of faecal material

2.4.1 Emptying and Transport of Faecal sludge

Mikhael et al. (2014) and Thye et al. (2011) outlined the appropriate methods for emptying sanitation systems is dependent on the characteristics of respective sludges. Such characteristics include the depth, and accessibility of the containment, its FS properties, disposal site and geography of the site. In urban areas a combination of mechanical and manual emptying methods is practiced for collection and transportation of FS, whereas rural and informal sectors primarily practice manual emptying.

2.4.1.1 Manual emptying

Manual pit emptying, as depicted in Figure 12, is typically performed using a shovel, bucket or manually operated pumps such as the Gulper, Manual Pit Emptying Technology (MAPET) and Vacutug (Thye et al., 2011). However, these techniques exhibit small carrying capacities and operate at lower speeds in comparison to vacuum trucks. This causes an increase in transport costs owing to the extensive distance between treatment plants and rural areas (Klingel et al., 2002). However, to combat this issue, the National Waste and Sewerage Corporation (NWSC) in Kampala (Uganda) suggested that, once FS has been collected through methods like Vacutug and MAPET, it should be transferred to a tanker, i.e., a mobile transfer station. This will allow for easier transportation to respective treatment plants (NWSC, 2008).

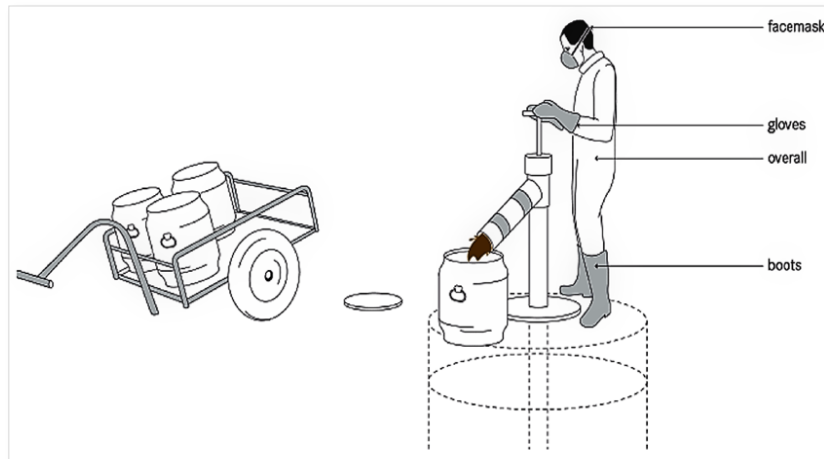


Figure 12: Systematic diagram of an example of manual emptying and transport method. Source: Tilley et al., (2014).

2.4.1.2 Mechanical emptying

Mechanical emptying as seen in Figure 13 uses fully mechanized equipment that are reliant on electricity, fuel or pneumatic systems. This type of emptying includes vacuum trucks, pumping systems or motorized augers (Strande et al. (2018); Thye et al. (2011)). Mechanical emptying poses fewer health risks as vacuums are typically mounted on trucks, limiting human contact (Chandana and Rao., 2021). However, the success of mechanical emptying is dependent on the viscosity of FS characteristics and therefore not suitable for emptying highly viscous sludges. They are instead better suited to liquid-slurry sludge, i.e., from STs, because of the more diluted septage (Thy et al., 2011).

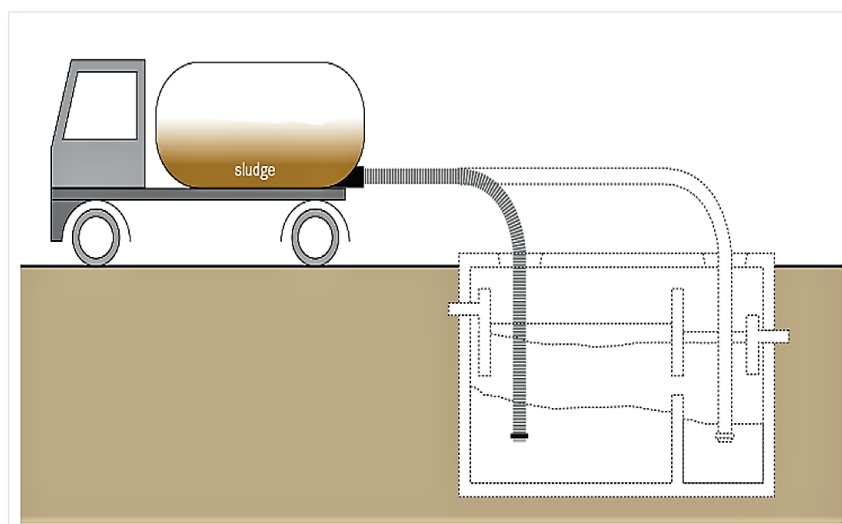


Figure 13: Schematic diagram of an example of mechanical emptying and transport. Source: Tilley et al., (2014).

2.4.2 Treatment and resource recovery options for faecal material

Koné and Strauss (2004) outlined that the appropriate selection of sludge treatment is dependent on sludge characteristics and their intended reuse. These characteristics vary on location, water content and the storage duration of the sludge, among other factors. In urban and high-income areas, the co-treatment of FS with sewage is feasible, particularly if the treatment plant is readily available and constructed to withstand a variety of FS types (Klingel et al., 2002). However, this becomes problematic in rural and low income urban areas, where the lack of adequate capital makes it challenging to support and maintain the proper functioning of centralised wastewater treatment plants.

This study assesses the potential treatment technologies for the optimal valorisation of raw faecal material from rural areas. Therefore, technologies discussed are appropriate low-cost technologies applicable to the varying characteristics of faecal material from different school OSS and suitability to rural areas.

2.4.2.1 Unplanted planted drying beds

Unplanted drying beds (UPDBs) are shallow filters containing sand or gravel and equipped with an under-drain for leachate collection (Strande et al., 2014). Tilley et al. (2014) explained that approximately 50-80% of the sludge volume is drained off as liquid or evaporates, hence they are commonly used for FS dehydration. Through dehydration a subsequent end- use product of FS could be building materials; this is owed to the alteration of the FS thermal properties (as it is controlled by moisture content). Septien et al. (2020) showed that a low thermal conductivity allows for dried solid waste to be a successful thermal insulator which can be a beneficial attribute for building materials. However, testing and questioning on the consistency of its properties and affordability is still being evaluated (Dieneret al., 2014). Other end-use products include fertilisers and biofuels.

Unplanted drying beds are better suited to sludges with a higher water content such as ST sludge. The process of drying is relatively quick ranging from days to weeks (Heinss et al., 1998). Small-scale sludge drying beds were implemented in Kumasi, (Ghana) and Cambérène treatment plant, Dakar, (Senegal), however, results showed that further stabilisation, drying or pathogen deduction may be required depending on the intended output of the FS (Strande et al., 2014).

UPDBs are applicable for rural areas, owing to its low costs, minimal energy requirements and large land space requirements (Singh et al., 2019). Furthermore, they can be developed with materials that are available locally, and operated by unskilled workers (Singh et al., 2019).

2.4.2.2 Planted drying beds

For Planted drying beds (PDBs) *Andropogon gayanus* and *Cymbopogon nardus* are the most used plants (Singh et al., 2019). FS is loaded into the PDBs, which serve to remove water and enhance stability by physical and biological mechanisms (Kadlec and Knight 1996). Edwards et al. (2001) outlined that, in summer seasons, PDBs achieved greater rates of water and solids reduction, as well as increased oxidation. This suggests that PDBs would be suitable for tropical countries which is characterised by minimal climate variation and higher solar radiation. This was supported by Koottatep et al., (2005), who described a successful operated PDB pilot scale facility in Thailand. Ideally the input FS should be a combination of fresh and stabilised sludge, with a relatively low solids concentration (~10%) and COD concentrations around 20-50 g/l (CDD Society, 2017).

Reeds are also a typical choice of plants for PDBs and subsequently referred to (“reed-beds”). Heins and Koottatep (1998) showed results from a pilot study in Bangkok (Thailand) whereby more than 50% TS was obtained in the dried septage layer. They further explained that sustainable valorisation of plant growth occurs at 25-35% TS. Among other options an overall reuse options for PDBs include stabilised biosolids that can be used as a soil conditioner for irrigating farmlands.

PDBs are suitable for rural areas owing to its cost-effective construction and maintenance, simple operation, robust performance and greater loading rate compared to UPDBs. Rural areas are often sparsely populated and less dense than urban cities and therefore can handle its large land requirement.

2.4.2.3 Anaerobic Digestion

Anaerobic digestion (AD) is a type of treatment used for the stabilisation of partially digested or fresh faecal sludge (Semiyaga et al., 2022). Faecal material with a high VS concentration (fresher faeces) has a greater potential for digestion, and release of biogas. The reuse potential of biogas from AD of FS can be used as source of renewable energy (Torondel, 2010; Strande and Brdjanovic, 2014; Buckley et al. 2008).

Design parameters for AD include a typical solids concentration between 3-20%, depending on the size of the digester being used. Monnet. (2003) described a low TS concentration as <10%, a medium TS as 15-20% and a high TS as > 20%. The higher TS will require a smaller digester volume due to the lower water content. Optimal organic concentrations should be between 60-80% VS and 20-50 g/l COD, with a higher VS% yielding a greater biogas production (Semiyaga et al., 2015; Lu, 2006; CDD Society, 2017). Typically, the biogas stabilisation reactors have a retention time of 5-15 days depending on the organic load input.

CDD Society. (2017) showed a successful pilot study faecal sludge treatment plant in Devanahalli, India that served 30 000 people and treated 6 m³ of FS daily. The site serves as a successful biogas production site, producing 45 m³ for 10 kL /day and 90 m³ for 20 Kl /day of biogas. While Porras and Gebresenbet, (2003) showed that human faeces produced a biogas of 0,02-0,28 m³ per kg of wet faeces.

Biogas production is influenced by COD, stability and temperature of faecal matter (Bakare et al., 2012). Faecal material stored for a short duration is preferred as a source of biogas, owing to its greater concentration of organic material, as the carbon content is reduced over time (due to ongoing biodegradation), subsequently lowering the VS% (Muspratt et al., 2014; Diener et al., 2014). Essentially, the ability in producing biomethane is reduced with time as the faecal matter is more stabilised, i.e., aged material is less desirable for the generation of biogas (Bakare et al., 2012).

AD can be easily scaled to fit the needs of smaller rural communities. The products yielded from this treatment can generate biogas, a clean energy which can be used for cooking or lighting in rural communities (CDD Society, 2017). Stabilised residue can further be used to improve soil fertility, which is particularly important in rural areas where agricultural productivity is essential to produce enough food for all the population (CDD Society, 2017; Singh et al., 2019).

2.4.2.4 Composting and vermicomposting

Composting of faecal sludge, according to Cofie et al. (2009) is associated with the aerobic degradation of FS. This process allows essential nutrients to be recovered, allowing FS to be used safely as an organic fertilizer (Cofie et al., 2016). Composting is a natural process that uses microorganisms to break down the FS organic material, whereas vermicomposting uses earthworms. Optimal operating parameters for composting require a C:N ratio of 25:1 to 30:1 and TS% of 30-40% (IWMI and SANDEC, 2002). A COD concentration between 2000 - 4000 mg/L enables the appropriate organic decomposition and pathogen reduction, which is achieved by temperature increases, microbial antagonistic effect, pH levels, nutrient depletion and moisture content (Kurniawan et al., 2010, Singh et al., 2017b, Manga et al., 2023).

Composting remains a good solution for rural areas where solid waste management poses a major challenge. Klingel et al. (2002) further explained that, owing to the low moisture content required for composting (50-60%), dewatering is needed prior to treatment, as FS and faeces typically exhibits high moisture contents (>85%). After composting and vermicomposting useful products formed include stabilised organic matter which can be used as fertiliser, earthworm biomass, and vermicompost, respectively (Singh et al., 2019).

2.4.2.5 Pyrolysis for Biochar

Pyrolysis is a thermal process whereby FS is heated in an oxygen-depleted environment to produce several products, such as biochar. FS is heated to temperatures between 300-700°C in a sealed reactor (without or extremely low levels of oxygen), allowing for the breakdown of organic material into biochar, gas, and bio-oil. Optimal operating parameters for pyrolysis requires ~ 70% TS. Despite the type of OSS, the FS will require pre-treatment to reduce the moisture content and increase its solids concentration. This is supported by Bridgwater (2018), Akhtar and Amin (2012) and Bridgwater et al. (1999) who showed that ideal moisture contents for FS should be between 10-30%. Pyrolysis for biochar has been successfully implemented in pilot studies in Senegal and Uganda respectively (Gwenzi et al., 2015).

While pyrolysis can be a high-tech process, it proves suitable for rural areas due to its scalability to meet the needs of small communities. This involves simplifying it into a lower-cost and rustic process, leading to reduced dependence on finite resources like water and a more space-efficient solution.

2.4.2.6 Solar drying in a greenhouse solar thermal system

Solar drying is a well-implemented and successful sludge treatment technology across Europe and Australia; however, pilot experiments are still being carried out in developing countries (Septien et al., 2019). The treatment harnesses the solar energy to evaporate moisture from FS, thereby reducing its volume and stabilising the sludge (Septien et al., 2019). Solar thermal systems convert the energy into heat for electricity, which can be used in cooking, drying, and lighting. Solar thermal energy is utilised by (1) solar thermal collection or (2) thermal concentration. Method 1 is used for heating whereas 2 is used for power generation (Septien et al., 2019).

Greenhouse thermal systems is one solar thermal drying technology among others, which operates when the sludge is applied in a thick layer above the ground. The sludge is frequently homogenised by either manual or mechanical techniques, the process if mixing is done to prohibit crust formation and ensure the distribution of the sludge (Septien et al., 2019). Herein the sludge absorbs solar radiation which passes through the clear walls of the greenhouse, the absorbed heat subsequently leads to moisture evaporation. However, small amounts of radiation can be lost by reflectance and absorbance on the clear wall. Greenhouses typically include ventilation systems and mixing fans for moisture evacuation and the creation of turbulence to reach higher drying rates (by increasing the mass transfer) respectively (Septien et al., 2019). Figure 14 illustrates a typical greenhouse solar drier. The primary consideration for a greenhouse design is based on drying rate (depends on solar radiance), the rate of production for sludge (depends on ventilation, mixing fan rates and thickness of beds) and the final dryness of the product (depends on initial solids concentration) (Seginer and Bux, 2006, Seginer et al., 2007).

Solar drying can remove the moisture from the faecal material to as low as 20%. Input FS should exhibit a ~ 20-30% TS to reach a final drying content between 60-95% TS (Socias, 2011, Stringel, 2020, Mathioudakis et al., 2013, Septien et al.).

The technology is suited to rural communities owing to its low energy requirements. It can be easily adapted to large- and small-scale processes, and is best placed in large, open spaces where the ecological footprint is lower (i.e., rural communities) (Magoya, 2012). The materials used in solar drying are simple, in turn lowering operational and maintenance costs.

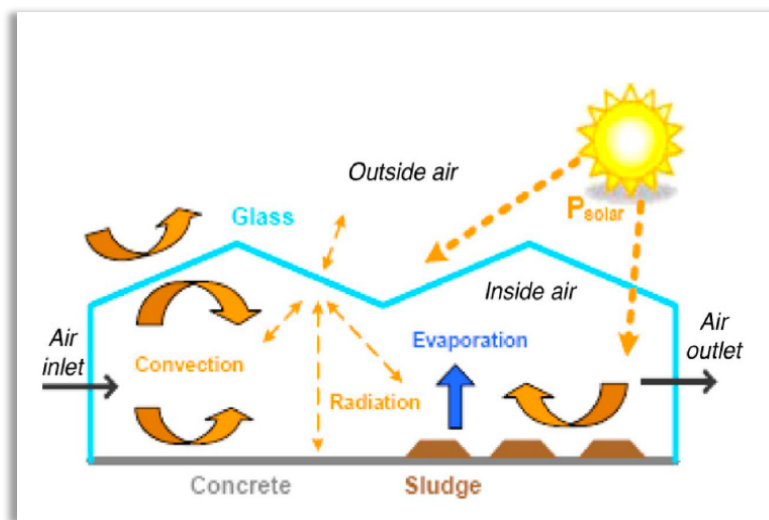


Figure 14: Schematic representation of the operation and structure of a greenhouse solar thermal system.
Source: (Septien et al.).

2.4.2.7 Deep row entrenchment

A possible option for utilising faecal material directly is through deep row entrenchments (DRE). The burying of sludge in deep trenches eliminates bad odours and decreases the possibility of exposure to pathogens, compared to indiscriminate open dumping of FS. Thereafter, high nitrogen requiring plants such as trees are planted on top the sludge for their successful growth. The anaerobic conditions within the trenches are responsible for minimising the nitrification and restricting the leaching of nitrates (Singh et al., 2019). Still and Taylor (2011) showed enhanced tree growth with no indication of groundwater contamination. Still et al. (2012) showed limited nitrate leeching and contamination of groundwater bodies from a DRE treatment site in Durban (South Africa). However, groundwater contamination is still being studied and should be evaluated on a case-by-case basis to ensure environmental protection (Strande et al., 2014). Important aspects to consider when practicing deep row entrenchment include soil permeability, water table level and distance to drinking water sources.

DRE is considered the most feasible FS treatment technology, especially in areas where there is extensive land space (Singh et al., 2019). It exhibits low operational and maintenance costs, requires no electricity or mechanical equipment, it can be developed using materials that are accessible in the local area, and operated by unskilled workers, making it highly suitable for rural areas (Singh et al., 2019).

2.5 Conclusions

The implications of inadequate sanitation facilities and FSM in rural schools is degenerating, leading to severe health and environmental consequences. To minimise these consequences and ensure safer FSM, this chapter examined existing literature to identify the different types of faecal sludge and faeces characteristics from VIPs, STs and MTs for its optimal valorisation and safe disposal. Sanitation systems significantly impact the faecal material characteristics. Sludge from STs and VIPs was outlined as stabilised faecal waste, while fresher faeces was outlined as high strength waste rich in organic content. This suggested that fresher faecal matter was better suited to generate end products like biogas, however, the more stabilised sludge (VIP and ST) can theoretically be dewatered more easily. Given the importance of faecal material characteristics on treatment and emptying pathways, the chapter examined the applicable treatment technologies and emptying options for rural areas. The following chapter will explore the methods used to characterise and collect faecal samples from rural school sanitation systems.

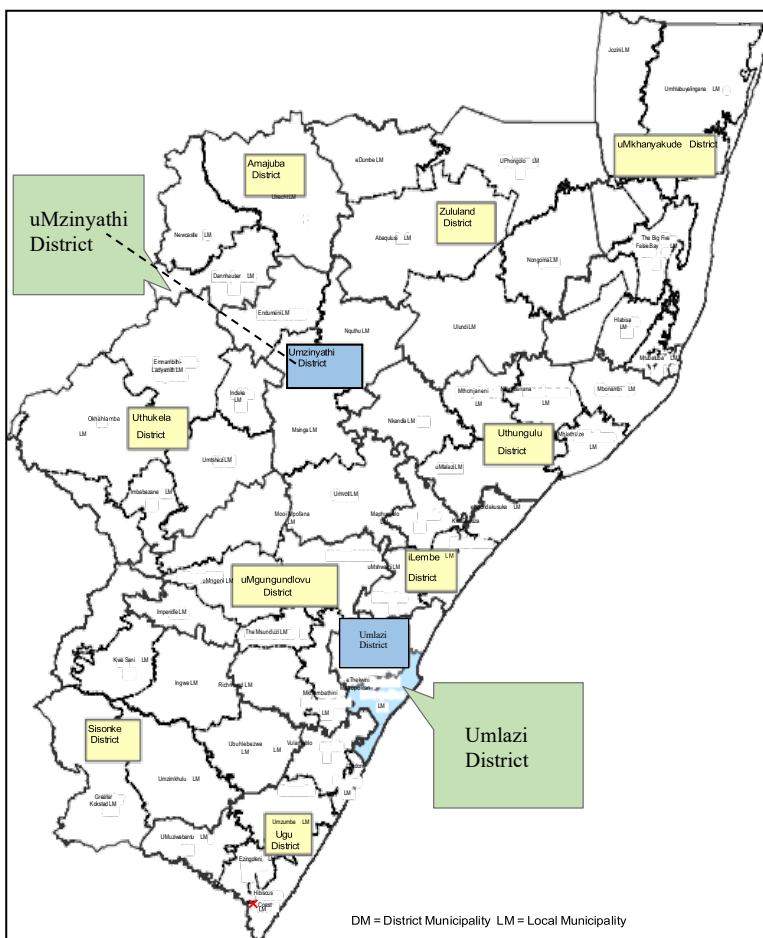
Chapter 3: Materials and Methods

This study investigated the characteristics of faecal material for its optimal valorisation and safe disposal. Samples were obtained from the most predominant onsite sanitation facilities within rural schools in the Umlazi and Pinetown district of KwaZulu-Natal. This chapter comprises a description of the feedstock, the research approach, and experimental work. It provides details on the statistical methods used for data analysis, (Kruskal-Wallis statistical test). Additionally, it outlines the methodology employed in the experiments, including the process of sampling.

3.1 Sampling site description

Samples were obtained from rural schools in Umbumbulu (Umlazi district) and uMzinyathi (KwaMashu) of KwaZulu-Natal. Figure 15 provides a map with the sample collection locations. Both locations exhibit a subtropical climate, which is characterized by warm temperatures and summer rainfall year-round.

District and Local Municipalities in KwaZulu-Natal



Rural School Location



Figure 15: Map of study area. Source: eThekweni GIS (2016).

3.2 Sampling Procedure

All research was conducted according to the Biomedical Research Ethics committee, from the University of KwaZulu-Natal research ethics office, Westville campus, Govan Mbeki building. The ethical clearance number BREC/00003672/2021. (Ethical clearance certificate provided in Appendix A).

Methods utilised for sampling involved the grab sampling technique at a specific location, depth, and time. This type of sampling method is appropriate for collecting a single sample at a particular site in a short time (Velkushanova et al., 2021). Details of the full method are provided in appendix F.

Faecal sludge and faeces were obtained from seven schools. Table 10 shows the breakdown of the sanitation system, school grades, learners age, number of toilets, and sludge types. Samples were collected with a cone-shaped device from 2-3 m within the system (Figure 16). One sample of 1 L was taken per toilet and placed in sampling containers. Each sampling bucket was labelled with a specific nomenclature and kept away from direct sunlight. The 1L airtight sampling buckets were then transported and stored in the UKZN-WASH R&D sanitation laboratory in a cold room at 4°C (avoiding physiochemical and biodegradation of material due to dehydration). Thereafter, faecal material was gently mixed with a metal spatula (to avoid changing structural properties, which is particularly important rheological and PSA tests) when analysed.

Onsite sanitation system	Grade of school	Learners age group	Number of toilets	Each sample tested	Type of samples
Mobile toilets	2- Primary school	~ 4-13	4	Triplicates	Fresh Faeces (< 1week old)
Pit latrines	1- High school	~ 14-18	3	Triplicates	Faecal Sludge
Septic tanks	2- Primary school 2- High school	~ 4-13 ~ 14-18	4	Triplicates	Faecal Sludge

Table 10: Type of sanitation system, schools and number of samples collected.

3.3 Testing procedures and analysis methods

All analytical tests performed for characterisation of faecal sludge and faeces were conducted according to the Methods for Faecal Sludge Analysis Velkushanova et al. (2021), and the standard operating procedures (SOPs) from the WASH R&D Centre, adopted from APHA (2017). The Link for Methods book and SOPs (with limitations) are attached in appendix F.

Quantitative research is associated with a variety of methods that involve the methodical investigation of social phenomena using statistical or numerical data approaches (Watson, 2014). The quantitative research of this study used SPSS version 28.0.0.0 (190) to test the significance of sanitation systems between data sets. A Kolmogorov- Smirnov test of normality was first conducted using SPSS. As the data was not normally distributed, a non-parametric Kruskal- Wallis one way analysis of variance was performed to test significance between data sets according to equation 2 (Fan and Zhang, 2012).

$$H = \frac{12}{N(N + 1)} \sum_{i=1}^k \frac{R_i^2}{n_i} - 3(N + 1), \quad \text{Equation 2}$$

Here n_i is the number of observations in the i^{th} group. N is the total number of observations (sum of all n_i) and R^i is the sum of the ranks of the n_i observations in the i^{th} group.

Thereafter, Microsoft Excel version 16.73 was used to establish relationships, display findings, and allow for experimental errors bars to be distinguished. Expected errors from the Kruskal- Wallis test is due to random factors. The statistical approach used to calculate error bars were done by the standard deviation method on excel, by quantifying how the data points deviated from the mean. All tests were conducted in triplicates.



Figure 16: Image of sampling tool and method for collection of faecal material.

The experiments performed on each sample are summarised in Table 11, and Table 12 provides a detailed summary on the significance of the analysed parameters.

Table 11: Characterization tests performed on each sample.

Chemical and Physico-Chemical Analysis	Mechanical Analysis	Nutrient Analysis	Thermal and Thermodynamic Analysis
pH	Density	Total Nitrogen	Thermal Conductivity
Electrical Conductivity	Sludge Volume	Total Phosphate	Calorimetry
Total Chemical Oxygen Demand	Index	Total Potassium	Water Activity
Solids (TSS; TS, TDS, VS)	Particle size Distribution	Ammonium	
	Rheology/ shear stress		

Table 12: Significance of the analysed parameters

Parameter	Significance to the study
pH	Assists in the understanding of water chemistry processes. The pH of a sample influences reaction rates, chemical speciation and biological processes.
EC	Measures the capability of an aqueous solution to conduct an electric current. Through measuring EC, the salinity of wastewater can be assessed, which is necessary for agricultural wastewater valorisation.
Total solids	Used for innovative design technologies and faecal sludge management decisions, e.g., on emptying and treatment
Total suspended solids	Used to determine the efficiency of solid/liquid separation technologies
Total dissolved solids	Indicate the presence of dissolved and colloid materials, which can implicate certain treatment technologies.
Volatile solids/ash	Volatile solids provide an indication of the organic material in the sample. The remaining ash corresponds to inorganic material from the post-oxidation of organic matter.
Chemical Oxygen Demand	Indicates the organic content and determines the degree of stabilisation in faecal material.
Nutrients <ul style="list-style-type: none"> - Nitrogen - Potassium - Phosphates - Ammonia 	Assesses potential end-use in agriculture.

C:N ratio	A balanced C and N ratio is necessary for aerobic/ anaerobic digestion, and biogas production.
Density	Relationship amongst mass and volume of a substance and indicates how compact a material is.
Shear yield stress	Determines flowability by determining the minimum force required for a material to flow.
Particle Size Distribution	Indicates faecal matter breakdown or the particle agglomeration and affects settleability and dewatering processes (assisting in the designing process). It is also a significant characteristic of the end product of FS treatment i.e., solid fuels.
Water Activity	Is indicative of the binding strength of moisture within faecal matter. It determines establishes the water available for biochemical and chemical reactions, as well as indicates microbial activity.
Calorific Value	Indicates the amount of heat energy released by faecal material when combusted. It evaluates the faecal sludge end-products e.g., biochar and biofuels.
Thermal Conductivity	Indicates the heat conduction capacity of faecal matter.

3.3.1 pH and Electrical conductivity

pH and Ec was measured with *HACH sensION MM374⁺* pH meter that comprised a glass electrode, reference electrode and magnetic stirrer with stir bar, and thermometer as shown in Figure 17. For dense faeces and faecal sludge, 1:10 dilution was executed. Samples were regulated till room temperature and poured into a beaker, ensuring complete immersion of the sensing element. Once the reading was stabilised the measurement and temperature were recorded. Limitations for this method involves strong acidic solutions with a < 1 pH yielding incorrect results.

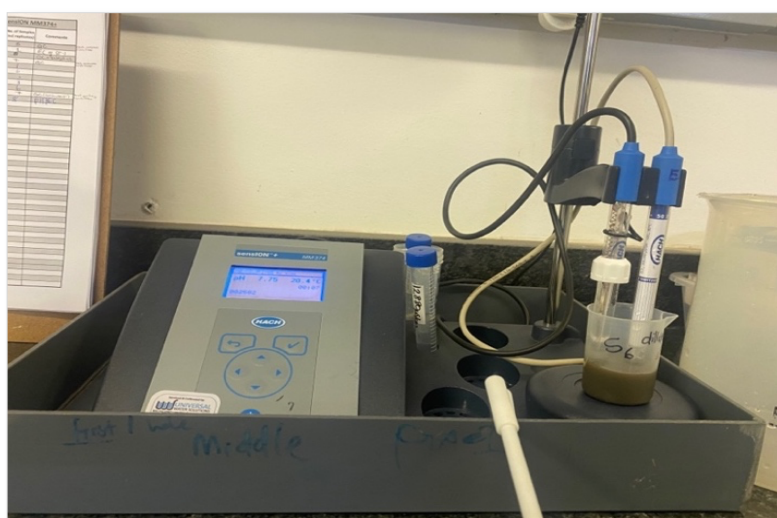


Figure 17: Apparatus used to measure pH and Ec

3.3.2 Solids testing

For all solids testing, larger inconsistent floating particles (hair, stones, maggots) should be removed as the inclusion of these will yield an incorrect result. No more than 20 g of sludge should be used, as this increases drying times and forms a crusty layer on the top of the crucible, trapping moisture within the sample. To ensure accurate measurements of the analytical balance and ovens must be calibrated weekly.

3.3.2.1 Total Solids

The gravimetric method using the oven drying technique was employed for both sludge and fresh faeces. A sample mass ranging from 10 to 20 g underwent drying in a *Scientific 276* oven for 24 hours at 105°C (Figure 18). Subsequently, it was re-measured on an analytical balance, and the total solids were calculated according to equation 3.

$$\text{Total Solids in Wet Sample} \left(\frac{g}{g} \right) = \frac{(W_2 - W_1)}{W_{\text{sample}}} \quad \text{Equation 3}$$

The provided notations are as follows, W_1 represents the mass (g) of crucible after oven drying, W_2 notates the mass (g) of crucible as well as the sample and W_{sample} = mass (g) of sample.



Figure 18: Faecal material after oven drying.

3.3.2.2 Volatile solids

Volatile solids were analysed according to the ignition method. The dried residue from total solids were weighed and placed in a *RJMSystems E-300 (DC) J4 (Economy LES-E Range)* muffle furnace at 550°C for 2 hours, to correspond to the samples organic content (See Figure 19). Samples were then placed in a desiccator to cool until ambient temperature was reached. Thereafter samples were measured on an analytical balance. The VS concentration on a dry basis was calculated according to equation 4 and 5.



Figure 19: Faecal material after furnace

$$\text{Volatile solids in wet sample } \left(\frac{g}{g} \right) = \frac{(w_3 - w_4)}{(w_2 - w_1)} \quad \text{Equation 4}$$

$$\text{Volatile solids in dry sample } \left(\frac{g}{g} \right) = \frac{(w_3 - w_4)}{(w_3 - w_1)} \quad \text{Equation 5}$$

The notations are as follows, W_1 represents the Crucible mass (g), W_2 is the Crucible mass and wet sample mass (g), W_3 represents the Crucible mass and the sample after drying (g) and W_4 is the Crucible mass and sample after incinerating (g). To determine the sample mass after drying in (g), the following equation is used ($W_3 - W_1$), and to determine the sample mass after incinerating the following equation is used ($W_4 - W_1$) = Sample mass after incinerating (g)

3.3.2.3 Total Suspended Solids

TSS was measured by oven drying method. Pre-weighed glass fibre filter (*Whatman No 4: 20-25 μ m*) with a diameter of 110 m was placed on a butcher funnel attached to a filtration flask. For slurry samples, 30 ml of the sample was vacuum filtered through the filter paper. For more solid samples (Fresh faeces), 1g of sludge was diluted into 100 ml of water ensuring the filter paper did not clog. Filter paper was then placed on an aluminium weighing boat and dried in a *Scientific 276* oven at 105°C for 1 hour, removing all moisture and then re-weighed. The increase of mass on the filter paper corresponds to the TSS. Equation 6 and was used to calculate TSS.

$$TSS = \frac{(w_2 - w_1)}{V_{\text{sample}}} (X DF) \quad \text{Equation 6}$$

The notations are as follows, W_1 represents the weight of filter paper and the Crucible/weighing boat before drying (g), W_2 is the weight of residue and crucible/weighing boat after drying (105°C) (g). DF represents the dilution factor and V_{sample} is the Volume of sample used (L).

3.3.2.4 Total dissolved solids

TDS was measured according to oven drying method using the filtrate from TSS. 30 ml of the remaining TSS residues (from the bottom of the funnel) was poured into weighed porcelain crucibles and placed in *Scientific 276* oven to dry for 24 hrs (Figure 20). Thereafter the total sample mass was measured on analytical balance.



Figure 20: Samples for the TDS determination

3.3.3 Moisture content

Moisture content was measured according to equation 7. Whereby the differences between the wet and dry solid masses measured from the total solid analysis, provided the moisture content of the sample.

$$\text{Moisture content in wet sample} = 1 - TS \quad \text{Equation 7}$$

The notations are as follows, TS represents the Total solids.

3.3.4 Spectrophotometric tests

All nutrient (N, P, K, NH_4) and COD tests were measured using a *Spectroquant Merck-Prove 300* and heated with *Lovibond RD 125* digester block. The *Spectroquant* had pre-defined test methods and corresponding concentrations programmed into it, depending on the nutrient test the targeted range was selected. For the sample preparation, 1-2 g of faecal material was liquified in a *Nutribullet* blender with a 100 ml of distilled water. For certain tests additional dilutions were performed ensuring concentrations were within measuring range of their respective test kits. However, all dilutions should be performed

according to accurate measurements based on the type of sludge, as turbid solutions may yield inaccurate readings.

3.3.4.1 Chemical oxygen demand

The COD test operates based on the principle that the oxidisable organic compounds are oxidised by $K_2Cr_2O_7$ (potassium dichromate) subsequently reducing $Cr_2O_7^{2-}$ to green Cr^{3+} . COD is then quantified spectrophotometrically. COD was measured using *Merck* chemical oxygen demand spectrophotometric test (Cat No.114555) with measuring range of 500-10 000 mg/l. Pre-treated sample was preheated in digestion block for 2 hours at 130°C, thereafter, cooled for 30 minutes and measured in spectrophotometer. Important quality control for measuring COD involves, that the value obtained from the Spectroquant only remains stable for 60 minutes therefore must be read in that timeframe. Further limitations are referred to manufactures protocol within the commercialised test kits and SOPs (appendix F).

3.3.4.2 Total Nitrogen

The Total Nitrogen (TN) test operates based on the oxidation of all nitrogenous compounds with persulfate digestion to NO_3^- , followed by a reduction process of Cd to measure the NO_3^- . TN is then quantified with spectrophotometrically. TN was measured using *Merck* total nitrogen spectrophotometric test (Cat. No. 1.14763) with a measuring range of 10 – 150 mg/l. Pre-treated sample was heated in a digester block at 150 °C for 30 minutes, thereafter, cooled and measured on spectrophotometer. Interferences in FS include chloride and bromide ions. Specific concentrations of these have been referred to in the manufactures instructions within the *Merck* total nitrogen spectrophotometric test.

3.3.4.3 Ammonium Test

The Ammonium test operates based on ammonia reacting with ClO^- (hypochlorite) to form NH_2Cl (monochloramine). Thereafter the NH_2Cl reacts with phenate forming 5-aminophenate which oxidises to indophenol and is catalysed by sodium nitroprusside. The Indophenol is thereafter quantified spectrophotometrically. The method measures ammonium ions and dissolved ammonia. Ammonium was measured using *Merck* Ammonium spectrophotometric test (Cat. No. 1.0068) with a measuring range of 5-150 mg/L. To separate faecal particles that can cause interference with the analysis and recover the liquid phase of the sample, a *HERMLE Z323* centrifuge was used at 5000 RPM for 15 minutes (Figure 21). The prepared solution was left to stand for 15 minutes before spectrophotometric measurement. Subsequently, it was transferred to a cuvette (Figure 22) and inserted into the spectrophotometer for reading. For precise concentrations, refer to the manufacturer's guidelines within

the Merck total nitrogen spectrophotometric test. Common interferences in FS include magnesium nitrite..



Figure 22: Centrifuge used to separate the liquid phase from the solid phase.

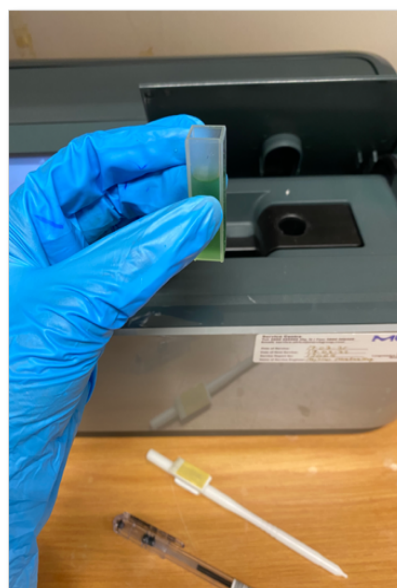


Figure 22: Cuvette with pre-treated sample

3.3.4.4 Phosphates

Phosphates was measured using *Merck* total phosphorus and orthophosphate spectrophotometric test (Cat. No. 1.14543) with a measuring range of 0,05-5 mg/L. The Pre-treated sample was thereafter preheated in digestion block for 30 minutes at 120°C, then it was regulated till room temperature, and was measured in a spectrophotometer. Operation of Phosphate test is based on the on the principle that within the H₂SO₄ (sulphuric acid solution), the -PO₄³⁻ (orthophosphate ions) react with M₂MoO₄ (molybdate ions) forming phosphomolybdic acid, which is reduced by ascorbic acid, and this has been quantified in spectrophotometer measuring total phosphorus. Typical interferences include ammonium, nitrites, sodium, and chemical oxygen demand. Further limitations are referred to in the manufactures for *Merck* total phosphorus protocol within the orthophosphate commercialised test kits and SOPs (appendix F).

3.3.4.5 Potassium

Potassium was measured using *Merck* Potassium Cell spectrophotometric test (Cat No. 114562) with a measuring range of 5-50 mg/L. Thereafter the pre-treated sample was preheated in digestion block for 30 minutes at 120°C. It was then regulated till room temperature and measured in spectrophotometer.

3.3.5 Carbon and Nitrogen Tests

The total carbon (C) and nitrogen (N) composition of faecal material was measured by a *LECO-TruMac-CNS Series 928* analyser. A well homogenized sample of ~ 0,2 g was pre-weighed and placed in a ceramic boat. The ceramic boat was loaded within the instruments purge chamber, which contained pure oxygen and facilitated combustion at a temperature of 1350 °C. The method is based on the principle that, the resultant gases in the form of carbon dioxide, NO_x and SO_x were split by chromatography and quantified within the instrument's thermal conductivity cell. The carbon and nitrogen as a proportion of wet sample mass was generated by allowing, for the interpretation of the elemental mass fraction.

3.3.6 Density displacement method using a Eureka can

A Eureka can was used to measure density (Figure 23), according to Archimedes' original method (Webb & Orr, 1997). ~30 g of sludge sample was placed in a measured crucible. The weight of crucible and sample was recorded. Thereafter, the Eureka can was filled with distilled water until the excess water flowed out the spout. The initial water level was recorded, subsequently the ~30 g of sample was added to the water-filled eureka can, until fully submerged. The water level increased due to displacement caused by the sample particles, and the new water level was recorded. The displaced volume was measured against the volume and mass of the sludge. The final density calculation was measured according to equation 8. Limitations for this method, assumes that the sample being measured does not dissolve within the water, and that there are no air bubbles on the objects surface.

The following calculation was used to measure final density.

$$D = \frac{m}{V} \quad \text{Equation 8}$$

The notations are as follows, *D* is the density (g/cm³), *m* is the mass of sludge (g) and *v* is the volume of water (ml)



Figure 23: Experimental setup used to measure the density of faecal material.

3.3.7 Particle Size Analysis

Particle size distribution was conducted using a *Malvern Mastersizer 3000* analyser that measures particle size using laser diffraction (Figure 24). Approximately a $\frac{1}{4}$ of a 4 ml spatula of faecal material was dispersed in 600 ml of water, until obscuration range (Between 10-20% for a wet dispersion unit, range was displayed on the computer screen and dependent on the sample) was reached. The method operates on the principle that the instrument employs a laser diffraction, by measuring the intensity of light scattered, when laser beam traverses through a dispersed particulate sample. Thereafter results were generated from the scattering pattern, forming a PSD curve of volume density (%) against particle size by the *Malvern Mastersizer 3000* software (Figure 25). The particle size range was 0,01-3500 microns. Interferences for this method includes foreign particles or suspended solids within the sample such as hair or stones. For a more detailed description on interferences, refer to the manufacture's guidelines.



Figure 25: Malvern Mastersizer 3000 used to determine particle size distribution.

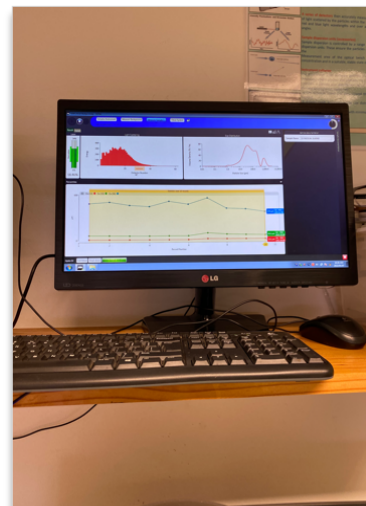


Figure 25: Description of generated results

3.3.8 Rheological properties

Shear stress tests were performed on faecal material at 20°C utilising an Anton Paar Rheometer MCR72 (Figure 26) and *Viscotherm 2* (for reducing temperatures). Shear stress tests were performed with 100-150 g of faecal matter. A vane-in-cup geometry (Figure 26) was used for the rheometer to sample shear rates ranging between 0,1-1000 s⁻¹ for 15 minutes. Each shear step was generally maintained for 1-2 s until the sample adapted itself to each measuring point. Data from the rheometer was captured through *Rheocompass software*TM which created rheograms that visually depict shear rate on a logarithmic scale.



Figure 26: Rheometer and vane in cup geometry used to determine shear stress.

3.3.9 Sludge Volume Index (SVI)

Approximately 1 g of room temperature faecal material was measured on a weighing boat and placed into a volumetric flask. Thereafter the volumetric flask was topped to 1 L and homogenised with the faecal material. The mixture was poured into a Imhoff cone until a 1 L marking and left to settle for 30-45 min at room temperature (Figure 27). After the time period, sludge volume of settled sludge and settling time were recorded. This method is based on the number of suspended solids that settle within 30-45 minutes. Interferences, such as disturbances to the cones (i.e., bumping) during the sludge settling process, can lead to inaccurate results. Additionally, due to settling behavior being influenced by temperature, it is advisable to ensure that the sludge reaches room temperature before measuring the Sludge Volume Index (SVI). Equation 9 was used to calculate settled sludge volume.

$$SVI \left(\frac{mL}{g} \right) = \frac{\text{Settled sludge volume} \left(\frac{mL}{L} \right)}{\text{Suspended solids} \left(\frac{g}{L} \right)} \quad \text{Equation 9}$$



Figure 27: Settled sludge volume in Imhoff cones during the SVI tests.

3.3.10 Calorific Value

The calorific value of faecal material was determined using a *Parr 6200* calorimeterTM. Samples were dried at 105°C for 24 hours prior to testing. Thereafter, 0,5-0,7 g of sample was weighed and placed in bomb calorimeter to obtain the calorific value. The method is based on the full combustion of the pre-weighed sample, with the calorific value representing the calorific energy or higher heating value (HHV) of a dry sample. The test should operate in a room between 20-25°C and the sample should not be over 1,1 g as the standard bomb cannot withstand combustible chargers > 8000 calories.

3.3.11 Water Activity

Water Activity was measured using *Aqualab Tunable Diode Laser- TDL*. Experiments were performed at a predetermined temperature of 25 °C (maintained by the instrument). Samples were gently mixed using a metal spatula and poured into given sample cups, to a demarcated level. Thereafter, a well-fitting lid was used to tightly seal each sample cup, which prohibited moisture loss while awaiting the measurement. The method operates on the principle that a hygroscopic polymer sensor within the water activity meter measured relative humidity of the air present in the space above the sample. The water activity is equivalent to the relative humidity measured in the cell at the thermodynamic equilibrium. The precision of the instrument was $\pm 0,005$ at a temperature of 25°C.

3.3.12 Thermal Conductivity

Thermal conductivity was measured with a *C-Therm-TCI™* thermal conductivity analyser (Figure 28 and 29). For these tests, 1,8 ml of homogenised sample was measured using 3 scoops of a 0,63 ml spatula. Density was determined by dividing the total mass of the sample in the spatula by its volume, thereafter the data was input into the software interface. The sample was spread as a thin layer on the sensor face (Figure 29) and covered with a blotter, which enclosed the sample during the test. The values for thermophysical properties were provided by the apparatus (*C-Therm TCI* Software) through indirect calculation. Thermal conductivity was measured by comparing the sensor (Figure 29) response with a factory calibration. The method operates on observing the rise in temperature within the sample, during and after exposure to brief heat stimulus. Typical interferences for this method include output cells (on results sheet, reported by *C-Therm TCI* Software) being red, or R^2 being $< 0,995$ rendering the value incorrect. Similarly, an orange output cell implies that the thermal conductivity is greater than the calibration range for that material.



Figure 28: C-Therm-TCI™ thermal conductivity analyser

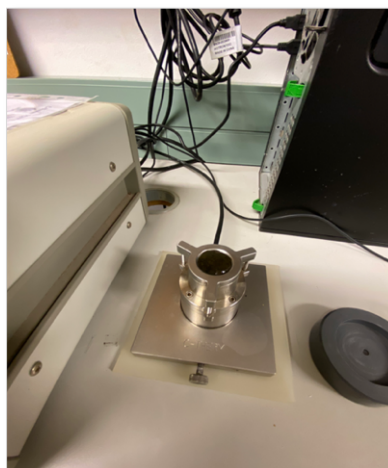


Figure 28: Sensor used for determining thermal conductivity

3.4 Conclusion

This chapter provided an overview of the methods and materials in this study. The chapter discussed the research site, sampling procedure, feedstock characteristics, testing procedures and statistical analysis used. The robustness of the methodology was enhanced through performing triplicate testing on samples, and well-established experimental methods. The following chapter will discuss and determine the significant relationships determined from the data analysis with the results being represented visually in the form of graphs and tables.

Chapter 4: Results and Discussion

This chapter discussed the experimental results obtained from the characterisation of faecal material from onsite school sanitation systems. Findings are presented in four sub-sections, including (1) physico-chemical, (2) thermal, (3) thermodynamic and (4) mechanical properties.

Summary of results

MTs displayed the highest concentration in almost every physico-chemical parameter analysed including, TS, VS, COD, and all nutrient tests (N, P, K and NH₄-N), indicating its high organic matter. Faecal sludge from STs demonstrated a higher degree of stabilisation compared to that from MTs. However, analysed sludge characteristics from school STs displayed slightly different physical characteristics when compared to literature and community-based sanitation systems. Faecal sludge from VIP latrines revealed the highest level of stabilised sludge, likely owing to the extended duration the sludge was maintained within its containment. This was supported by laboratory findings which showed VIPs exhibited the lowest VS% (45,5%) and COD (72,6 g/L) concentrations. The higher degree of stabilisation of sludge from VIPs requires less intensive stabilisation treatment processes, such as MTs prior to environmental discharge, subsequently making it a less costly FS to handle. Faecal sludge and faeces from all school sanitation systems (MTs, VIPs and STs) exhibited a water activity of ~1 a_w. Similarly, thermal conductivity for each OSS (MTs, VIPs, STs) represented a narrow ranged between 0,48-0,59 W/K/m. Whereas, calorific value on a dry basis was highest from MTs containing fresher faeces at 23,2 (MJ/kg dry solid). Particle size for school toilets ranged between 0,7- 2046,7 µm 0,6 - 1202,3 µm and 0,7-1492,5 µm for VIPs, MTs, and STs, respectively. Ultimately faeces from MTs were considered to have high strength. VIPs demonstrated the highest TSS, TDS and lowest SVI at 3,6 g/L, 1422 mg/L and 75 ml/g respectively

4.1 Physico-chemical results of faecal material from school sanitation systems

4.1.1 Solids concentration

4.1.1.1 Total solids concentration

The total solids concentration from school onsite sanitation systems are illustrated in Figure 30. The box and whisker plot provide the minimum, maximum, median, 75th and 25th quartile values for school Septic Tanks, Mobile Toilets, and Ventilated Improved Pit Latrines sanitation systems.

TS findings showed that fresher faeces from MTs exhibited the highest TS% concentration with an average of 17,4% ± 2,8 and a range of 14,4 - 20,5%. This finding was within the range 14-37%

supported by Rose et al. (2015), Penn et al., 2017, and Wignarajah et al. (2006). STs showed a mean TS of $10,2\% \pm 4,3$ and a range of 5 – 15% which was higher than the reported results from Velkushanova et al. (2021) who outlined a $<5\%$ TS concentration from community-based septic tanks. The higher-than-average TS concentrations from STs in school sanitation systems are likely due to the lower than usual “flush water” entering the STs, owed to prolonged water shortages experienced in rural communities. VIPs exhibited a mean TS concentration of $8,5\% \pm 1,9$ and a range of 6,8 – 12,2% which corresponded to findings by Zuma et al. (2015) and Velkushanova. (2014), however was lower than the reported range 20-30% by Septiens et al. (2020, 2018) and Naidoo et al. (2020). This is likely caused by a higher water infiltration within pit containments (can be used to extend the lifespan of a VIP) in school VIPs, compared to samples collected from household latrines.

Statistical analysis showed that TS% is significantly affected by the sanitation facility type ($P < 0,001$ for $N=33$). Which highlights the importance of considering type of onsite sanitation system when assessing TS%.

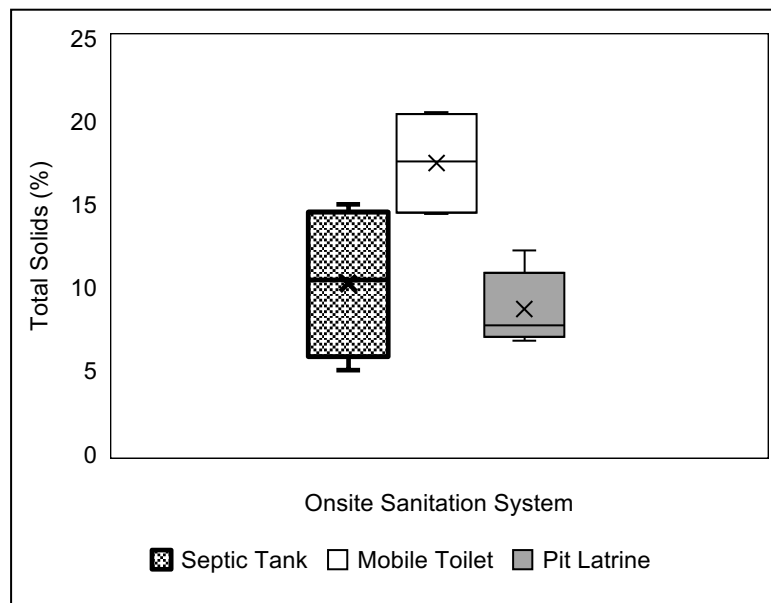


Figure 29: Box and Whisker plot showing TS% from school OSS.

4.1.1.2 Total Suspended and Dissolved solids

The TSS and TDS for school MTs, STs and VIPs are presented in Figure 31 and 32. The box and whisker plot provide the minimum, maximum, median, 75th and 25th quartile values for school ST, MT, and VIP sanitation systems. Table 20 (Appendix b) further provides a comprehensive summary of the physico-chemical findings obtained in this study. Figure 31 showed fresh faeces from MTs illustrated

the highest average TSS of 121,6 g/L, which agree with the results from Zewde et al. (2021) and Dangol (2013), however higher than results reported by Koottatep et al. (2005) and Heinss et al. (1998). The high TSS in MTs are likely due to the increased solid matter and greasier texture exhibited in fresher faeces. VIPs and STs exhibited TSS concentrations of 107 g/L, and 77,9 g/L respectively, which were within the range reported by Jayathilake et al. (2019), Junglen et al. (2020), and Ziebell et al. (2016). Figure 32 shows VIP sludge exhibited the highest mean TDS content 42,3 g/L within the range of Olatunji and Oladepo (2013), followed by MTs with an average of 33,4 g/L, and STs with an average of 9,6 g/L which agree with results from Osei et al. (2015). Statistical analysis showed that TSS was not statistically significant i.e., the distribution was similar across sanitation systems (P=0,55 for N=33). For TDS, statistical analysis showed that results were significantly affected by the sanitation facility type (P=0, 022 for N=33).

Shirin & Yadav (2014) and Kutty et al. (2011) outlined that TSS and TDS play a key role in wastewater and faecal sludge treatment. High TDS values (500 – 1000 mg/L) as seen in VIPs and MTs, exhibit several implications such as reduced dewaterability, owing to dissolved solids impeding the settling and filtration processes through increased viscosity, subsequently impacting the efficiency of coagulation and flocculation processes used for treatment. Research by Wu and Maskaly (2018) showed that TDS over 3000 mg/L led to reduced COD concentrations.

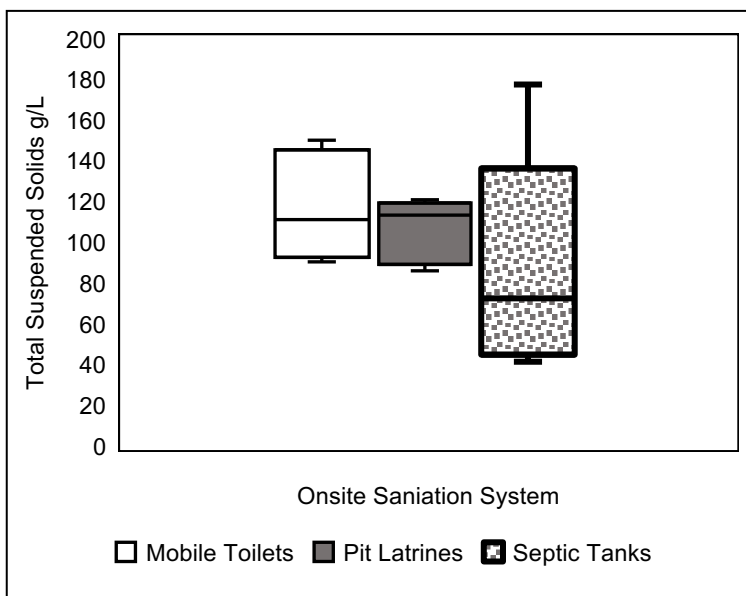


Figure 31: Box and Whisker plot showing TSS from school OSS.

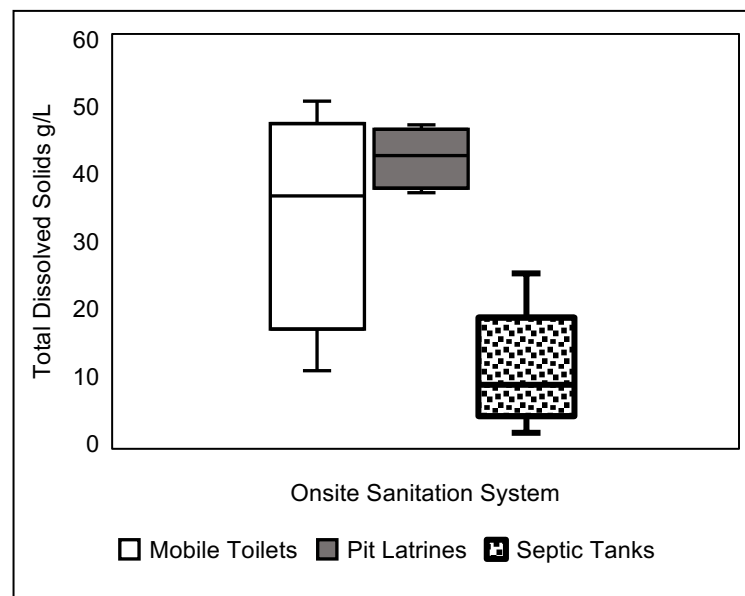


Figure 30: Box and Whisker plot showing TDS from school OSS.

4.1.2 Organic composition

4.1.2.1 Volatile solids

The volatile solids concentration, which signifies the organic composition of faecal material from school MTs, VIPs and STs are illustrated by a box and whisker plot in Figure 33. The data shows the minimum, maximum, median, 75th and 25th quartile values.

The organic composition and variation in biodegradable organics can be explained by the storage time of faecal material within the OSS. Shorter retention times do not allow for significant stabilisation, while a longer retention time does. Faeces from MTs were collected on a regular basis (weekly) and therefore exhibited relatively fresh faecal material undergoing little degradation, this was substantiated by the high average VS content of $78,4\% \pm 7,7$. The VS% for MTs were comparable to the organic composition of faeces outlined by Rose et al. (2015) (~85% VS). STs exhibited an average VS concentration of $77,4\% \pm 9,8$ similarly implying minimal degradation, whilst FS from VIPs displayed the lowest VS content at $45,5\% \pm 5,6$ indicative of the advanced stabilisation and trash disposal within the pits. The VS for VIPs were in agreement with those by Buckley et al. (2008). However, Heinss et al. (1999) outlined a higher VS concentration from VIPs (58-69%). The lower VS in this study could be the result of a prolonged storage period, resulting in a lower organic matter content.

Statistical analysis showed that VS% is significantly affected by the sanitation facility type ($P < 0,001$ for $N=33$).

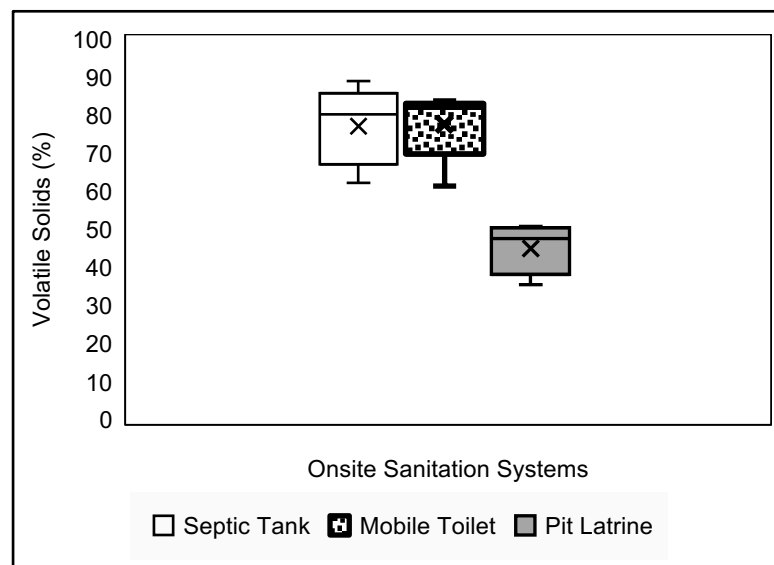


Figure 32: Box and whisker plot showing VS% from school OSS.

4.1.2.2 Chemical oxygen demand

Figure 34 shows a box and whisker plot of the COD for school sanitation systems, these data visually inform the minimum, maximum, median, 75th and 25th quartile values. Raw data values for, the COD are also presented in Table 20 (See Appendix B).

MTs exhibited the highest COD values at 242 g/L, supported by Gaillard (2002), Elmitwalli et al. (2006), and Luostatinen et al. (2007). The COD concentration of MTs suggest a high strength sludge. STs exhibited an average COD of 119,2 g/L, which is comparable to findings by Heinss et al. (1998). Lastly, VIPs exhibited the lowest COD concentration of 72,6 g/L falling within the range 49- 103, g/L found by Heinss et al. (1999), Doku (2003), Chaggu (2004), Coetzee et al. (2011), Hawkins (1982), Foxon et al. (2009), Salisbury et al. (2009), Buckley et al. (2008) and Norris (2000). Statistical analysis showed that COD is significantly affected by the sanitation facility type ($P < 0,001$ for $N=33$).

The analysis of organic composition indicates sludge from VIPs are of low strength. The low strength is attributed to the extensive degradation prior to desludging (Strauss et al., 2000). While fresher faeces from MTs illustrated a sludge of “high strength”, which is consistent with work from Strande et al. (2014) and Buckley et al. (2008). The high strength sludge, (as seen in fresher faeces from MTs), is attributed by partial to almost no degradation and high organic matter content, in turn can be degraded more easily (Strauss et al., 1997). Whereas low strength FS (as observed in VIPs) suggests that there is a lower amount organic matter and significant degradation, resulting in low degradability potential for this sludge type (Strauss et al., 1997).

COD findings also showed a strong correlation ($R^2 = 0,8$ and $R = 0,9$) to TS%, summarising that as COD concentration increased so did the TS% (see Figure 35). The correlation of TS and COD has important implications for FS treatment and emptying as high TS% can indicate the presence of high organic matter within the sludge (as seen in MTs). This finding was supported by Chanadana and Rao (2021), who showed COD concentration was imperative for selecting emptying techniques.

To determine organic content for biological degradation and treatment efficiency VS and COD were correlated. The relationship between COD and VS% is presented in Figure 36, which suggested a moderate positive correlation ($R = 0,5$) between VS and COD, indicating that as COD increases there is an expected increase in VS%. However, the coefficient of determination (R^2) suggests only 29% of the variability can be explained by the linear relationship with COD, which could be due data heterogeneity and outliers. High organic content values of sludge and faeces in MTs and STs indicates anaerobic treatment is a suitable treatment method, preventing active microorganisms from lysing (Zhen et al., 2017).

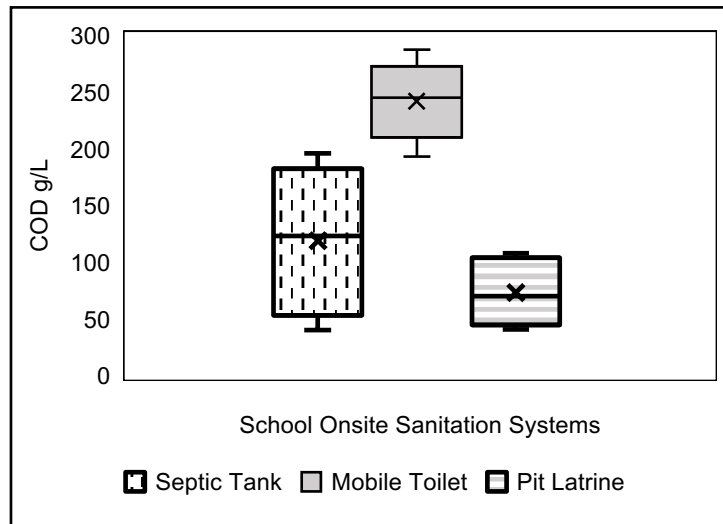


Figure 33: Box and whisker plot showing COD from school OSS.

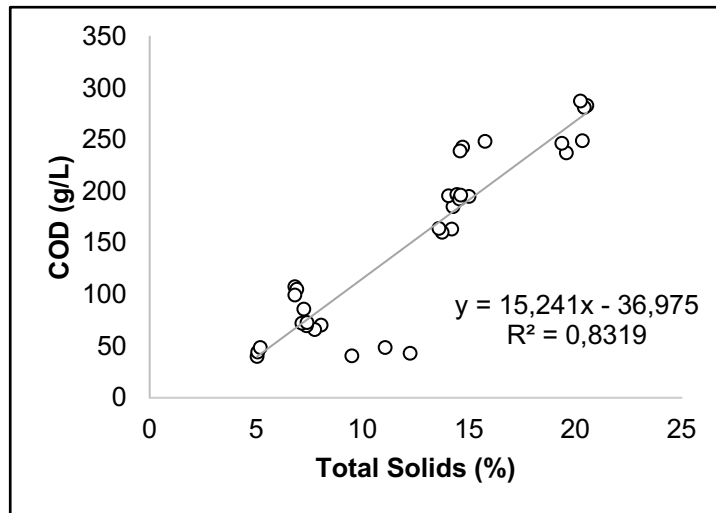


Figure 34: Relationship between COD and TS%

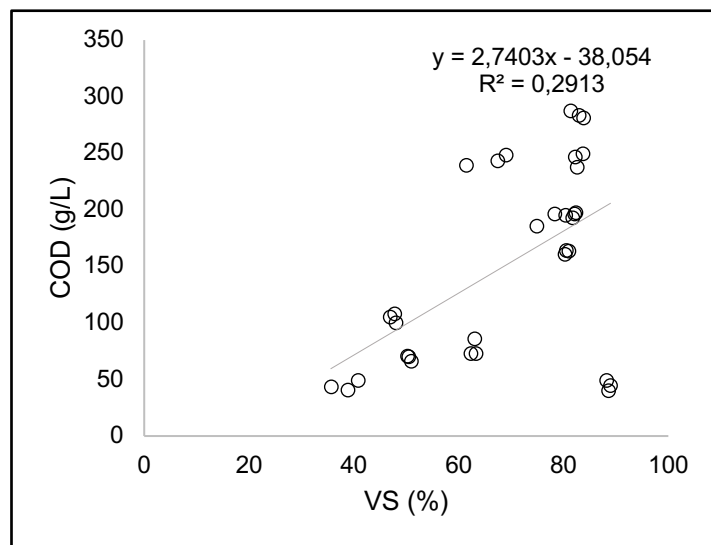


Figure 35: Relationship between COD and VS%

4.1.3 pH and Electrical conductivity

4.1.3.1 pH

The pH for school MTs, STs and VIPs are presented in a bar graph as seen in Figure 37. STs and VIPs showed pH measurements of $7,3 \pm 0,3$ and $8,5 \pm 0,09$ respectively, these findings were supported by Heinss et al. (1999). In contrast to this, the fresher faeces from MTs were more acidic (5,63), which could be accounted for by the chemical additive used in the toilet as an odour suppressant. Cioabla et al. (2012) showed pH values between 6,8- 7,2 are optimal for AD, as a result, sludge from STs school systems are suitable for AD treatment (High VS and suitable pH). Similarly, Velkushanova et al. (2021) outlined that other biological treatment and pre-treatment techniques needed for dewatering required pH levels between 6-9. Hence values outside this range, as seen in MTs (owing to the chemical additives), contributes to disputes within biological processes or corrosion.

Statistical analysis showed that pH is significantly affected by the sanitation facility ($P < 0,001$ for $N=33$).

4.1.3.2 EC

The EC values for school OSS sludges are shown in Figure 37, which presents a line graph showing EC concentrations of $7675 \mu\text{S}/\text{cm}$ and $23341 \mu\text{S}/\text{cm}$ for MTs and STs respectively, whilst VIPs illustrated the highest EC at $30583,3 \mu\text{S}/\text{cm}$. These findings were equivalent to those from Ahamed et al. (2019). Rusydi (2018) explains that the high EC in VIPs is likely due to the increased TDS (particularly ions), which must be considered for operational activities. Figure 38 explored the correlation between TDS and EC and found a positive correlation ($R=0,61$). However, the R^2 suggests 38% of the variability can be explained by the linear relationship with TDS, however, this could be due to data heterogeneity and outliers such as temperature, pressure, and the presence of other chemicals. These findings were in agreement with McNeil and Cox (2000). Hence the ionic content of faecal sludge from VIPs is a possible cause of the high EC values observed in this system (Ahmed et al., 2019).

Statistical analysis showed that EC is significantly affected by the sanitation facility type ($P < 0,001$ for $N=33$).

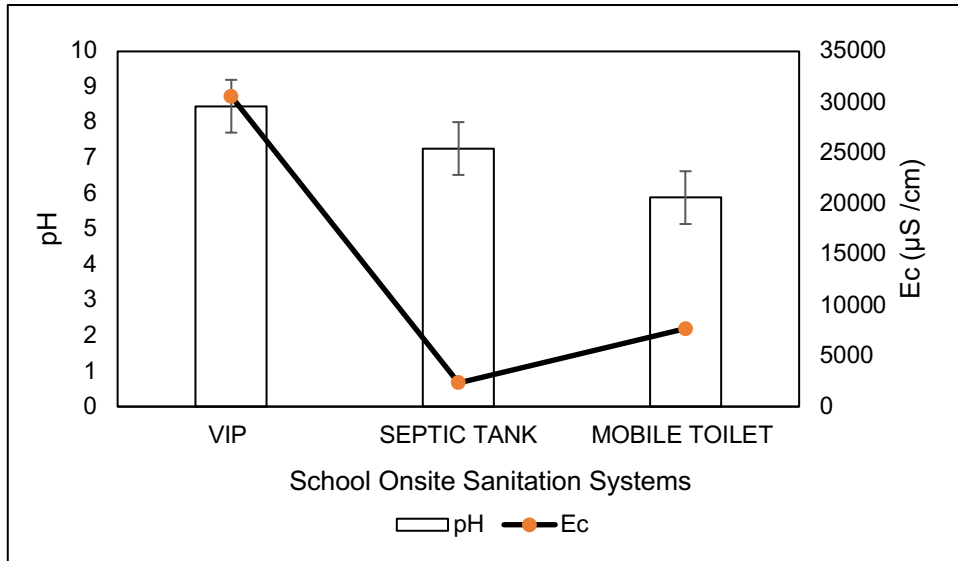


Figure 36: Graph showing pH and EC of school OSS.

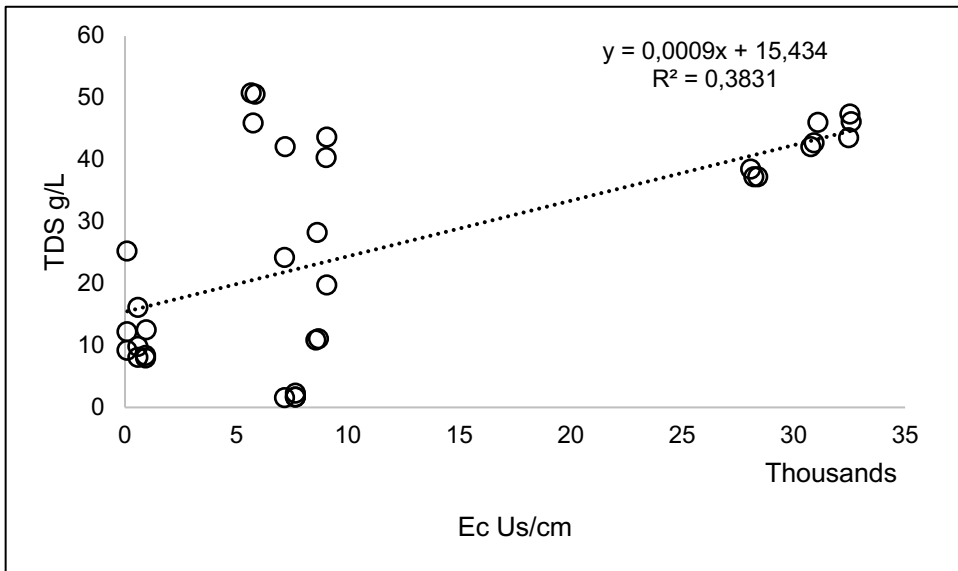


Figure 37: Relationship between EC and pH

4.1.4 Nutrient properties

Figure 39 shows a box and whisker plot of the nutrient concentrations in faecal material for school sanitation systems, these data visually inform the minimum, maximum, median, 75th and 25th quartile values. Results were presented on a dry basis to avoid the dilution factor and variability caused by water and moisture content, in turn providing a more accurate assessment on nutrient content. Additionally, raw data values for the nutrient concentration were also calculated on a wet basis and presented in Table 20 (See Appendix B). The results showed MTs had the highest mean nutrient composition overall. Nitrogen from MTs exhibited the highest average concentration of 54,8 g/kg, followed by VIPs at 34,2

g/kg, and STs at 23,78 g/kg, respectively. Potassium showed similar trends with the highest concentrations reported from MTs at 18,4 g/kg, followed by VIPs at 11,31g/kg and STs at 8 g/kg, respectively. However, highest mean Ammonia and Phosphorus concentrations were observed from VIPs at 25,9 g/kg and 13,2 g/kg, followed by MTs at 19,4 g/kg and 12,65 g/kg, and STs at 7,24 g/kg and 6,4 g/kg. All values were within the ranges supported by Krueger et al. (2021a) and Septien et al. (2020).

The combination of high nitrogen and ammonia concentrations established from fresher faeces (in MTs) presents itself problematic for environmental discharge, poses risks on crops (as they contain high levels of nitrate) and increase the risk of groundwater contamination (contamination of nitrate and nitrite) (Dich et al., 1996, Heinonen-Tanski and van Wijk-Sijbesma, 2005). Therefore, when quantifying permissible FS loads nitrogen and ammonia is a significant design parameter. Hence, treatment needs to be carefully managed and implemented to ensure these levels are brought down to the regulatory limit (1,0 mg/l) for safe valorisation.

Additionally, the percentage of N-NH₄ in the total N concentration of faeces and sludge, provides information about the nitrogen composition and nutrient availability for agriculture uptake. Hence, nitrogen composition was determined by examining the percentage of (N-NH₄) in total (N) for school sanitation systems in Figure 40. VIPs illustrated the highest percentage of N-NH₄⁺ in total N, with 76,5% this was followed by MTs and STs at 35,2% and 31,9% respectively.

Statistical analysis showed that nutrient concentration is significantly affected by the sanitation facility (P<0, 001 for N=33).

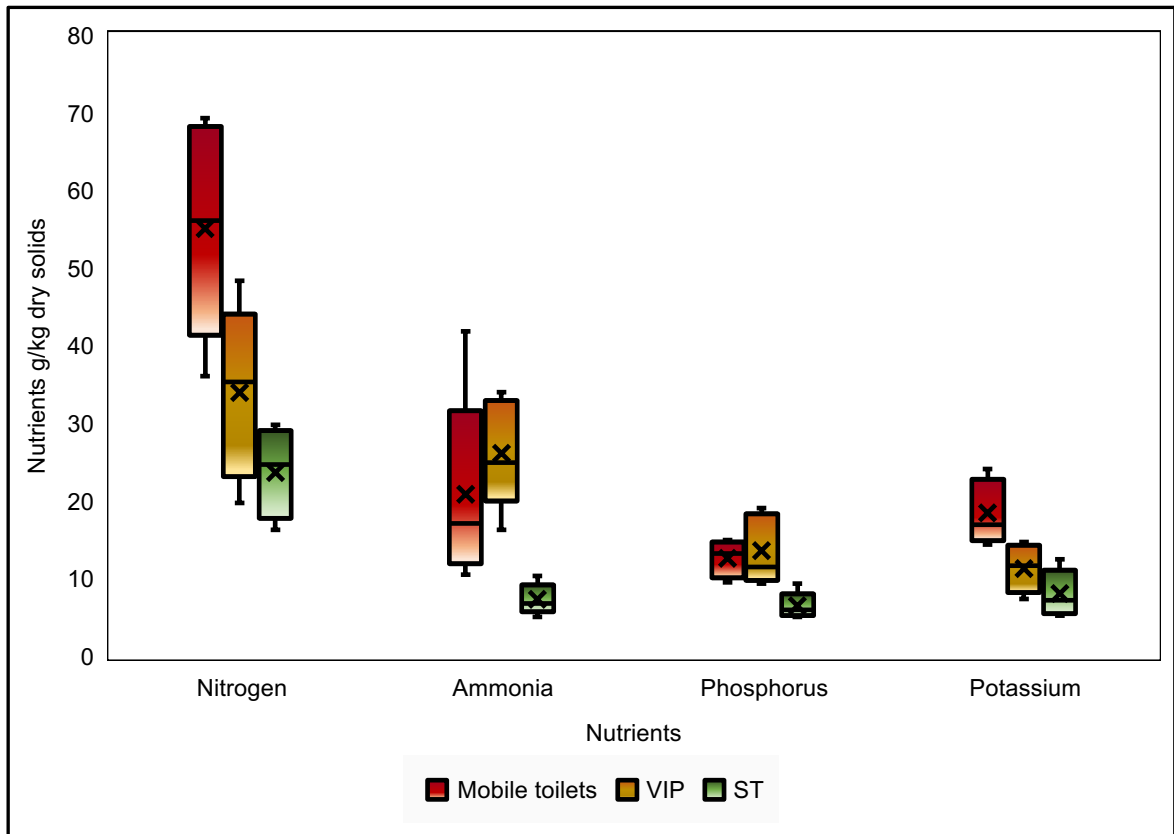


Figure 38: Box and Whisker Plot showing nutrient concentration as dry basis

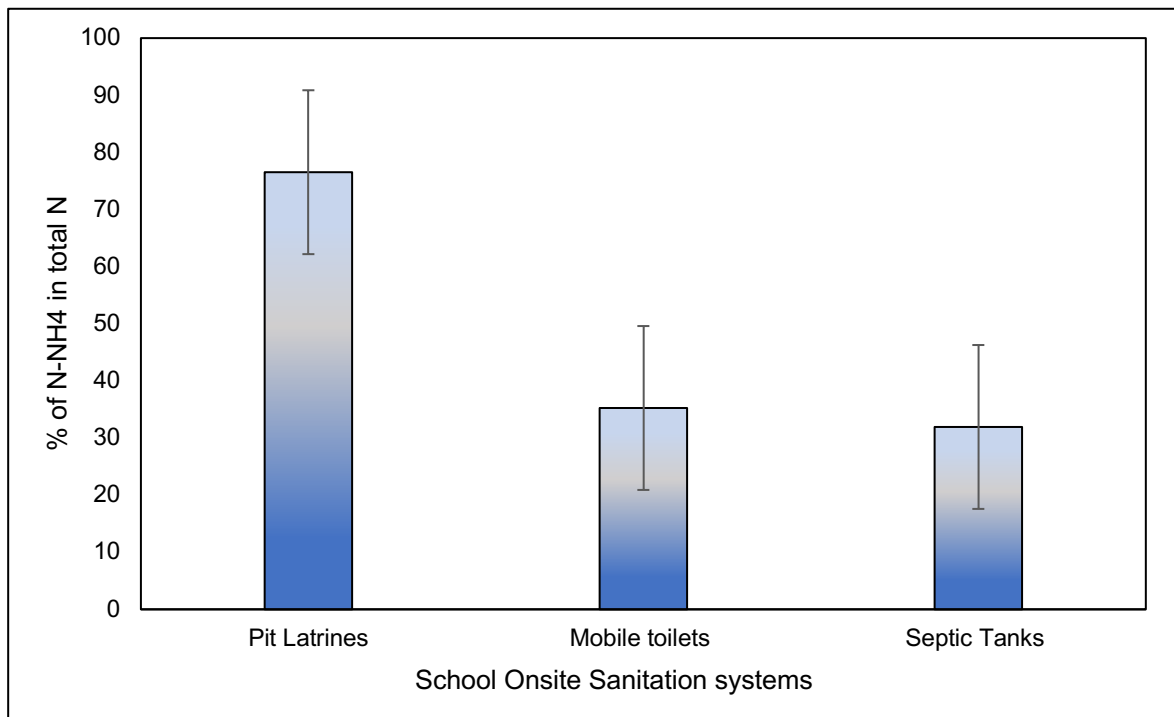


Figure 39: Bar graph showing percentage of NH4 in total N

4.1.5 C: N

The carbon-to-nitrogen ratio was determined by dividing the total carbon content to the total nitrogen content. Table 20 (Appendix B) shows STs illustrated the highest C:N ratio of 9,7 followed by VIPs at 8,3 and MTs at 4,9, findings were comparable to those by Hadiyanto et al. (2020) and Miller et al. (2015). Nevertheless, sludge from all systems were still lower than the optimal composting ranges. Ideal C:N ratio for composting ranges between 20-30, this is based on the ratio that microbes consume carbon and nitrogen throughout their production. The low C:N ratio observed in MTs can be explained by Strande et al. (2014) who outlined that a C:N less than 20 implies there is a surplus of nitrogen within the system, which will eventually be lost through mineralisation, caused by nitrate leaching or ammonia volatilisation. Statistical analysis showed that Carbon and Nitrogen is significantly affected by the sanitation facility type ($P= 0,017$ and $P= 0,018$ for $N=33$ respectively).

4.2 Thermal and Thermodynamic properties of faecal material from school sanitation systems

4.2.1 Thermal conductivity

Thermal properties such as thermal conductivity and heat capacity of FS, are likely to impact drying behaviour and various treatment pathways based on thermal process such as pyrolysis (Deering et al., 2018). Figure 41 shows a box and whisker plot of the thermal conductivity, from faecal material in varying school toilets. The data visually informs the minimum, maximum, median, 75th and 25th quartile values.

Thermal conductivity for each OSS (MTs, VIPs, STs) represented a narrow ranged between 0,48-0,59 W/K/m, with STs illustrating the highest value. The thermal conductivities from this study were closely resembled the value pure water (0,6 WK/m), indicative of the high-water content within faecal material. Therefore, the highest thermal conductivity from STs is likely due to the higher moisture content (~ 90%) (sorption isotherm). These findings are supported by Zuma et al. (2015), Pandarum et al. (2019); Hanson et al. (2000) and Bart-Plange et al. (2009).

Statistical analysis showed that thermal conductivity is significantly affected by the sanitation facility type ($P<0, 001$ for $N=33$).

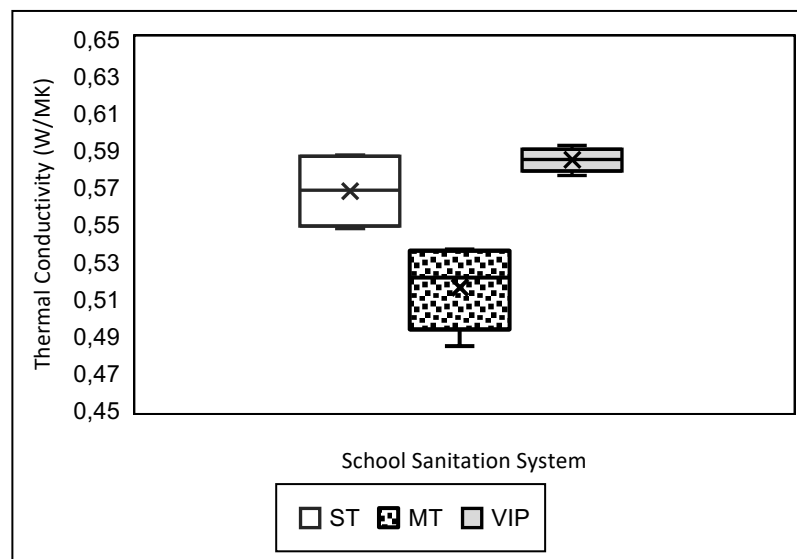


Figure 40: Box and whisker plots showing thermal conductivity from different school OSS.

4.2.2 Calorific Value

A possible end-use application of faecal material includes biofuel (for heating and a source of energy) which is determined by the calorific content of faecal material. Figure 42 presents a box and whisker plot of the calorific value from varying OSS, the data informs on the minimum, maximum, median, 75th and 25th quartile values.

Calorific value on a dry basis was highest from MTs containing fresher faeces at 23,2 (MJ/kg dry solid). These findings were similar to the results from Onabanjo et al. (2016) who reported a calorific value of dried faeces to be around ~25 MJ/kg dry solid. The calorific value of VIPs displayed minimal variation and a mean of 18,4 MJ/kg dry solid, comparable to Muspratt et al. (2014) who found averaging calorific value of 16,2 MJ/kg dry solid, but slightly higher than Zuma et al. (2015) who reported 14,3 MJ/kg dry solid. Similarly to VIPs, STs illustrated a calorific value of 18,9 MJ/kg dry solid. Calorific value of FS and faeces from school sanitation systems, were similar to wood and various coal ranks including lignite, bituminous, peat and coal (14-25 MJ/kg), suggesting that FS presents the potential to be used as a fuel source.

Figure 43 shows that the COD (organic content) of faecal material directly affects the calorific value ($R^2=0.7$). This can be explained as the organic material in fresh faeces is less degraded compared to digested FS, which leads to fresher faeces exhibiting higher proportions of combustible substances i.e., carbohydrates and fats, which can be readily converted into energy upon combustion. In contrast, the lower values observed in VIPs and STs are likely due to the prolonged storage period of FS, indicating a more digested FS with a lower organic concentration i.e., less combustible material present. Calorific Value was further correlated with VS (see Appendix G), which suggested a moderate positive correlation ($R=0,5$) between VS and Calorific Value, indicating that as Calorific Value increases there is an expected increase in VS%. However, the coefficient of determination (R^2) suggests only 29% of

the variability can be explained by the linear relationship with VS, which could be due data heterogeneity and outliers.

Statistical analysis showed that calorific value is significantly affected by the sanitation facility type ($P < 0,001$ for $N=33$).

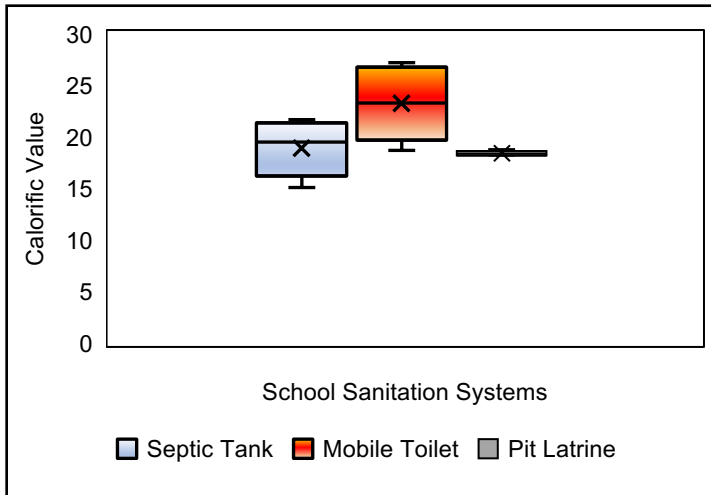


Figure 42: calorific value of faecal material from school OSS.

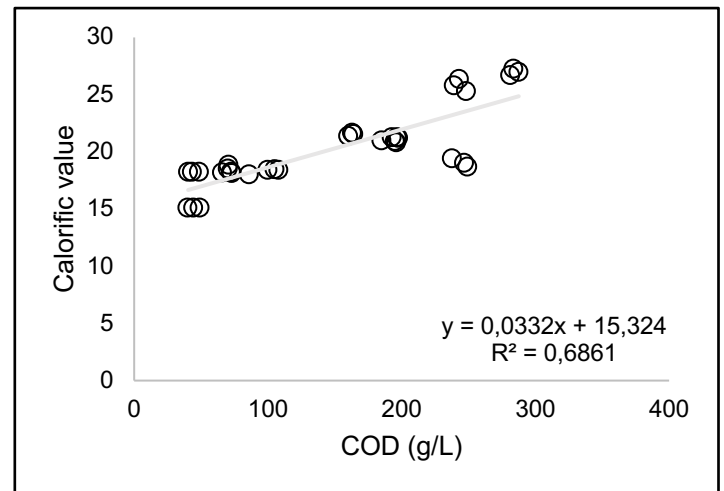


Figure 41: Correlation of calorific value and COD of faecal material from school OSS.

4.2.3 Water Activity

Both fresh faeces (from MT) and faecal sludge (from VIP and ST) consist predominantly of water, which is either bound or in an unbound form. Therefore, examining water activity in faecal material is important for the implementation of onsite treatment technologies, which is decided by the proportion of unbound and bound water within FS or faeces.

Faecal sludge and faeces from all school sanitation systems (MTs, VIPs and STs) exhibited a water activity of $\sim 1 a_w$. These findings were comparable to studies from Sagar & Kumar (2010), and Strande & Brdjanovic (2014). The high-water activity ($\sim 1 a_w$) of faecal material across school sanitation systems signifies unbound moisture in FS that is less difficult to remove i.e., good dewaterability potential. Statistical analysis showed that onsite sanitation systems had no significant impact on the water activity.

To improve faecal sludge transport and reduce costs, faecal material could include an initial dewatering phase, where the liquid portion (supernatant) is directed to a sewer, and the dewatered sludge is transported to treatment facility (Strande, 2017). Findings from faecal material in all school sanitation systems exhibited a high-water content (water activity $\sim 1 a_w$), implying dewatering would be necessary for certain treatments, like pyrolysis (Beik et al., 2023). Possible dewatering methods include UPDBs, PDB, gravity thickening, filtration, centrifugation, and belt filter presses (Semiyyaga et al., 2017).

4.3 Settleability results of faecal material from school sanitation systems

4.3.1 Sludge Volume index

Sludge volume index provides an indication on the settleability of faecal material over time and assists in determining the potential removal frequency of faecal material from settling tanks, ponds, and grit chambers. Figure 44 presents the SVI of faecal material from school toilets. Samhan et al. (1990), Abdel-Magid et al. (1997) and Heinss et al. (1999) outlined that an SVI between 40-80 ml/g presents an excellent sludge settling. Whereas faecal material with an SVI between 100-200 ml/g typically indicates fair settleability, and sludge or faeces with an SVI of > 250 ml/g exhibits poor settleability and compacts slowly.

Faecal sludge from STs presented the highest SVI averaging at 16,4 ml/g, within the ranges reported from GITONGA (2021) and Bassan et al. (2013), followed by MTs at 10 ml/g and finally VIP sludge with an SVI of 9,9 ml/g. The low SVI found in VIPs and STs were similar to results reported by Ward et al. (2019), who showed improved settleability for liquid FS compared to wastewater sludge. Similarly, Mercer et al., (2021) found that fresher faeces settled significantly better than wastewater sludge.

SVI results implied that sludge from school sanitation systems exhibits excellent settleability. (As seen in figure 27). Chandana & Rao (2021) suggest that the SVI of sludge decreases with age, also explaining the lower SVI from VIPs (containing the most aged and over oxidised FS).

Statistical analysis showed that SVI is not significantly affected by the sanitation facility type i.e., distribution was similar across sanitation types ($P=0,055$ for $N=33$).

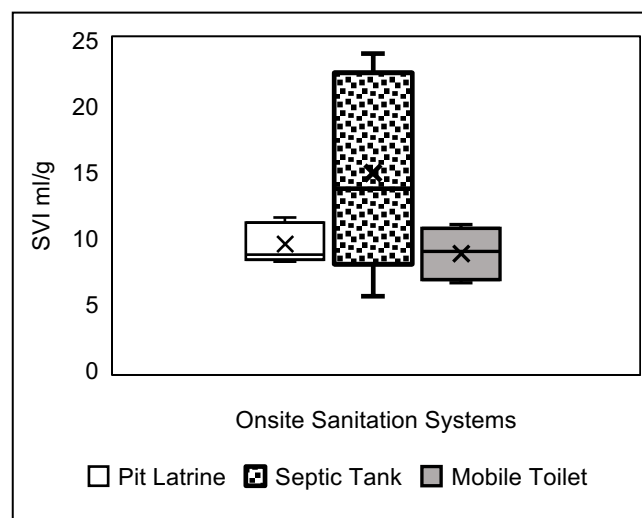


Figure 43: SVI of faecal material from school OSS.

4.4 Mechanical results of faecal material from school sanitation systems

4.4.1 Particle size analysis

Figures 45-47 graphically shows the particle size distribution of faecal material against cumulative volume density (%), obtained from STs, MTs, and VIPs. Furthermore, Table 13 further shows the average D_v 10, 50 and 90 (μm) for each school sanitation system. The median particle diameter D_v (50 μm) was 82,9 μm , 38,9 μm and 118,4 μm for STs, MTs, and VIPs respectively. D_v (50 μm) values from school OSS FS corresponded to studies by (Ward et al., 2021, Sam et al., 2022, Houghton and Stephenson, 2002, Yu et al., 2016, Ward et al., 2023). The particle size for school sanitation systems ranged between 0,7- 2046,7 μm 0,6 -1202,3 μm and 0,7-1492,5 μm for VIPs, MTs, and STs, respectively.

Particle size is crucial, particularly for pre-treatment and subsequent treatment methods such as dewatering. Faecal sludge or faeces containing a significant proportion of small particles ($< 10 \mu\text{m}$) and a smaller median aggregate size (D_v 50 μm) show lower dewatering performance. Conversely, sludge or faeces with higher fractions of larger particles ($>100 \mu\text{m}$) exhibit better dewaterability performance (Ward et al., 2023). Ward et al. (2023) further shows that stabilised sludge exhibits better dewatering performance and contains lower concentration of small particles. This finding corresponded to Rasmussen et al. (1994) who observed that particle fractions from 0,45-10 μm was accompanied with poor filtration. Hence it is reasonable to suppose that faecal sludge samples with an excessive proportion particles consolidated within larger aggregate sizes exhibits fewer suspended smaller particles that can contribute to clogging for emptying of OSS (Ward et al., 2023).

Table 13: Particle size distribution of faecal material from school OSS

Sanitation System	DV (10 μm)	DV(50 μm)	DV(90 μm)
Septic Tank	9.2	82.9	632.8
Mobile Toilet	2.9	38.9	486.8
Pit Latrine	10.2	118.4	791.7

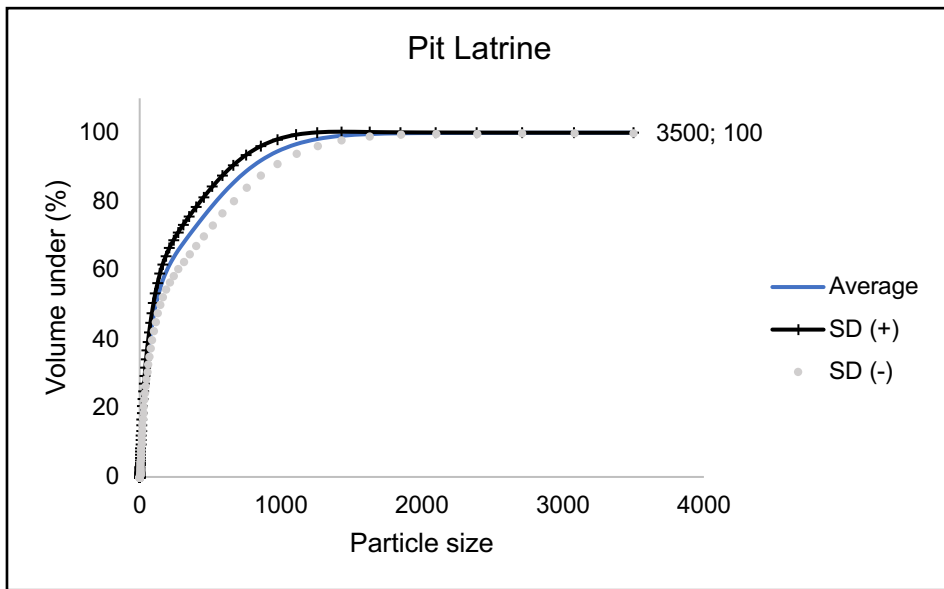


Figure 46: Particle size analysis of faeces from VIPs

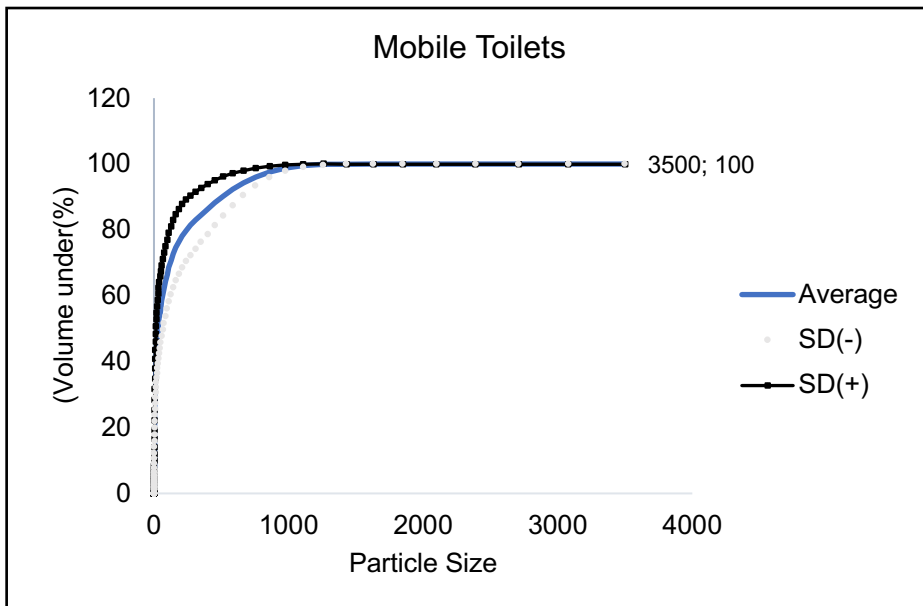


Figure 45: Particle size analysis of faeces from MTs

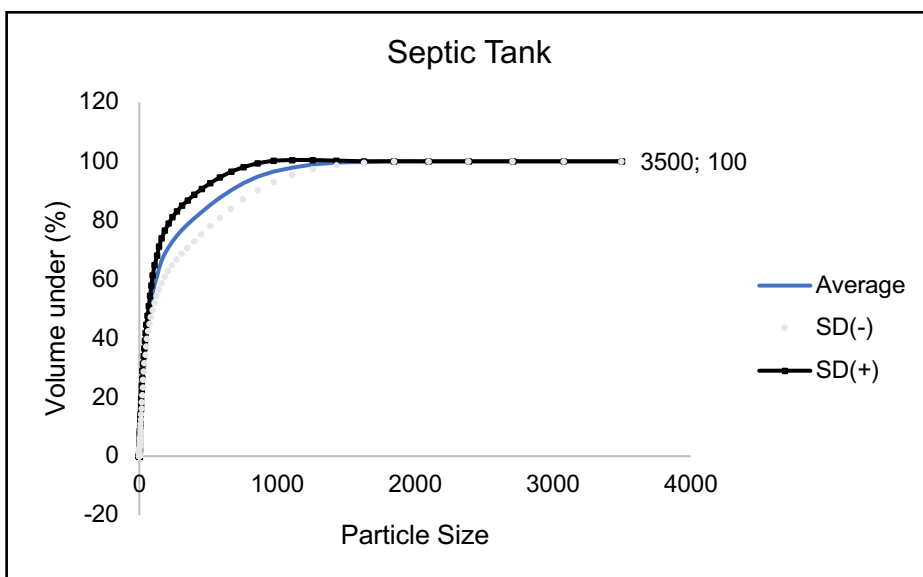


Figure 44: Particle size analysis for FS from STs

4.4.2 Density

Densities for faecal sludge and faeces from school OSS are illustrated as box and whisker plots in figure 48, showing the minimum, maximum, median, 75th and 25th quartile values. MTs (containing fresh faeces), STs and VIPs illustrated similar densities averaging at 1,047 g/cm³, 1,039 g/cm³ and 1,026 g/cm³ suggesting that there is no statistical significance between sanitation systems. All density values were within the range supported by R Penn et al. (2018) and Brown et al. (1996). Faecal sludge density values from school sanitation systems were slightly higher than the density of water (~0,998 g/cm³ for T=20°C) reported by Velkushanova et al. (2021). The higher density in FS can be a result of organic content, suspended solids, or salts dissolved in the liquid fraction, resulting in a more condensed faecal material owing to electrostatic interactions. Density of FS and faeces can vary depending on moisture content, health of humans and presence of non-organic materials.

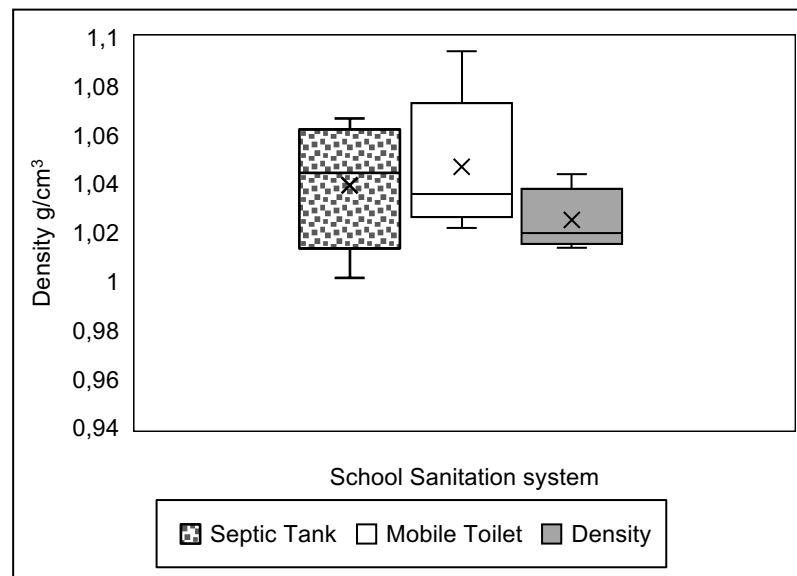


Figure 47: Box and Whisker plots showing density of faecal material from school OSS.

4.4.3 Shear yield stress

Yield stress is outlined as the minimum shear stress applied to a solid to initiate flow. Together with density and particle size, rheological properties provide insights on the pumpability of the materials from the sanitation systems. Sludge material can flow when it undergoes shear thinning, where the increase in shear rate is expected to improve the pumpability (Septien et al., 2018). Figure 49 compares the yield stress change as function of TS between aged FS and fresher faeces obtained from VIPs, STs and MTs in school sanitation systems to the sludge and faeces data reported by Mercer et al. (2021) and Septien et al. (2018).

Faecal sludge from STs illustrated the highest mean shear stress at 349,9 Pa, followed by VIP sludge and fresher faeces from MTs. Results for faecal sludge and fresh faeces in this study were within range (500 to 1000 Pa) of those reported by Septiens et al. (2018). Findings implied that fresher material depicts a higher flowability at the same solid concentration, thereby requiring a lower volume of water and reducing pumping head for emptying technologies. This was supported by Mercer et al. (2021) who showed faecal rheological behaviour can be predicted by TS concentration, and proved that fresher faeces required lower shear stress for flowing compared to aged sludge, at the same TS. Fresher faeces further demonstrates thixotropic behaviour with most of its structural breakdown at a low shear rate (Woolley et al., 2014, Mercer et al., 2021). Septien et al. (2018) further showed shear stress diminished from 1000-10 Pa by increasing moisture content from 77 to 90%, which subsequently improves pit emptying and reduces the viscosity of sludge, allowing for improved pumpability and pit emptying (Septien et al., 2018).

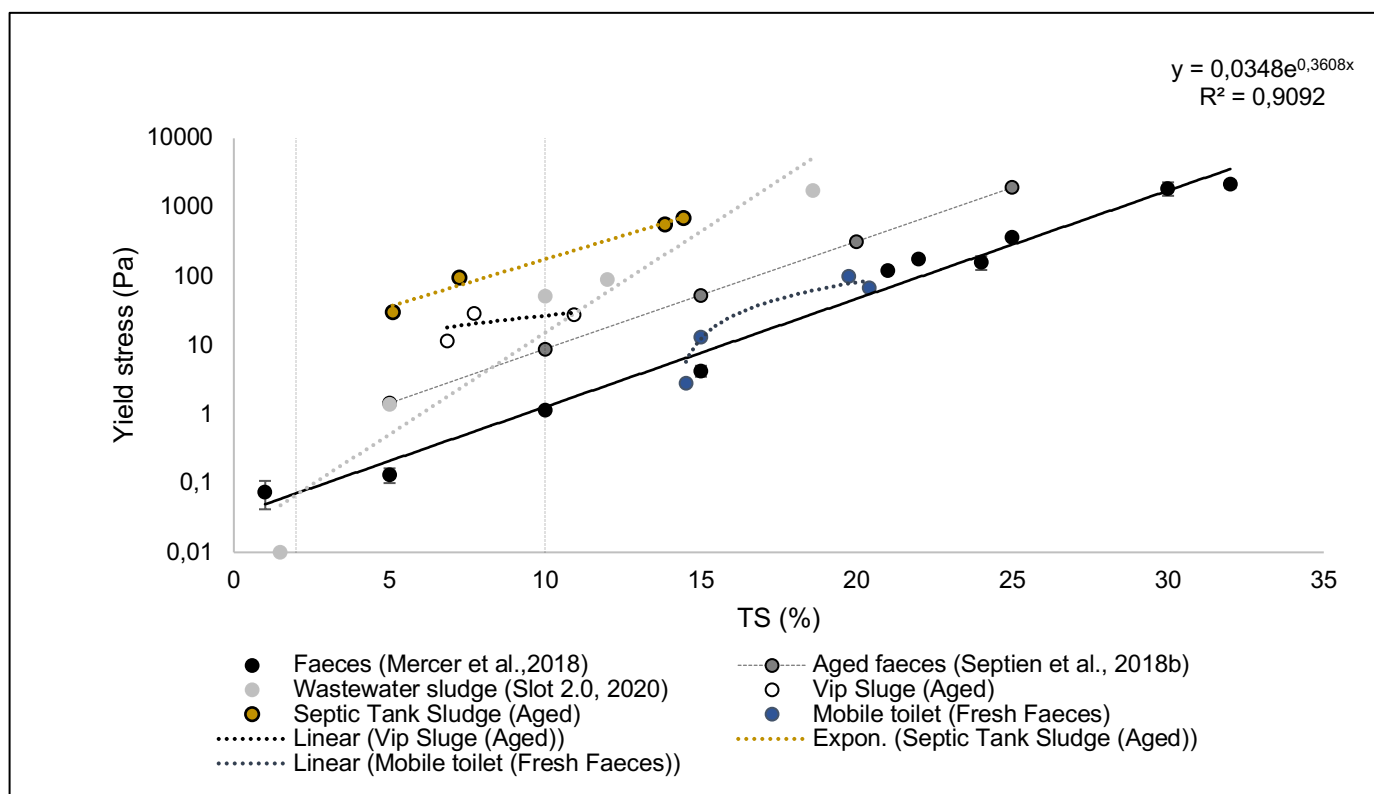


Figure 48: Comparison of fresh faeces, aged faecal sludge, wastewater sludge and faecal sludge from school sanitation systems shear yield stress as a function of total solids. Data obtained from Septien et al., 2018; Mercer et al., 2021; Slot 2020.

4.5 Decision Matrices showing possible pit emptying and treatment pathways of faecal material for rural schools.

4.5.1 Influence of faecal material characteristics on emptying technologies for rural areas

Figure 50 provides a key criterion to determine possible options for faecal material emptying from rural school OSS. The decision can be made based on the total solids concentration, type of OSS, presence of non-biodegradable material such as trash, and accessibility of the sanitation system. Rural school sanitation systems all exhibited relatively easy access for sample collection apart from certain schools having concrete sealed VIPs. Part A of the tree is for the user to predict the likely characteristics and the presence of trash within the system. Part B looks at the type of sanitation system and relationship to its TS. For instance, STs exhibit more watered-down and less viscous FS due to an influx of “flush water”, compared to MTs that operate by a waterless system. Lastly Part C examines the accessibility of the OSS. If the system is difficult to access, certain emptying methods should be avoided.

The decision tree is based on the following assumptions.

- It is more difficult to deal with and empty denser faecal sludge (high TS%) than watery sludge (Thye et al., 2011).
- Extremely dense and viscous faecal sludge can only be emptied through manual emptying (Thye et al., 2011).
- Direct emptying of faecal sludge is limited by the pit depth, i.e., the gulper can only empty FS less than 1m.

Emptying of OSS is dependent on the TS concentration and presence of trash in the containment (Strande et al., 2014). Velkushanova et al. (2021) explained that liquid faecal sludge exhibits a total solids concentration of < 5% and usually obtained from wet containments such as STs and wet/leached VIPs. These systems can be emptied through mechanical techniques, such as vacuum pumping. Slurry like faecal sludge is slightly thicker than liquid but still has liquid/muddy consistency and exhibits a TS concentration between 5-15% (Velkushanova et al., 2021). These types of samples are often collected from improved/unimproved pit latrines or at the bottom of STs. Semi-solid faeces are studied to have a soft paste like texture, and often obtained from MTs. This type of the faecal material exhibits difficulty when pumping, hence, should be collected by spade or other manual techniques (Velkushanova et al., 2021).

Hence, high TS range (14,4- 20,6%) observed from MTs, requires the sludge to be emptied using manual methods, ensuring no blockage or breakage of the emptying technologies (Ahmed et al., 2018). Whereas the slurry range for some STs and VIPs in school sanitation systems allows for a wider array

of emptying technologies as shown in the decision tree. Table 21 and 22 (in Appendix C) provides a summary on manual and mechanically operated emptying technologies, their pumping head and flow rates used in Figure 50 (the decision tree).

Buckley et al. (2008) and (Gaulke et al., 2008) show that, within a VIP containment system, recently deposited FS (typically in the top layers of pits) exhibits higher water and organic content compared to sludge at deeper layers. Material that is older, deeper, and more digested i.e., at the bottom of containment systems, exhibits a thicker texture. The thicker texture sludge requires an addition of water to facilitate pumping, otherwise it is advisable for the sludge to be emptied manually (Linda et al., 2008).

Hence typical VIP sanitation systems (TS 20-30 %) and OSS which exhibit FS older than two years likely contain denser and more viscous sludge owing to biodegradation, stabilisation, and dehydration processes. Figure 35 supported this assumption, by showing COD findings were strongly correlated ($R= 0,90$) to TS%. Findings were supported by Chanadana and Rao (2021) who further elaborated on the importance of COD for distinguishing emptying techniques. Subsequently, this makes it difficult to use mechanical vacuuming for collecting sludge at bottom layers, promoting manual emptying. Nonetheless, desludging between 2-2,5 years can evade manual emptying by limiting COD and TS build-up. However, as there was a higher water infiltration in school VIPs (TS 6-12,5 %) faecal material was less viscous, subsequently increasing the options for emptying technologies.

Overall faecal sludge emptying remains a major difficulty especially in rural localities where there are many full/overflowing sanitation systems. An all-round solution to FS emptying in such places is still to be developed with the main challenges comprising the following:

- Traditional mechanical emptying technologies such as vacuum pumps cannot be used for emptying dense and viscous sludges.
- The characteristics of faecal material vary significantly depending on its containment; therefore, requires a wide range or combination of emptying technologies to insure efficient emptying for varied sludge types.
- The method of emptying developed needs to address subsequent faecal sludge components of FSM such as transport (due to the high costs associated with the long distances between toilets in rural settings), treatment (technology should economically be feasible in a rural context), and disposal where necessary.

Based on the FS sample characteristics collected from rural school sanitation systems, Table 14 shows proposed emptying methods to ensure safe emptying of school toilet facilities.

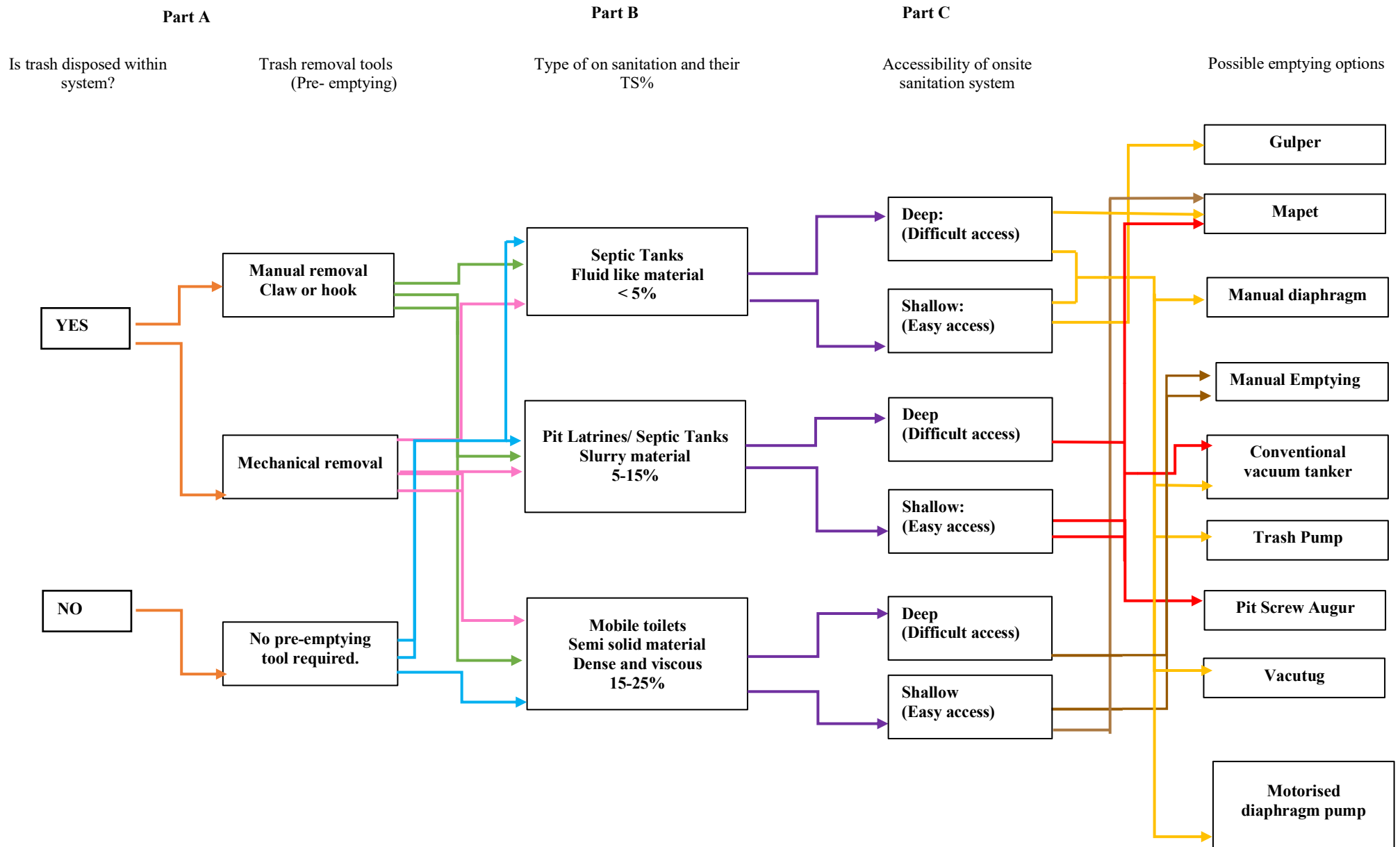


Figure 49: Decision tree showing possible emptying pathways for faecal material.

Table 14: Proposed emptying methods for rural school sanitation systems.

Sanitation Type	Proposed Emptying Methods	Explanation
Rural school Pit Latrines	Gulper Vacuum Tanker	TS ranged between 6-12%, Gulper works well with less viscous sludge, and allows for varying depths. Toilets had a fairly large superstructure, and low trash in some VIPs.
Pit latrines (TS > 20%)	Manual Emptying	
Septic Tanks	Nibbler Conventional Vacuum tanker Gulper MAPET Motorised diaphragm Vacutug Pit Screw Augur	TS ranged between 5-15%. Certain schools experienced prolonged periods of water shortages, which resulted in STs not operating under usual conditions. Nibbler works well with more viscous sludge, while vacuum tanker is more optimal for low viscous sludges (for those schools who had normally operated STs). Systems exhibited minimal trash.
Mobile Toilets	Manual Diaphragm MAPET emptying. Manual emptying	TS ranged between 14-20%, MTs exhibited presence of viscous sludge, with high contents of non-biodegradable materials. These systems therefore require a pre-emptying (trash removal stage).

Source:(Strande and Brdjanovic, 2014)

4.5.2 Influence of faecal material characteristics on treatment technologies for rural areas

Kone and Strauss (2004) outlined that the selection of treatment technology is dependent on the sludge characteristics and its reuse or disposal option, such as land application, biogas production or landfilling. The challenges faced by rural communities are different to those experienced in urban areas (Klingel et al., 2002). The population density in urban areas generates a greater volume of faecal material (Singh et al., 2017a). The greater volume of generated sludge requires more complex treatment techniques such as centralised faecal sludge treatment plants (Singh et al., 2017a). These types of treatment plants require large capital investments and more resources such as electricity and water, in turn making it easier to manage these complex systems (Ingallinella et al., 2002).

In contrast, rural areas have a significantly lower population density, water scarcity challenges, minimal resources and capital investment (Singh et al., 2017a). This makes it difficult to implement large complex treatment systems. In these areas, decentralised and low-cost treatment technologies are therefore a better option for a more sustainable output, owing to their lower cost and lower energy requirements (Singh et al., 2017a). Additionally treatment options used in rural areas should be easily scaled to fit the needs of rural communities, and should be successfully operated by the locals with

minimal training (Harada et al., 2016). The reasons for selected treatment technologies are provided in table 15.

Table 16 provides a treatment implementation matrix to determine the criteria selection and appropriateness of faecal sludge treatment technologies for rural areas. The scoring is ranked from 1-3, where (1) indicates high suitability, (2) indicates medium suitability and (3) implies low suitability. The matrix criteria are provided in Appendix D. Similarly, Table 17-19 provides a decision-making tool to determine treatment pathways for faecal sludge from rural school toilets based on the analysed characteristics. The scoring for sludge treatment methods is ranked from 0-2, where (0) implies the sludge is unsuitable for the treatment; (1) indicating the sludge is suitable following pre-treatment and (2) the sludge is suitable in its original form. The criteria for treatment matrices are provided in Appendix E.

Based on the studied characteristics, aspects to consider when designing FS treatment technologies involve an initial sludge pre-treatment, whereby there is a preliminary removal of moisture, or sludge contaminants, such as sand, trash, or other solids. Pre-treatment is used to avoid blockages, formation of increased solid loading, and extend the lifespan of faecal sludge treatment technologies (Tilley et al., 2014). The type of pre-treatment is specific to the sludge characteristics and desired outcomes. Two main pre-treatment methods often used includes dewatering and drying (drying can also serve as a core treatment). However, pre-treatment is not limited to this. Figure 51 provides an overview of pre-treatment technologies. Details on operating parameters can be found in (Tilley et al., 2014).

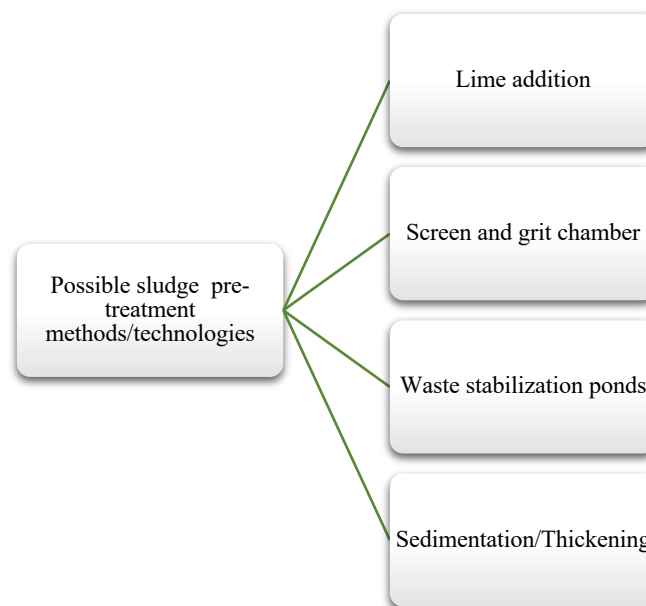


Figure 50: Overview of faecal sludge pre-treatment technologies.

Table 15: Suitability of selected Faecal sludge technologies for rural areas.

Selected technology	Description	Suitability for rural areas
Unplanted Drying beds	An Unplanted drying bed is a basin with drainage at the bottom, it is used for dewatering and drying of FS at the surface. When the bed is loaded with FS, it collects the percolated leachate and allows the sludge to dry by percolation and evaporation. The process reduces the water content of the sludge and kills pathogens.	<ul style="list-style-type: none"> • Simple and low-cost technology • Does not require any electricity and mechanical equipment • Beds can be constructed using locally available materials • Easy to operate and maintain • Low skills required • Large land space available
Planted drying beds	<p>Planted drying beds are similar to unplanted drying beds, however, includes an advantage of evapotranspiration from vegetation that has been planted in the sludge. Typical vegetation types include Reeds, Cattails, Antelope grass (<i>Phragmites sp</i>; <i>Typha sp.</i> and <i>Echinochloa sp.</i> respectively). However other, non-invasive species that reproduce quickly, grow well in warm environments and are typically resistant to salt water can also be used.</p> <p>The key advancement of PDB over UPDBs is that the faecal sludge only needs removal after every 3-5 years and not after every drying cycle, allowing fresh sludge to be compiled onto previous layer.</p>	<ul style="list-style-type: none"> • Simple and low-cost technology • Does not require any electricity and mechanical equipment • Easy to operate and maintain • Low skills required • Beds can be constructed using locally available materials • Crop yield from bed can generate income for locals • Large land space available
Composting	Composting treatment is a method for treating FS that involve mixing the sludge with other organic materials, such as sawdust, vegetable waste or straw, to create a compost mixture. Essentially it refers to a process whereby biodegradable components are biologically decomposed by bacteria or fungi under aerobic conditions. WHO (2006) and Niwagaba et al., (2009) further explained that composting is commonly used for the treating of source-separated faeces.	<ul style="list-style-type: none"> • High pathogenic reduction • Beneficial soil conditioner • Low water required. • Large land space available • Generates income with compost that can be used as a fertiliser
Deep row entrenchment	Deep row entrenchment are deep trenches filled with faecal sludge and covered with soil. Trenches are lined with impermeable material that prevents the sludge from leaching into the soil. In addition, trees are grown on the sludge allowing for further stabilisation. The use of sludge buried in deep rows, enable the organic nutrients within the sludge to be used beneficially through its natural decomposition overtime.	<ul style="list-style-type: none"> • Low-cost technology • Large land space available • Does not require electricity or mechanical equipment • Can be constructed using locally available materials • Low operation and maintenance skills • The trees used in DRE Increases potential of wood production for rural communities.

Vermicomposting	Similar to composting, vermicomposting is a method of treating faecal sludge with the addition of plant nutrients, soil enzymes, microorganisms or earthworms. The worms consume the organic matter and excrete worm castings, which is classified as a nutrient rich fertiliser.	<ul style="list-style-type: none"> • Does not require any electrical or mechanical equipment Can be performed on a small scale and confined area i.e., allowing it to be suitable for small communities or households. • Low maintenance skills
In-Situ treatment (By Tiger worms)	Involves the introduction of specific earthworm species, commonly known as tiger worms (<i>Eisenia fetida</i> or <i>Eisenia andrei</i>), to facilitate vermicomposting within the latrine. These worms are added directly to the pit, whereby they assist in breaking down organic waste. As the worms consume and digest organic matter, they enhance the decomposition process and accelerate the transformation of human waste into nutrient-rich vermicompost.	<ul style="list-style-type: none"> • Low cost cost-effective solution, requiring minimal investment in infrastructure and technology. • Low technical expertise • Reduced water reliance • Adaptability to local contexts • Environmentally friendly- avoids the use of chemicals
Solar drying in greenhouse (solar thermal system).	Drying is the process whereby moisture is removed from sludge and pathogens within the faecal material are killed. Within the greenhouse (solar thermal technology) faecal material is spread as a thick layer on the surface of the ground. Thereafter, the sludge is homogenised to avoid crusty formations. The sludge then absorbs the solar radiation passing through the clear walls of the greenhouse and the captivated heat leads to the moisture evaporation of the sludge.	<ul style="list-style-type: none"> • Uses free energy source (solar energy), compared to electric processes. • Drying occurs at a faster rate compared to drying beds • Suitable for areas where water is scarce. • Dried sludge can be used as a source of fuel or fertiliser.
Anaerobic Digestion	The process of degradation and stabilisation of organic compounds through microorganisms that break down organic matter in the absence of oxygen. This type of treatment produces biogas (methane and carbon dioxide), and stabilised residue suitable for use as fertiliser or soil conditioner.	<ul style="list-style-type: none"> • The process generates clean energy which can be used for cooking or lighting. • Stabilised residue produced by the process can be used to improve soil fertility, which is particularly important in rural areas. • Easily scaled to fit the needs of communities
Pyrolysis	Pyrolysis is a thermal treatment process whereby faecal sludge is heated between 300-700°C, allowing the organic matter in the sludge to break down into various products, such as biochar.	<ul style="list-style-type: none"> • Can be scaled to fit the needs of small communities • Produced biochar and biogas for fertiliser and source of energy • Processes requires minimal water • Less space requirements (good for smaller rural communities)

Sources: Koné and Strauss, 2004; Mbouendeu et al., 2022; Eawag (2008); WHO (2006); Niwagaba et al., (2009); Strande et al., (2014); Tchobanoglous et al. (2004).

Table 16: Decision Matrix 1

			Technology Implementation decision matrix							
Criterion	Legend		Unplanted drying bed	Planted drying bed	Anaerobic Digestion	Pyrolysis for Biochar Recovery	Vermicomposting	Composting	Solar Drying	Deep row entrenchment
Skill requirement / Ease of operation	1 2 3	High Medium Low	3	3	2	1	2	2	2	3
CAPEX	1 2 3	High Cost Medium Cost Low Cost	3	3	2	1	2	2	2	3
OPEX	1 2 3	High Cost Medium Cost Low Cost	3	3	3	1	2	2	2	3
Land requirement	1 2 3	Low Medium High	3	3	1	1	3	3	3	3
Energy requirement	3 2 1	Low Medium High	3	3	2	1	3	3	2	3
Reusability of treated waste	1 2 3	Low Medium High	2	2	2	3	2	3	3	1
Robustness of the process	1 2 3	Low Medium High	3	3	3	2	3	3	2	3
Level of implementation	1 2 3	Tested in lab. Pilot tested. Tested in SA	3	2	3	2	3	3	2	3
Total			23	23	18	12	20	21	18	22

Source: Singh et al. (2017b); Tilley et al. (2014).

Table 17: Decision Matrix 2

Parameter	Legend	Selection criteria	Treatment: Mobile Toilets							
			Unplanted drying bed	Planted drying bed.	Anaerobic Digestion	Pyrolysis for Biochar Recovery	Vermicomposting	Composting	Solar Drying	Deep row entrenchment
TS (%total mass)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	1	2	2	1	1	1	1	1
VS (%)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	-	1	2	2	2	1	-	2
TSS concentration (mg/L)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	1	1	2	1	1	1	-	1
COD (mg/L)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	-	1	1	-	1	1	-	1
Nutrient Concentrations Nitrogen (mg/L)?	0 1 2	Not Suitable Suitable after pre-treatment Suitable	1	1	1	-	1	1	-	1
Nutrient concentration: Ammonia (mg/L)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	-	-	2	-	1	1	-	1
Nutrient Concentration: Potassium (mg/L)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	1	1	-	-	1	1	-	1
Nutrient Concentration: Phosphorus (mg/L)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	-	1	-	-	1	1	-	1
C: N	0 1 2	Not Suitable Suitable after pre-treatment Suitable	-	1	1	-	1	1	-	1
pH	0 1 2	Not Suitable Suitable after pre-treatment Suitable	2	2	2	2	2	2	2	2
Total			6/10 60%	11/18 61,1%	13/16 81,3%	6/8 75%	12/20 60%	11/20 55%	3/4 75%	12/20 60%

Table 18: Decision Matrix 3

Constraint	Legend	Selection criteria	Treatment: Septic tanks							
			Unplanted drying bed	Planted drying bed.	Anaerobic Digestion	Pyrolysis for Biochar Recovery	Vermicomposting	Composting	Solar Drying	Deep row entrenchment
TS (%total mass)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	1	2	2	1	1	1	1	1
VS (%)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	-	1	2	2	2	1	-	2
TSS concentration (mg/L)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	1	2	2	1	1	1	-	1
COD (mg/L)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	-	1	1	-	1	1	-	1
Nutrient Concentrations Nitrogen (mg/L)?	0 1 2	Not Suitable Suitable after pre-treatment Suitable	1	1	2	-	2	1	-	1
Nutrient concentration: Ammonia (mg/L)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	-	1	2	-	2	2	-	2
Nutrient Concentration: Potassium (mg/L)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	1	-	-	-	2	2	-	1
Nutrient Concentration: Phosphorus (mg/L)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	-	1	-	-	2	1	-	1
C: N	0 1 2	Not Suitable Suitable after pre-treatment Suitable	-	1	1	-	1	1	-	1
pH	0 1 2	Not Suitable Suitable after pre-treatment Suitable	2	2	2	2	2	2	2	2
Total			6/10 60%	11/18 61,1%	14/16 87,5	6/8 75%	16/20 80%	13/20 65%	3/4 75%	13/20 65%

Table 19: Decision Matrix 4

			Treatment: Pit Latrines							
Constraint	Legend	Selection criteria	Unplanted drying bed	Planted drying bed.	Anaerobic Digestion	Pyrolysis for Biochar Recovery	Vermicomposting	Composting	Solar Drying	Deep row entrenchment
TS (%total mass)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	1	2	2	1	1	1	1	1
VS (%)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	-	2	1	1	2	2	-	2
TSS concentration (mg/L)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	1	1	1	1	2	2	-	1
COD (mg/L)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	-	1	1	-	1	1	-	1
Nutrient Concentrations Nitrogen (mg/L)?	0 1 2	Not Suitable Suitable after pre-treatment Suitable	1	1	2	-	2	1	-	1
Nutrient concentration: Ammonia (mg/L)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	-	-	2	-	2	2	-	1
Nutrient Concentration: Potassium (mg/L)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	1	1	-	-	2	1	-	1
Nutrient Concentration: Phosphorus (mg/L)	0 1 2	Not Suitable Suitable after pre-treatment Suitable	-	1	-	-	2	1	-	1
C: N	0 1 2	Not Suitable Suitable after pre-treatment Suitable	-	1	1	-	1	1	-	1
pH	0 1 2	Not Suitable Suitable after pre-treatment Suitable	2	2	2	2	2	2	2	2
Total			6/10 60%	12/18 66.6%	12/16 75%	5/8 62,5%	17/20 85%	14/20 70%	3/4 75%	12/20 60%

4.5.2.1 Unplanted and planted drying beds.

Mobile toilets exhibited high strength faeces, rich in organic matter, nutrients, and TS content. Although MTs displayed highest TS% compared to STs and PTs, it was still lower than optimal performance levels for UPDBs and PDBs. A minimum TS of 20-30% for UPDBs is considered ideal, however, a higher range (35-40%) is often preferred for a drier product (Silva et al., 2023, Seck et al., 2015, Uggetti et al., 2009, Kengne et al., 2014, Al-Malack et al., 2002, Dodane and Ronteltap, 2014a, Strande and Brdjanovic, 2014, Cofie et al., 2006). Conversely, MTs, VIPs and STs showed optimal TS% for PDBs, as suggested in the studies by Seck et al. (2015) and Sonko et al. (2015) who show that planted drying beds require a lower range of ~7-20% TS. The lower TS% required in PDBs is owed to plants assisting in the moisture reduction of sludge (Elbaz et al., 2020, Kengne et al., 2014).

Kuffour (2015), Elbaz et al. (2020) and Krithika et al. (2017) show that typical TSS concentrations for UPDBs is around 1000-1500 mg/L and 500-2000 mg/L for PDBs. However for optimal treatment it is advisable to have a lower TSS for planted drying beds (~500 mg/l), as high levels are shown to reduce the amount of available water for plant growth, negatively impacting the PDB performance (Jain et al., 2022, Englund and Strande, 2019). All sanitation systems exhibited high TSS concentrations, therefore require pre-treatment for peak performances of UPDBs and PDBs.

CDD society (2017) reports an ideal operating COD of 20 000-50 000 mg/L for PDBs. This implies all sanitation systems would require pre-treatment to lower the COD concentration. Ahmed (2019) explains that treating un-stabilised sludge on PDBs can lead to drying difficulties, process upsets, pathogen growth, and increased treatment costs, eventually producing a poor-quality final product which may not be suitable for fertiliser or soil.

Studies show that optimal nutrient concentrations for PDBs are difficult to define as they vary based the drying bed/plants utilised (Kengne et al., 2014, Kuffour, 2015). However, general guidelines presented in Dodane and Ronteltap (2014a), Singh et al. (2017a) and Kuffour (2015) show that nitrogen in the form of ammonium and organic nitrogen should be between 1-3% on a dry weight basis, phosphorus between 0,2- 0,5% and potassium 1-2%. The combination of nitrogen, ammonia, phosphorus, and potassium are vital nutrients for plant growth and, as such, can be used in fertilisers, however, excessively high concentrations can pose adverse environmental discharge impacts (Smith and Schindler, 2009).

School sanitation systems found that fresher faeces in MTs had the highest nutrient concentrations compared to digested FS from VIPs and STs that stabilised within its containment.

Septic tanks exhibited the highest C:N ratio, followed by VIPs and MTs. The low C:N ratio in MTs can be explained to its high nitrogen content. Nikiema et al. (2020a) explains that optimal C:N ratios for PDBs range between 20:1- 30:1. Typically a lower C:N ratio is more favoured (i.e., 20:1) for PDBs as increased nitrogen will assist with increased plant growth, conversely minimising nitrogen being lost through denitrification (Kadlec, 2012, Lusk et al., 2017).

For UPDBs, all sanitation systems exhibited a performance sensitivity score of (60%). Whilst VIPs illustrated the highest performance sensitivity score (66,6%) for PDBs, followed by STs and MTs at 61,1%.

4.5.2.2 Anaerobic Digestion

For AD, the typical solids concentration ranges between 3-20%, depending on the size of the digester being used. Monnet. (2003) described that a low TS% concentration is typically <10% and a medium TS% ranges between 15-20%, so a higher TS will require a smaller digester volume due to the lower water content, however an excessively high TS% (> 40%) can cause blockages within the digestion tank. All school sanitation systems had ideal total solids content for direct treatment.

The ideal organic content is reported between 60-80% with a higher VS% yielding a higher biogas production (Semiyaga et al. 2015; Lu, 2006). According to CDD Society (2017), optimal COD concentrations ranges between ~ 20 000-50 000 mg/L for high strength AD, and around 500-3 000 mg/L for low strength AD (Tchobanoglus et al., 2003; Shivendra et al., 2016; Gold et al., 2018). Hence, a stabilisation pre-treatment, is necessary to lower the COD of all FS from school sanitation systems prior to direct application.

Lu (2006) and Sheng et al. (2013) show that ammonium-nitrogen should range between 2000-5000 mg/L to promote quicker biodegradation for a higher biogas production. However, developers and design engineers should consider that, the characteristics of AD vary based on the specific anaerobic digestion system, process configuration, and operating conditions.

Anaerobic digestion scored the highest performance sensitivity across all three sanitation systems with 87,5% from STs, 81,3% from MTs and 75% VIPs, supporting a high operation success documented in literature.

4.5.2.3 Pyrolysis

Faecal sludge obtained from all school systems (MTs, VIPs and STs) requires pre-treatment before performing pyrolysis, to reduce the moisture content and increase solids concentration (all school systems illustrated a TS < 20%). Bridgwater (2018), Akhtar and Amin (2012) and Bridgwater et al. (1999) found that for effective pyrolysis it is important for moisture to be below 10%. All school sanitation systems exhibited a MC of above 83% which can interfere with the heating process and critically reduce the energy efficiency of the pyrolysis system.

Although TS% is the most important parameter regarding pyrolysis, Junglen et al., (2020), Nansubuga, (2015) and Piskorz et al. (1986) showed for a high quality biochar typical organic matter content ranges between 60-80%. Hence, high organic matter produces a biochar rich in carbon, whereas a lower organic content leads to the formation of biochar rich in inorganics (He et al., 2022). For cases where organic content is high, biochar can be used as a source of biofuel, whereas in the case of low organic content, biochar can be used as an agricultural product (Nikiema and Cofie, 2014). School sanitation systems showed that FS from STs and MTs were within optimal ranges to produce high quality biochar for reuse as biofuel, while VIPs exhibited a much lower organic content and can subsequently be used as an agricultural product.

Pyrolysis scored a good to moderate performance sensitivity score across all sanitation systems, with MTs and STs scoring (75%), followed by VIPs at (62,5%), supporting a moderate operation success documented in literature for VIPs and a high operation success for STs and MTs.

4.5.2.4 Composting and vermicomposting

Contreras-Ramos et al. (2005), Nsiah-Gyambibi et al. (2021) and Manga et al. (2023) showed that a suitable C:N ratio for vermicomposting and composting ranged between 20:1- 30:1. The low C:N ratio observed from all school sanitation systems can be balanced with an incorporation of carbon rich materials such as leaves, vegetative waste or sawdust prior to treatment.

Both vermicomposting and composting require a higher TS% than reported from school sanitation systems. For vermicomposting, the ideal TS% should range between 20-30% whereas composting is ideal for FS with 30-40% TS (Cofie et al., 2009, Manga et al., 2021, Lalander et al., 2013). All sanitation systems showed higher TSS concentrations than optimal ranges according to Debnath (2018) and Nikiema et al. (2020b), for both vermicomposting and composting. This subsequently requires a pre-treatment to lower TSS levels which allows for sufficient microbial activity to break down the organic matter, while also maintaining proper aeration for the compost pile.

Additionally, COD is an essential characteristic to consider for composting and vermicomposting, owing to high COD concentrations inhibiting microbial activity and low COD concentrations providing limited organic matter for the microorganisms/ earthworms to break down. Kharrazi et al. (2014), Debnath (2018) and Singh et al. (2019) reported an optimal vermicomposting COD range between 2000 - 4000 mg/L and Nikiema et al. (2020b) showed an average COD of ~21 757 mg/l for successful composting. These ranges allow for proper decomposition, pathogen reduction and ensures there are adequate nutrients for the microorganisms to breakdown organic matter within the compost pile. All sanitation systems exhibited higher COD concentrations than the ideal range, which stipulates the need for pre-treatment.

It is important to consider nutrient content when treating sludge through composting/vermicomposting, as an excessively high nutrient concentration can cause environmental issues such as contamination of water bodies and eutrophication (Niwaigaba et al., 2014b). Moreover, factors contributing to higher nutrient concentrations are also consequences of chemical additives (common to MTs and VIPs), which may pose risks to earthworms and disrupt the vermicomposting process. Typically, Yadav et al., (2012) showed that vermicomposting exhibited higher nitrogen, ammonium, potassium, and phosphorus thresholds compared to traditional composting (Nikiema et al., 2020b). This is likely due to the worms in vermicomposting requiring a higher concentration of nutrients to sustain their growth and reproduction, compared to the microorganisms which exhibit lower nutrient requirements (Adhikary, 2012, Rostami, 2011, Fornes et al., 2012).

Hence, despite MTs demonstrating the highest nutrient content, FS from VIPs and STs were better suited for vermicomposting and composting. This preference is justified by the potential adverse effects from high nutrient concentrations. Overall, vermicomposting scored a performance sensitivity of 85% and 80% for VIPs and STs respectively, and 60% for MTs. Similarly for composting, VIPs and STs exhibited scored the highest performance sensitivity score of 70% and 65%, followed by MTs at 55%.

4.5.2.5 Solar Drying

Typically, solar drying is best suited for viscous sludges with a TS% > 15 which allows for efficient removal of moisture, resulting in a hygienic dry-end product (Mugauri, 2019, Mawejje, 2021). MTs had a TS of 17,2% supporting the treatment criteria, STs exhibited a slightly lower TS% of 10,2% hence not requiring an intensive pre-treatment compared to VIPs (TS% 8,5%).

Solar drying scored a good performance sensitivity score across all sanitation systems with all three systems scoring 75% supporting a high operation success documented in literature.

4.5.2.6 Deep row entrenchment

Deep row entrenchment (DRE) is specific to the local regulations and site conditions. For optimal treatment performance, Ronteltap et al. (2014), Adadzi (2012) and Still et al. (2012b) report the TS% of FS should range between 30-40%, a TSS of 1000-2000 mg/L, and a VS of ~60-70%. Optimal COD concentrations were outlined as 20 000 – 50 000 mg/L (Still et al., 2012b). These findings imply that FS from all sanitation systems will require a pre-treatment to reduce its TSS content, ensuring a good treatment efficiency. Similarly, all systems will require pre-treatment to increase TS%, and lower COD content for optimal treatment efficiency. Sanitation systems reported a performance sensitivity score of 65% for STs and 60% for MTs and VIPs.

4.5.3 Summary of optimal treatment pathways for rural school sanitation systems

Overall, Figure 52 presents a combination of treatment processes to maximise resource recovery from rural school VIPs, STs, and MTs.

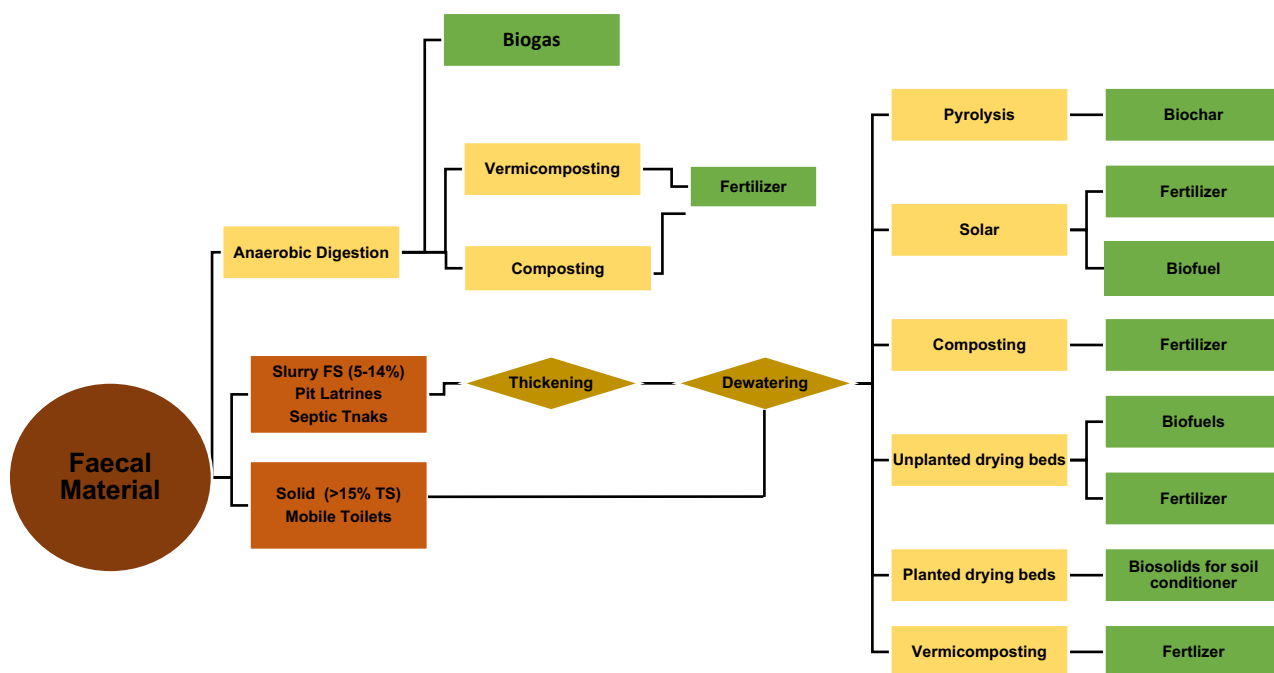


Figure 51: Combination of treatment processes to maximise resource recovery from rural school sanitation systems

Legend

	Output
	Treatment Process
	Type of faecal sludge
	Pre-treatment

4.6 Conclusion

This chapter analysed and discussed the characterisation results of faecal material from rural school OSS. Optimal treatment and emptying pathways were identified for each sanitation system. The implementation and feasibility of each technology was examined, including, operating parameters, spatial requirements, technical skill expertise, and capital/operating expenditures for rural areas. The following chapter will present the overall conclusions for this dissertation.

Chapter 5: Conclusions and Recommendations

Inadequate disposal, management and valorisation options for faecal material in rural school toilets have led to rising environmental and health risks on the South African youth. With minimal studies reporting on the severity of this problem, this study undertook a detailed chemical, physical, mechanical, and thermal characterisation of faecal sludge and faeces from rural schools' sanitation systems (STs, MTs, and VIPs) in Durban, South Africa. Variation of faecal matter properties were observed across different sanitation systems. This research suggested emptying and treatment methods that could be implemented for each sanitation system.

5.1 Conclusions of suitable emptying and treatment technologies for rural school toilets based on analysed faecal sludge characteristics.

5.1.1 Mobile toilets

Mobile toilets were emptied routinely on a weekly basis, resulting in the presence of relatively fresh faeces upon sample collection. Laboratory analysis indicated that faeces from MTs displayed the highest concentration in almost every physico-chemical parameter analysed including, TS, VS, COD, and all nutrient tests (N, P, K and $\text{NH}_4\text{-N}$). The high COD findings (242 g/L) showed a strong correlation ($R= 0,90$) to TS% (17,4%), which explained that an increase in COD subsequently increased TS%. Therefore, a high TS% could possibly indicate high organic matter presence, correspondingly posing treatment implications. Similarly, high the volatile solids and nutrient concentration was indicative of the less stabilised faecal material and high organic content.

Rheological properties revealed that fresher faeces from MTs required a lower shear yield stress at the same solid's concentration, compared to a more degraded faecal sludge from VIPs and STs, in turn improving flowability, and reducing pump head required for treatment processes. Thermodynamic tests revealed that faeces from MTs illustrated the highest calorific value making it suitable for biofuel and biogas production. Therefore, faeces from MTs were considered to have "high strength" (high strength sludge referring to a higher concentration of nutrients and organic matter, which allows for the greatest potential for resource recovery), however requiring multiple treatment stages to ensure environmental discharge standards are met.

The decision tree used for determining possible emptying pathways, revealed that manual emptying methods such as (MAPET and manual diaphragm) are most appropriate for mobile toilet systems, this was based on MTs illustrating shallow containment, large presence of trash, high TS% and denser faecal material. However, the lower shear yield for MTs further implies mechanical methods can be explored

after a pre-emptying stage whereby the presence of trash is lowered, or faecal material watered down to prevent pump blockages.

Additionally, decision matrices for optimal valorisation showed that faecal characteristics from MTs indicated a good performance sensitivity score for most treatments' technologies and were highest for UPDB, PDB, AD, pyrolysis and solar drying. Overall resource products that can be obtained from MTs included, fuel-biogas, biochar, and soil conditioners, correspondingly fresh faeces from MTs demonstrated a good potential to enable circular economy opportunities.

5.1.2 Septic Tanks

Faecal sludge from STs demonstrated a higher degree of stabilisation compared to that from MTs. However, analysed sludge characteristics from school STs displayed slightly different physical characteristics when compared to literature and community-based sanitation systems. A relatively higher average TS% (10,2) was observed from rural school STs which was unusual, however, this was likely due to the water shortages experienced at rural schools.

Septic tank FS showed lower concentrations of nitrogen and ammonia compared to fresher faeces from MTs, thus reducing the need for certain treatment requirements prior to environmental discharge. Compared to MTs, STs exhibited a lower strength of FS, which reduces intensive pre-treatment requirements for pathogenic reduction, but lowers the potential for optimal resource recovery.

Rural schools where STs operated normally (without prolonged water shortages) demonstrated FS with a higher fluidity (TS% 5-15%), due to the high-water content, which subsequently dilutes resource recovery potential, and rendering these systems more amenable to various emptying methods. The decision tree showed that STs FS could be emptied by seven possible techniques whereas MTs were characteristically suited to one. However, it is important to note, when comparing faecal material at the same solid's concentration, fresher faeces (from MTs) exhibited a lower yield stress.

Treatment decision matrices showed that, STs sludges obtained high performance sensitivity scores for composting, vermicomposting, pyrolysis, AD, solar drying, UPDBs, and PDBs. This was largely due to the optimal nutrient concentration and better stabilised waste, when compared to MTs that illustrated excessively high nutrient concentrations, which can pose environmental threats.

Overall end use products that can be obtained from rural STs sludge include biochar, fertiliser, biogas-fuel, and soil conditioners.

5.1.3 Pit latrines

Faecal sludge from VIP latrines revealed the highest level of stabilised sludge, likely owing to the extended duration the sludge was maintained within its containment. This was supported by laboratory findings which showed VIPs exhibited the lowest VS% (45,5%) and COD (72,6 g/L) concentrations. The higher degree of stabilisation of sludge from VIPs requires less intensive stabilisation treatment processes, such as MTs prior to environmental discharge, subsequently making it a less costly FS to handle.

School pit latrine sludge exhibited a slurry texture this was supported by a TS of (8, 5%) and shear stress of 22,7 Pa, compared to typical VIP sludge (TS 20-30%), rendering it applicable to three types of emptying methods, including conventional vacuum tankers, MAPET emptying and pit screw augurs. Additionally, due to the problem of trash build-up in pit latrines, it is recommended to implement effective waste management strategies, such as introducing dedicated bins for non-biodegradable waste and conducting educational programs to raise awareness among users about proper disposal practices.

Treatment decision matrices showed that VIP sludge scored a high-performance sensitivity to composting and vermicomposting, owing to its optimal nutrient concentration and degree of stabilisation. Solar drying, UPDBs, PDBs, and DRE scored a moderate performance sensitivity score. However, for effective pyrolysis treatment, VIP sludge would require a longer drying and dewatering period to increase TS% compared to MTs or STs. Obtainable end-use products from rural school VIP sludge include biochar for an agricultural product and fertiliser.

5.1.4 Key conclusion for rural school sanitation systems, emptying and treatment technologies.

Overall faecal material from MTs and VIPs, obtained from rural school sanitation systems were comparable faecal material from community-based sanitation systems. However, some schools presented VIPs that exhibited a lower-than-average, TS due to a high infiltration of water in the system. Conversely, the FS from STs illustrated a higher-than average TS% owing to certain schools experiencing pro-longed water shortages, hindering the normal performance of STs, and limiting the amount of “flush-water” entering the system.

In conclusion, the way forward for rural school sanitation is based on contextual factors. Ideally the adoption of water-borne sanitation (i.e., STs), where possible (if water availability allows it) should be embraced, however, challenges such as high infrastructure costs, dependence on reliable water supply, and technical complexity need to be carefully considered. Nevertheless a transition towards these

systems will greatly prohibit the spread of diseases associated with dry systems such as MTs and VIPs and make female use of school toilets more comfortable. In addition, this change will drastically reduce the deaths that have spurred from primary school children drowning in VIPs due to their unsafe nature. The move towards water-born sanitation systems would facilitate the use mechanical emptying methods, with a plethora of emptying options, such as vacuum tankers, vacutug, and motorised diaphragm pump. Although treatment pathways for STs were not as optimal for direct resource recovery as fresh faeces in MTs, viable options do exist following pre-treatment such as, PDBs, UPDBs, SD, AD, pyrolysis, and composting.

Moreover, progressive options for areas with no water access, include, contained-based sanitation or re-invented toilets such as composting toilets. This will allow for faeces to be valorised relatively fresh, obtaining direct resource recovery, for high quality end-use products like biogas, and biochar for a fuel source.

5.2 Recommendations for sanitation systems, emptying and treatment of faecal material.

5.2.1 Recommendations for sanitation systems at schools

- Upgrading of MTs and VIPs to improve performance and safety. Currently MTs act as a temporary containment solution, with limited maintenance and negative user perception. Therefore, consideration to improve user experience and safety is as critical to the final treatment, by maintaining a high standard of hygiene through providing amenities like toilet paper, hand-washing water and soap, menstruation bins and frequent cleaning.
- Design and construction: Improving the design the of novel MTs and VIPs to ensure they are safe, durable, and environmentally appropriate. This can include the use of materials that are resistance to corrosion and degradation, designing for ease of maintenance and emptying, and ensuring proper ventilation and drainage. Or a design that promotes some degree of treatment during storage.
- Novel designs of STs should incorporate improved filtration systems which can remove pollutants or harmful substances from ST sludge in turn protecting groundwater contamination. Similarly, back-up water tanks should be incorporated into these systems ensuring there is a sufficient flush water if water shortages occur.
- Septic tanks could be integrated with other systems like greywater systems, or rainwater harvesting systems to create efficient systems for managing wastewater and faecal sludge.

- Implementing effective maintenance and operation programs that ensure MTs, VIPs, and STs are properly managed, cleaned, and emptied regularly. This can include the development of appropriate regulations, monitoring systems, and training programs.
- Whereby water-based sanitation cannot be employed, innovative toilets that can be used to replace MTs and VIPs include composting toilets, urine-diverting toilets, vacuum toilets, ecological sanitation systems or container-based sanitation (these systems operate similarly to MTs, however, exhibit better management whereby toilets collect the excreta in cartridges and thereafter transferred to a treatment facility for adequate treatment and resource recovery).

5.2.2 Recommendations for treatment and emptying.

- Improved stakeholder engagement. A greater emphasis should be placed on engaging the private sector for developing and implementing FSM across the entire sanitation chain, subsequently this could build the support and increase sustainability on a long-term basis.
- Capacity building through community outreach and education, more work should be done on conducting community outreach, to raise awareness about the potential of FSM and potential health and environmental risks associated with poor practices.
- Involving local leaders, such as community and religious leaders in the educational outreach to build support. This is owed to religious leaders being seen as more relevant to the cultural and social norms of rural communities, and therefore better trusted compared to outside sources of information, such as scientists or government officials.
- There should be regular monitoring and analysis of FS characteristics to understand how it changes over time to make informed decisions about treatment technologies over time.

Chapter 6: References

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Appendices

Appendix A: Ethical Clearance Certificate



24 May 2022

Miss Estella Zandile Jingxi (221089153)
School Of Engineering
Howard College

Dear Miss Jingxi,

Protocol reference number: BREC/00003672/2021

Project title: Characterisation of faeces, urine and faecal sludge from school sanitation facilities

Degree: PhD

We wish to advise you that your application for amendments listed below received on 12 May 2022 to add a new co-investigator (Ms Yuri Ramruthan) for the above study has been noted and approved by a sub-committee of the Biomedical Research Ethics Committee.

The committee will be notified of the above approval at its next meeting to be held on 14 June 2022.

Yours sincerely

[Redacted signature]

Ms A Marimuthu

(for) Prof D Wassenaar

Chair: Biomedical Research Ethics Committee

15 December 2021

Miss Estella Zandile Jingxi (221089153)
School Of Engineering
Howard College

Dear Miss Jingxi,

Protocol reference number: BREC/00003672/2021

Project title: Characterisation of faeces, urine and faecal sludge from school sanitation facilities

Degree: PhD

EXPEDITED APPLICATION: APPROVAL LETTER

A sub-committee of the Biomedical Research Ethics Committee has considered and noted your application.

The conditions have been met and the study is given full ethics approval and may begin as from 15 December 2021. Please ensure that any outstanding site permissions are obtained and forwarded to BREC for approval before commencing research at a site.

This approval is subject to national and UKZN lockdown regulations, see (http://research.ukzn.ac.za/Libraries/BREC/BREC_Amended_Lockdown_Level_1_Guidelines.sflb.ashx). Based on feedback from some sites, we urge PIs to show sensitivity and exercise appropriate consideration at sites where personnel and service users appear stressed or overloaded.

This approval is valid for one year from 15 December 2021. To ensure uninterrupted approval of this study beyond the approval expiry date, an application for recertification must be submitted to BREC on the appropriate BREC form 2-3 months before the expiry date.

Any amendments to this study, unless urgently required to ensure safety of participants, must be approved by BREC prior to implementation.

Your acceptance of this approval denotes your compliance with South African National Research Ethics Guidelines (2015), South African National Good Clinical Practice Guidelines (2020) (if applicable) and with UKZN BREC ethics requirements as contained in the UKZN BREC Terms of Reference and Standard Operating Procedures, all available at <http://research.ukzn.ac.za/Research-Ethics/Biomedical-Research-Ethics.aspx>.

BREC is registered with the South African National Health Research Ethics Council (REC-290408-009). BREC has US Office for Human Research Protections (OHRP) Federal-wide Assurance (FWA 678).

The sub-committee's decision will be noted by a full Committee at its next meeting taking place on 08 February 2022.

Yours sincerely,



Prof D Wassenaar
Chair: Biomedical Research Ethics Committee

Biomedical Research Ethics Committee
Chair: Professor D R Wassenaar
UKZN Research Ethics Office Westville Campus, Govan Mbeki Building
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Founding Campuses:  Edgewood  Howard College  Medical School  Pietermaritzburg  Westville

INSPIRING GREATNESS

Appendix B: Summary of physico-chemical property results

Table 20: Physico-chemical results

Parameter	Unit	Mobile Toilets				Septic Tank				Pit Latrine			
		Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
TS	%Dry mass	17,4	2,8	14,4	20,6	10,2	4,3	5	15	8,5	1,9	6,8	12,2
TSS	g/L	121,6	24,3	90,4	150,4	77,9	38,4	41,6	177,7	107,	13,9	113,4	119,1
TDS	g/L	33,4	14,4	10,9	50,7	9,7	6,7	1,7	25,2	42,3	3,9	37,2	47,3
VS	%	78,4	7,7	61,4	83,8	77,4	9,8	62,3	88,9	45,5	5,6	35,6	50,9
Moisture content	%	82,6	2,8	79,5	85,6	89,8	4,2	85,0	95,0	91,5	2,0	87,8	93,2
COD	g/L	242,1	32,9	193	287,6	119,2	63,1	40,4	196,2	72,6	26,3	40,7	108,1
pH	-	5,9	0,2	5,6	6,4	7,3	0,3	6,9	7,6	8,5	0,1	8,3	8,5
EC	µs/cm	7675	1356,9	5690	9080	2334,1	3236,3	105,1	7675	30583,3	1885,3	28100	32600
TN	g/L	9,7	1,493	7,7	3,5	2,7	1,6	1,0	4,6	2,8	0,3	2,5	3,5
NH ₄ -N	g/L	3,4	1,5	2,0	6,3	0,7	0,3	0,4	1,5	2,2	0,2	1,9	2,5
K	g/L	3,3	0,7	2,2	4,4	0,8	0,2	0,4	0,9	0,9	0,1	0,9	1,0
P	g/L	2,2	0,2	2,1	2,6	0,6	0,2	0,4	0,9	1,1	0,2	0,8	1,4
C:N		4,9	1,8	2,2	6,1	8,2	1,7	7,8	11,6	9,7	3	4,9	10,4

Appendix C: Operating Parameters for manual and mechanical emptying techniques

Table 21: Operating and suitability parameters for manual emptying technologies.

Type of emptying Technology	Suitability	Maximum pumping head	Maximum Flow rate	Constraints
Gulper	<ul style="list-style-type: none"> • Applicable for less viscous sludge, with low pumping requirements • Easily accessible to rural areas • Can be constructed with locally available materials 	Maximum Pumping head usually 1 m, but dependent on design (Mikhael et al., 2014)	Average flow rates of 30 L/min (Mikhael et al., 2014)	<ul style="list-style-type: none"> • Cannot access bottom parts of the sanitation system, i.e., if the pit is deep. • Requires increased time to empty. • Clogging if non-biodegradable material is present.
Manual diaphragm	<ul style="list-style-type: none"> • Applicable for less viscous sludge, with low pumping requirements • Easy to use. • Affordable 	Maximum pumping head of 3.5 m – 4.5 m (Mikhael et al., 2014)	Maximum flow rate of 100 L/min (Mikhael et al., 2014)	<ul style="list-style-type: none"> • Easily clogged if non-biodegradable material is present. • Difficulty keeping a tight seal reducing functionality. • Difficulty sourcing materials in local areas (Muller and Rijnsburger 1992) • Difficulty emptying sludge at great depths
MAPET	<ul style="list-style-type: none"> • Suitable for thicker sludge • Most advanced manually driven with mechanical system. • Cheaper than motorized equipment 	Maximum pumping head 3 m (Mikhael et al., 2014).	Maximum flow rates of between 10 and 40L/min (Mikhael et al., 2014)	<ul style="list-style-type: none"> • Strong reliance on institution service providers. • Requires importation for key materials. • Can be costly, as MAPET service providers cannot cover transport from emptying fees (Waste Consultants, 1993)

Strande et al.(2014).

Table 22: Operating and suitability parameters for mechanical emptying technologies

Type of emptying Technology	suitability of sludge	Maximum pumping head	Maximum Flow rate	Constraints
Pit screw augur	<ul style="list-style-type: none"> • Suited for liquid sludge. • Can handle small presence of non-biodegradable material. 	Maximum pumping head 3 m	Flow rates of over 50 L/min	<ul style="list-style-type: none"> • Difficulty emptying sludge at great depths • Difficulty emptying dry sludge • Clogging if non-biodegradable material is present • Difficulty in moving around owing to weight and size
Vacutug	<ul style="list-style-type: none"> • Suited for sludge with low viscosity. • Can handle small presence of non- biodegradable material • Easily accessible to rural areas 	-	-	<ul style="list-style-type: none"> • Difficult to empty highly viscous FS • Expensive to transport over long distances. • Can only withstand small volumes of sludge. • Handles 500-1900 L of sludge
Motorised diaphragm pump	<ul style="list-style-type: none"> • Suited for more liquid sludge • Can reach great depths 	Maximum pumping head of 15 m	Maximum flow rate of 300 to 330 L/min (Mikhael et al., 2014)	<ul style="list-style-type: none"> • Driven by petrol • Expensive to manufacture and maintain • Clogging if non- biodegradable material is present
Trash Pump	<ul style="list-style-type: none"> • Suited for more liquid sludge 	Maximum pumping head i.e., pumping head of 25- 30 m.	Maximum flow rates of approximately 1,200 L/min (Mikhael et al., 2014).	<ul style="list-style-type: none"> • Clogging if non-biodegradable material is present. • Difficult to source materials.
Conventional Vacuum Tanker	<ul style="list-style-type: none"> • Suited for sludges with low viscosity • Ideal for long haul transport 	Pumping head is dependent on design and pump model utilized.	-	<ul style="list-style-type: none"> • Expensive • Difficult to enter densely populated areas • High militance and operational costs for low-income countries • Some materials need to be internationally sourced

(Strande and Brdjanovic, 2014).

Appendix D: Criteria selection for decision matrix 1

Criterion	Definitions	Legend	Selection criteria
Skill requirement / Ease of operation	The level of skill needed, associated with the amount of knowledge required to operate the treatment technology.	+ ++ +++	Low: No special skills/knowledge needed Medium: Some special skills required High: Only trained operators
CAPEX	Considers the major long-term expenses, i.e., your physical assets needed, such as building, equipment in *However is subject to the cost of land	+ ++ +++	Low Cost: low land prices, constructed with local materials and filter/sand is readily available, minimal electrical dependence. Medium Cost: Not excessively dependent on technology, limited skill required for design, can source energy from free sources i.e., the sun High Cost: Expert Design is required, high tech dependence, high dependence on electricity.
OPEX	Considers the operating costs, and small term expenses, i.e., staff salaries, property rent and taxes.	+ ++ +++	Low Cost: ~ < 1 000 000 Medium Cost: ~ R 1000 000 – 5 000 000 High Cost: ~ > 5 000 000
Land requirement	Refers to the land-use intensiveness, the amount of available land present per spatial unit. Rural areas are characterized by large open spaces, compared to high density urban areas. Hence, technologies, requiring large spaces will be highly suitable	+ ++ +++	Low: Requires minimal area Medium: Medium area required High: Requires a large land space for technology to operate effectively
Energy requirement	Refers to the intensity of energy required by the treatment technology.	+++ +	Low: Not fully mechanised, and require little energy input, i.e., composting treatments. High: Highly mechanised technologies, and dependent on a constant source of energy, i.e., bioreactors in AD.
Reusability of treated waste	The potential of directly reusing the sludge post treatment.	+ ++ +++	Low: No products can be reused Medium: Some further treatment is required for direct reuse High: Products can be directly reused
Ground water table (shallow/deep)	The depth of the groundwater table, this factor considered the possibility of groundwater pollution.	+ +++	Shallow: Less than 6m, with a high groundwater table Deep: Greater than 6m, lower groundwater table
Level of implementation	How widely tested is the treatment technology.	+ ++ +++	Tested in lab: Only understood in theory and practiced within laboratory confinements. Pilot tested: Practiced, but not on a large scale. Deployed in SA: Proven and widely used in South Africa

Source : (Shende and Pophali, 2022); (CSTEP2016); (Neethling, 2023).

Appendix E: Criteria Selection for decision matrix 2- 4

Ideal performance criteria for treatment methods								
Constraint	Unplanted drying bed	Planted drying bed	Anaerobic Digestion	Pyrolysis for Biochar Recovery	Vermicomposting	Composting	Solar Drying	Deep row entrenchment
TS (%total mass)	~ 7-30% (higher idea for drier product i.e., 35%) (Silva et al., 2023, Seck et al., 2015, Uggetti et al., 2009, Kengne et al., 2014, Al-Malack et al., 2002, Strande et al., 2014, Cofie et al., 2006, Dodane and Ronteltap (2014b))	~ 7-20% (Lower TS% as plants assist in the moisture reduction of sludge) Seck et al. (2015) and Sonko et al. (2015)	~ 3-20% (depending on the size of the digester) Small Digester: <10% Medium: 15-20% Monnet, (2003)	~ 70% Yuan et al. (2022)	20-30% (Lalander et al., 2013)	30-40% Best done after drying beds. (Cofie et al., 2009) (Manga et al., 2021) (Strande et al., 2014)	TS > 15% (Mugauri, 2019)	30-40% Ronteltap et al. (2014), Adadzi (2012) and Still et al. (2012b)
VS (%)	-	~ 40-60% Slightly lower than UPDBs (Sonko et al., 2014a)	~ 60-80% (Higher organic content yields a higher biogas production) (Semiyaga et al., 2015, Lu, 2006)	~ 40-80% (Junglen et al., 2020) Piskorz et al. (1986) (Nansubuga, 2015) (Gold et al., 2018a)	~ 40-80% (Yadav et al., 2012) (Acquah et al., 2021)	- ~ 40-65% (Debnath, 2018)	-	~ 50-70% (Still et al., 2015) Ronteltap et al. (2014), Adadzi (2012) and Still et al. (2012b)
TSS concentration (mg/L)	~ 1000-1500 Kuffour (2015) Elbaz et al. (2020); Krithika et al. (2017)	~ 500-2000 (Sonko et al., 2014b) Kuffour (2015) Elbaz et al., (2020); Krithika et al. (2017)	~ 2000-8000 High strength > 30, 000 mg/l (Hastuti et al., 2021)	-	Average ~ 41750 (Debnath, 2018)	average: ~ 17,792 (Nikiema et al., 2020b)	-	~ 1000-2000 Ronteltap et al. (2014), Adadzi (2012) and Still et al. (2012b)
COD (mg/L)	-	20000-50000 (CDD Society, 2017) (Sonko et al., 2014) (Strande et al., 2014), also reported a successful outcome around ~ 16g/L	High Strength AD: ~ 20000-50000 Low strength: AD 500-3000 (Tchobanoglus et al., 2003) (Shivendra et al., 2016) (Gold et al., 2018b)	~ 25407- 90,000 High COD make for a good indicative on biogas generation. (Ahmed et al., 2019)	~ 2000 – 40000 (Kharrazi et al., 2014) (Debnath, 2018) Singh et al. (2019)	Average ~ 21,757 (Nikiema et al., 2020b) (Debnath, 2018)	-	~ 20,000 to 50,000 mg/L (Matthews, 2022)
Nutrient Concentrations Nitrogen (mg/L)	-	~ 1-3% 350-1000 mg/L	2000-7000	-	~ 42000 Yadav et al., 2012)	~ 500-1000 (Nikiema et al., 2020b)	-	~ 500-1000

		Dodane and Ronteltap (2014a); (Singh et al., 2017a) and (Kuffour, 2015)						WRC Report No. TT 883/22
Nutrient concentration: Ammonia (mg/L)	-	-	~ 2000-5000 Lu (2006); Sheng et al. (2013)	-	~ 1000-2000	~ 1000-2000	-	~ 500-800
Nutrient Concentration: Potassium (mg/L)	-	1-2% Dodane and Ronteltap (2014); (Singh et al., 2017) and (Kuffour, 2015)	-	-	~ 26700 (Yadav et al., 2012) Study used results that produced vermicompost suitable for agriculture	~ 200-800 (Nikiema et al., 2020b)	-	~ 10-50
Nutrient Concentration: Phosphorus (mg/L)	-	~ 0,2- 0,5% ~ 18-200 mg/L (Sonko et al., 2014) Dodane and Ronteltap (2014); (Singh et al., 2017) and (Kuffour, 2015)	-		~12400 (Yadav et al., 2012)	~ 100- 600 (Nikiema et al., 2020b)	-	~ 50-100
C: N	-	20:1 -30:1 (Lower is ideal, as it will favour plant growth) Nikiema et al. (2020a)	~20:1 to 30:1 Sanghal and Sharma (2022)	20:1 -30:1 Nuagah et al., 2020)	~20:1 -30:1 (Butt, 1993)Ndegwa and Thompson (2001) Contreras-Ramos et al. (2005); Nsiah-Gyambibi et al. (2021)	Low Medium High	-	~20:1 -30:1 (Still et al., 2012b)
pH	~ 6-7.5 Strande et al., (2014)	~6.5-7.5 (Jain et al., 2022) (However, can vary depending on the type of plant used)	~6.5-7.5 (Too low can inhibit microbial processes i.e., reduce biological performance) Sanghal and Sharma (2022)	5.5 -7.5 (Nuagah et al., 2020)	~5-8 (Lohri et al., 2017)	6-8 (Rynk 1992). (Lohri et al., 2017)	6-7.5 (Mudasar and Kim, 2017) (Colón et al., 2015)	~6-7.5 (Still et al., 2012b)

Appendix F: Method utilized for sampling and SOPs for Lab testing.

Full description of method

Method 3.4.1: Grab Sampling

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SOPs

<http://prg.ukzn.ac.za/laboratory-facilities/standard-operating-procedures>

Appendix G: Additional Figures

1) Calorific Value correlated with VS.

