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Treatment of human waste in small-scale facilities: a prospective review

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A sustainable approach to the small-scale treatment of portable toilet and septic tank waste is in need to minimise the associated public health and environmental risks. Treatment of human wastes is highly regulated by legislation in many countries given the high content of organic components and pathogens. In conventional wastewater treatment plants, sewage undergoes a series of treatment stages (primary, secondary and tertiary) before effluent discharge. Delivering this approach on a small scale, to meet demands of temporary events, public gatherings or remote locations provides benefits in reducing transport and bulk handling. For a small-scale treatment process to effectively work, the process steps need to be simplified and minimised. The potential of a new treatment scheme is reviewed where the first step is the solid–liquid fraction separation, followed by anaerobic digestion (AD) of the solid fraction, including energy recovery in a combined heat and power unit. The liquid fraction undergoes a series of filtration and disinfection steps to comply with effluent regulations. Digestate from AD is burned on site to provide local domestic/office heating. This has a great potential for application in different locations where inputs may be sporadic, such as outdoor festivals, disaster response scenarios and construction sites.

Keywords: sustainability/UN SDG 6: Clean water and sanitation/waste management & disposal/waste valorisation

1. Introduction

The provision of portable toilets for music festivals, agricultural shows, weddings and constructions sites is typically through small-scale contractors who also offer waste removal and disposal and provide septic tank emptying services. Currently, the waste collected is mainly transported to wastewater treatment plants (WWTP) away from the production source for full treatment and disposal (Bonifazi *et al.*, 2019).

A more sustainable approach to waste management is needed in order to reduce the carbon footprint and cost associated with disposal in a WWTP. This paper presents a critical appraisal of the best available techniques for human waste treatment on a small scale. The potential to treat the waste onsite and potential for energy production and by-product recovery are also evaluated. The aim is to provide a state-of-the-art knowledge of human waste treatment for a future plant design and development. This same approach can be followed for other scenarios with localised sporadic inputs, such as disaster response (Potangaroa *et al.*, 2011).

Blackwater is defined as wastewater from toilets or urinals, and it is mainly composed of human excreta (urine and faecal matter), flush water and toilet paper (Florentino *et al.*, 2019). Contributions considered as accidental, such as cleaning chemicals in toilets, cigarette butts and so on, are also included. Portable toilets and septic tanks can also contain small quantities of inert and hard-to-degrade materials such as stones, rags and plastics. Blackwater is referred to as faecal sludge (FS) when it is collected from septic tanks, pit latrines, dry toilets or generally any technology not connected to the sewer, such as portable toilets (Fakkaew *et al.*, 2018; Penn *et al.*, 2018; Septien *et al.*, 2018a).

The parameters that should be considered for the characterisation of FS are the same as those considered for wastewater characterisation: total solid (TS) concentration, chemical oxygen demand (COD), biological oxygen demand (BOD), nutrients, metals and pathogens. Additional parameters such as calorific value and volatile solids (VS) are of interest for waste-to-energy conversion technologies (Strande *et al.*, 2018; Ward *et al.*, 2019). Standard values for FS characterisation are difficult to determine, as they are greatly dependent on onsite sanitation technology, geographical location, climate, frequency of collection and inflow/infiltration (Englund *et al.*, 2020). Table 1 presents FS characteristics from literature, where the high variability in the parameters can be noticed.

The length of time that FS is stored will greatly affect its characteristics mainly due to the digestion of organic matter that occurs during storage (López *et al.*, 2019; Singh *et al.*, 2017). Faeces sludge from septic tanks shows lower values of BOD and ammonia (NH_4^+–N) than those from public toilets (Table 1). Waste from old septic tanks has an inert composition of 10% whereas waste from fresh septic tanks has a much greater digestible material. As biological degradation rates are temperature dependent, the local climate will greatly influence the degradation rates in septic and storage tanks.

The aim of this study is to review the current state of the art in human waste treatment and to propose a new small-scale process scheme to treat FS from portable and/or sporadic sources as an alternative to its treatment is large-scale WWTP.

Table 1. Reported parameters of FS from onsite sanitation facilities

	FS	
Parameter	**Public toilet**	**Septic tank**
Density: kg/l		1–2.2
Dewatering rate	11% TS in the dewatered cake	
TS: mg/l	30 000–52 500	12 000–35 000
VS (as % of TS)	65–70	45–75
pH	6–9	
COD: mg/l	20 000–50 000	1200–10 000
BOD: mg/l	7600	840–2600
Total nitrogen, TN: mg/l	50–1500	
Ammonium nitrogen, $NH_4^+ - N$: mg/l	2000–5000	400–1200
Total phosphorous, TP: mg/l	450	150
Faecal coliforms (cfu/100 ml)[a]	10^5	
Calorific value: MJ/kg	11–19	

[a]cfu, colony-forming units, number of viable microorganisms
Source: Penn *et al.* (2018) and Strande *et al.* (2014)

Figure 1. Manual bar screen

2. Waste treatment options

The main aim of the treatment is to ensure the protection of the environment and human health (Jurga *et al.*, 2019; Oarga-Mulec *et al.*, 2017; Paulo *et al.*, 2013). Maximum energy recovery from the waste is the secondary treatment target (Gao *et al.*, 2019).

2.1 Solid–liquid separation

The first treatment step consists of a solid–liquid separation, also referred to as dewatering. Solid–liquid separation can be achieved through gravity separation (settling–thickening tanks), filtration, evaporation, centrifugation or drying. The resulting fractions both require further treatment. Most of the organic matter and pathogens present in the waste will be contained mainly in the solid fraction.

2.1.1 Screening

A preliminary treatment for the removal of coarse materials such as plastics, stones, wood or textiles is required (USEPA, 2003). These solids are trapped by vertical or inclined bar racks (Figure 1). Bars set at 30–70° are preferred to facilitate cleaning. In small-scale plants, the screening bars are usually hand cleaned. The distance between the bars is usually 30–50 mm allowing the liquid fraction to flow while larger particles are trapped (le Hyaric *et al.*, 2009). A minimum flow velocity of 0.45 m/s is required to prevent fine solid deposition and maximum flow velocity should not exceed 1 m/s to avoid washout of coarse waste through the bars. The approach channel should be straight for at least 60 cm ahead of the screen bars to produce uniform flow and prevent eddies around the screen (Moran, 2018). Stop-gates are recommended ahead and behind the bars for maintenance. Screenings can be disposed of along with municipal solid waste or can be incinerated.

2.1.2 Settling–thickening tanks

This is a robust and resilient technology where the waste is introduced at the top of a rectangular or circular tank and the liquid (supernatant) exits at the opposite side. The solids settle at the bottom of the tank whereas lighter particles float to the surface (Figure 2) (Kim and Pipes, 1999). The loading periods of the tanks range from 1 week to 1 month depending on the tank volume and the solid fraction (thickened sludge) that can be pumped out (Bajcar *et al.*, 2011; Patziger *et al.*, 2012). Both the incoming and thickened sludge are pathogenic; therefore, operators should be equipped with proper protection. The settling tank performance is measured by BOD, TS and phosphorus removal efficiencies. These efficiencies typically range between 25 and 35% for BOD, 50–65% for TSS and 5–10% for nitrogen and phosphorus. The main parameters in the design of settling tanks are surface overflow rate, hydraulic retention time, settling velocity and temperature (Bajcar *et al.*, 2011; Jover-Smet *et al.*, 2017; Kim and Pipes, 1999; Patziger *et al.*, 2012).

The main advantages of this technology are the relatively low capital and operational costs, no electrical energy requirements and no skilled operators are needed. The main disadvantage is that the settling process will be affected by the precipitation as rainfall will be mixed with the tank contents with the liquid fraction increasing and impacting on downstream processes flows (Voutchkov, 2005).

2.1.3 Filtration

Drying beds are commonly used in wastewater and FS treatment but as seen for settling–thickening tanks, the climate conditions will be the major drawback (Al-Nozaily *et al.*, 2013). Drying beds are made of porous media (gravel and sand) and can be unplanted (acting as a sand/gravel filter) or planted emergent macrophytes. The dying beds combine different physical and biological mechanisms of action: percolation, evapotranspiration and mineralisation (Kengne and Tilley, 2014).

The techniques described here are well established for sewage sludge dewatering and may be applied to FS treatment.

In belt filter presses, the sludge is squeezed between two filter cloths. The filter belts pass under successive rollers increasing the pressure and compressing the sludge cake. Belt filter presses are suitable for small-scale applications (Schaum and Lux, 2011) but they will require large quantities of rinse water for filter cleaning. Belt filter presses can achieve final product of 15–32% TS.

Frame filter presses consist of a series of vertical frames covered with filter cloths creating a chamber between each two successive frames. The sludge is fed into the chambers and dewatered under pressure with the water passing through the filter cloths and draining off. By releasing the pressure on the frames, the filter cake can be recovered (Cui *et al.*, 2022).

Frame filter presses achieve higher final solids concentrations around 40% (Wakeman, 2007); however, the moisture content reduction depends on the compressibility properties of the sludge and the addition of conditioning chemicals (Valderrama-Bravo *et al.*, 2022).

The main disadvantage of mechanical filtration is the energy requirements and the higher capital and operating costs. The main advantages when compared with the settling tanks are the speed of dewatering and the small physical footprint.

2.1.4 Centrifugation

During centrifugation, FS is squeezed outwards against the surface of a rotating cylinder due to the centrifugal forces

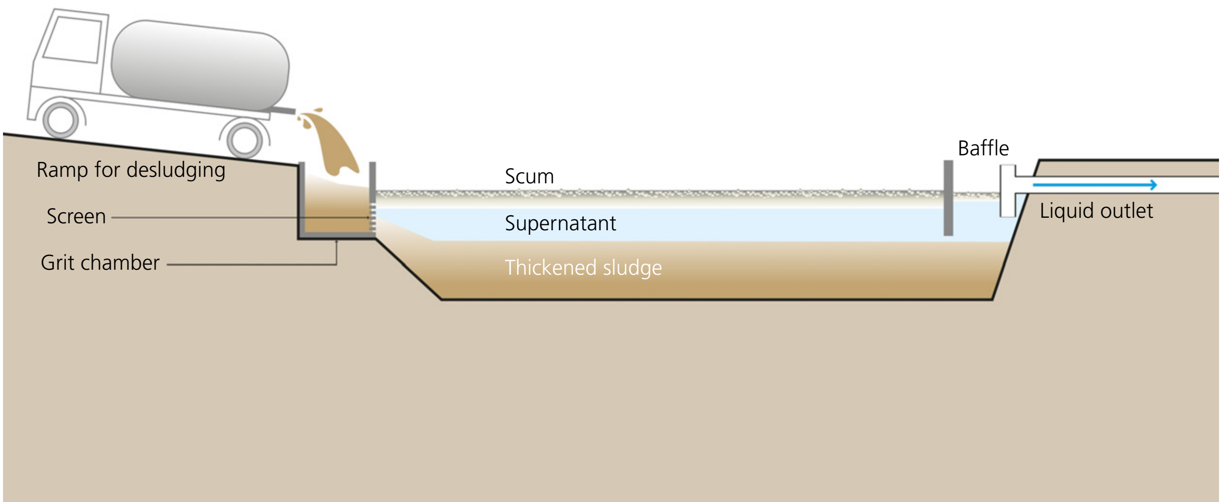

Figure 2. Sedimentation and thickening tank (source: Tilley *et al.* (2014))

applied (100–600 times the force of gravity). Basic centrifuge design includes a rotating cylindrical bowl with an interior screw conveyor. Solids content, chemical conditioning and feed rates can be varied to influence centrifuge's performance (Samal *et al.*, 2022).

Centrifugation is mainly used for sludge dewatering (after anaerobic digestion (AD) treatment) where a moist cake of 20–30% solids can be achieved (Wakeman, 2007). Raw waste is more difficult to centrifuge efficiently since the organic solids are difficult to clarify. The main disadvantage of centrifugation is the high capital cost and high-energy requirements (Ginisty *et al.*, 2021).

2.1.5 Polymer dosing

Filtration and centrifugation technologies often require prior chemical conditioning. Chemical conditioning uses inorganic chemicals or water-soluble polymers. Inorganic additives include chemicals such as lime that also acts as pathogen removal agent (see Section 2.2.2). The addition of a polymer results in the aggregation of the particles and this improves the solid–liquid separation properties (Hnamte and Pulikkal, 2022). These polymers can be natural or synthetic-based compounds that help to form larger flocs. The choice of polymer and dosage depends on the following factors (Saravanan *et al.*, 2022).

(*a*) Desired dewatering efficiency.
(*b*) Sludge/FS properties (VS concentration, electrical charge, protein and polysaccharides content).
(*c*) Dewatering devices.
(*d*) Space for storage and handling of the product.
(*e*) Cost effectiveness.
(*f*) Requirements for polymer make up, ageing and feeding equipment.
(*g*) Safety considerations.

The polymer dosage required can be determined at lab scale and must be verified in full-scale trials. Polymer–FS mixing is essential for a proper conditioning process and the mixing intensity should be high enough to achieve homogeneity in optimum contact time but must not be too high to break the aggregated material. The FS should reach the dewatering unit as soon as possible after polymer addition (Ali *et al.*, 2022).

If the raw waste should be chemically conditioned before biological processing, caution should be exercised when choosing the polymer and its effect on the downstream processes. Studies carried out on the effect of polymers on AD are few and contradictory. The conclusions vary from negative effects on the anaerobic degradation (Liu *et al.*, 2021; Wang *et al.*, 2018) to no significant effects (Cobbledick *et al.*, 2017).

2.2 Waste stabilisation

Stabilisation of FS is usually carried out by biological or chemical processes. Biological treatments of waste help its stabilisation by reducing organic matter content and pathogens. Raw sludge (FS or sludge after solid–liquid separation) that has not undergone stabilisation contains high percentage of easily degradable compounds such as sugars, carbohydrates and proteins (Mahapatra *et al.*, 2022). Stabilisation reduces the volume of the waste and produces a more stable sludge composed mainly of inorganic matter and less degradable components such as cellulose and lignin. During biological treatment, the organic matter is degraded by the action of living microbial organisms (Goodarzi *et al.*, 2022).

There are different types of biological treatments depending on the process conditions with temperature and oxygen availability being the most important parameters (Sohoo *et al.*, 2021).

2.2.1 Anaerobic digestion

AD of organic matter is characterised by the absence of oxygen with the main products being digestate and biogas.

The biogas reactors can be constructed on site or can be prefabricated tanks and they can be installed above ground or totally or partially underground (Figure 3). Biogas reactors normally operate at 35–38°C and the minimum hydraulic retention time (the time the waste remains in the reactor) is 25 days.

Biogas is a mixture of methane (55–75%), carbon dioxide (30–45%) and other trace gases. As methane is a combustible gas, biogas can be converted into heat and/or electricity. The methane content of biogas is an important parameter that influences the combustion of biogas and thus the energy recovery from the system. The energy content of 1 N m^3 of biogas is 5–7.5 kWh and its electrical output is 1.5–3 kWh_{el} and this variability in the figures depends on the methane content of biogas (Rutz *et al.*, 2015).

The main uses of biogas from AD at industrial scale are the heat and electricity generation in a combined heat and power (CHP) unit. Biogas should be dried and cleaned before its combustion in a CHP unit to remove water vapour and hydrogen sulfide. CHP units have efficiencies up to 90% with products typically being 35% electricity and 65% heat (Yin *et al.*, 2021). A fraction of this heat generated (20–40%) is needed for digester heating and the rest can be used for heating or drying applications (e.g. building heating) (Ishikawa *et al.*, 2021). Since the exhaust gas temperature from a CHP unit in biogas plants is about 450–520°C; it can be used for hot water supply and residential heating applications where the required

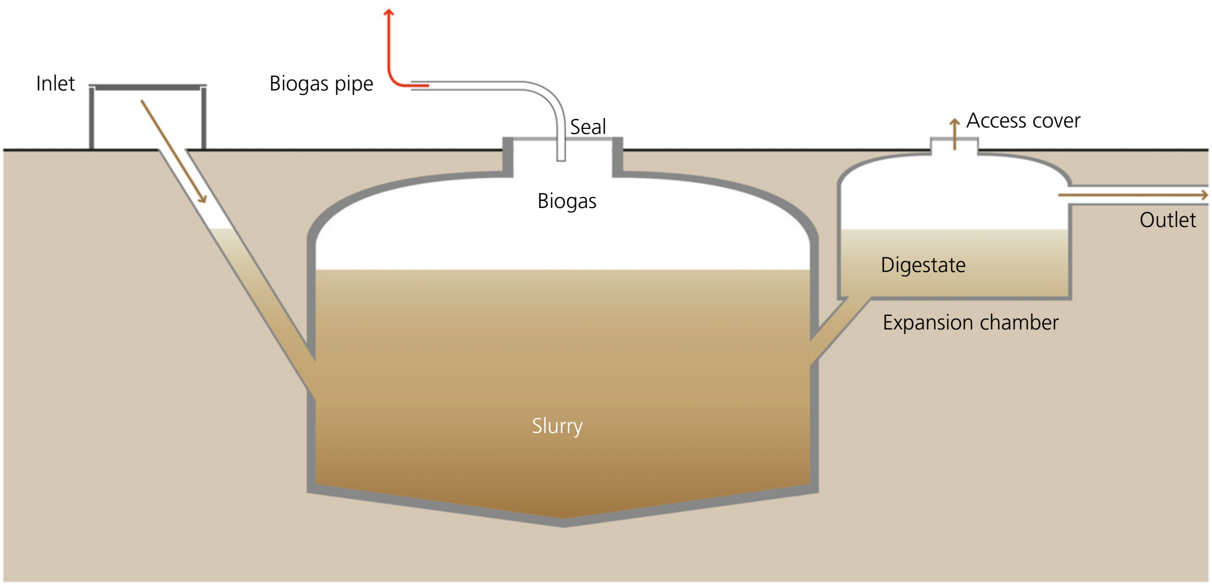

Figure 3. Underground biogas reactor (source: Tilley *et al.* (2014))

temperature is 50–80°C. Drying processes can also be sustained by the CHP exhaust gas as the temperature required is 60–150°C (El-Mesery and El-khawaga, 2022).

Since the C : N ratio is an important parameter for the degradation of the organic matter. Most authors consider an optimal C : N in the range 10–30 (Ajeej *et al.*, 2015; Rodriguez *et al.*, 2018a). As human excreta C : N ratio varies between 6 and 10, the ease of digestion can be greatly improved by co-digestion with another feedstock as a source of carbon. Food and agriculture waste such corn stover, cucumber residue (Zhang *et al.*, 2019a), switchgrass (Rodriguez *et al.*, 2017a; Zhong *et al.*, 2020), olive and grape pomaces (Aylin Alagöz *et al.*, 2018), wheat straw (Zahan and Othman, 2019), coconut, cassava and coffee ground have (Abouelenien *et al.*, 2014) a high carbon content, so it will balance the nutrient requirements in the reactor (Kim *et al.*, 2019; Rajagopal *et al.*, 2013) and improve the COD and VS removal (Wendland *et al.*, 2007; Zhang *et al.*, 2019b).

Pathogens such as *Escherichia coli* and *Ascaris lumbricoides* can be effectively deactivated by applying high concentrations of non-dissociated volatile fatty acids (VFAs) during the anaerobic stabilisation (Riungu *et al.*, 2018). Although this effect can be desirable to sterilise waste; high levels of VFA also lead to a reduction in methane potential (Rodriguez *et al.*, 2015, 2017b, 2018b).

Digestate is the slurry after treatment, and it is partially sanitised but depending on its end use, further treatment might be required. Currently, the Waste and Resources Action Programme (Wrap), a UK-based climate action NGO working around the globe to tackle the causes of the climate crisis, does not allow the certification of anaerobic digestate as soil conditioner from any kind of sewage sludge (Wrap, 2014). Non-agricultural land application could be an alternative for digestate disposal.

The storage, transport and handling of digestate results in significant cost due to the large volume and low solids content (3–10%). To reduce the costs, digestate needs further processing. Solid–liquid separation systems discussed in Section 2.1 can be used for digestate processing. The liquid fraction from digestate can be mixed with the liquid fraction from the raw waste for further processing (Section 2.3). Solid fraction of the digestate can be diverted for composting (Section 2.2.3) or can be incinerated (Section 2.2.4).

2.2.2 Chemical treatment

Alkaline additives (the most common is lime/quick lime ($CaO/Ca(OH)_2$)) are used for FS stabilisation. The addition of lime causes a rise in pH which stops the microbial activity resulting in a reduction of pathogens and odours (Li *et al.*, 2022). Reactivation of pathogens can occur over time. The lime addition can be carried out with or without solid–liquid separation, but if the lime is added to the raw waste then a higher dose is required.

Although no specific information was found for lime-treated FS, it is unlikely that it can be used as soil conditioner in countries with restrictive policies. The certification requirements are expected to be similar to those for compost (Wrap and

Environment Agency, 2007) and digestate (Wrap, 2014) although due to waste provenance, may require more comprehensive screening of pathogens in particular.

2.2.3 Digestate composting

Composting refers to the biodegradation of the organic matter under aerobic conditions (in the presence of oxygen) by bacteria and fungi. The resulting product; the compost, is a dark, friable humus-like matter (Song *et al.*, 2022). The process takes 10–18 weeks (Piveteau *et al.*, 2022); however, for the destruction of pathogens it is needed to maintain thermophilic conditions (50–70°C) over a minimum period of 3 weeks. In cold climates, the composting process should be indoors to ensure that low temperatures do not impede the microbial action (Zaman *et al.*, 2022).

Composting can be performed in outdoor piles or windrows or in an enclosed chamber. The composting chamber requires a ventilation system to provide oxygen (from air) and to allow gases produced (carbon dioxide and water vapour) to escape (Xie *et al.*, 2023). Both piles and chamber require a leachate collection system that can be mixed with the liquid fraction from solid–liquid separation (Section 2.1).

Composting is controlled by the moisture content, the oxygen supply and the carbon/nitrogen ratio (C : N). As human waste has relatively high nitrogen and moisture content, this waste can be mixed with other organic solid waste such as the organic fraction of the municipal solid waste (OFMSW) or agricultural wastes to increase the carbon content of the mixture (Manga *et al.*, 2021). This process is named as co-composting. Reported mixing ratios of FS and solid organic waste range from 1 : 2 to 1 : 3 for dewatered sludge and 1 : 5 to 1 : 10 for raw sludge (Cofie *et al.*, 2016, 2009). Chemical or biological additives could also be used to enhance the composting process (Abdellah *et al.*, 2022).

Currently, Wrap does not allow the certification of compost as soil conditioner from any kind of sewage sludge (Wrap and Environment Agency, 2007).

2.2.4 Digestate incineration

Depending on the solid contents after separation, the solid fraction of the digestate usually should be further dried before incineration (Figure 4). The water contents (by volume) should be reduced below 40% before incineration (Zeng *et al.*, 2022). If the digestate is to be dried at the premises, heat from the CHP unit can be used in a belt dryer where the exhaust gas flows over the digestate evaporating the water but a careful analysis must be carried out for the purpose of optimisation of the energy streams.

The calorific value of dried digestate increases up to 15 MJ/kg (Lamolinara *et al.*, 2022) making it an interesting fuel source for incineration plants as well as for power or cement plants. The end products are stable and in granular form, allowing easier storage and/or transport. The main constraints of digestate drying at the premises are the high-energy requirements, the potential risks of fire or explosion due to the gas and dust in the system, and the high maintenance requirements (Cathcart *et al.*, 2021).

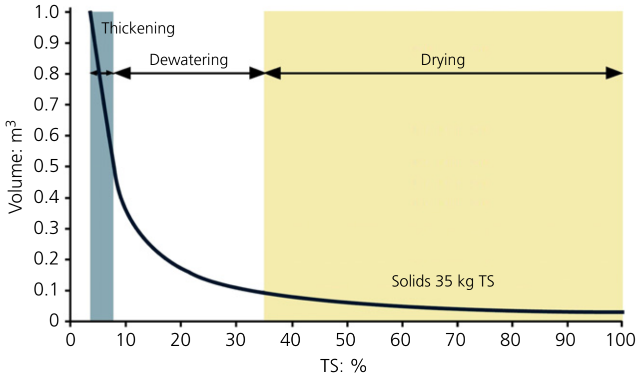

Figure 4. Volume reduction of 1 m^3 sewage sludge with initially 3.5% TS (source: Schaum and Lux (2011))

Since landfill disposal is being discontinued in many countries, the ash produced from incineration could potentially be used in construction or as cover material for dry toilets.

2.2.5 Latrine dehydration and pasteurisation system

The latrine dehydration and pasteurisation (LaDePa) technology is a new system developed by Durban municipality, South Africa (Septien *et al.*, 2018b). LaDePa converts FS into fertilising pellets of low nutrient content. Pellets are formed by extrusion then the extruded sludge passes through a heating zone to remove moisture and a radiation zone (infrared radiation) for pasteurisation (Figure 5). LaDePa process is relatively energy intensive and requires a constant source of energy. The produced pellets could be sold as fuel since the fertilisation option is not permitted in some countries. The pellets have an average calorific content of 18 $MJ/kg_{dry\ solid}$, similar to the values for wood, peat and lignite (14–25 MJ/kg) and those from sewage sludge (10–20 MJ/kg). The profitability of the system will depend on the sale price of the pellets.

2.3 Liquid fraction treatment

Liquid fractions from solid–liquid separation (Section 2.1) and digestate dewatering (Section 2.2.1) need to be further processed as residual constituents are still present. These constituents can be divided into four categories; the different categories and their effect on the environment are listed below:

(*a*) *Suspended and colloidal matter:* Affect the effluent turbidity and can shield microorganisms.

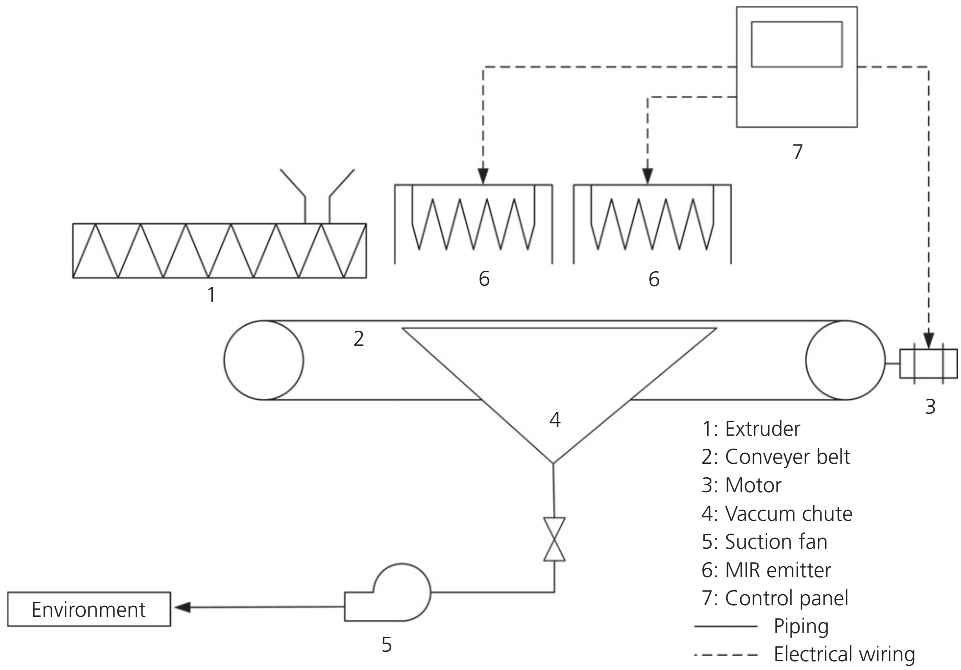

Figure 5. Schematic representation of the LaDePa pelletiser (source: Septien *et al.* (2018b))

(*b*) *Dissolved organic matter:* Toxic to humans, negative impact on aquatic species.
(*c*) *Dissolved inorganic matter:* Stimulates algal growth, increases water hardness, interferes with agricultural processes.
(*d*) *Biological:* May cause diseases to human and fauna.

There are two main types of processes for the treatment of the liquid fraction: mass transfer-based processes and chemical and biological transformation processes (disinfection). In mass transfer-based processes, the residual contaminants are transferred from one phase to another (absorption, adsorption, ion exchange, precipitation) or concentrated within a phase (membrane filtration) (Qi *et al.*, 2021). Chemical and biological processes transform or destroy trace constituents through a series of chemical reactions, usually oxidation and reduction reactions. These reactions can be carried out with a chemical oxidant (with ozone and chlorine being the most common) (Zhu *et al.*, 2022) or through ultraviolet (UV) radiation.

2.3.1 Membrane filtration

Membrane separation processes are based on the presence of semipermeable membranes. The membrane acts as specific filter that will let water flow through while it withholds the residual constituents. Depending on the pore size of the membrane and starting by the largest, membrane processes are classified as microfiltration (MF), ultrafiltration (UF), nanofiltration (NF) and reverse osmosis (RO) (Qi *et al.*, 2021). Decreasing the membrane pore size, more constituents can be eliminated from the inflow stream; however, higher pressures are needed for increasing energy requirements.

The selection of the membrane filtration system will depend on the local regulatory standards for effluent discharges to surface waters for example as set by the Scottish Environment Protection Agency (SEPA) in Scotland (SEPA, 2017, 2018). If NF or RO treatment is needed, the influent to these units should be pre-treated, with the pre-treatment usually being MF or UF process.

2.3.2 Disinfection

After membrane filtration, the liquid effluent should be disinfected to render it safe for dispersal in the environment or for reuse in different applications. Disinfection is the process used to achieve a given level of destruction or inactivation of pathogens. Disinfection is most commonly accomplished by chemical agents and non-ionising radiation. Disinfection chemical agents include chlorine and its compounds, ozone, bromine, iodine, alcohols, phenolic compounds, synthetic detergents, hydrogen peroxide and various alkalis and acids (Somers *et al.*, 2020). UV light is the most common form of non-ionising radiation used for wastewater disinfection. This review focuses on the most common disinfection methods for small-scale operation.

Chlorine is the most used chemical disinfectant and it can be used in different forms as chlorine (Cl_2), sodium hypochlorite ($NaOCl$) or calcium hypochlorite [$Ca(OCl)_2$]. The latter is

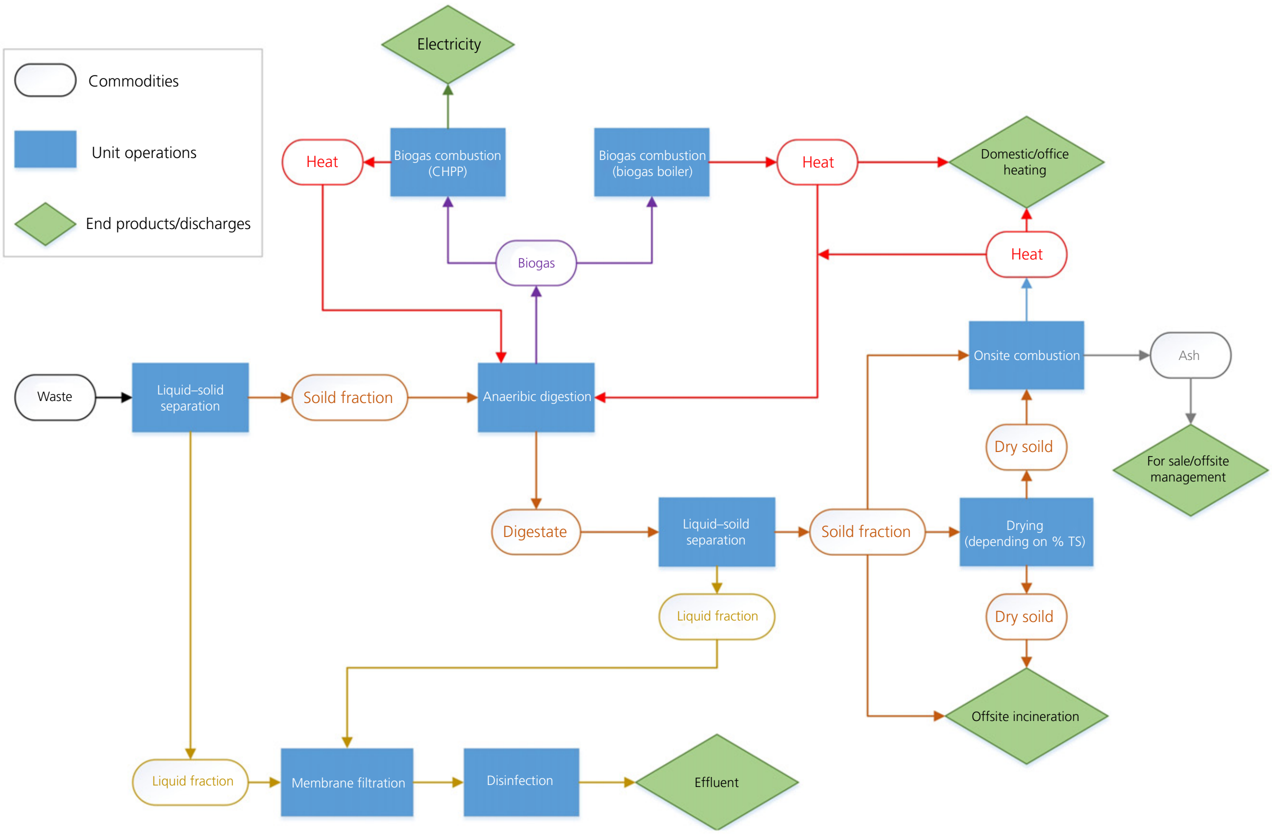

Figure 6. Proposed process diagram for small-scale treatment of human waste

recommended for small treatment plants due to its ease of operation and handling for plant operators, and its safe storage. Calcium hypochlorite is available commercially in either wet or dry form of granules, briquettes, sticks or pellets; and readily soluble in water (Yu *et al.*, 2021). The chlorine dosage depends on the influent stream pathogens concentration and the standards for effluent discharge.

UV radiation has proven to be an effective disinfectant for pathogenic organisms. UV lamps are isolated from water by housing them in a quartz glass sleeve inside the water chamber (Cuerda-Correa *et al.*, 2020).

2.4 Nutrient recovery

FS contains significant concentrations of nutrients such as nitrogen and phosphorous (Table 1) that can be harnessed for beneficial resources recovery (Wei *et al.*, 2022). However, if the nutrient recovery is not properly managed it can result in negative environmental consequences such as contamination of drinking water and eutrophication.

Theoretically, the quantity of FS produced yearly by an adult contains nearly enough macro and micronutrients to grow the quantity of food they require in a year. The simplest form of nutrient recovery is achieved through plant uptake directly from the sludge and subsequent harvesting. Further investigation is needed on non-agricultural land applications such as non-edible crops or forestry (Orner *et al.*, 2021). Deep row entrenchment for forestry applications eliminates the odours and reduces the risk of exposure to pathogens; however, there is a risk of groundwater contamination, and this application needs to be considered on a case-by-case basis depending on the area.

During membrane filtration (Section 2.3.1), ammonium, phosphate and potassium can be recovered from the liquid fraction. The major drawback for nutrient recovery from membrane systems is membrane fouling (van Puffelen *et al.*, 2022). The viability of the membrane systems for nutrient recovery in the present process will need further study alongside the commercial market opportunity for these nutrients.

3. Conclusions

The most suitable available technologies to treat waste from portable toilet and septic tanks waste (FS) on a small scale were assessed. Treatment of human wastes is highly regulated by legislation in many countries given the high content of organic components and pathogens. In conventional WWTP, sewage undergoes a series of treatment stages (primary, secondary and tertiary) before effluent discharge. The management of FS in developing countries is common but policies for waste final use or disposal are less restrictive than other more developed jurisdictions such as the UK, so further steps need to be added to liquid effluents to comply with more stringent regulations.

Combining the most suitable options reviewed above, an optimum conceptual model is proposed and is shown in Figure 6. The first step consists of a solid–liquid separation (Section 2.1), including screening to remove large solids. The use of polymer will need further investigation to study its effects on AD. The solid fraction will be anaerobically treated to produce biogas (Section 2.2.1). Biogas can be burned either in a CHP unit to produce electricity and heat or in a conventional boiler. Part of the heat generated will be used to maintain the temperature of the AD reactor and the excess can be used for domestic heating applications. Digestate will be separated into two fractions: a liquid fraction that will be further treated alongside the first liquid effluent from the first solid–liquid separation and a solid fraction of the digestate that will be used as a solid fuel. Depending on its dry matter content, the solid fraction may require further drying prior to combustion (Section 2.2.4). The liquid fractions from primary and digestate separations will be filtered in a series of steps depending on the influent composition and effluent discharge regulations (Section 2.3.1), and disinfected (Section 2.3.2) before final discharge.

Further research needs to focus on the detailed mass and energy balances to derive a more defined process design. The specific local legislation on waste treatment and discharge regulations need to be considered, as it will affect final output conditions of the various waste streams and compliance checks.

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This article is dedicated to the memory of Dr Zaki El-Hassan.

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