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REVIEW ARTICLE

Technologies for pollutant removal and resource recovery from blackwater: a review

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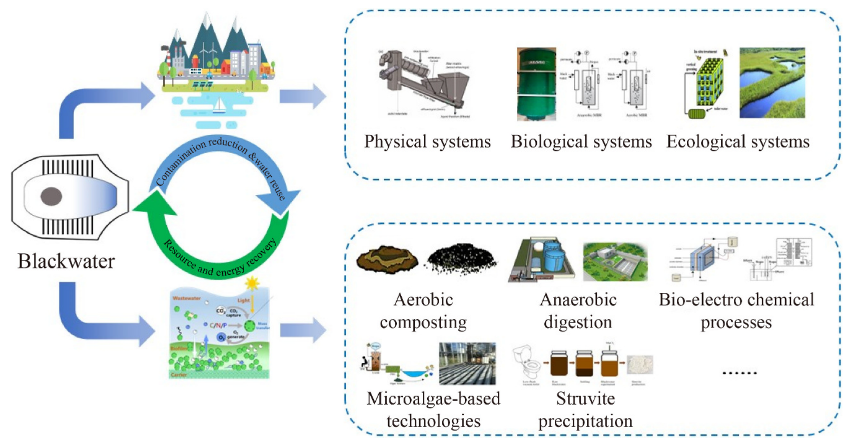
HIGHLIGHTS

- Blackwater is the main source of organics and nutrients in domestic wastewater.
- Various treatment methods can be applied for resource recovery from blackwater.
- Blackwater treatment systems of high integration and efficiency are the future trend.
- More research is needed for the practical use of blackwater treatment systems.

GRAPHIC ABSTRACT



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ABSTRACT

Blackwater (BW), consisting of feces, urine, flushing water and toilet paper, makes up an important portion of domestic wastewater. The improper disposal of BW may lead to environmental pollution and disease transmission, threatening the sustainable development of the world. Rich in nutrients and organic matter, BW could be treated for resource recovery and reuse through various approaches. Aimed at providing guidance for the future development of BW treatment and resource recovery, this paper presented a literature review of BWs produced in different countries and types of toilets, including their physiochemical characteristics, and current treatment and resource recovery strategies. The degradation and utilization of carbon (C), nitrogen (N) and phosphorus (P) within BW are underlined. The performance of different systems was classified and summarized. Among all the treating systems, biological and ecological systems have been long and widely applied for BW treatment, showing their universality and operability in nutrients and energy recovery, but they are either slow or ineffective in removal of some refractory pollutants. Novel processes, especially advanced oxidation processes (AOPs), are becoming increasingly extensively studied in BW treatment because of their high efficiency, especially for the removal of micropollutants and pathogens. This review could serve as an instructive guidance for the design and optimization of BW treatment technologies, aiming to help in the fulfilment of sustainable human excreta management.

1 Introduction

The sharply growing world population has made human excreta an outstanding problem. The safe disposal of human excreta is essential not only to human health, but also to the environment, as poor excreta management will lead to the contamination of water bodies and soils (Rose et al., 2015; Schmitt et al., 2017). Currently, up to 4.5 billion people in the world still lack access to safely managed sanitation services (Zhou et al., 2021). An

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estimated 850,000 people die annually because of a lack of access to clean water, sanitation and hygiene (Reynaert et al., 2020). One of the key solutions to this dilemma is the improved sanitation systems and sustainable treatment technologies for toilet wastewater.

In 2015, 193 member states of the United Nations approved 17 sustainable development goals (SDGs) and 169 targets that will drive the development of the world in the next 15 years, covering social, economic and environmental aspects (Orner and Mihelcic, 2018). Sanitation was included in SDGs, namely, SDG 6, to "ensure availability and sustainable management of water and sanitation for all." Included in this goal, Targets 6.2 and 6.3 aim to provide adequate and equitable sanitation for all, and improve the water quality and halve the untreated wastewater by 2030, respectively (UN, 2015). The proposal of SDG has triggered the world, especially developing countries, to devote manpower and financial resources to developing innovative and sustainable sanitation techniques in these areas (Nhamo et al., 2019; Pathak and Chakravarty, 2019; Zhou et al., 2018). Based on SDGs, the Chinese government has been actively advocating the "toilet revolution" and "living environment upgrade" in rural areas in recent years (Cheng et al., 2018), aiming to provide favorable and improved sanitation to hygienically protect people from contact with excreta. In 2010, the percentage of the population in China who did not have access to improved sanitation, such as ventilated pit latrines or flushing toilets, was 36% (Yang et al., 2012). When it comes to 2015, the coverage of sanitary toilets in rural areas increased to 78.5% (Cheng et al., 2018). Through the "toilet revolution," the indoor hygienic conditions of household toilets have been significantly improved, along with the alleviation of odor. However, most of the excreta are still stored in a subsurface container made of bricks and concrete, transferred several years later by a sludge suction truck, and finally end in treatment sites or abandoned areas (Schmitt et al., 2017). This approach has led to considerable pollution to the environment and waste of resources. Therefore, technologies for reliable excreta management still need progress.

Both dry and water-flushing sanitary systems are used worldwide. Dry sanitary systems are generally easier to operate and require less energy (Orner and Mihelcic, 2018; Aburto-Medina et al., 2020). However, for most residents, water-flushing toilets better meet their demand for comfort, but in turn increase the water consumption and production of concentrated wastewater. In 2015, cities in China produced an average of 14.28 million tons of fecal sludge every day, and the majority remained untreated (Cheng et al., 2018). Generating from water-flushing toilets, BW consists of a mixture of urine, feces, toilet paper and flushing water. As an indispensable stream of domestic wastewater, BW is always labeled "high nutrient concentrations," and "high pathogen concentrations" as well, being a potential pollution source and risk (Gros et al., 2020; Odey et al., 2017). However, BW is a valuable resource as the organics can be utilized to produce biogas and electricity, nutrients can be applied to agricultural use, and water can be reclaimed (Butkovskyi et al., 2016; Harder et al., 2019; Ziemba et al., 2019). The application of BW in agriculture has a long history, but rapid urbanization and development of agricultural modernization have decreased the need for organic fertilizer. To reduce the dependence on energy- and infrastructure-intensive wastewater treatment technologies of lower income countries and address water shortages, there has been an urgent need for intelligent, synergetic and decentralized systems (Odegaard, 2016; Hawkins et al., 2017). The fundamental principle for a more efficient wastewater treatment is source separation, which divides BW from greywater (GW) generated from bathtubs, laundries, and kitchen sinks (Boyjoo et al., 2013; Kozminykh et al., 2016). Compared to GW, BW is smaller in volume, but much higher in pollutant concentrations including nutrients and pathogenic microorganisms (Paulo et al., 2019; Vuppaladadiyam et al., 2019).

There have been several reviews regarding the characteristics of GW, as well as its treatment and reuse methods and scenarios (Boyjoo et al., 2013; Oteng-Peprah et al., 2018; Cecconet et al., 2019; Vuppaladadiyam et al., 2019). The characteristics and nutrient recycling pathways of human excreta have also been reviewed (Rose et al., 2015; Harder et al., 2019). However, the treatment and reuse strategies of BW generated from water-flushing sanitation systems have not been systematically summarized. Accordingly, the objective of this paper is to review the literature to determine the application and adaptability of existing technologies for BW treatment and reuse, as well as resource recovery. In this paper, the quantitative and qualitative characteristics of BW are discussed. A comparison of BW characteristics is made among different countries. The nutrient recovery potential is analyzed from the prospective of elemental composition. Subsequently, different treatment processes aimed at pollution control and nutrients and energy recovery are presented and their application prospects are discussed. Shortcomings hindering the practical application of BW treatment strategies are also analyzed. We hope the information presented in this work will provide guidance on the development and revolution of technologies that recover resources and purify water, especially for practical use in countries where water and energy resources are extremely scarce.

2 Properties of blackwater

2.1 Production

BW is the effluent collected by water-flushing sanitation systems, such as sewered toilets and free-standing toilets. In sewered toilets, the stream is transported through gravity or vacuum systems, for the latter water-saving is a prominent advantage. Differences in collection systems often lead to distinct options for treatment processes. Generally, BW in sewered system always ends in a wastewater treatment plant or small centralized treatment facilities built for several households. In comparison, on-site treatment processes are more preferrable for BW in free-standing toilets.

As a main part of domestic sewage, BW makes up 12%–33% of the total volume annually (Zha et al., 2019). The volume and concentration of BW mainly depend on toilet types, or flushing water consumption specifically. Conventional toilets (CT) consume 6–9 L water per flush, which is 6–15 times more than the vacuum toilets (VT) and 2–3 times the volume water-conserving toilets (WCT) use. The production of BW may vary significantly among countries depending on their geographical location and climate, living standards, custom and dietary habits, etc. Among individuals, age or body size has an inevitable effect on the production of excreta. A healthy individual excretes 51–796 g feces every day, with an average value of 128 g (Rose et al., 2015). The mean weight of daily feces for a child (3–18 years) has been recorded between 75 and 374 g, while infants (1–4 years) were shown to have a mean stool weight of 85 g/(cap·d) with no significant difference found between ages (Rose et al., 2015). For urine, an adult produces 1–1.5 L of urine per day in 4–5 times. The urinary output of a child is about half less than that of an adult (Karak and Bhattacharyya, 2011). Compared to industrial wastewater, the production of domestic wastewater is generally more stable. However, in some rural areas (e.g., in China), due to the seasonal population migration (Wang et al., 2017), the amount of BW may show great volatility. Additionally, on account of the different frequencies of urination and defecation, BW is more concentrated in residential areas whereas the one from workplaces or tourist areas is generally more diluted (Pedrouso et al., 2020).

2.2 Characteristics

The main components within BW include human excreta and flushing water. Toilet paper, with insoluble fiber (i.e., cellulose, hemicellulose, and lignin) being the main component, may also be contained in BW (Li et al., 2019). In many cases, however, toilet paper is always collected separately from excreta and removed from the source (Knerr et al., 2011). Although the volume ratio of BW is not large in domestic wastewater, it is rich in organics, N and P. For this reason, it is suggested by the "source separation" principle that BW be treated and reused separately from GW (Lam et al., 2015; Andersson et al., 2018). Additionally, as most of the pathogens and nutrients existing in domestic wastewater originate from BW, it is of great necessity to characterize BW with respect to its physical and chemical parameters, and microbial indicators.

2.2.1 Physicochemical parameters, elemental composition and resource recovery potential

Parameters mainly concerned in BW include suspended solids (SS), total nitrogen (TN), ammonia (NH_3/NH_4^+), total phosphorus (TP), chemical oxygen demand (COD) and biochemical oxygen demand (BOD). The parameter values depend closely on the type of sanitary systems. Generally, BW shows a weak alkalinity. The total COD and BOD_5 were within the range of 200–10000 and 100–1500 mg/L for CTBW and WCTBW, respectively (Table 1). For VTBW, the total COD could be as high as 30000 mg/L (Gao et al., 2019b). According to Todt et al. (2015), BW is the major contributor to the total load of organic matter and nutrients, with a low COD/N ratio and high content of free ammonia. Apart from flushing consumption, living standards and dietary habits in different countries may also affect BW quality through an impact on the excreta composition (Rose et al., 2015; Simha and Ganesapillai, 2017). The characteristics of BW collected from different countries are presented in Table 1, in which GDP per capita is applied as a parameter to characterize the average development level.

From the prospective of elemental composition, the primary elements contained in the dried solids of BW are carbon (C), nitrogen (N), phosphorus (P) and potassium (K), where C represents organic matter, while N, P and K represent nutrients. Contents of the above four elements imply the resource recovery potential of BW. The characteristics of feces, urine and excreta (mixture of urine and feces) are displayed in Table 2 (Meinzinger and Oldenburg, 2009). Data for excreta differ in some parameters from the sum of feces and urine, which might be attributed to the variability in the data sources. For urine, the dominant solute is urea, accounting for more than half of the organic compounds (Rose et al., 2015). The dried solids in urine contain approximately 13% C, 14%–18% N, 3.7% P, and 3.7% K (Rose et al., 2015). Carbon and nitrogen species in urea can be broken down into bicarbonate (HCO_3^-) and carbonate (CO_3^{2-}), and ammonium (NH_4^+) and ammonia (NH_3), respectively (Udert et al., 2003; Rao and Mogili, 2021). Feces contains 25% of solids by weight, and the remainder is water (Harder et al., 2019). N, P, and K make up 5%–7%, 3%–5.4%, and 1%–2.5% of the dried solids respectively (Rose et al., 2015) (Fig. 1). In general, the majority of C comes from feces, while urine contains most of the N. Apart from these elements, heavy metals are also present in BW, mostly attributed to feces (Tervahauta et al., 2014a). Compared to nutrients, heavy metal concentrations in BW are extremely low, but they may be critical when BW is reused for agricultural application.

Table 1 Characteristics of BW collected from several countries

Parameter	Unit	Australia (Tannock and Clarke, 2016)	Canada (CT) (Gao et al., 2019b)	Canada (VT) (Gao et al., 2019b)	China (Liu et al., 2017)	Germany (Wendland et al., 2007)	India (Welling et al., 2020)	Netherlands (De Graaff et al., 2010)	South Africa (Sahondo et al., 2020)[b]	Sweden (Palmquist and Hanaeus, 2005)	Turkey (Murat Hocaoglu et al., 2010)	USA (Hawkins et al., 2019)
GDP per capita[a]	US dollars in thousands	63.5	52.1	52.1	12.4	50.8	2.19	58.3	6.95	60.0	9.53	69.2
Flushing volume	L/flush		9	1		0.7–1	6		6		9	
pH			8.4	8.6	7.3–8.6		7.2–8.1	8.6 ± 0.53	8.0 ± 0.4	8.9–9.1	8.0 ± 0.3	8.9–9.0
Turbidity	NTU						100–600		97 ± 57			248–461
SS	mg /L				46–161		374–1030					180–667
NH_3-N	mg /L		96.4	1040	18–130	1100 ± 140	80–300	850 ± 150			147 ± 18	
TN	mg /L	275 ± 30	190	1700	49–191	1500 ± 250	100–350	1200 ± 180	186 ± 49	130–180		
TP	g /L	40 ± 14	38	330	67–95	202		150 ± 64	17 ± 9	21–58	25 ± 9	
COD total	g /L	1.44 ± 0.57	2.58	29.52	0.26–1.57	8.70 ± 0.40	0.28–2.82	7.7 ± 2.5	0.4 ± 0.095	0.81–3.14	1.23 ± 0.56	0.86–1.82
COD ss	g /L		1.54	19.32				4.9 ± 2.0				
COD sol	g /L	0.40 ± 0.13	0.89	8.88		2.40 ± 0.65		2.3 ± 0.81			0.41 ± 0.12	
COD col	g /L		0.15	1.32				0.50 ± 0.22				
BOD	mg /L									410–1400	338 ± 155	
TOC	mg /L					2500 ± 950						
TS	mg /L	1441 ± 435	2390	17140					63 ± 30[c]	920–4320	625 ± 437[c]	2001–2634
VS	mg /L	896 ± 325	1847	14200		4500 ± 2680[d]				420–3660	529 ± 377[d]	

Notes: a) GDP per capita in 2021. Data provided by International Monetary Fund; b) Data obtained after a solid/liquid separator; c) TSS; d) VSS.

Table 2 Organics and nutrients contained in human excreta (Meinzinger and Oldenburg, 2009)

Parameter		Unit	Feces	Urine	Excreta	Sum of urine&feces
Volume		L/(p·d)	0.14	1.37	1.25	1.51
Organics	TSS	g/(p·d)	38	57	51	95
	BOD_5	g/(p·d)	20	5	32	25
	COD	g/(p·d)	60	10	50	70
Nutrients	N	g/(p·d)	1.5	10.4	11.9	11.9
	P	g/(p·d)	0.5	1.0	1.5	1.5
	K	g/(p·d)	0.7	2.5	2.0	3.2
	S	g/(p·d)	0.2	0.7	0.19	0.9

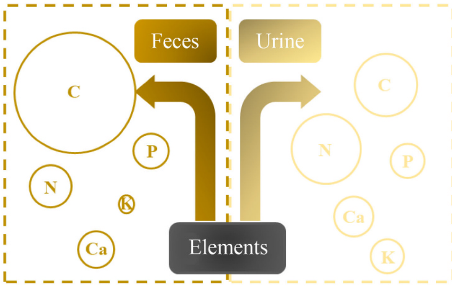

Fig. 1 Elemental composition of dried solids in feces and urine. Note that the area of the circles is proportional to the elemental contents.

2.2.2 Pathogens in blackwater

Microorganic pathogens are nearly the most concerning issue when BW is reused, especially in epidemic cases (Schoen and Garland, 2017; Lai et al., 2018; Li et al., 2022). BW is thought to contain extremely high levels of microorganisms and is considered liable to spread enteric microorganisms among people (Odey et al., 2017). SARS-CoV-2 RNA has also been detected in human feces (Makhmalbaf et al., 2022). Exposure to untreated BW presents a great risk to human health and may lead to potential water-borne diseases. Pathogens contained in BW are mainly from feces, while urine contains few pathogens, especially for a healthy person. According to Odey et al. (2017), pathogens within untreated BW are classified into four categories, namely, bacteria, viruses, helminths eggs and protozoa. Apart from these microorganisms existing in raw BW, insects might act as a transmission media for diseases through breeding. The microorganisms most commonly monitored and used as indicators include coliform bacteria, *Salmonella*, and fecal streptococci (Boyjoo et al., 2013; Magri et al., 2015; Huang et al., 2016). Some typical viruses that are present in feces include Hepatitis A virus, Poliovirus, Rotavirus, etc. (Odey et al., 2017). Viruses in feces mainly lead to health hazards including enteric diseases such as diarrhea, abdominal pain, and sometimes fever as well (Wigginton et al., 2015). Among protozoas, *Entamoeba histolytica* and *Giardia intestinalis* could lead to amoebiasis and giardiasis, respectively, being a considerable public health burden for countries with poor hygienic conditions (Haque et al., 2003). A list of BW-related pathogens and the related symptoms has been reported elsewhere (Odey et al., 2017).

3 Treatment technologies of blackwater

In general, the objective of BW treatment can be classified into two aspects (Fig. 2). One is the removal of pollutants to meet the corresponding discharging or reuse standards. The other is the application of certain methods to recover organic matter or nutrients to produce energy, fertilizer, or other high value-added products. Prior to further treatment, a pretreatment step has always been recommended to decrease the subsequent processing load, or remove the insoluble or suspended solids. For water or nutrient reuse, a post-treatment process should be employed to meet the requirement for microbial or pathogenic indicators (Cid et al., 2022).

3.1 Blackwater treatment technologies for contamination reduction and water reuse

3.1.1 Physical systems

Owing to high pollutant concentrations, BW pretreatment prior to further treatment is a suitable and recommended option. As both solid (feces and toilet paper) and liquid (urine and flushing water) exist in BW, physical processes, such as filtration and sedimentation, have always been applied as a pretreatment step. In a filtration process, not only could suspended solids (SS) be removed to decrease the subsequent treatment loads and avoid system blockage, but attachment of pathogens or shedding of disinfectants could also be disturbed (Pype et al., 2016). Filters that have been reported include peat and sawdust (Todt et al., 2014), sand and nonwoven textile (Tao et al., 2011), activated carbon (Sahondo et al., 2020) and more commonly, membranes (van Voorthuizen et al., 2005; van Voorthuizen et al., 2008; Kamranvand et al., 2020). The application of organic percolation filters is limited due to the severe clogging (Todt and Jenssen, 2015). A novel mechanical filter device combining traditional screening with a new type of counterflow filter using an organic media was therefore developed by Todt et al. (2014) and applied in mountain cabins. They reported the employment of a mixing filter composed of peat and sawdust to the treatment of settled BW, and a TSS reduction of 60%–75% was achieved. The result was satisfactory compared to the initial high fraction of small particles existing in the BW sample, but the effluent TSS concentration was still up to 200–300 mg/L, implying the need for further treatment.

Membrane filtration processes (including microfiltra-

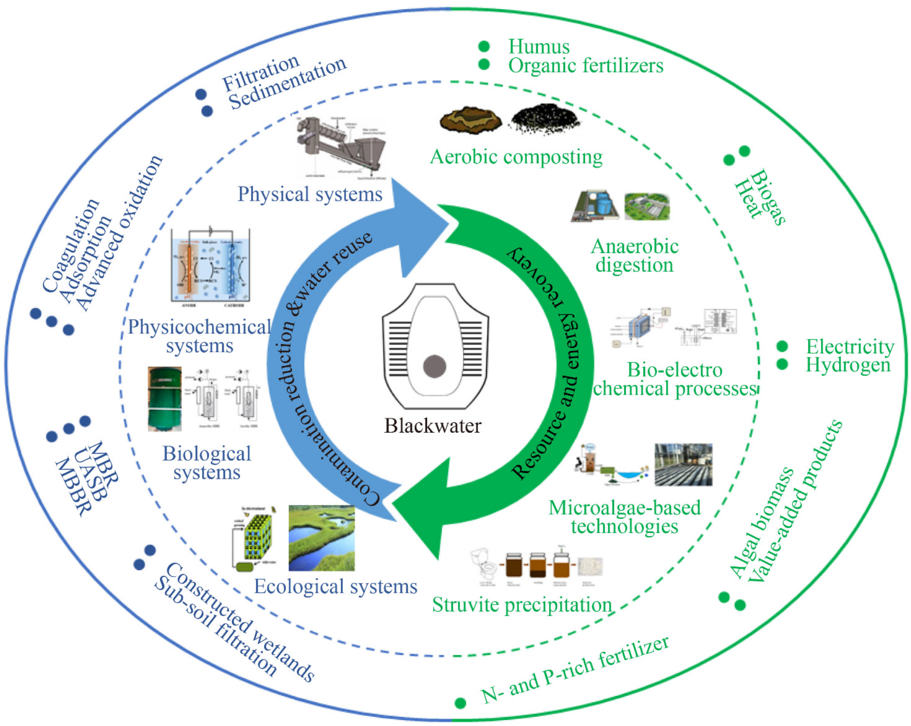

Fig. 2 A general view of BW treatment technologies (Todt and Jenssen (2015); Huang et al. (2016); Dorji et al. (2022); van Voorthuizen et al. (2008); Jin et al. (2018b); Sharma and Mutnuri (2019); Zamalloa et al. (2013); Vasconcelos Fernandes et al. (2015); Silva et al. (2019); Sun et al. (2020)).

tion, ultrafiltration, nanofiltration, reverse osmosis and forward osmosis) can produce effluent of high quality, so they are sometimes essential in water reuse projects (Kamranvand et al., 2020; Shi et al., 2021). Specific to BW treatment, membrane technology could be used to recover nitrogen and phosphorus in the biological effluent for agricultural reuse as well, with the permeate being reused for toilet flushing (van Voorthuizen et al., 2005). Additionally, membrane separation could be combined with conventional thermal driving processes to facilitate distillation from lower quality heat sources (Kamranvand et al., 2020). Membrane distillation (MD) is a nonisothermal membrane-based separation process with the aid of diffusive and convective transportation of vapor across a membrane (Tun et al., 2016). It is regarded as a desirable method for water reclamation, because the process can be performed under controlled conditions and no further treatment is needed (Teoh et al., 2011). In toilet wastewater treatment, MD is suitable for urine dewatering, while the introduction of upstream interventions such as source separation, post-flush source separation or prefiltration could alleviate the side effects of fecal contamination that reduce the permeate quality and constrain the membrane flux (Kamranvand et al., 2018; Khumalo et al., 2019). Although membranes have shown high removal efficiency of SS and total organic matter, a main drawback associated with this process lies in the occurrence of membrane fouling, which exists as an obstacle to operational stability (Vuppaladadiyam et al., 2019).

3.1.2 Biological systems

Due to the advantages of high nutrient concentrations and suitable BOD/COD ratios in BW, biological methods have always exhibited good potential in BW treatment, including both aerobic and anaerobic processes. Systems typically reported are membrane bioreactors (MBRs) (Murat Hocaoglu et al., 2013; Knerr et al., 2011; Whitton et al., 2018; Ren et al., 2022), upflow anaerobic sludge blankets (UASBs) (Luostarinen and Rintala, 2005; De Graaff et al., 2010; Gao et al., 2019a), and septic tanks. In addition, biofilm technology, a key solution to rural wastewater treatment, is also considered applicable in BW treatment (Hyun and Lee, 2009; Todt and Dorsch, 2015; Jin et al., 2020; Zhou et al., 2020).

Biological treatment could be considered as a central process in BW treatment, which is usually set after coarse filtration and followed by a sedimentation/filtration process to separate the biosolids or sludge and a disinfection process to meet the microbial requirements. In some cases, when submerged MBRs are used, biological degradation and solid/liquid separation proceed in the same chamber; therefore, space occupation and construction costs will be reduced. The efficiencies of biological systems are mainly influenced by factors including solids content, energy density, concentrations of protein and fat in feces, and urea concentrations in urine (Rose et al., 2015). The reaction rates of aerobic treatment are generally higher than those of anaerobic processes, leading to a smaller volume of reactors. However, the high requirement for oxygen might lead to great demand for energy. In addition, although reclaimed water with acceptable quality is obtained, the resources are mostly wasted from a Circular Economy perspective (Robles et al., 2020).

To efficiently remove pollutants from BW, activated sludge method, as well as its modification, is widely used. As a part of the activated sludge method, MBR technology has a history of nearly 30 years in water treatment and reclamation (Fig. 3). Murat Hocaoglu et al. (2011) reported a pilot-scale MBR for the treatment of BW, which was pretreated through a series of 6- and 3-mm screens. The COD reduction for the effluent was 97%, while the average removal for TN was 73%. The result was obtained with a low DO range of 0.15–0.35 mg/L and revealed the advantages of MBR over activated sludge processes. However, the loss of alkalinity in the MBR caused by a high influent nitrogen concentration must be fulfilled for a stable nitrification efficiency (Knerr et al., 2011). The accumulation of refractory substances and high salinity in a complete-cycled MBR may lead to restriction to biological activities (Knerr et al., 2011); therefore, desalination and post-oxidation units are always needed, which also assist in effluent color and pathogen removal. Compared with the traditional aerobic biological treatment processes, the initial investment of MBR treatment is higher, as well as the maintenance cost due to membrane fouling (Cecconet et al., 2019). However, to achieve the same treatment effect as MBR, the increased process and operation cost are also high (Gao et al., 2022). Although the application of MBRs is mature in BW treatment, the technology still faces some problems in practical use, such as membrane fouling caused by the accumulation of calcium and magnesium salts and other refractory organic compounds (Kamranvand et al., 2018). Membrane fouling is considered the Achilles' heel for MBR systems (Cecconet et al., 2019). A cake layer is more likely to form in the anaerobic MBR, as well as aerobic MBR with extremely high organic loading (van Voorthuizen et al., 2008). It was revealed in their work that the irreversible fouling increased in the start-up stage and then became relatively constant in an anaerobic MBR. This fouling phenomenon could not be prevented, but combined relaxation and backwashing could hinder its further increase. Reversible fouling, in comparison, was tightly linked with the concentration of soluble compounds in BW, especially colloids (van Voorthuizen et al., 2008). Membrane fouling may lead to a short membrane cleaning cycle and complicated operation and management. Under this condition, traditional cleaning method may be less effective in membrane flux recovery. In addition, the long-term operation stability of the MBR system, and the disposal of excess sludge will still be a difficult problem in practical application (Hamedi et al., 2019). Hence, the research and development of effective and economical antibacterial membrane and the reduction in operation and maintenance costs will be the bottleneck to be overcome .

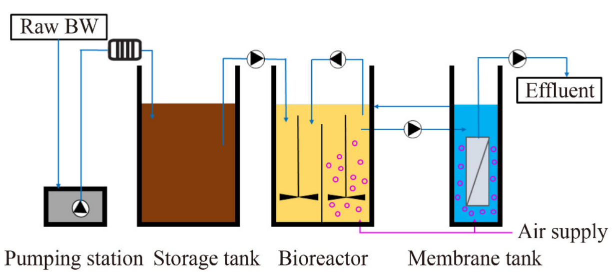

Fig. 3 Schematic diagram of a pilot blackwater treatment plant (Knerr et al., 2011).

UASB has been intensively investigated as an anaerobic system for BW treatment in recent years. Due to the efficient gas/liquid/solid separation, long SRT could be maintained at a relatively short HRT (Adhikari and Lohani, 2019; Gao et al., 2019a). Compared to other types of anaerobic treatment processes, the reactor volume could be significantly reduced (De Graaff et al., 2010). An average COD removal of 78% was achieved using a UASB at HRT = 8.7 d for VTBW (De Graaff et al., 2010). The efficiency of UASB is closely linked with the reaction temperature. Kujawa-Roeleveld et al. (2006) found that COD removal of 78% was achieved in UASB for BW treatment at 25°C, while it decreased to 61% at 15°C. The economy and compactness of UASB makes it potential and suitable for replacing the traditional septic tanks (Adhikari and Lohani, 2019; Vuppaladadiyam et al., 2019). Very recently, a pilot-scale UASB (200 L/d) for on-site BW treatment in Bhutan has been reported (Fig. 4). In this case, UASB was applied as an alternative to the septic tank, showing BOD_5, COD and TSS removal of over 70%. The authors stated that the UASB only provided primary treatment of BW, and the main drawback lied in the need for further treatment for organic removal. In addition, high free ammonia concentration may lead to inhibition of microorganism activity. Moreover, many persistent organic micropollutants, including pharmaceuticals, estrogens and personal care products are hardly biodegradable in the reactor under anaerobic conditions (Butkovskyi et al., 2018).

Worldwide, a great volume of BW is discharged into water bodies via septic tanks as a primary pretreatment step to reject solids and partially degrade organics, especially in rural areas without perfect drainage piping systems (Guo et al., 2014; Singh et al., 2019). Due to its simple structure and convenient maintenance, septic tank is regarded as a common option for the onsite treatment

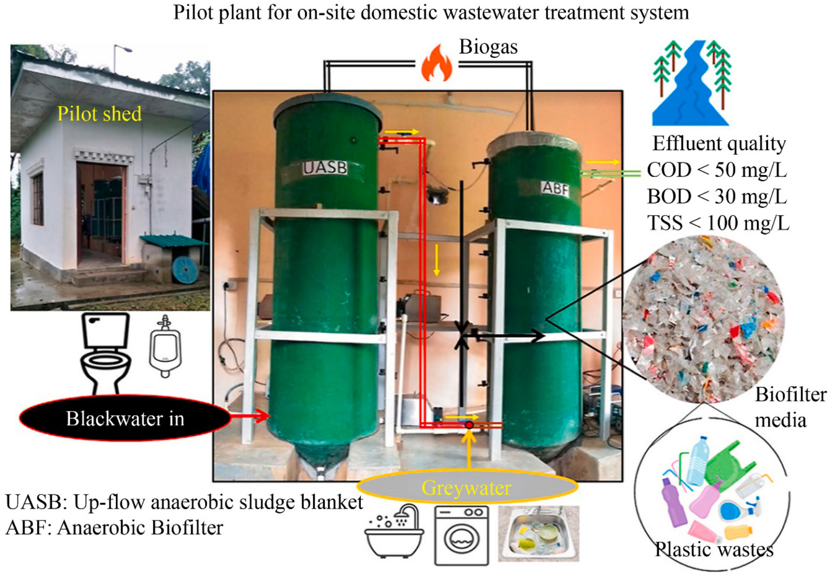

Fig. 4 A pilot-scale UASB in BW treatment (Dorji et al., 2022).

of domestic wastewater in cases where water quality is not strictly required (Hong et al., 2019). Compared to developing countries, septic tank systems are relatively less used in Europe and United States. Twenty-five percent of the households in the United States and 26% in Europe rely on septic tanks for onsite sanitation (Withers et al., 2014). Among various types of septic tanks, the three-chambered type is most widely used. In the first chamber, BW undergoes a process of fecal decomposition and solid-liquid separation by a difference in specific gravity. The effluent, containing small amounts of large particles and pathogens, undergoes a further fermentation and separation process in the second chamber, and is stored in the third chamber. The effluent quality of a septic tank relies closely on the reaction duration of the biochemical processes. Within a conventional three-chambered septic tank, the digestion and removal of pollutants are not efficient as it can only partially convert organic nitrogen and phosphorus in the influent into ammonium and soluble ortho-P, leading to a relatively low pollutant removal. In the absence of greywater introduction, the effluent of the third chamber could meet the requirements for drainage into sewage systems, but mainly not for discharging into water bodies or irrigation unless equipped with advanced nutrient removal (Withers et al., 2014). Therefore, more information on effluent quality and septic tank modification is urgently needed. In practice, natural treatment technologies always acted as an additive to meet certain discharging standards in rural areas (Jin et al., 2018a; Saeed et al., 2021) (Fig. 5). With the stricter discharging requirements and the development of anaerobic technologies, enhancement and modification have been performed on septic tanks, including upflow operation to improve the contact between anaerobic sludge and wastewater (Luostarinen and Rintala, 2005; Luostarinen and Rintala, 2007; Al-Jamal and Mahmoud, 2009; Santiago-Díaz et al., 2019); membrane module introduction for effective rejection of suspended solids and associated particulate organic matter (Khalid et al., 2017); four-chamber septic tank with orifice plate and filter (Singh et al., 2019); combination with microbial electrolysis cell (MEC) to enhance biogas production and reduce the discharge of phosphorus and H_2S (Zamalloa et al., 2013); and solar energy-assisted septic tank to raise the in-tank temperature and promote solids degradation (Connelly et al., 2019).

3.1.3 Ecological systems

Ecological systems represent an alternative to biological and physical systems, in which both physical removal and biological degradation processes are likely to occur (Boyjoo et al., 2013). Compared to biological treatment processes mentioned in former sections, ecological systems including constructed wetlands (CWs) and subsoil filtration offer the advantages of low cost and energy consumption, high treatment capacity, convenient maintenance, and user-friendliness (Masi et al., 2010; Paulo et al., 2013; Jin et al., 2018b). They may also gain economic benefits through the harvest of crops as well. Therefore, ecological systems are particularly common in rural areas of developing countries. Normally, an ecological system always undergoes both physical processes through a filtration medium, and biological processes via microorganisms within the system (Boyjoo et al., 2013). Processes including chemical precipitation, adsorption, and microbial interactions and uptake by vegetation are also believed to occur in the system (Kivaisi, 2001).

Masi et al. (2010) introduced several pilot CW systems with different pretreatments located in Egypt, Morocco,

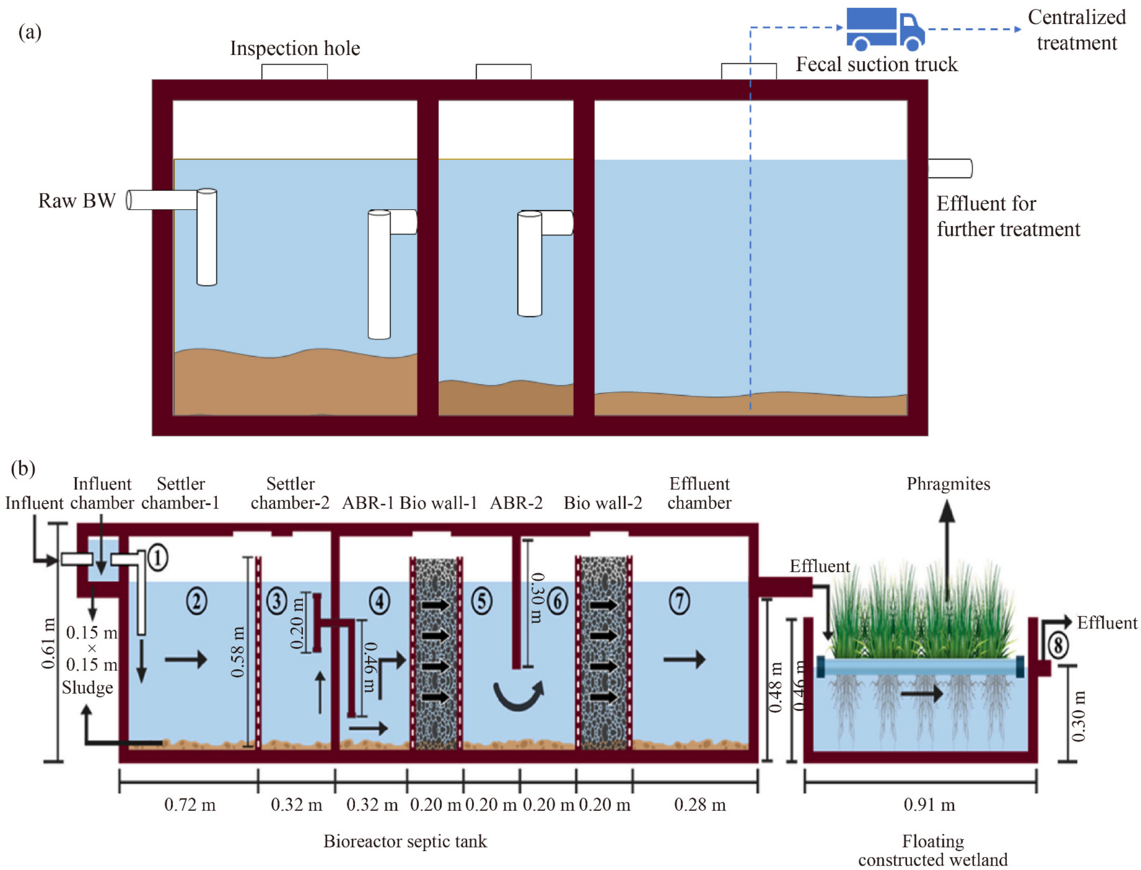

Fig. 5 Schematic diagram of (a) a conventional three-chambered septic tank, and (b) a bioreactor septic tank integrated with floating constructed wetland (Saeed et al., 2021).

Tunisia and Turkey. Multistage CW configurations were tested, consisting of horizontal and vertical subsurface flow CW for secondary treatment and free water system as the tertiary stage. The results indicated that all systems obtained efficient COD reduction (up to 98%) and nitrification (92%–99%) for BW. The effluent BOD_5, NO_3^--N and NH_4^+-N concentrations were below 15, 1 and 0.5 mg/L, respectively, suggesting the feasibility of CW application in the Mediterranean area.

Paulo et al. (2013) used a natural system for the treatment and reuse of source segregated domestic wastewater. The BW fraction was treated by an evapotranspiration tank (TEvap) and digester. The effluent, as well as the GW fraction, was treated subsequently by a compact setup including a grease trap, sedimentation tank and two CWs. The results showed that this ecological and low-cost system realized the reuse of greywater and nutrients in BW, and was essentially maintenance free.

Although labeled low cost, low mechanical operation and low energy consumption, ecological systems also suffer from problems such as large land occupation, bad odors and long HRT (Boyjoo et al., 2013; Lansing et al., 2017). In the meantime, although CWs, especially hybrid wetland systems, have shown activity in pathogen and fecal indicator removal (Wu et al., 2016), ecological systems are generally not efficient in pathogen removal. It may be a potential risk for the transmission of airborne and waterborne diseases (Walton, 2012), and a disinfection process is still needed or recommended for an effluent containing few pathogens.

3.1.4 Physicochemical systems

The physicochemical treatment of BW is now attracting increasing interest for its relatively high efficiency and independence on environmental conditions. Reported processes include coagulation (Kozminykh et al., 2016), adsorption (Moges et al., 2018; Florentino et al., 2019b; Hawkins et al., 2019), ammonia or lime treatment (Fidjeland et al., 2015), yeast sanitization (Odey et al., 2017) and advanced oxidation processes (AOPs) such as electrochemical catalysis (Chun et al., 2018; Cid et al., 2018b; Rogers et al., 2018; Haupt et al., 2019), ozonation (Zhang et al., 2013; Haupt et al., 2019), and peroxidation (Englehardt et al., 2013; Zhang et al., 2013). Conventional methods, such as coagulation, are generally not capable of meeting the discharge requirements and can only be introduced as additional steps applied to reduce the subsequent loading, or for sludge thickening and nutrient conservation.

Ammonia has been proven to have the potential to sanitize BW before its reuse in agriculture (Fidjeland et al., 2015). NH_3 entering the pathogen cells can take up intracellular protons to form NH_4^+, leading to disordered organism functioning, and therefore deactivating the pathogens (Odey et al., 2017). Moreover, ammonia is highly soluble in water and lipids, making it easy to transport over cell walls and other barriers. The sharp rise in internal pH caused by the solution of ammonia could lead to the irreversible change in cell proteins, and eventually result in the overall pathogen destruction (Pecson and Nelson, 2005; Pecson et al., 2007). This treatment process is generally more applicable in cases of urine diverting toilets with little flushing water. When BW is highly diluted because of the large volumes of flushing water used, urea can be added to increase the ammonia concentration for sanitation (Odey et al., 2017).

AOPs are always used as a final step to remove micropollutants, chroma and pathogens from BW, following biological systems. When the treated water is to be reused, AOPs for micropollutants removal are always indispensable (Meng et al., 2021). Compared to biological treatment processes, AOPs exhibit higher reaction rate and efficiency while covering a smaller space (Mawioo et al., 2016). Generally, when some AOPs are used, disinfection process could be omitted as they are efficient in pathogen removal; therefore, they are widely applied in BW reuse as well.

Among advanced oxidation processes, electrochemical treatment provides a reliable method for pollutant and pathogen removal. Electrochemical processes are characterized by the capability to adjust to influent composition variations, modular design and small footprints (Anglada et al., 2009) and are identified by Radjenovic and Sedlak (2015) as potential "next-generation technologies" for the treatment of contaminated water. Advantages associated with electrochemical technologies include versatility, high energy efficiency, amenability to automation and cost-effectiveness (Feng et al., 2016). In general, a typical electrochemical treatment process can be defined as the removal of pollutants or inactivation of microorganisms with the aid of an external electric current passed through water using appropriate electrodes (Gonzalez-Rivas et al., 2019). Electrochemical process is especially efficient in BW treatment as high concentration of chloride exists in urine. Reactive chorine radical species (RCS) including Cl_2, HOCl, ClO^-, and ·Cl could therefore generate and serve as primary disinfectants (Fig. 6). In that case, reactive oxygen species (ROS), including hydroxyl radicals (·OH), superoxide anion radicals (·O_2^-), ozone (O_3) and hydrogen peroxide (H_2O_2), act as enhancers of the electrochemical process (Huang et al., 2016). Owing to these oxidants, electrochemical processes are always used for BW disinfection. Haupt et al. (2019) achieved a COD degradation efficiency of 38.1% for the treatment of synthetic VTBW, which was higher

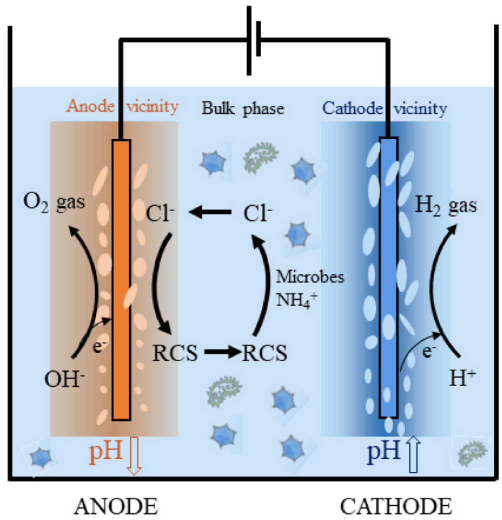

Fig. 6 Schematic for electrochemical disinfection of BW based on the study of Huang et al. (2016).

than the efficiencies obtained by ozonation (17.0%) and peroxone (25.7%), and a lower specific energy demand (93.6 kWh/kg mCOD). In practical applications, however, the scaling and erosion of electrodes still remain a key problem, due to the complex composition of BW (Thostenson et al., 2018). Furthermore, the formation of disinfection byproducts during the chlorination process is another concern owing to their possible association with adverse reproductive outcomes (Huang et al., 2016).

Microwave (MW) technology has emerged as another fast and efficient option for BW treatment, characterized by instant and accurate control of the power input as well as fast and uniform heating throughout the target material (Zhang et al., 2022). Studies have revealed the application of MW in the treatment of various types of sludge, showing its good performance in pathogen deactivation and volume reduction (Wojciechowska, 2005; Mawioo et al., 2016; Kocbek et al., 2020). The destruction of pathogens by MW was attributed to both the nonthermal (electromagnetic radiation to break hydrogen bonds) and thermal (temperature rise to cause water boiling and cell rupture) effects (Banik et al., 2003). In a laboratory-scale MW unit for fresh BW sludge treatment, a volume reduction of over 70% was achieved, as well as a complete reduction of *E. coli*, at a contact time of 1 min (MW energy = 8 Wh, temperature = 71°C), showing great potential in BW treatment and disinfection for toilets in emergencies (Mawioo et al., 2016). However, to the best of our knowledge, the pilot-scale application of MW in toilet wastewater treatment is rare. The main obstacle for this situation lies in the fact that energy in microwaves cannot be efficiently transferred into BW in practice. The safety of workers during operation is another limiting factor. Performance of several BW treatment processes has been listed in Table 3.

3.2 Blackwater treatment technologies for resource and energy recovery

3.2.1 Fertilizer and chemical production

Due to the high organic concentration in BW, complete conversion into CO_2 and H_2O in traditional WWTPs calls for high oxygen and energy input. In comparison, aerobic composting of BW is a suitable option to obtain humus and organic fertilizers, especially for areas without connection to sewers (Oarga Mulec et al., 2016). It could also be applied as a reliable method for sanitizing fecal sludge (Manga et al., 2021). The aerobic composting of BW has long been applied worldwide, mostly to the solid fraction in VTBW. According to the Chinese standard "Hygienic requirements for harmless disposal of night soil" (GB 7959-2012), the artificial composting should last for at least 10 d at temperatures above 50°C or 5 d at above 60°C, while the mechanical composting should last for 2 d at above 50°C. For the obtained compost, the death rate of Ascaris eggs should be more than 95%, while the fecal coliform value should be over 10^{-2} g (or mL) and *Salmonella* should not be detected. Compared to the initial solid waste, the composting process has the potential to halve the volume and mass (Odey et al., 2017). The high initial ammonia levels, along with the high temperature in pile, are effective in the alleviation of microorganism contamination (*E. coli* and enterobacteria) (Oarga Mulec et al., 2016). Aerobic composting process usually includes pretreatment, primary fermentation, secondary fermentation and solid fertilizer processing. Among them, the temperature of one fermentation compost is generally controlled at 55–60°C, and the air ventilation rate of pile material per unit volume is generally 0.05–0.20 m^3/min, to meet the oxygen demand of microorganisms. In terms of nutrient content, a compost product obtained by Oarga Mulec et al. (2016) using the solid fraction in VTBW exhibiting N:P:K = 3.0:3.2:1.6, along with a favorable ratio of C:N = 10, indicating its biological stability. In practical applications, vermin and odor may make the technology less acceptable (Odey et al., 2017), thus good operation and management are needed.

Microalgae-based technologies have also been used in BW treatment (Cai et al., 2013; Vasconcelos Fernandes et al., 2015; Silva et al., 2019), and a pilot reactor is shown in Fig. 7. As microalgae can also utilize CO_2 in the atmosphere through photosynthesis, they have shown promising potential as a sustainable and economical wastewater treatment technology (You et al., 2022). In this process, algal biomass could be simultaneously obtained with treated water by utilizing the nutrients in wastewater. Biomass produced could be further used for

Table 3 Processes for BW treatment

Reference	Process	Treatment type	COD (g/L)		TN (mg/L)		TP (mg/L)		*E. coli* ($\log_{10}$ CFU/mL)	
			In	Out	In	Out	In	Out	In	Out
Todt and Jenssen (2015)	Mechanical wood-shavings filter	Physical	19	4			310	210		
Davey et al. (2022)	UF (0.03 μm)	Physical	4.10	2.15	2000	1900	166	85	3.44	< −1
	RO (post-UF)	Physical	2.15	0.09	1900	583	85	0.537	< −1	ND
Kamranvand et al. (2020)	MD (0.1 μm)	Physical	7.3	0.075 ± 0.035	398[a]	75 ± 94[a]			5.78	< −2
Knerr et al. (2011)	Rotary screen +MBR	Physical +Biological	2.89 ± 0.79	0.14	477	49	34.4 ± 6	29.2 ± 2.7	4.23	ND
De Graaff et al. (2010)	UASB, HRT=8.7 d	Biological	9.8 ± 2.6	2.4 ± 0.84	1900 ± 190	1800 ± 220	220 ± 67	130 ± 15		
Zha et al. (2019)	ABR, HRT=48 h	Biological	1.93 ± 0.25	0.11 ± 0.02	189 ± 14	134 ± 6	37.4 ± 5.3	25.1 ± 3.8		
Sakurai et al. (2021)	UASB+CW	Biological +Ecological	1.22	0.066	273.2	125	33.2	10	N.A.	2.23[b]
Haupt et al. (2019)	Ozonation (81 mg/L, 4 h)	Chemical	4.67	3.88						
	Ozonation (81 mg/L, 4 h)+H_2O_2	Chemical	4.79	3.56						
Rogers et al. (2018)	Granular activated carbon	Chemical	2.71 ± 0.37	1.10 ± 0.25						
Sahondo et al. (2020)	Solid/liquid separation +Electrochemical +GAC	Physical +Chemical	0.4 ± 0.095[c]	0.061 ± 0.049	186 ± 49[c]	102 ± 20	17 ± 9[c]	14 ± 12		

Notes: a) Ammonia nitrogen; b) Fecal Coliform; c) Water quality after solid/liquid separation; N.A.: Not available.

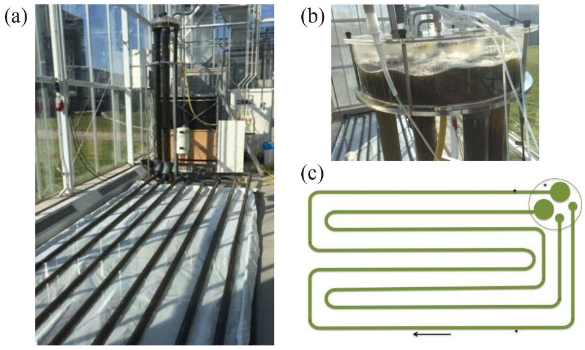

Fig. 7 Nutrients recycle from BW using microalgae. Reproduced from Silva et al. (2019) with permission. (a) Pilot Photobioreactor (PBR). (b) Close-up of the bubble columns and mixing box. (c) Schematic image of the pilot Photobioreactor (PBR) from an overhead perspective.

energy, livestock or agricultural uses, or be subsequently applied to produce high value-added products, such as fatty acids and vitamins (Robles et al., 2020). As microalgae cultivation is always assisted by light, threats caused by bacteria and pathogens could be minimized with aid of ultraviolet radiation and photooxidation (Zitnik et al., 2019). Some substances excreted by microalgae have also been proven to inhibit the growth of pathogenic bacteria (Skulberg, 2000). Zitnik et al. (2019) investigated the impact of environmental parameters and *C. vulgaris* on *E. coli* removal in a microalgae-based BW treatment system. The results showed that electric conductivity (EC) affected *E. coli* inhibition the most. When the EC was above 1569 μS/cm, a mutualistic relationship between *C. vulgaris* and *E. coli* was observed. Studies have revealed that the green microalgae *Chlorella sorokiniana* was able to fully remove the phosphorus and nitrogen in BW or anaerobically treated BW (Vasconcelos Fernandes et al., 2015; Vasconcelos Fernandes et al., 2017). The results indicated that P was completely removed from BW in 4 days, but another 8 days were needed for N removal, due to the high N:P ratio of the BW, as well as a relatively slow N utilization by the microalgae. The biomass/concentration obtained at the end of the experiment was 11.5 g/L, with a P content of approximately 1% and a N content of 7.6%, showing the potential of microalgae in BW treatment and posttreatment (Vasconcelos Fernandes et al., 2015).

Struvite ($MgNH_4PO_4 \cdot 6H_2O$), initially observed in pipes and pumps in WWTPs, has once been regarded as a scaling problem (Ma et al., 2018). This precipitation problem, along with its potential use as a fertilizer, has motivated the development of the technology for nutrient recovery (Robles et al., 2020). In recent years, struvite preparation from raw or anaerobically digested BW (Gell et al., 2011; Tervahauta et al., 2014b; Yee et al., 2019; Sun et al., 2020) or urine (Wilsenach et al., 2007; Latifian et al., 2014; Wei et al., 2018) has become a research interest, as both wastewater fractions are rich in N and P (Fig. 8). Compared to conventional P precipitation or biological processes, the recovery of N and P could be realized simultaneously based on the additional introduction of Mg^{2+}. The crystallization reaction is affected by factors including pH, temperature, Mg/P ratio, stirring speed. The pH value determines the crystallization process by influencing the morphology of NH_4^+ and PO_4^{3-}. When the pH is between 8 and 9, the crystallization of MAP is favorable (Rahman et al., 2014). The Mg:P mole ratio was another decisive factor affecting the crystallization of the reaction. The results showed that when this ratio is between 1.1 and 1.3:1, the recovery rate of phosphorus could reach over 90% (Tao et al., 2016). In comparison, the recovery of nitrogen was sometimes unsatisfactory due to the imbalance of N/P content ratio in BW and the MAP product (Patel et al., 2020). A high reaction temperature will lead to an increase in solubility product constant of MAP and ammonia evolution, which is also not conducive to crystallization. Gell et al. (2011) obtained struvite products from both urine and anaerobically digested BW with the addition of $MgCl_2$ at pH 8.6, followed by a separate precipitation tank and air drying at 40°C. The concentrations of P (12%), N (5%) and Mg (11%–14%) for both products were all suitable for use as fertilizers, meeting the local requirements for heavy metals and pathogens as well. The improvements in maize growth and harvest were comparable to soluble phosphorus fertilizer, suggesting the effectiveness of struvite as a slow-release fertilizer. Sun et al. (2020) conducted a series of batch experiments using different raw BW flushed with different amounts of tap water to examine the P recovery potential. The results revealed that owing to moderate phosphate, high ammonia, and relatively low calcium concentration, along with strong buffering capacity and ideal pH, a P recovery efficiency over 90% was achieved. Without the need for pH adjustment, the obtained struvite products had high purity of nearly 95% and low concentrations of heavy metals. However, the large-scale application of struvite is limited, mainly because of shortcomings in product quality,

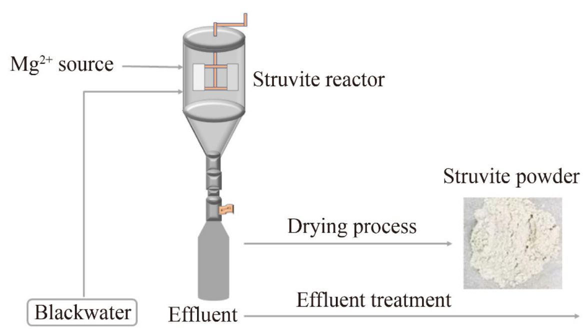

Fig. 8 Schematic for struvite production from BW (modified from Nagy et al. (2019) and Sun et al. (2020))

energy consumption, economic feasibility, and footprint (Robles et al., 2020; Zahed et al., 2022). Most laboratory-scale investigations on struvite formation have been conducted in aqueous solutions containing ions that are present in struvite (i.e., NH_4^+, PO_4^{3-}, Mg^{2+}) in ratios similar to their proportions in struvite crystals. However, in practical applications, the presence of other ions (Ca^{2+}, Fe^{2+}, etc.) may affect the dynamics of the interactions and ion clustering (Tansel et al., 2018). Additionally, if not well controlled, an overdose of chemicals and their terrible residue in effluent could be problems. In addition to struvite, precipitated hydroxyapatite ($Ca_5(PO_4)_3OH$) could also be obtained through the recovery of P and used as a low solubility phosphorus-rich fertilizer (Cid et al., 2018a).

3.2.2 Biogas and energy generation

Anaerobic treatment is regarded as the core technology for energy and nutrient recovery from source separated BW because it converts organic matter to methane, which can be used to produce electricity and heat, with low yields of excess sludge (De Graaff et al., 2010). It has been regarded as the most promising approach by research communities based on the principle of sustainability (Sawatdeenarunat et al., 2016). However, for BW, especially collected by vacuum, the extremely high free ammonia concentration has been reported to have significant negative impact on BW digestibility (Florentino et al., 2019a; Gao et al., 2019a). Additionally, the COD to ammonia nitrogen ratio (COD/NH_4-N) for this BW lies in the range of 100:12–14 (Wendland et al., 2007), which is much lower than the recommended value for anaerobic digestion.

To promote the digestibility of BW, a solution of kitchen waste (KW) co-digestion has been proposed by many researchers (Luostarinen and Rintala, 2007; Wendland et al., 2007; Tannock and Clarke, 2016; Zhang et al., 2021), which is characterized by a high COD/N ratio (100:(0.07–0.18)). Zhang et al. (2019a) investigated the effects of mixing ratios on anaerobic co-digestion of BW and kitchen organic waste. The biological methane potential (BMP) increased by over 150% at both volatile solids (VS) ratios of 1:2 and 1:3, and the methane production yield increased by 50%, along with an enhanced hydrolysis efficiency. The results suggested the improved energy recovery by co-digesting BW with KW. Apart from co-digestion, ultrasonication (Zhang et al., 2019b) was speculated to promote the digestion of BW by breaking down cell walls in the influent biomass, leading to its partial decomposition. A MEC-assisted anaerobic digester was developed by Huang et al. (2021), who demonstrated a positive correlation between known electroactive bacteria and electrotrophic methanogens, and the compensation of MEC power consumption by biomethane. Meanwhile, zero valent ion (ZVI) (Xu et al., 2019) and granular activated carbon (Florentino et al., 2019b; Zhang et al., 2021) could promote BW digestion by enriching electroactive microorganisms tolerant to high ammonia concentrations or stimulating the growth of hydrogenotrophic methanogens and lowering the oxidation–reduction potential (ORP) of the treatment substrate, respectively.

The relatively low biomass growth rate within the anaerobic processes might lead to a long sludge retention time (SRT), which in turn causes a large reactor volume and therefore restricts its wide application. The combination of membrane technology with anaerobic digestion to form AnMBR decouples the hydraulic retention time (HRT) and SRT (Monino et al., 2016; Pan et al., 2022), making it possible to operate the process in a relatively low volume while the SRT is still kept long.

From the perspective of energy generation, bio-electrochemical processes are also applied to BW treatment, showing enhanced efficiency compared to traditional biological processes (Vogl et al., 2016; Liu et al., 2017; Liu et al., 2020). Bio-electrochemical processes include microbial fuel cells (MFCs) aimed at simultaneous organics removal and energy generation, and microbial electrolysis cells (MECs) aimed at pollutant removal and hydrogen production. These processes are designed to remove pollutants and generate energy, and they can simultaneously be used to recover the N, P and K (Castro et al., 2014; Walter et al., 2017; Sharma and Mutnuri, 2019). With the aid of microorganisms that provide electrons at the anode, substrates in BW could be oxidized and combine with oxygen and protons surrounding the cathode to form water (Liu et al., 2017). MECs are able to obtain hydrogen gas in the cathode using carbon sources and ammonium in the substrate, and degrade pollutants with electric current stimulation (Gil-Carrera et al., 2013; Escapa et al., 2016; Barbosa et al., 2019). These bio-electrochemical processes are characterized by energy self-sufficiency and low sludge yields (Zamalloa et al., 2013). In addition, they have also shown potential in the efficient removal of recalcitrant pollutants (Huang et al., 2016; Yang et al., 2022). Processes for nutrient and energy recovery from BW are summarized in Table 4.

4 Perspectives

Water crises and environmental pollution are challenges that continue to threaten the world. The effective treatment and reuse of BW could contribute to both aspects. While BW is high in nutrients and pathogens, numerous technologies have been proposed for the treatment of BW. As outlined in Table 3, Table 4, and Fig. 9, a broad range of technologies and pathways to facilitate nutrient recovery, energy production or water reclamation from BW is available. Treatment strategies of

Table 4 Nutrients and energy recovery from BW

Processes	Products	Recovery					Applications
		C	N	P	K	Energy	
Aerobic composting	Organic fertilizer		√	√	√		Agricultural
Anaerobic digestion	Biogas Nutrient-rich digestate	√	√	√		√√	Agricultural or industrial Agricultural
Microbial fuel cells	Electricity Struvite		√	√		√	Electricity production for lighting Struvite utilization
Microalgae-based process	Microalgae-based bio-fertilizer High Value-added chemicals	√	√	√	√		Agricultural Chemical extraction
Precipitation	Struvite or similar products		√	√√	√		Fertilizer manufacturing

√: Able to recover; √√: High recovery.

different adaptability have their own advantages. For example, aerobic composting is mainly used for biosolid treatment, while filtration processes are only used for wastewater treatment. In the past, biological and ecological systems were popularly used in the world as the main treatment process. Ecological systems, with CWs being a typical strategy, has a long history in domestic wastewater treatment, due to their low cost and energy consumption, as well as decorative and economic benefits. However, traditional biological and ecological processes suffer from relatively slow reaction rates and large land occupation, which exist as the main obstacles for their use in cities. At the same time, considering the odor and pathogens contained in the raw BW and the potential contact between the water body and humans, CW should be built away from residential areas. Additional post-treatment such as a disinfection process is always needed to meet the regulation standard of reuse.

Among biological systems, MBRs display effluent of good quality and a smaller occupation of land area compared to the conventional activated sludge process. It is also effective for pathogen removal. The main drawback for MBRs lies in the high capital cost and the requirement for a long-term professional maintenance because of membrane fouling, hindering their large-scale application in low-income areas. Due to the high level of organics in BW, anaerobic digestion is widely studied for the generation of renewable energy, and UASB is the most applied technology. One disadvantage associated with anaerobic digestion is the low degradation efficiency for some micropollutants, which calls for a reliable disposal for sludge. Moreover, for VTBW, the inhibition effect of high free ammonia concentration on the methanogenesis step is an issue to be considered.

Many innovative and efficient BW treatment strategies, such as membrane and advanced oxidation processes, have emerged in recent years. In addition, due to the increasing awareness of energy and resource crises, water

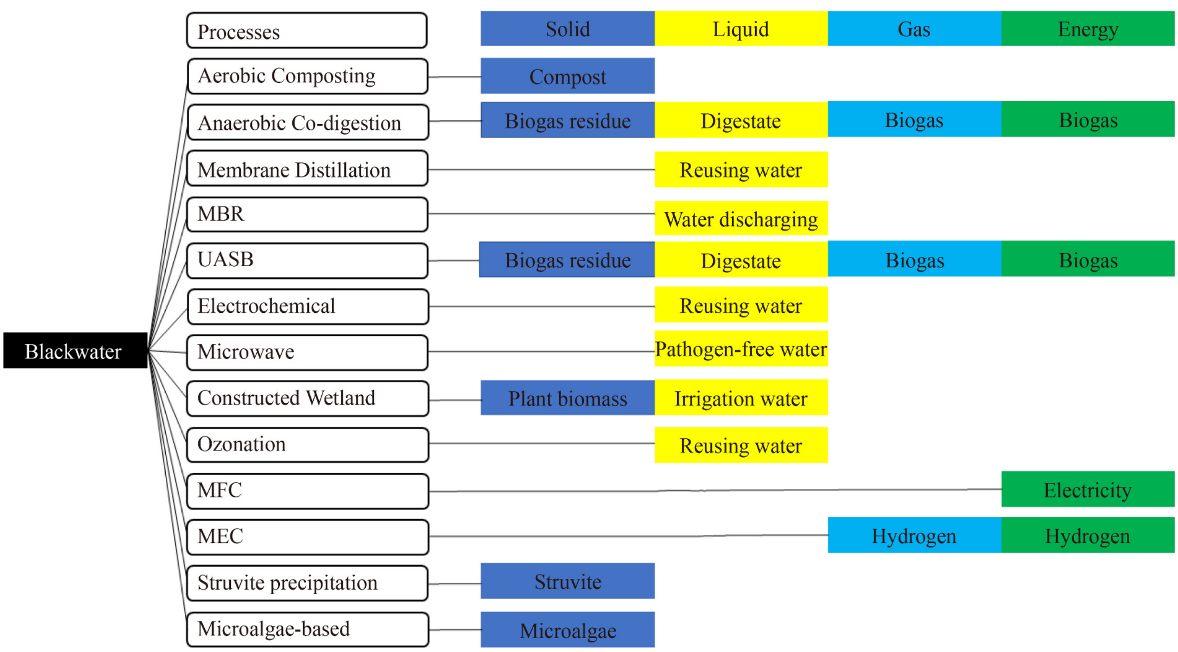

Fig. 9 Typical BW treatment processes and the corresponding products.

and resource recovery from BW has become a new focus. Because raw BW has a complex composition and high pollutant concentration, and membrane separation alone cannot eliminate pollutants, the membrane process is only used as an additional treatment. However, for urine, which is much lower in SS and COD, membrane filtration and membrane distillation are intensively used for dewatering. More commonly, membranes are used as pre- or post-treating processes, showing good performance in SS removal and existing as a barrier for bacteria and viruses.

Chemical and physicochemical treatment processes including AOPs, adsorption and struvite precipitation, are attracting great interest, especially since the Bill & Melinda Gates Foundation (BMGF) called on researchers to "Reinvent the Toilet." Under the cost constraint of $0.05 per person per day including capital and operational costs, electricity-based advanced oxidation processes were intensively reported. These technologies are capable of treating and recycling water at a relatively low cost and avoiding the use and erosion of pipe systems. More importantly, they are extremely effective in pathogen removal, acting as an efficient disinfection process. However, the development of AOPs is currently immature. Most of these AOPs are restricted to the bench scale and are not affordable for governments. More research is needed to promote the practical application of advanced treatment methods and reduce the cost.

In the above description, it can be concluded that BW treatment technologies can be divided into laboratory-scale or practical-scale ones. Drawbacks hindering scale application could be complex composition of real BW (Tansel et al., 2018), cold climate limitation in some areas (Oarga Mulec et al., 2019), as well as high energy consumption of certain technologies (Yu et al., 2021). Due to the influent fluctuation, pipe scaling, temperature variation, the recovery of energy and nutrients is less efficient; therefore technology optimization is necessary for practical application. Indeed, no single technology can ideally solve the problem of BW treatment and countries of different development levels have their own preferences. Therefore, it is difficult to identify the best BW treatment system. Generally, we recommend that a typical BW treatment process include a primary treatment to partially remove influent SS and insoluble substances, biological or chemical processes to degrade most of the pollutants, and a further ecological process to meet the discharging standard, or a disinfection process for advanced treatment. Specific process options should be made based on the need for water discharge, reuse, or nutrient recovery. From the perspective of the development trend of BW treatment technology, considering the increasingly scarce resources and energy, we speculate that the future development trend of BW treatment will be a more integrated system. Nutrients and energy recovery will be more emphasized.

In terms of economy, the cost for certain technology is closely linked with treatment purposes (discharging, reuse for irrigation or toilet flushing) and user number, so it might not be suitable to make a direct comparison (Dorji et al., 2022). Conventional biological and ecological processes are cheap options in terms of both capital expenditure and operating expenditure, making them suitable for water treatment and discharging in developing countries, especially in temperate regions (Boyjoo et al., 2013). In contrast, the capital expenditure of novel technologies, such as electrochemical and membrane technology, are higher due to their relatively immature development and the need for electrode and membrane replacement. Their operating expenditures are also high because the work of professional staff is needed, so they are mostly applied for reuse in centralized areas, such as office blocks (Sahondo et al., 2020; Welling et al., 2020).

5 Conclusions

In this work, the qualitative and quantitative characteristics of BW has been presented. Processes aimed at contamination reduction and water reuse, or resource and energy recovery were reviewed, with technologies covering the whole process from pretreatment to advanced treatment. Each of the treatment systems (physical, biological, chemical or ecological) has respective advantages and drawbacks due to different treatment purposes or development levels. Apart from efficiency and reliability, the selection for a certain technology for BW should take many other factors into consideration: social acceptance, environmental and health impact, economic feasibility and relative regulation support. The selection of certain technologies should meet the local needs and situations. Therefore, the adaptation of a certain technology is the product of all the above factors.

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