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Integrating recent scientific advances to enhance non-sewered sanitation in urban areas

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Half of the world's population is now served by non-sewered sanitation, yet the field remains fragmented, with a focus on individual research agendas, and prevalence of imprecise terminology that hinders scientific learnings and leads to misconceptions. The field is at a decisive juncture, with scientific knowledge taking off that holds the potential to fulfil the urgent need for inclusive sanitation in a rapidly urbanizing world. In this critical Review, relevant and diverse research results are assembled with findings translated to one consistent terminology, to provide scientific evidence to draw out interlinkages and learnings, debunk common misconceptions and identify key research needs. Properties of non-sewered wastewater are highly variable, and degradation during storage has a direct impact on greenhouse gas emissions and downstream treatment processes, which facilitate different resource recovery. New technologies and wastewater-based epidemiology can help to address the lack of monitoring. The findings are presented by wastewater properties, biological processes during storage, treatment processes and monitoring.

There has been discourse for over 20 years around a paradigm shift from sewers towards alternative options for non-sewered sanitation[1]. The reality is, the time for implementation is now, as is nearly half (46%) of the world's population is not connected to a sewer, and that number is increasing twice as fast as the population with connections[2]. In urban areas worldwide, 37% of the population is not connected to sewers, with a higher prevalence in lower- and middle-income countries (LMICs) of 60–90%, compared with 5–30% in upper-income countries[2]. Scientific and operating knowledge around non-sewered sanitation is still relatively young, but it is an exciting time of growth[3], with research that has been motivated by improving sanitation in LMICs having global relevance. Sanitation solutions that do not rely on 50-year infrastructure cycles could be more resilient to extreme weather events, reduce energy consumption and transport distances to treatment, be quicker to deploy in humanitarian settings, and increase the capacity of existing infrastructure. A multifaceted approach for globally relevant sanitation treatment chains is needed with options to select from along the treatment chains, to design solutions fitting to the specific context. However, the research remains fragmented, with organizations and research groups focusing on one technology or service provision model. If non-sewered sanitation is to be brought to the readiness level of drinking water provision, sewer-based collection and treatment facilities, and rural land-based treatment, it requires a more holistic approach establishing linkages at the interfaces of recent advances. To fill this urgent need, misconceptions and gaps in knowledge need to be immediately addressed, what is known about how systems are performing has to be established and, from there, knowledge can continue to be rapidly built up. This Review focuses on exploring interlinkages in recent scientific advances to present the current state of knowledge and identify research needs for non-sewered sanitation in urban areas. The Review is organized by the topics of wastewater properties, biological processes during storage, treatment processes, monitoring and outlook.

As illustrated in Fig. 1, complete service chains for non-sewered sanitation in urban areas are defined by wastewater flows being

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managed, the nature of containment and storage duration, and the complexities of transport to treatment[4]. One difficulty in establishing interlinkages in research and implementation is the different terminologies and understandings of terminology that are in use. In an attempt at unifying research findings, in this Review, as outlined in Fig. 1, streams of wastewater are referred to by their constituents, including excreta (that is, urine and faeces), blackwater (that is, wastewater from toilets, including cleansing materials), greywater (that is, wastewater with no excreta) and wastewater (that is, mixed blackwater and greywater), and are specified whether or not wastewater streams are stored, and if stored for short-term (<1 week) or longer-term (>2 weeks) periods of time. The term 'non-sewered sanitation' is in itself problematic, as it describes only what is not present and does not capture these intricacies, and is used in this Review as a blanket term.

Two main drivers of research include developing immediate solutions to the global sanitation crisis, with 3.5 billion people still lacking safely managed sanitation[2], and, in parallel, looking towards more sustainable, globally relevant options for the future. Currently, in urban areas of LMICs, 30–66% of the population is served by 'faecal sludge management' (Box 1), where wastewater is stored at the source of production in various forms of containments[2,5–7] (Box 2). Although this is 'non-sewered sanitation', it still relies on energy-intensive and expensive transport of liquids to treatment, most often by vacuum trucks through heavily congested urban areas[4]. Reducing volumes of clean water that are used for energy-intensive pumping of excreta through sewers or road-based transport of excreta will go a long way towards increasing sustainability. An obvious way to advance this is water-saving toilets[8]; others are treatment directly at source[9–11], or separation of waste streams such as urine[12] and greywater[13]. Another is a service provision model implemented in informal settlements that utilizes portable storage containers for short-term storage with reduced transport distances and separate collection of greywater and urine (for example, community-scale treatment)[14]. Treatment directly at the source of production, with no storage or transport, is already implemented in many locations with greywater, such as Hamburg, Germany; Hanoi, Vietnam; Helsinborg and Malmo, Sweden; Odisha, India; and San Francisco, USA[4]. Research is actively exploring whether this can be achieved at the same level with excreta and blackwater[11].

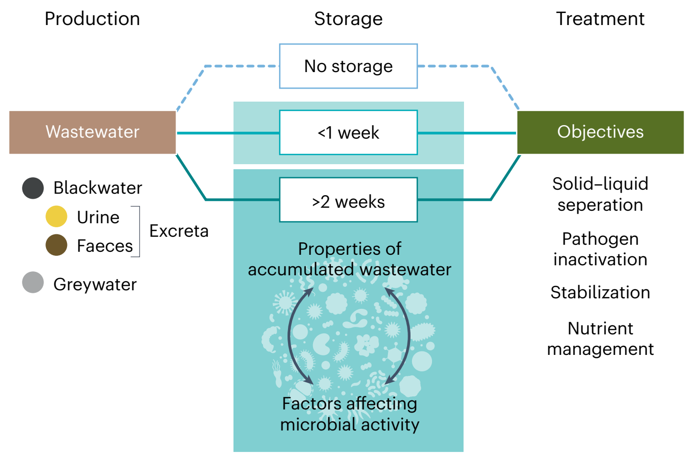

Fig. 1 | Overview of wastewater flows that are produced in non-sewered sanitation. Wastewater is either treated directly at source of production, or stored in containment for short-term (<1 week) or longer-term (>2 weeks) periods and then transported to treatment. Storage of wastewater in containment is not a form of treatment, although degradation of wastewater that occurs in storage will impact GHG emissions, downstream treatment processes and possibilities for resource recovery.

Wastewater properties

Wastewater production in non-sewered sanitation is more variable than in sewer-based systems due to the different flows of wastewater being captured, differing volumes of water usage and variations at the level of individual user. In addition, as shown in Fig. 2, what accumulates in storage and is then delivered to treatment is even more variable due to the wide range of storage and construction quality (Box 2), the biological, chemical or physical processes that take place during storage in containment (Fig. 1), and the fact that no homogenization occurs during transport, as wastewater in individual containments is typically collected and then transported batch-wise to treatment[15,16].

Excreta

Many studies have reported characteristics of urine, including nutrients (9,300 mg l^{-1} to 23,300 mg l^{-1} of urea, 470 mg l^{-1} to 1,070 mg l^{-1} of total phosphorus, 750 mg l^{-1} to 2,610 mg l^{-1} of potassium)[17], salts (1,170 mg l^{-1} to 4,390 mg l^{-1} of sodium, 1,870 mg l^{-1} to 8,400 mg l^{-1} of chloride)[17], organic fractions[18] and micropollutants[19]. Urine makes up less than 1% of municipal wastewater flows, but contains 80% of the nitrogen, 56% of the phosphorus and 63% of the potassium[17]. Although faeces is a main component of wastewater, and an essential daily life process, there is surprisingly little information available. Gastroenterology textbooks do not contain a range of characteristics of faeces from healthy individuals, and the main sources of knowledge include mass balances by NASA from the 1960s[20], characteristics specific to diseases[21] and gut microbiome studies form the 2000s[22]. A review of faeces focused towards non-sewered sanitation reports 92% volatile solids (VS) as total solids (TS), and covers ranges of microbial biomass (25–54% of solid organic matter), protein and nitrogenous matter (2–25%), carbohydrates and non-nitrogenous undigested matter (25%), undigested lipids (2–15%), and trace constituents such as secretions and inorganic fractions in faeces[23]. More recent research confirms similar ranges of characteristics with lipids (20.9% as dry mass), protein (20.1%) and fibres (27.9%), compared with non-fibre carbohydrate (1.7%)[24], and 83–86% (ref. 25), 84–90% (ref. 26) and 60–90% (ref. 27) VS of TS. Owing to the high variability in faeces, synthetic recipes have been developed for research purposes. Although they can replicate viscosity, water and VS content, adequate knowledge is lacking to reproduce dewatering performance, or chemical, biological and thermal properties together[28].

There has been a focus on the role of diet in sanitation solutions[23], but it is not supported by adequate scientific evidence. Referenced studies actually observe the same total faeces production within study groups[29], or use the construct of race as a biological category[30], which is problematic[31]. It is in general difficult to draw conclusions on the effect of diet on faeces owing to highly variable reported intestinal transit times, from 24–48 hours[32] to 40–60 hours[33], with transit time and faeces properties dependent on several confounding factors, including total water and fibre consumption, physical activity, age and body mass index[34,35]. The result is differing faeces among people eating similar diets, and classifications such as 'vegetarian' or 'low income' in reality representing a wide range of diets. Globally relevant solutions need to move past a focus on differences or implying different solutions for different groups of people, and rather focus on universal properties relevant to treatment.

Wastewater

Wastewater stored in containments is typically less than 5% TS[36,37]. Water consumption is rapidly increasing, and impacts blackwater with toilet flushing and greywater from bathing, cooking and cleaning. There is a prevalence in societal demand for flush toilets[2], with flush toilets resulting in less direct pathogen exposure to users over 'improved pit latrines'[38,39]. Increased water consumption has important ramifications on the entire service chain from safe containment, to collection and transport technologies, appropriate treatment technologies, and options for resource recovery. Reported accumulation rates of wastewater in storage (Fig. 1) range from 100 l per capita

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BOX 1

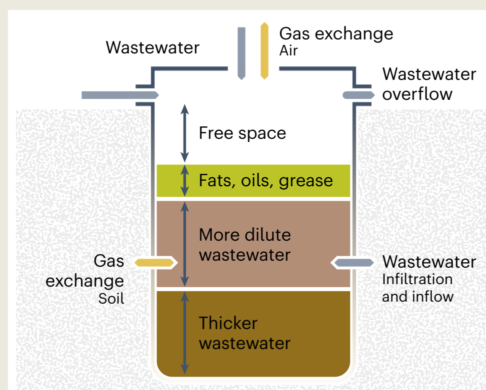
What is 'faecal sludge'?

'Faecal sludge' is a term that was coined in the 1990s to bring attention to the urgent need for improved sanitation in urban areas of Asia, Africa and Latin America. The use of 'sludge' most likely stems from the appearance of excreta that accumulates in dry pit latrines in rural areas. However, 'faecal sludge' is very widely defined as wastewater that has not been transported through a sewer. The term 'faecal sludge' has mainly been used for wastewater that is stored in containments at the source of production in urban areas of LMICs. In urban areas, containments are often erroneously referred to as 'pit latrines' or 'septic tanks' (Box 2). The wastewater ('faecal sludge') that is produced and accumulates during storage is quite dilute, as it is the result of all wastewater streams going into the containment (for example, blackwater, greywater). The resulting wastewater is partially digested, and can be a liquid, slurry or semisolid depending on inputs and losses due to overflows and leaching, and is highly variable in consistency, quantity and concentration (this definition is compiled from refs. 5–7,161). The term 'faecal sludge management', or 'FSM', refers to the service chain, and includes the storage, collection, transport, treatment, and safe end use or disposal. This mode of sanitation service provision is widespread in urban areas of LMICs countries, where it now accounts for 30–66% of coverage[2]. However, the use of 'faecal sludge' is problematic, as it has been used only in the development context of LMICs, implying that there are separate sanitation solutions for 'poor' or 'southern' contexts[4]. There is growing awareness that this special terminology has led to a number of misconceptions[4], many of which are addressed in this Review. This includes that 'faecal sludge' is mainly faeces, whereas in reality the majority of 'faecal sludge' in urban areas is <5% total solids[36,37], and that it provides a simple, low-cost solution, where in reality the management of the service chain is much more complicated than sewer-based approaches. FSM can, in theory, provide a sustainable sanitation solution. But owing to the complexities of containment and transport, the majority of faecal sludge is not yet safely managed, often resulting in spillage or dumping directly into the urban environment, resulting in significant harm to public health[141]. This Review refrains from the use of 'faecal sludge' and 'faecal sludge management', other than for clarification in comparison with literature.

BOX 2

Pit latrines and septic tanks are in fact not pit latrines and septic tanks

In dense urban areas, containments for storage of wastewater are often incorrectly labelled as 'pit latrines' or 'septic tanks'. This labelling is inappropriate, as the safe management of excreta and wastewater in pit latrines and septic tanks requires adequate land availability for treatment in engineered drain fields or controlled leaching to soils. This is possible in rural areas to the urban periphery, but is not possible in dense urban areas. The mislabelling of 'pit latrines' and 'septic tanks' implies that the wastewater is safely contained or safely managed, whereas in addition to lack of land-based treatment, they are often in reality a chaotic mixture of inappropriately and haphazardly constructed containments, with no level of standardization[50]. Safely managed sanitation requires moving away from inadequately defined labels that imply safe containment of wastewater. As illustrated in the figure, a typology for all types of containment should instead include what is more important from a management perspective, such as aspects of construction that will affect operational and boundary conditions. This includes whether containments have an overflow or not, and whether they are lined (that is, fully sealed), partially lined or unlined. Fully sealed containments (or 'storage tanks') could prevent leakage of wastewater directly into the urban environment, if a fully functional service chain is in place (FSM, Box 1). This Review refrains from the use of 'pit latrine' and 'septic tank', other than for clarification in comparison with literature, or in the case where they are used as intended in land-based treatment.



Typology for onsite containment of stored wastewater. Boundary conditions of containments for storage of wastewater in urban areas include whether containments are fully sealed or impermeable, and whether or not there is an overflow. These features are important to adequately describe the implementation of safely managed sanitation. Depths of layers within containments indicated by double arrows are meant for illustrative purposes, and could range from zero to entire containment depth.

per year up to 57,000 l per capita per year[40,41]. Accumulation rates are less than total water consumption owing to liquid fractions going directly into the environment via leaching, evaporation, overflow, spillage and dumping (Boxes 1 and 2).

As shown in Fig. 2, the characteristics of stored wastewater do not follow a normal distribution[41], and can be one to two orders of magnitude more concentrated in total and suspended solids, organic matter and nutrients, with varying levels of stabilization compared with municipal wastewater. Reported median values for stored wastewater include 800–13,600 mg l^{-1} of TS[36,40], 580–121,100 mg l^{-1} of chemical oxygen demand (COD)[40,42], 183–3,110 mg l^{-1} of ammonium nitrogen[36] and electrical conductivity of 1.8–14.5 mS cm^{-1} (ref. 42), with a general lack of reported values for phosphorus, pH and temperature. In comparison, sewer-based wastewater has 300–750 mg l^{-1} of COD[43,44], indicating it could prove more useful to look to other wastewater systems for comparison, such as paper production with high concentrations of cellulose, dredging with silt, loam or clay slurries[45], swineries with high ammonia, or the food industry where the biological oxygen demand and COD range from a thousand to several hundreds of thousands milligrams per litre of COD[43]. More detailed characterization data are becoming available in open-access datasets (for example, for over 900 on-site containments from Ghana, India, Kenya, Lebanon, Sierra Leone,

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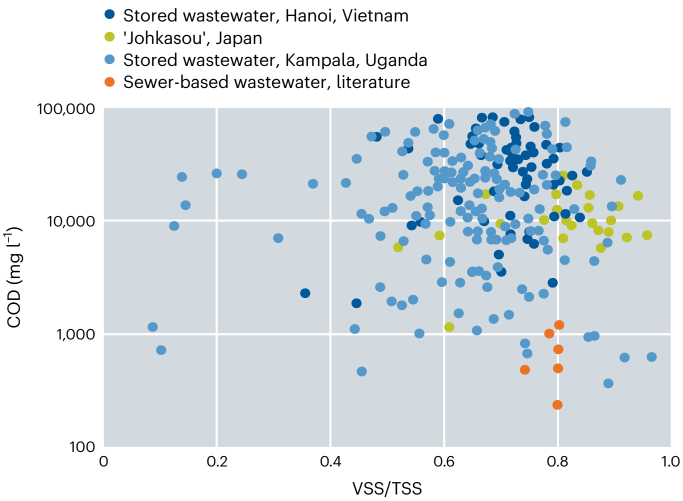

Fig. 2 | Different characteristics of non-sewered and sewer-based wastewater arriving at treatment plants. The volatile suspended solids (VSS) to total suspended solids (TSS) ratio, versus the COD values from Hanoi, Vietnam[16], five cities in Japan[36], and Kampala, Uganda[16,36] are from non-sewered sanitation. 'Johkasou' are small-scale treatment units for at-source treatment of blackwater and wastewater[135]. The influent for sewer-based wastewater treatment are compiled from the literature[44,158,159]. Figure adapted with permission from ref. 160 under a Creative Commons licence CC BY 4.0.

Uganda and Zambia[16,40,42,46–48]), and characterization is increasing to include particle size distribution[46], fibres[26,49], simple carbohydrates[49], proteins[36,49] and lipids[26,49].

The inherent variability of non-sewered wastewater greatly complicates attempts to make reasonably accurate projections of quantities and qualities (Q&Q) that accumulate in containments at community to city-wide scales[50]. Simple averages are not valid, as normal distributions are not applicable (Fig. 2), and estimates cannot be projected directly by population due to widely varying conditions. Normalizing non-sewered wastewater production or accumulation to per capita or population equivalents is also complicated by the fact that people spend most of their waking time outside the household where they sleep at night, with daytime commuting doubling the population of cities[15]. Although monitoring data mainly come from households, schools and healthcare facilities, in urban areas around 50% of wastewater can be expected to come from non-household sources, such as offices, restaurants, markets, malls, small-scale manufacturing and hotels[15,44]. People are using different toilets throughout the day, and toilet usage patterns differ from day to night, so normalizing wastewater production or accumulation by the total number of users of each toilet is not representative and overestimates per capita usage[47].

From an empirical perspective, to improve estimates at community- to city-wide scales, demographic, environmental and technical forms of data can be used to make relatively simple projection models[41,51–53], or with more advanced models[16]. It is observed that Q&Q of stored wastewater can be distinctly different for types of demographic (for example, income level), environmental (for example, groundwater level) and technical (for example, containment type) data[41]. Income level is not itself a direct cause of characteristics of stored wastewater, but with sufficient empirical data to establish statistical relationships it can be used as a predictor for groupings of Q&Q[15,41]. For example, in Kampala, Uganda, ranges of TS in stored wastewater are significantly different by the groupings income level, containment type, domestic or commercial source, flush type, or number of users[15]. Containment type, toilet type and water source (for example, on-site water connection) have been observed to be the most universal categories for predicting ranges of Q&Q[40]. In addition, public ablution blocks generally have higher water consumption (for example, 60% of water demand due to laundry, reported in Durban, South Africa[54]), and wastewater from public market toilets has higher ammonia concentrations from urine than from households (for example, 1,950 mg l^{-1} of ammonium nitrogen, compared with 1,320 mg l^{-1} of ammonium nitrogen, reported in Accra, Ghana)[55]. Improved projections are important for the appropriate design and sizing of management solutions.

Biological processes during storage

The level of degradation of wastewater occurring during storage has a direct impact on the design of downstream treatment processes and greenhouse gas (GHG) emissions. It is a common misconception that storage is equivalent to treatment, or that complete stabilization occurs during storage, or that time since last emptied (or 'storage time') is an indicator of level of stabilization, none of which hold up to scientific investigation.

Level of degradation

The majority of degradation occurs with readily biodegradable organic matter within the first one to two weeks of storage, and then levels off. Field studies confirm this; for example, an evaluation of stored wastewater from 221 containments in Kenya, Senegal, Uganda and Zambia revealed no statistical relation between soluble COD and time since last emptied, and readily degradable protein-like fractions of extracellular polymeric substances (EPS) were still present in containments that had not been emptied for years[42,46,48,56]. In Uganda and Kenya, time since last emptied was also not a predictor of particle size distribution (that is, median particle size or per cent of solids less than 10 µm)[46]. In South Africa, discernable patterns of VS or COD were not found between containments of excreta ('dry latrines'), with degradation stopping at depths greater than 1 m (ref. 57). In South Africa and Tanzania, much slower rates of in situ degradation were observed than predicted, with faster degradation occurring when samples were diluted in laboratory studies[27,58]. In Ghana, 5-day biochemical-oxygen-demand concentrations of 9,125 mg l^{-1} were observed in bottom layers of pit latrines, with a particulate-to-soluble COD ratio of 2.9 (ref. 59). Although statistical patterns for time since emptied can sometimes be observed, it does not confirm causation[60]. Laboratory studies have also confirmed field observations with conditions mimicking storage. For example, during the first weeks, a 20% VS and 30% COD reduction was observed[46,56], with EPS and aggregates originating from faeces also being broken down during this time[46,56]. The majority of reduction of mass during storage of faeces alone (total of 72% over 16 weeks) was also accounted to dehydration and not biological activity (based on energy and nutrient content)[25].

Stabilization metrics

Although reported VS:TS ratios are quite variable, a regression of 0.49 ($R^2 = 0.87$) has been reported for stored wastewater from 1,206 containments with a wide range of COD and TS (data from Burkina Faso, Ghana, India, Kenya, Lebanon, Japan, Sierra Leone, Switzerland, Tanzania, Uganda, Vietnam and Zambia)[40]. It is not known whether degradation of fractions of organic matter stop during storage due to rate-limiting conditions[27], such as high TS, volatile fatty acids, ammonia inhibition or other constituents[61], or whether humics and highly processed biopolymers are building up due to long retention times. Although commonly used metrics of stabilization such as the VS:TS ratio and the carbon-to-nitrogen ratio can provide information on the level of readily available organics that have been degraded, they are not useful for comparison of wastewater stored in containments beyond this, due to the wide range of degradable to non-biodegradable or inert organic matter making up pools of total organic matter when measured as VS, COD or total organic carbon. The TS in stored wastewater can also contain varying amounts of inert inorganic matter from soil or rubbish. Regionally specific correlations of COD:TS ratios are sometimes observed[7] and sometimes not[40], and the VS:TS ratio is not a predictor of biogas production[62]. Metrics that are closer to potential for biological degradation such as soluble COD, biomethane potential, soluble oxygen uptake rate, biological oxygen demand or respirometry, will probably be more accurate than total pools for understanding GHG emissions and designing treatment. Another possibility is colour, which is observed to change from light brown–green to more dark brown–black with stabilization[42] (Box 3).

Microbial communities

So far, there have been only a handful of studies looking at microbial communities in stored wastewater in Canada[63], Malawi[64], Senegal[48], South Africa[58,65], Tanzania[48,66,67], Thailand[68], Uganda[48] and Vietnam[67], and in comparison with faeces[26]. It is generally thought that aerobic degradation takes place at the very thin surface layer in containments, with anaerobic pathways of degradation otherwise occurring[69,70] (Fig. 3). However, anoxic conditions[71,72], and the presence of both aerobic and anaerobic bacteria together, have been observed within containments[46,58,67]. A wide range of oxygen reduction potential is observed, including −247 mV to 65.9 mV in storage of wastewater ('pit latrines') in Uganda[71], and −600 mV to −200 mV in storage of blackwater ('septic tanks') in Vietnam[60,73]. A study in rural Tanzania found a linear gradient of beta diversity with increasing depth in containments, with a shift in microbial community structure at the taxonomic level of family from gut-associated families in the top layer to environmental- and sewer-based wastewater-associated taxa at greater depths[66]. However, in a study in Malawi focusing in at the genera level, 55% of the microbial communities in containments were unique compared with municipal wastewater systems and human gut microbiomes[64], with a change in metabolic pathways but not overall microbial communities with depth[64]. Microbial community composition can be expected to change with zones of active, slow and passive pools of available organic matter, potentially with microbial communities in deeper regions being more established and adapted, and the upper regions characterized by more dynamic changes[74,75]. In addition to microorganisms that carry out enzymatic reactions, the role of invertebrates, worms and fungi in the cycling of nutrients and organic matter during storage cannot be overlooked.

Conceptual model

Microbial activity taking place during storage cannot be compared with the stabilization processes occurring at sewer-based treatment facilities[76], where process control includes mixing, recirculation, redox conditions and temperature, for the maximum stabilization of organic matter and nutrients in the smallest possible areas[77]. As proposed in Fig. 3, improved models of degradation with no-mixing, continual feeding with new substrates and no outlet (of solids) are needed. Potentially the exchange between pools of soil organic matter could provide a more useful model. Soils are also continuously fed (with diverse inputs of organic matter), with very long retention times and no outlet (of solids). Similar to storage of wastewater, the top layers of soils are expected to be aerobic to microaerobic[78], with 'deep soils' more than 20–30 cm below the surface containing highly processed organic matter[79]. It is being challenged whether humus fractions in deep soil are composed of complex, long-chain molecules, or rather a mix of shorter-chain biopolymers[80]. Organic matter becomes non-available for degradation when 'locked' up inside of aggregates, bound to minerals or aggregates limiting enzymatic activity, or as complex, stable organic matter[79]. 'Priming' occurs as labile or available organic matter and nutrients are delivered into the existing ecosystem at a rate that depends on substrate quality, number of enzymatic steps and redox conditions[81]. Substrates and nutrients that were previously limiting then become available, allowing microorganisms to degrade the stable or 'locked' organic fractions in the 'deep soil', analogous to what could be expected during storage in containments.

BOX 3

Harnessing the power of new technology

Rapidly advancing new technologies such as 3D-printing, open-source microcontrollers and low-cost sensors, are opening up possibilities for low-investment, high-accessibility solutions[162]. The internet provides a space where specialized communities can work together for the design of globally relevant applications. Localized small- and medium-sized enterprises can utilize and apply global knowledge through initiatives such as free open-source hardware and do-it-together strategies, which aim to democratize and distribute the design and production of technology[163]. In manufacturing, small- and medium-sized enterprises can further develop and adapt technologies to local contexts. The use of openly available knowledge for local production can help alleviate problems with international supply chains such as cost, availability, and ability to adapt and repair, with distributed production at the point of usage[164]. To advance the power of new technology, scientists and publishers need to commit to open-access publishing, and to openly sharing all relevant data, including information on experimental set-ups (for example, bill of materials, CAD drawings, software, calibration and validation). Scientists and practitioners also need to make a strong commitment to calibration and validation, which is required for adequate scientific evidence for the implementation of open-source hardware.

Self-made sensors are proving to be useful for monitoring in river research[165], and low-cost sensors and machine learning have been validated for prediction of microbial water quality and to ensure compliance for the decentralized reuse of treated greywater[166,167]. Another example is the Sludge Snap App, which uses colour and texture data from smartphone pictures to estimate characteristics and dewatering performance of stored wastewater. The app uses a machine learning model that was built from pictures and laboratory data from stored wastewater in 421 containments in Lusaka, Zambia, and could reduce time and costs of expensive laboratory methods and reliance on chemicals. Technology advances could lead to the use of low-cost sensors for real-time monitoring or process control of decentralized, small-scale treatment solutions. However, obvious limitations need to be addressed. Any type of reasonably accurate model requires high-quality data acquisition, including rigorous quality assurance and quality control. The massive amount of data generated by sensors needs to be cleaned and stored. Measuring instruments based on open-source design must be calibrated and validated, which not all laboratories are equipped for[162]. This is the case with solutions such as the Sludge Snap App, which, although validated for Lusaka, cannot be transferred to other locations without the required expensive collection of data for model validation.

Treatment processes

An overview of the research in this Review in relation to treatment is presented in Fig. 4. Although research groups often focus on different stages of treatment, by mapping them in one framework their interrelatedness becomes clear. Wastewater streams that require treatment will be directly impacted by usage patterns, storage and service provision models that reduce volumes of water consumption, or separately manage different wastewater streams. These then have a direct impact on treatment technologies and facilitate different forms of resource recovery.

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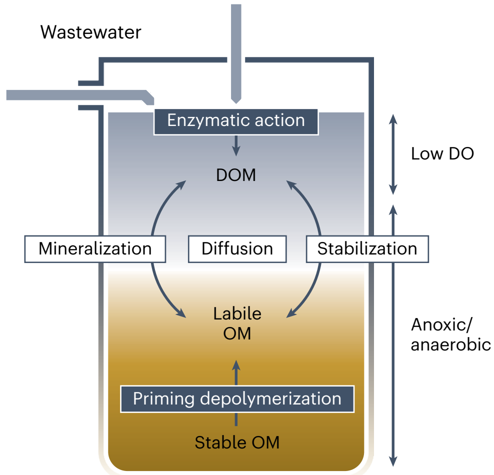

Fig. 3 | Conceptual model for biological processes occurring during storage of wastewater in containments. During storage there are continuous inputs of fresh organic matter and no output (of solids). Enzymatic actions release dissolved organic matter (DOM). The relative location or depth of hydrolysis are illustrative, as the actual depth of reactions are not yet known. Levels of dissolved oxygen (DO) are thought to rapidly decrease from the surface layer. The input of labile, or readily degradable, organic matter (OM) and nutrients results in priming with subsequent degradation of fractions of stable OM.

Wastewater streams requiring treatment

The level of stabilization of wastewater is relevant for dewatering, and whether treatment and resource recovery can take advantage of readily available organic matter. For example, anaerobic digestion and biogas production is most suitable for wastewater and faeces that have not been stored, or stored for <1 week, such as with portable containment[14,62]. Laboratory studies comparing anaerobic digestion of fresh faeces and stored wastewater confirm an order of magnitude higher methane yield with short-term storage of wastewater (<1 week)[61,62]. This is due to not only the availability of readily degradable organic matter but also the fraction of COD remaining as lignin, cellulose and hemi-cellulose that will not be biodegraded[82]. This is also confirmed in field operating conditions, where mesophilic anaerobic digestion of wastewater that had undergone longer-term storage achieved less than 40% COD removal for treatment of stored wastewater in Tanzania[83]. Biogas production could be increased with anaerobic co-digestion of other organic waste streams, or with physical, chemical or thermal pre-treatment, but it needs to be evaluated whether additional costs and complexities make them an attractive option[84]. Biological treatment with black soldier flies or worms is another way to harness readily available organic matter for biomass production from faeces, wastewater treated at source or with short-term (<1 week) storage before treatment[24]. Vermicompost (for example, from tiger worms) can effectively treat blackwater directly at the source of production[10], although further research is required to establish reliable nutrient and pathogen concentrations in liquid streams following treatment[10].

Separating waste streams

Separation of urine, faeces and greywater can result in smaller volumes of less contaminated water, enhance resource recovery and simplify subsequent treatment steps. Scientific knowledge around the treatment and resource recovery from separately collected urine is well established, including struvite (magnesium ammonium phosphate) precipitation, storage, acidification, alkalinization, nitrification and ultraviolet oxidation[12,85]. In addition to facilitating resource recovery of nutrients, separate collection of urine could alleviate problems such as the need for nitrogen removal, and biological inhibition in storage or treatment due to high ammonia concentrations. However, management of trace contaminants and pathogens should not be overlooked[86]. Treatment of blackwater with vermicomposting[87] also benefits from more carbon-rich wastewater. The separate management of greywater opens up possibilities for nature-based solutions[13], and on-site water reclamation for toilet flushing, bathing or hand washing at the household or building scale[88]. However, there remains a need for relevant water reuse frameworks and monitoring to co-evolve with emerging small-scale technologies[88].

Improved dewatering

Dewatering is defined as the solid–liquid separation of wastewater, and entails the removal of unbound water[7]. Dewatering performance of stored wastewater measured as filtration and turbidity is directly linked to level stabilization, with highly concentrated wastewater with very short emptying intervals not dewatering well[89]. Specifically, dewatering improves following 1–2 weeks of storage, and then levels out as with overall changes in degradation during storage[26,46,48,56]. Semi-centralized drying beds have been the cornerstone of treatment facilities, often preceded by settling-thickening tanks[6,90]. Incremental improvements are steadily being made, such as covers to increase solar drying[91], or optimizing macrophyte selection in planted drying beds[92]. However, these technologies still require transport of liquids, are land intensive for dense urban areas, and performance remains unpredictable. Clogging of filters (drying beds) and insufficient removal of suspended solids in supernatant (settling-thickening tanks) remain more problematic than the fractions of bound water associated with solids, with or without storage[48,93]. For example, even faeces (macerated, no storage) can reach cake solids of ~50% TS following 5 to 550 hours of settling, with cake compression behaviour more similar to sediment slurries than municipal wastewater sludge[94,95].

Investigations into EPS, particle size distribution, fibres, lipids and cations are shedding light on governing mechanisms, and how stabilization affects dewatering. Reported concentrations of EPS include 25–442 mg l^{-1} in stored wastewater (‘faecal sludge’ with a wide range of emptying frequencies), 340–553 mg l^{-1} in fresh blackwater (‘faecal sludge’ that has not undergone storage) and 465–1,308 mg l^{-1} for faeces (no storage), compared with 173 mg l^{-1} for anaerobically digested sludge from a sewer-based wastewater treatment facility[26,46,48,56]. The EPS in stored wastewater is composed of a higher proportion of humic-like substances than in sewer-based systems[48,56], and acts as colloidal material, resulting in more suspended particles, clogging of filters and increasing turbidity[48,56]. EPS does not increase aggregate formation or water binding during storage, and the behaviour of aggregates in stored wastewater is not comparable to floc behaviour in activated sludge[46,48]. The destruction of EPS in stored wastewater increases hydrophobicity and decreases bound water content, sludge viscosity and compressibility[96]. As EPS is degraded, suspended small particles are released, which can also be drivers of poor dewatering, with worse filtration performance observed with higher concentrations of true colloidal particles (<10 µm)[46]. However, overall performance is still improved with stabilization and destruction of EPS. The particle size distribution of stored wastewater is much broader than sewer-based wastewater sludges (that is, anaerobically digested waste activated sludge, anaerobically digested primary sludge and waste activated sludge), with higher proportions of very large (>300 µm) and very small (<10 µm) particles[46]. Increased lipids increase turbidity, and increased concentrations of fibres improve filtration, reduce turbidity and increase cake solids[26]. Wastewater that has undergone storage has greater concentrations of simple carbohydrates, lignin, cellulose and hemi-cellulose than sewer-based wastewater, and lipids (oil, grease)[97], possibly due to bio-concentration following degradation of more readily biodegradable compounds. Protein concentrations of stored wastewater have a similar range to digested primary wastewater sludge[97]. The role of mono- and divalent cations in dewatering performance is not

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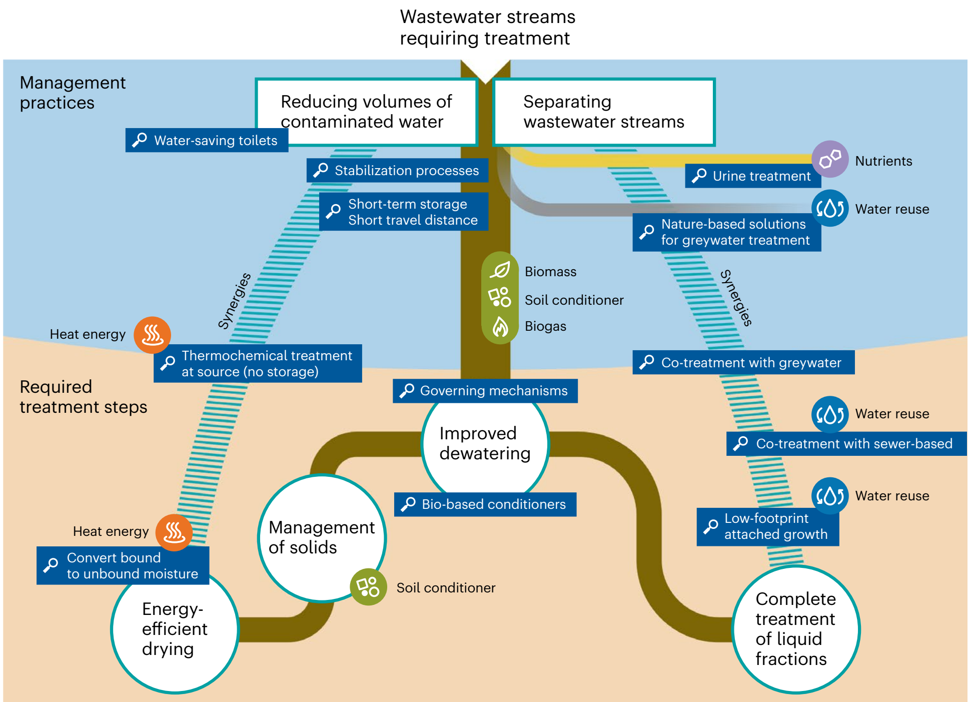

Fig. 4 | Landscape illustrating interrelatedness of research covered in this Review. Findings of research focused on different parts of the service chain are interrelated, when viewed cumulatively the synergies become apparent, including management practices that affect wastewater flows and have relevance to downstream treatment. The resulting wastewater streams and appropriate treatment technologies will more readily facilitate different forms of resource recovery.

yet clear, but the divalent cation bridging mechanism that decreases turbidity in municipal wastewater[98] is not occurring with wastewater that has undergone storage[26,48].

Fast and easy to implement metrics of stabilization could be used to identify which wastewater (for example, <1 or >2 weeks storage) requires pre-stabilization for improved separation of particulate and colloidal matter, to reduce clogging and supernatant turbidity. Dewatering of stored wastewater with less EPS and smaller particles can be improved with the use of conditioners (coagulants and flocculants)[99–103]. The use of conditioners opens up the possibility for low-energy solutions such as manually operated mechanical dewatering, which greatly increases dewatering performance. Smaller-footprint, more efficient treatment facilities could lead to community-level treatment in dense urban areas, which could greatly reduce transport distances. Remaining research hurdles for the reliable implementation of conditioners includes real-time optimal dosage to highly variable wastewater, to avoid under- or over-dosing (Box 3), understanding of the resulting floc strength and resistance to shear, supply chain issues and costs of conditioners, and environmental impacts. Bio-based conditioners are quite promising with equivalent settling performance to synthetic, commercially available products[102,104], but resulting floc strength appears to be weaker. Another approach is co-treatment, for example, with sawdust, which absorbs water and serves as a skeleton builder or filter aid[105].

Complete treatment of liquid fractions

Following dewatering, the complete treatment of remaining liquid fractions is a critical but neglected research topic. Reported concentrations of organic matter and nutrients are as high or higher than municipal wastewater influent, for example, as reported for drying bed leachate 840–8,620 mg l^{-1} of TS, 3,600–6,701 mg l^{-1} of COD and 640–890 mg l^{-1} of ammonium nitrogen[89,106,107], and frequently elevated levels of salinity (reported as electrical conductivity)[103]. Improved solid–liquid separation also opens up the possibility for more efficient and smaller-footprint treatment of liquid streams. For example, following conditioning and settling, the COD in supernatant is mainly present as soluble COD, with removal of colloidal fractions still to be quantified[104]. In sewer-based wastewater treatment, effluent total suspended solids requirements are often met via the colloidal fraction being readily enmeshed with flocs[108], but it is not known yet for stored wastewater what the hydrolysis mechanism is or how colloidal matter is enmeshed with solids.

Where sewer-based wastewater treatment is also in place, co-treatment of supernatant following dewatering of non-sewered wastewater is possible. However, there are not many implementations to draw on[6], and fractionations of organic matter for design and modelling purposes of co-treatment remain purely theoretical[109]. It is clear that based on the humic, protein and polysaccharide breakdown of stored wastewater, COD fractioning is expected to have distinct patterns or 'fingerprints' from municipal wastewater treatment models[110].

Although respirometric characterization in total, settleable, colloidal or soluble fractions of wastewater following storage has not yet been reported, blackwater with no storage (for example, cruise ships, vacuum flush toilets) has a greater fraction of settleable COD, and the hydrolysis mechanism is 40% slower than the hydrolysis of slowly biodegradable substrate in sewer-based wastewater[111–113]. There is need for further research into fractionating the soluble COD and potentially adjusting effluent requirements, as biological oxygen demand <30 mg l^{-1} can reliably be met with liquid streams, but limits of COD < 100 mg l^{-1} cannot[83,114], which could be due to higher concentrations of less biodegradable compounds such as lignin and cellulose in stored wastewater[97].

Management of solids

There is a general lack of guidelines on what to do with the remaining solids following dewatering, with a focus mainly on end use or disposal, which includes soil conditioner or deep row entrenchment[6,90] (see 'Monitoring' section). Although co-composting is well established, detailed knowledge of pathogen die-off and scientific studies remain rare[115,116]. A huge gap is consideration of trade-offs in selecting end use and resource recovery options from solids that need to be factored in, such as comparison of GHG emissions, displaced nutrients (for example, nitrogen, phosphorus and potassium), chemical fertilizers and overall soil health[117].

Energy-efficient drying

Further drying of bound water in dewatered solids can inactivate pathogens[118], and resource recovery as a dry fuel or as a soil amendment can take place with carbonized or non-carbonized processes[119,120]. Key to optimizing these processes is an improved understanding of bound water and water binding strength. Research is showing that following the removal of free water, the remaining moisture and drying kinetics are similar, regardless of the type of containment and categories of wastewater (for example, urine-diverting dry toilet, septic tank, improved pit latrine)[121]. Research into low-cost, low-energy methods to release the remaining bound water for drying have revealed that the kinetics of pellet drying fit the Page model, adapted from the food industry[122]. And that solar and wind energy can provide adequate heating and air convection for the drying process[123]. For carbonization, pyrolysis requires more pre-drying than hydrothermal carbonization[120], and to optimize resource recovery from pyrolysis, faeces and urine should be collected separately[124]. Co-management with agricultural and food waste also improves the quality of end products, including fixed nitrogen and carbon[119,125]. Reported ranges for calorific value and ash content for faeces include 17.2–24.7 MJ kg^{-1} and 7.5–18.3% dry weight, and for wastewater that has undergone a range of storage conditions 8.3–19.1 MJ kg^{-1} and 15.7 to 58.5% dry weight[120]. Calorific values suggest viable use as a fuel, but are limited by a high ash content[119]. Another technology transfer from the food industry is the potential use of microwave-assisted drying for emergency response or humanitarian settings, which greatly reduces the sludge volume, and maintains nutrients in the dried sludge (80–84 mg g^{-1} of total nitrogen TS, 26–27 mg g^{-1} of total phosphorus TS)[100,126].

Reduced volumes of contaminated water

Mechanical, physical and thermochemical processes, including liquid–solid separation, hydrothermal carbonization, combustion and electrochemical treatment, could also potentially be employed directly at the source of production of faeces, excreta and blackwater, and would greatly reduce volumes of contaminated water and the need for transport[11]. Recent research in this area has led to an improved understanding of pyrolysis[127], combustion[128], gasification[129] and hydrothermal oxidation[130] of faeces. Here there are clear synergies in knowledge advancement with energy-efficient drying, and supernatant treatment, which is applicable for complete treatment of liquid fractions. For example, electrochemical treatment for supernatant from anaerobic storage of macerated toilet waste[131], and thermal distillation of excreta with different membrane configurations[132]. Other research for at-source treatment of blackwater (flush toilets) includes solid–liquid separation through diversion and settling, followed by filtration through activated carbon with disinfection by electrochemical oxidation[133], and electrochemical oxidation of residual organic and inorganic constituents with disinfection via in situ chlorine generation[131]. Field testing has indicated that performance would be improved with biological pre-treatment (anaerobic and aerobic)[131], linking to learnings on stabilization during storage and governing mechanisms of dewatering. Other process flows with biological treatment for stabilization include an anaerobic membrane bioreactor, followed by nutrient capture with ion exchange and carbon sorption, and electrochlorination of effluent[9]; and biological oxidation, nitrification and partial denitrification in a sequencing batch reactor, followed by electrochemical oxidation of supernatant[134]. This research has undergone proof-of-concept field testing, but is not yet ready for full-scale implementation. Treatment chains including biological treatment are more similar to the small-scale treatment at-source units employed in Japan, termed 'Johkasou'[135].

Monitoring

There is a general lack of information and monitoring on the removal of pathogens in non-sewered sanitation. It is a common misconception that pathogen inactivation occurs during storage of wastewater, whereas in reality storage is not treatment and enteric pathogens are reported to not vary with depth in containment[136], and viable helminth eggs are reported following years of storage[7]. Design guidelines for treatment following storage also typically do not contain information on expected pathogen removals, with a focus rather on getting solutions in place based on management of organic matter and nitrogen, and operations and maintenance[6,114,137]. In existing treatment facilities, consistent pathogen removal during treatment processes is also not observed[114]. Owing to a lack of monitoring data, for resource recovery or end use, a multiple barrier, risk-based approach to exposure is therefore generally recommended[90].

Safely managed sanitation

Ensuring excreta flows are safely managed requires collecting systematic data to measure and monitor adherence to guidelines. One way to fill this gap is rapidly increasing possibilities for implementation of real-time operation and monitoring data with advances in new technologies (Box 3). However, this needs to be balanced with a realistic evaluation within the realities of high rates of failures of sanitation implementations[138]. Public health risks of exposure from inadequately managed sanitation can also be monitored directly in the environment. For example, the GloWPa-Crypto C1 global model of *Cryptosporidium* concentrations in rivers has identified that point sources from human faeces are the dominant source of oocysts worldwide[139]. Methods for monitoring exposure risks and identifying areas for priority interventions in dense urban areas include SaniPath, with environmental sampling and behavioural exposure data combined in a quantitative microbial risk assessment[140], and HyCRISTAL, a flood model that incorporates faecal waste contributions[141]. Evidence-based public health tools are informative for identifying areas for priority interventions, or to evaluate effectiveness of an intervention. Learnings could also be applied for real-time community-level epidiomological monitoring. Wastewater-based epidiomology has been effective in monitoring for pathogens or drug-use prevalence at the community level, with a rapid expansion for the detection of SARS-CoV-2 during the COVID-19 pandemic[142]. However, the focus of wastewater-based epidiomology has been in sewer-based systems, and usefulness in non-sewered areas has not yet been demonstrated. A study in Malawi has validated that bacteria and protozoa in containments ('pit latrines') are useful for community monitoring, but not for tracking of individual health due to different survival rates of pathogens in storage[39]. However, community monitoring at the level of individual containments would be too resource intensive and invasive. It is known based on SaniPath studies, that the most important exposure points to untreated wastewater in urban areas are often street drains, floodwaters, street food and uncooked produce[140,141,143]. In which case, environmental monitoring could be used for community-level epidiomology. Potentially, informal drainage channels will be as effective for monitoring as treatment plant effluents, and could be scaled to estimate disease levels in catchment populations with topological maps, drainage patterns[144] and Q&Q methodologies[41]. Studies in Bangladesh and Thailand have also indicated that monitoring from hotspots such as public markets would be effective[145,146]. Transmission of antimicrobial resistance could also be similarly monitored in urban areas[147], and with advances in portable, faster, less-expensive molecular methods, robust diagnostic field testing is becoming a possibility for resource-limited settings[148].

GHG emissions

Although emissions from sewer-based municipal wastewater treatment are relatively well studied, values cannot be directly transferred to non-sewered systems due to the different service chains, treatment technologies and varying degradation in on-site storage[149]. GHG emissions emanating from storage of wastewater in dense urban areas is directly tied to the level of degradation occurring during storage, but only a handful of studies have tried to estimate GHG potential[60,73,150,151]. In situ measurements in Vietnam and Ireland indicate that the Intergovernmental Panel on Climate Change (IPCC) methodology greatly overestimates GHG emissions[60,73,152]. The IPCC values for emission factors have not been updated with current learnings, and assume a much higher level of anaerobic degradation during storage than what is in actually taking place (see 'Biological processes during storage' section). Scaling emissions for city-wide estimates of GHG emissions are exacerbated by the same challenges facing estimates for Q&Q in general, and the developed methodologies for projection models can also be applied for GHGs (see 'Wastewater properties' section). To scale estimates for GHG emissions in Kampala, Uganda, one study estimated the total number of toilets and normalized that by the estimated daytime population, assuming equal usage of all toilet types[151]. A typology of containments was used based on whether they were lined or unlined (Box 2), and the water content[151]. However, the assumptions for methane correction factors and emission factors were much higher than what has been quantified with gas measurements in field studies[60,73,152], and were as high as maximum laboratory values observed in biomethane potential tests at 37 °C with mixing and an inoculum (0.01–0.0012 g g^{-1} of methane COD)[62]. It is also not known whether the readily biodegradable organic matter that is degraded during the first 1–2 weeks of storage is taking place in aerobic, anoxic or anaerobic zones, which will affect levels of carbon dioxide, nitrous oxide and methane emissions. To make more reliable estimates and develop mitigation strategies, field monitoring and laboratory measurements are needed to understand the volumes of GHGs being emitted, and how values from storage of excreta[150], blackwater[60,73] or wastewater[152] are comparable. Very short-term storage followed by aerobic treatment, or anaerobic treatment that captures methane, could offset emissions, but increased road transport would need to factored in. A mass balance in Haiti based on short-term storage in portable containments with urine separation followed by co-composting, demonstrated that this model would substantially mitigate sanitation-related emissions[150]. Another possibility could be on-site aerobic treatment (for example, vermicomposting), where carbon dioxide is generated rather than methane, or separate management of urine, which could potentially reduce nitrous oxide emissions. Although organic matter is undergoing less degradation during storage than previously thought, it is not fully understood at which level it becomes available for degradation when emptied, mixed and exposed to different environmental conditions[27].

Outlook

Global goals for safely managed sanitation will not be met in dense urban areas with one technology solution alone, be it sewered or non-sewered[153]. A knowledge base of diverse solutions needs to be built up, with multiple technologies and processes along different treatment chains. Engineering design approaches based on treatment objectives for resource recovery, taking into consideration the specific characteristics of wastewater streams, can then be applied in a modular fashion to assemble solutions that are adapted to the specific context. It is a key moment in time, where rapid advances in scientific knowledge are starting to take off, as they are crucially needed to fulfil climate-resilient sanitation solutions in rapidly growing cities. It is critical at this juncture that the sector commits to a unified and scientifically robust terminology that is descriptive of wastewater constituents, storage conditions, transport to treatment[4], and treatment processes and objectives, to ensure the accurate translation of findings. This includes moving away from misconceptions that wastewater from human excreta are all the same, as it is well established that properties of non-sewered wastewater are diverse and different to that of sewer-based wastewater. It also includes moving beyond a focus on individual solutions, research agendas and funding mandates, to synthesize scientific and engineering knowledge and develop a cumulative understanding and approach to scientific learnings. In addition, uninformative labels, such as 'faecal sludge management', 'dewats' and 'eco-san' must be surpassed, as they hinder progress and lead to misconceptions, such as storage being a simple solution that is analogous to treatment processes[4]. Technical advances will not deliver effective urban sanitation without an integrated approach to knowledge generation and sharing. Standard methods of analysis need to be developed so that results are comparable[7], and then similar metrics need to be compared, such as metrics of stabilization (for example, biomethane potential, soluble oxygen uptake rate, biological oxygen demand) and dewatering performance (for example, filtration, settling, water holding capacity). Reliable approaches for scaling estimates of Q&Q of non-sewered wastewater production and accumulation to city-wide levels are needed for planning and management purposes, including GHG mitigation. The development of reliable guidelines for design and operation, including pathogen removal, will require continued trials in field-testing platforms[131,133], and risk-based piloting approaches with municipalities and research institutes working together[118]. Sanitation is not a standalone solution, and requires intersectoral thinking, with management of solid waste together with complete urban water flows and cycles[3]. It is well known that rubbish can clog and disrupt processes, and that flooding and runoff are significant environmental pathways of exposure impacting public health in urban areas[141].

Ideally, community-scale treatment solutions implementing recent scientific advances and new technology will be more climate resilient, as they are less reliant on road- or sewer-based infrastructure for transport to treatment. However, the realities of implementation are that sanitation systems have very high failure rates due to lack of municipal engagement, community buy-in, unaddressed sanitation priorities, and inadequate resources and technical support for operations and maintenance[138]. Engineering solutions cannot be developed outside the context of realities of implementation, as it results in over- or under-designed systems that fail. Research developments have to be considered in the context of clear governance (roles and responsibilities), institutional support (technical and financial), systematic policies and monitoring, public awareness and community engagement and service provision models[3]. These difficulties are endemic, and are not unique to LMICs, as seen by the lack of adequate or equitable water supply and sanitation provision in high-income countries around the world[154], and problems with ageing sewer-based infrastructure such as in the United States. Technology alone is obviously no solution, and how to fulfil these remaining gaps blocking sustainable technology implementations is the greatest obstacle remaining to be overcome.

Inadequate sanitation is a global problem, not a rich or poor problem, directly linked with challenges to global health that are currently being faced with climate change, species extinction, antimicrobial resistance, rapid urbanization and pandemics. Sanitation is already globally directly impacted by climate change in coastal areas, flood-prone areas and water-scarce cities. Addressing these challenges requires local government action, together with collective action, as they cannot be overcome within individual political boundaries. Although soon over half the world's population will be served by non-sewered sanitation, the vast majority of engineering curriculum in high-income countries still focuses solely on sewer-based solutions. There is a wealth of experience in non-sewered sanitation in urban areas of LMICs, but the majority of the engineering literature that is published focuses on high-income contexts[155], indicative of the lack of authorship equity in science, and the obstacles faced by researchers and practitioners in LMICs[156]. The equitable involvement of globally representative researchers in knowledge generation will enable impartial and relevant solutions, which will more readily bridge the science–policy interface based on local credibility and legitimacy[157]. Despite a pronounced increase in scientific advances in non-sewered sanitation over the past decade, for the sector to mature, and if sustainable non-sewered sanitation is to move beyond a discourse to a reality, these barriers need to be overcome.

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