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Review

Prospects of hybrid rainwater-greywater decentralised system for water recycling and reuse: A review

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ABSTRACT

This review examines the prospects of a decentralised hybrid rainwater-greywater system to simultaneously alleviate water scarcity and address technical, environmental, and cost constraints. This includes (1) a review on the physicochemical and microbiological characteristics of rainwater and greywater to determine the necessary treatment options; (2) a review of individual components and potential treatment trains for hybrid systems; and (3) an evaluation of configurations for hybrid systems. The literature review reveals that both untreated rainwater and greywater are highly variable in quality and quantity, and so an equalisation basin is proposed to normalise influent into a hybrid system. Both rainwater and greywater should not be reused without treatment due to the presence of pathogens such as *Aeromonas*, *Salmonella*, *Pseudomonas*, and *Staphylococcus*. Based on the literature, hybrid systems are categorised under three configurations: (a) separate treatments of rainwater and greywater; (b) rainwater reused in washing machines prior to reuse as greywater; and (c) combined treatment of rainwater and greywater. In all three designs, rainwater requires only first-flush diversion and disinfection. Combined rainwater-greywater mixtures should be treated as greywater. Greywater requires chemical, biological, and physical treatment to meet non-potable reuse standards. Chemical processes are effective at removing solids, organics, and surfactants in light greywater, whereas aerobic biological processes are effective at organics removal in mixed and dark greywaters with high organic strength. Physical processes, particularly membrane filtration, are recommended for polishing effluents from chemical or biological treatment as membranes foul frequently and are costly. Subsequently, a combination of ozone or UV with chlorine is recommended to eradicate chlorine-resistant *Cryptosporidium* oocysts from hybrid rainwater-greywater systems and prevent microbial regrowth.

Contents

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1. Introduction

Increasing water demand, diminishing water quality, and insufficient water infrastructure all contribute to the growing problem of urban water scarcity (Falkenmark et al., 2007). Traditional responses to urban water scarcity have been to construct additional water infrastructure, such as dams, in order to increase mains water supplies. The resource-intensive nature of these solutions have prompted diversification of water resources, as high quality mains water is not required for non-potable activities which comprise more than 20% of a building's mains water demand (Campisano and Modica, 2010).

In an urban setting, both rainwater and greywater are alternative water resources that can be collected, treated and distributed via decentralised rainwater harvesting and greywater recycling systems. Greywater is defined as wastewater sourced from kitchen sinks, showers or baths, and washing machines, excluding sewage or blackwater contributions from toilets (Eriksson et al., 2002), whereas rainwater is defined as rainfall precipitation collected from a roof catchment (Thomas and Greene, 1993). Both systems however, suffer individual drawbacks when installed separately. Separate implementation of either decentralised rainwater or greywater systems increases the overall capital costs: Ghisi and Ferreira (2007) estimated rainwater and greywater systems alone to cost approximately USD 1224 per system, whereas a hybrid system costs approximately USD 1929. Furthermore, rainwater supply is limited by low rainfall precipitation in arid climates, small catchment areas common to high-rise buildings in urban areas, and small storage tanks in rainwater harvesting systems (Aladenola and Adeboye, 2010). Rainwater harvesting systems alone supplied water to meet only 10% of non-potable toilet flushing demand in California due to the arid climate (Loux et al., 2012), and 14.7–17.7% of non-potable water demand in Brazil due to the limited tank size (Ghisi and Ferreira, 2007).

A hybrid rainwater-greywater system is proposed to overcome this drawback. Greywater recycling systems have high water saving potential in urban areas with high population densities and high daily per capita mains water consumption (Ghisi and Ferreira, 2007; Ghisi and Mengotti De Oliveira, 2007). Furthermore, addition of a greywater recycling system to an existing rainwater harvesting system will help offset the seasonal nature of rainfall as greywater is generated independent of climate conditions (Loux et al., 2012). When compared to conventional rainwater or greywater systems, hybrid systems are less sensitive to changes in occupancy that would severely affect greywater production, and are less sensitive to climate changes, as they can switch from greywater reuse in dry seasons to rainwater reuse in monsoon seasons. Hybrid systems present the highest water saving potential and shortest payback period, followed by greywater recycling and then by rainwater harvesting (Ghisi and Ferreira, 2007). Additionally, hybrid systems simultaneously manage stormwater at its source, thus reducing surface runoff and flooding risks (Zhang et al., 2010; Kim and Yoo, 2009), in addition to reducing wastewater volumes by concentrating pollutants to centralised wastewater treatment plants (Penn et al., 2013).

This review paper examines the prospects of a hybrid rainwater-greywater decentralised system. The objectives of this review article were to (1) evaluate the quality of rainwater and greywater in order to recommend several water treatment design options for hybrid systems, (2) evaluate existing material selection and flow options for hybrid systems, and (3) review available treatment trains for combined or separate treatment of rainwater and greywater in hybrid systems.

2. Quality of rainwater and greywater

2.1. Physicochemical characteristics

Table 1 summarises the physicochemical characteristics of rainwater, light greywater, dark greywater, and mixed greywater. Light greywater is sourced from showers, baths and washbasins, while dark greywater is sourced from kitchen sinks, washing machines and dishwashers. These characteristics have been benchmarked against the Malaysian drinking water standards (DWS) in order to illustrate how rainwater and greywater typically exceed the maximum standard limits. The Malaysian DWS is selected as non-potable reuse standards have limited parameters. The Malaysian DWS listed in Table 1 are identical to, or more stringent than the World Health Organization (WHO) DWS with the exceptions of barium (0.7 mg/L), boron (2.4 mg/L), and nitrite (0.9 mg/L), and will henceforth be referred to as 'DWS'.

Most physicochemical parameters for raw rainwater fall within the DWS, with the exception of pH and heavy metals concentrations (Huston et al., 2009; Yaziz et al., 1989). Table 1 shows that the pH of rainwater can range from weakly acidic (pH 3.1) to weakly alkaline (pH 11.4). The pH of rainwater occasionally exceeds the DWS range of 6.5–9.0. In contrast, greywater has a smaller pH range, with values falling near pH 7.0.

Biological oxygen demand (BOD_5), chemical oxygen demand (COD) and total organic carbon (TOC) indicate the organic strength

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Table 1
Physicochemical characteristics of rainwater, light greywater, dark greywater, mixed greywater, and Malaysian drinking water standards (DWS).

Parameter	Unit	Rainwater[a]		Greywater (light)[b]		Greywater (dark)[c]		Greywater (mixed)[d]		Malaysia DWS
		min	max	min	max	min	max	min	max	
pH	–	3.10	11.40	6.40	8.10	6.48	10.00	6.06	8.38	6.5–9.0
Alkalinity	mg/L	0.50	61.00	24.00	43.00	83.00	200.00	0.00	0.00	
Hardness	mg/L	0.00	270.00	0.00	0.00	0.00	0.00	0.00	0.00	500.0
Conductivity at 25 °C	μS/cm	3.00	1017.00	82.00	1565.00	190.00	2457.00	2097.00	2097.00	
Turbidity	NTU	0.20	303.50	60.00	240.00	50.00	210.00	37.00	173.00	5
Colour	Pt-Co	0.40	310.50	60.00	100.00	50.00	70.00	206.00	550.00	15.0
Total dissolved solids (TDS)	mg/L	1.00	750.00	520.00	787.00	590.00	1396.00	280.00	350.00	1000.0
Total suspended solids (TSS)	mg/L	1.00	153.00	40.00	303.00	68.00	625.00	16.00	2850.00	
Total solids (TS)	mg/L	20.00	200.00	208.00	1090.00	658.00	2021.00	570.00	700.00	
Biological Oxygen Demand (BOD_5)	mg/L	0.00	3.00	33.00	424.00	48.00	890.00	1354.00	1354.00	
Chemical Oxygen Demand (COD)	mg/L	8.74	23.83	76.00	645.00	725.00	1340.00	92.00	5470.00	
Total organic carbon (TOC)	mg/L	0.00	0.00	40.00	120.00	110.00	582.00	940.00	940.00	
Total nitrogen (N)	mg/L	0.45	1.92	0.00	0.00	0.00	0.00	8.00	11.00	
Total phosphorous (P)	mg/L	0.21	50.00	0.11	1.80	0.06	42.00	0.89	11.00	
Ammoniacal-nitrogen (NH_3-N)	mg/L	0.00	0.00	0.10	15.00	0.10	10.70	0.60	26.00	1.5
Nitrate-nitrogen (NO_3-N)	mg/L	0.00	72.40	0.34	0.90	0.45	1.60	0.50	3.10	10.0
Nitrite-nitrogen (NO_2-N)	mg/L	0.00	2.45	0.00	0.00	0.00	0.00	0.00	0.00	
Total Kiedjahl nitrogen (TKN)	mg/L	0.00	0.00	4.60	20.00	1.00	40.00	0.00	0.00	
Dissolved oxygen (DO)	mg/L	4.41	6.79	0.00	0.00	0.00	0.00	0.19	1.60	
Aluminium (Al)	μg/L	80.20	336.00	1000.00	1000.00	1000.00	21,000.00	1480.00	3390.00	200.0
Ammonium (NH_4^+)	μg/L	0.00	35,400.00	0.00	0.00	0.00	0.00	0.00	0.00	
Arsenic (As)	μg/L	0.00	27.10	1.00	1.00	1.00	7.00	0.00	0.00	10.0
Boron (B)	μg/L	11.00	56.00	350.00	440.00	20.00	400.00	0.00	0.00	
Barium (Ba)	μg/L	0.00	11.20	0.00	0.00	0.00	0.00	15.50	21.80	
Cadmium (Cd)	μg/L	0.00	0.40	0.01	10.00	0.52	10.00	0.01	0.22	3.0
Calcium (Ca)	μg/L	0.00	31,150.00	3500.00	7900.00	3900.00	12,000.00	31,600.00	38,000.00	
Chromium (Cr)	μg/L	0.00	4.80	0.00	0.00	0.00	0.00	2.06	5.46	50.0
Chloride (Cl^-)	μg/L	0.00	164,000.00	9000.00	284,000.00	9000.00	450,000.00	0.00	0.00	250,000.0
Copper (Cu)	μg/L	1.10	4500.00	60.00	120.00	50.00	322.00	47.00	70.20	1000.0
Fluoride (F)	μg/L	0.00	1000.00	0.00	0.00	0.00	0.00	0.00	0.00	600.0
Iron (Fe)	μg/L	0.00	1390.00	340.00	1100.00	290.00	1000.00	180.00	570.00	300.0
Lead (Pb)	μg/L	2.00	271.00	3.00	10.20	5.00	33.00	2.14	10.00	10.0
Magnesium (Mg)	μg/L	0.00	9350.00	1400.00	2300.00	1100.00	2900.00	5300.00	6220.00	150,000.0
Manganese (Mn)	μg/L	0.50	533.00	0.00	0.00	0.00	0.00	9.55	14.30	100.0
Mercury (Hg)	μg/L	0.00	0.00	0.00	0.00	0.00	0.00	0.02	36.00	1.0
Nickel (Ni)	μg/L	0.00	12.20	5.15	26.50	0.00	0.00	4.45	28.10	20.0
Potassium (K)	μg/L	0.00	8730.00	1.63	5200.00	15.60	17,000.00	7690.00	8850.00	
Phosphates (PO_4-P)	μg/L	0.00	620.00	400.00	15,000.00	400.00	169,000.00	0.00	0.00	
Sodium (Na)	μg/L	0.00	32,320.00	7400.00	151,000.00	49,000.00	530,000.00	61,400.00	92,400.00	200,000.0
Zinc (Zn)	μg/L	0.50	3200.00	59.00	6300.00	90.00	320.00	55.30	77.80	3000.0

[a] **Rainwater** data compiled from: Yaziz et al. (1989), Simmons et al. (2001), Zhu et al. (2004), Adeniyi and Olabanji (2005), May and Prado (2006), Sazakli et al. (2007), Abdulla and Al-Shareef (2009), Ward (2010), Vialle et al. (2011), Moon et al. (2012).
[b] **Light greywater** data compiled from: Christova-Boal et al. (1996), Surendran and Wheatley (1998), Friedler (2004), Revitt et al. (2011), Santos et al. (2012), Antonopoulou et al. (2013).
[c] **Dark greywater** data compiled from: Antonopoulou et al. (2013), Christova-Boal et al. (1996), Friedler (2004), Surendran and Wheatley (1998).
[d] **Mixed greywater** data compiled from: Palmquist and Hanæus (2005), Eriksson et al. (2009), Katukiza et al. (2014), Bani-Melhem et al. (2015).

of a wastewater, while total solids consist of total suspended solids (TSS) and total dissolved solids (TDS). Dark greywater has the highest organics and solids contents, followed by light greywater, mixed greywater, and rainwater. Additionally, both light and dark greywaters are often unable to meet non-potable and potable water quality standards without further treatment because of their high organics and solids contents (Eriksson et al., 2002).

Out of all the heavy metals, lead (Pb) most frequently exceeds the DWS limit of 10 μg/L in raw rainwater. For instance, Pb concentrations exceeded 10 μg/L in all rainwater samples in a Malaysian study (Yaziz et al., 1989), whilst a previous study in Auckland, New Zealand showed that 18 rainwater samples (14.4% of total samples) had Pb concentrations exceeding New Zealand's water standards (Simmons et al., 2001). Similar results were reported in Brisbane, Australia where 25 rainwater samples (14.2%) exceeded DWS (Huston et al., 2009) and 33% of 49 rainwater tanks in Melbourne, Australia had Pb concentrations exceeding the DWS (Magyar et al., 2008). Huston et al. (2012) attributed Pb sources in rainwater tanks to Pb flashing or paint (58%), plumbing (16%) and atmospheric deposition (21%). Their study highlighted that good rainwater harvesting system design is paramount in minimising health risks associated with rainwater harvesting and use, as prolonged consumption and exposure to Pb may result in learning and behavioural disorders (Needleman, 2004).

Comparatively, fewer studies have reported heavy metals contamination in greywater. The highest Pb concentration of 33 μg/L in laundry greywater was reported by Surendran and Wheatley (1998), while a significantly lower concentration range of 0.61–10.2 μg/L reported in bathroom or shower greywaters (Table 1). Potential sources of Pb in greywater include plumbing materials, cutlery and dental fillings (Eriksson et al., 2009; Revitt et al., 2011). However, heavy metals are not always present in greywater: Friedler (2004) found that silver, cadmium, chromium, copper, manganese, nickel and zinc were below detection limits. The high variability of heavy metal concentrations indicates that greywater quality is dependent on both source water quality and the inhabitants' lifestyles.

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2.2. Microbiological characteristics

2.2.1. Faecal indicators

Physicochemical water quality parameters are usually of secondary importance to microbiological quality. This is because waterborne pathogens pose a significant health threat in non-potable reuse applications with risks of cross-contamination, ingestion or body contact (Meera and Ahammed, 2006). Despite this, it is not economically feasible to test for all pathogens in source waters. Thus, common faecal indicators such as *Escherichia coli (E. coli),* faecal coliforms, *Streptococci* and *Enterococci* are used to indicate faecal contamination and dictate the possibility that other pathogens may be present.

Tables 2 and 3 summarise the microbiological quality of rainwater and greywater. Table 2 shows that rainwater contains large variation in *E. coli* concentrations (up to 4.2 $\log_{10}$ CFU/100 mL) and prevalence. In Australia, Ahmed et al. (2014) found that 76% of 72 rainwater samples from Southeast Queensland were tested positive for *E. coli*, as compared to 33% of 49 samples from Victoria (Spinks et al., 2006b) and 30% of 974 samples in Adelaide (Rodrigo et al., 2011). A similar observation was made for different countries, whereby: 79% of 14 rainwater samples tested positive for *E. coli* in the Netherlands (Albrechtsen, 2002), as compared to 41% of 156 samples in Greece (Sazakli et al., 2007) and 72% of 180 samples in South Korea (Lee et al., 2010). The high frequency of rainwater samples which tested positive for *E. coli* indicates that rainwater is often contaminated by faeces from birds and small mammals (Despins et al., 2009). This shows the need for robust disinfection measures capable of reducing *E. coli* concentration to 0 CFU/100 mL in order to minimise potential health risks from rainwater.

Gilboa and Friedler (2008) found that all 44 greywater samples contained faecal coliforms, whilst mixed greywater had tested positive for at least one faecal indicator (Table 3). Additionally, Table 3 shows that *E. coli* concentrations are significantly higher in greywater than rainwater, ranging up to 6.2 $\log_{10}$ CFU/100 mL. Greywater sourced from showers and baths has higher faecal indicators than laundry discharge (Dixon et al., 2000; Rose et al., 1991). The number of samples containing *E. coli* are not typically reported in previous greywater studies (Friedler et al., 2011). The high prevalence and concentrations of faecal indicators indicate that faecal contamination of greywater is expected (Winward et al., 2009) and similar robust disinfection measures are required.

2.2.2. Bacterial and protozoan pathogens

Tables 2 and 3 presents the prevalence and concentration of bacterial pathogens in rainwater and greywater respectively. The most frequently reported bacterial pathogens in rainwater are *Aeromonas* (43 samples), *Salmonella* (24 samples) and *Legionella* (22 samples). Meanwhile, the most frequently reported bacterial pathogens in greywater are *Pseudomonas* (51 samples),

Table 2
Faecal indicators and pathogens in rainwater. First row of each microbe indicates positive samples (total samples), whereas the second row indicates concentration in $\log_{10}$ CFU/100 mL, unless stated otherwise. NR = not reported.

	Yaziz et al. (1989)	Crabtree et al. (1996)	Uba and Aghogho (2000)	Albrechtsen (2002)	Simmons et al. (2001)	Birks et al. (2004)	Sazakli et al. (2007)	Ward et al. (2010)	Ahmed et al. (2010)	Ahmed et al. (2014)
Location	Malaysia	U.S. Virgin Islands	Nigeria	Denmark	New Zealand	UK	Greece	UK	Australia	Australia
Faecal indicators										
E. coli				11 (14) 0.6–3.0		NR (NR) <0.0–4.2	63 (156) <0.0–2.4			53 (72) 0.5–3.7
Faecal coliforms	NR (24) <0.0–1.1	9 (26) <0.0–2.9	NR (NR) <0.0		70 (125) <0.0–2.9			NR (32) <0.0–3.2		
Total coliforms	NR (24) 1.4–1.9	15 (26) <0.0–3.5	NR (NR) 0.5–1.6		NR (125) 0.0–4.3	NR (NR) <0.0–6.4	125 (156) <0.0–2.8	NR (32) <0.0–3.4		
Enterococci					NR (125) 0.0–3.7	NR (NR) <0.0–2.8	45 (156) <0.0–1.5	NR A (32) <0.0–3.2		68 (72) 0.3–3.6
Streptococci			NR (NR) <0.0							
Bacterial pathogens										
Aeromonas				2 (14) 1.0–1.5	20 (125) NR					21 (72) 2.1–4.5
Campylobacter				2 (17) ND	0 (115) NR	0 (2) 0			1 (214) ND	
Clostridium							0 (156) 0			
Legionella				5 (21) 0	0 (23) NR	0 (2) 0			12 (214) 2.8–3.2	5 (72) 3.2–4.0
Mycobacteria				1 (14) 0						
Pseudomonas			NR (NR) <0.0–2.9	1 (14) 0–1.3			0 (156) 0			9 (72) 1.4–4.3
Salmonella			NR (NR) <0.0–2.9		1 (115) NR	0 (2) 0		0 (2) 0	23 (214) 2.8–3.6	
Shigella			NR (NR) <0.0–2.3							
Staphylococcus										11 (72) 2.8–4.6
Vibrio			NR (NR) 1.0–3.0							
Protozoan pathogens										
Cryptosporidium oocysts/L		22 (45) 0–0.7		6 (17) 0–50	2 (50) NR	0 (2) 0		0 (2) 0	0 (214) 0	
Giardia cysts/L		12 (45) 0–0.04		0 (17) 0	0 (50) NR	1 (2) 0.2			21 (214) 0.6–3.6	

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Table 3
Faecal indicators and pathogens in greywater. First row of each microbe indicates positive samples (total samples), whereas the second row indicates concentration in $\log_{10}$ CFU/100 mL, unless stated otherwise. LA = laundry; WB = washbasin; SH = shower; NR = not reported.

	Birks et al. (2004)	Birks and Hills (2007)	Casanova et al. (2001)	Gilboa and Friedler (2008)	Winward (2007)
Location	UK	UK	Arizona, U.S.A.	Israel	UK
Source	WB	LA, WB, SH	LA, WB, SH	SH, BA, WB	SH, BA, WB
Faecal indicators					
E. coli	NR (NR)	NR (NR)			NR (58)
	<0.0–6.4	5.6			2.8
Faecal coliforms			NR (20)	44 (44)	
			3.5–6.9	3.6	
Total coliforms	NR (NR)	NR (NR)	NR (20)		NR (57)
	3.4–6.4	7.3	5.8–8.3		5.4
Enterococci	NR (NR)	NR (NR)			NR (56)
	<0.0–4.3	3.4			2.8
Streptococci			NR (20)		
			0.9–3.0		
Bacterial pathogens					
Aeromonas					
Campylobacter	0 (3)	0 (8)			6 (9)
	0	0			0
Clostridium				6 (7)	NR (12)
				0.7	3.1
Legionella	3 (6)	0 (8)			
	4.2–4.9	0			
Mycobacteria					
Pseudomonas			NR (20)	42 (45)	9 (9)
			2.3–5.2	3.5	4.4
Salmonella	0 (3)	1 (8)			7 (13)
	ND	NR			0
Shigella	0 (3)				
	ND				
Staphylococcus			0 (20)	34 (40)	8 (8)
			0	4	3.4
Vibrio					
Protozoan pathogens					
Cryptosporidium	2 (3)	0 (8)			
oocysts/L	0.4–1.21	0			
Giardia	2 (3)	5 (8)			
cysts/L	0.61–1.21	0.5–1.5			

Staphyloccocus (42 samples) and *Salmonella* (9 samples). Faecal indicators were correlated to *Aeromonas hydrophila*, *Staphylococcus aureus*, *Pseudomonas aeruginosa*, and *Legionella pneumophila* (Ahmed et al., 2014). These findings suggest that it may be possible to use the most frequently occurring bacterial pathogen (i.e. *Aeromonas* for rainwater, *Pseudomonas* for greywater) as a surrogate for microbial contamination rather than the faecal indicators.

At present, limited information is available on protozoan parasites such as *Cryptosporidium* and *Giardia* in both rainwater and greywater. Table 2 shows that 9.1% of 330 samples and 10.4% of 328 samples had tested positive for *Cryptosporidium* and *Giardia* in rainwater. On the other hand, Table 3 shows that 18.2% and 38.9% of 11 samples from a similar study had tested positive for *Cryptosporidium* and *Giardia* in greywater. Despite their low prevalence in rainwater, high concentrations have been reported for both *Cryptosporidium* (up to 70 oocysts/100L) (Crabtree et al., 1996) and *Giardia* (up to 360 cysts/100L) (Ahmed et al., 2010). Ahmed et al. (2011) attributed the differences in *Cryptosporidium* and *Giardia* prevalence to local fauna, such as possums. Birks and Hills (2007) reported that 63% of all greywater samples were tested positive for *Giardia*, ranging from 0.5 to 1.5 cysts/L. In contrast, Casanova et al. (2001) reported no *Cryptosporidium* or *Giardia* in greywater.

2.2.3. Viruses

Dixon et al. (2000) discussed that the number of viruses found in greywater is dependent on the health of the population generating the greywater. Thus, an increase in the occupancy rate of a building would result in a higher likelihood of viral infection. This is supported by Gilboa and Friedler (2008), where neither F-RNA nor somatic coliphages was found in raw greywater supplied from non-infected hosts. Similarly, O'toole et al. (2012) found that 18% of 111 greywater samples contained enterovirus, norovirus, or rotavirus. These results suggest that while viruses may not be as prevalent as bacterial pathogens or faecal indicators in greywater, disinfection systems are required to reduce viruses to 0 PFU/100 mL.

3. Hybrid rainwater-greywater system components

Hybrid systems incorporate both rainwater harvesting and greywater recycling systems, and thus require separate design considerations for both rainwater and greywater components.

3.1. Rainwater harvesting system components

A rainwater harvesting system consists of four major components: (1) roof catchment, (2) pre-treatment, (3) storage tank, and (4) additional treatment. Fig. 1(a) illustrates a generic rainwater harvesting system used to supply rainwater for toilet flushing and irrigation purposes. A mains water top-up system with level controller ensures that the plumbed rainwater tank does not run dry during periods without rainfall.

3.1.1. Catchment

Rainwater is collected via a roof catchment. Both the roof catchment incline angle and material selection are important design parameters in rainwater harvesting systems. Smooth, sloping roofs with higher incline angles and thus higher runoff coefficients (RC) may harvest up to 50% more rainwater than flat,

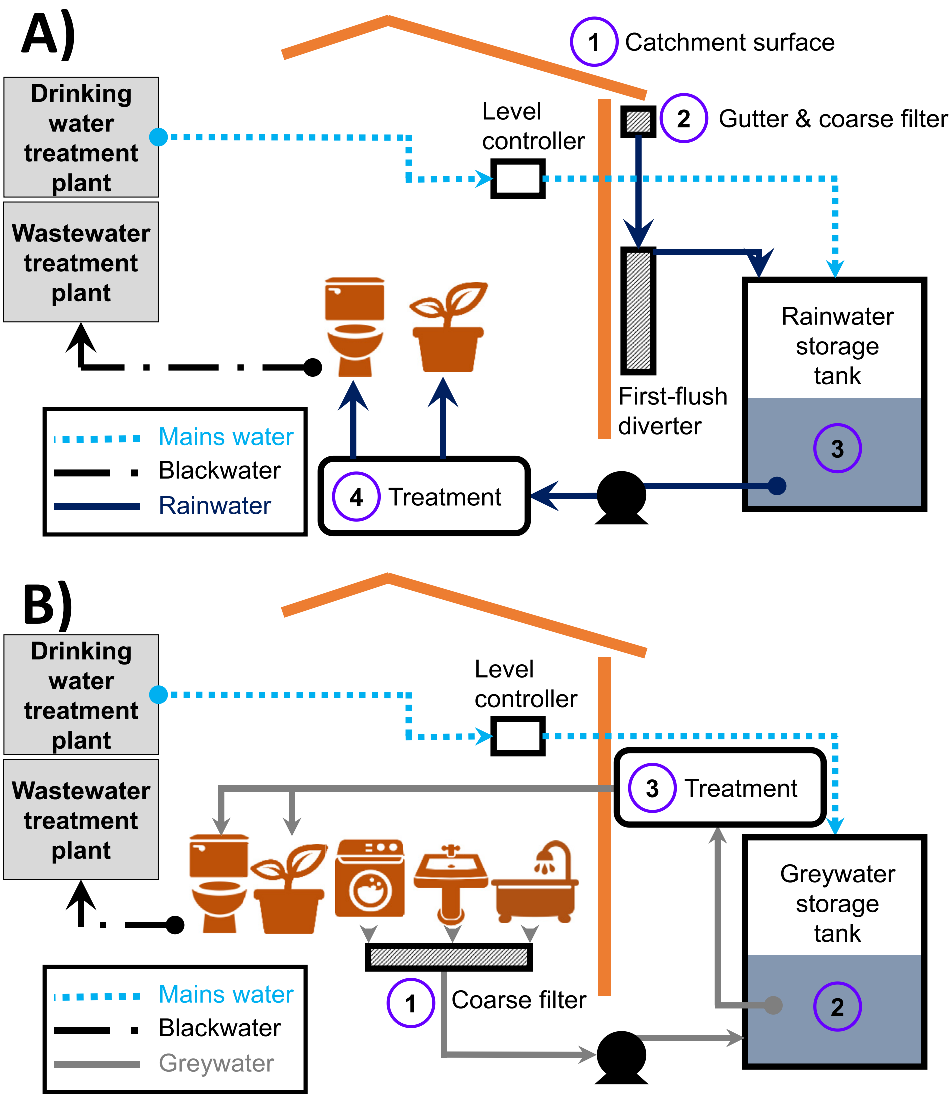

Fig. 1. (a) Rainwater harvesting system and (b) Greywater recycling system.

rough roofs, and generally have rainwater of better quality (Farreny et al., 2011). Catchment material selection determines which pollutants leach from the catchment into rainwater. Concrete catchments significantly increase rainwater pH due to the dissolution and leaching of calcium carbonate (Yaziz et al., 1989; Zhu et al., 2004). Similarly, galvanised iron roofs are known to result in higher Zn concentrations in rainwater (Yaziz et al., 1989; Thomas and Greene, 1993), while elevated Cu and Pb concentrations result from Cu and Pb flashings on roofs (Förster, 1996; Magyar et al., 2007). Runoff from metal roofs contains significantly lower total coliform and faecal coliform concentrations than any other roof material, likely due to higher surface temperatures inactivating coliforms (Mendez et al., 2011). Wood shingle roofs presented the poorest rainwater quality due to high concentration of dissolved heavy metals, likely due to the rough surface and low incline angle of the wooden roof (Chang et al., 2004). Green roofs, which are a new trend in contemporary roof catchment design, generate runoff containing significantly higher amounts of dissolved ions and dissolved organic carbon (DOC) concentrations than conventional roofing materials (Mendez et al., 2011; Zhang et al., 2014). Runoff with DOC concentrations exceeding water standard limits of 3.5 mg/L are not recommended for treatment systems with chlorination, as health-detrimental disinfection by-products (DBPs) may form (Mendez et al., 2011). Sloped metal and concrete tile roofs with RCs close to 1 are therefore recommended for rainwater harvesting systems as they produce better quality rainwater.

3.1.2. Pre-treatment

Pre-treatment is carried out after rainwater is collected from the roof catchment and diverted through the gutter (Fig. 1(a)). Pre-treatment encompasses coarse filtration and first-flush diversion. Coarse filters, such as leaf and gutter screens, are first in line as they remove large solids (e.g. twigs, leaves). Coarse filters prevent accumulation of organic materials in the rainwater tank, lower rates of corrosion-erosion to the piping system, reduce likelihood of pump damage, and reduce the cleaning frequency of downstream filters (Brewer et al., 2001).

The "first-flush" phenomenon describes initial volumes of rainwater containing high turbidity, conductivity and total solids due to debris accumulation on the roof from antecedent dry periods (Yaziz et al., 1989). First-flush diverters discard this polluted initial volume, thereby allowing for maximised rainwater quality in the tank without compromising its water saving efficiency. First-flush diverters are best installed after coarse filters in order to prevent clogging. To date, there is no consensus on the minimum first-flush volume. Yaziz et al. (1989) found that rainwater collected after the initial 0.3 mm (i.e. 30 L for 100 m^2 roof area) of rainfall in Malaysia contained no faecal coliforms, although Förster (1999) defines the first flush as the initial 1–2 mm of rainfall in Germany. In contrast, Gikas and Tsihrintzis (2012) found that rainwater collected after the initial 0.11–0.13 mm contained significantly lower TSS, magnesium, calcium and sodium in Greece. The wide variation between the first-flush volumes suggests that rainwater quality is highly site-specific due to variation in rainfall intensity, frequency and duration. A minimum value of 0.11 mm is recommended to maximise rainwater quality without diminishing its water saving potential.

3.1.3. Storage tank

After the first-flush diverter, rainwater is stored in a storage tank. Despins et al. (2009) showed that the contribution of storage tank materials to rainwater quality is significantly less than that of catchment materials, although concrete tanks can increase the pH of stored rainwater due to leaching of calcium carbonate (Zhu et al., 2004). Tank design is a key parameter affecting both the quality and quantity of stored rainwater (Magyar et al., 2011). For instance, the presence of covers and screens on rainwater tanks resulted in lower total coliforms (Dillaha Iii and Zolan, 1985). Therefore, a tightly sealed rainwater tank is recommended to improve microbiological quality and prevent mosquito breeding (Gould and Nissen-Petersen, 1999). Moreover, tank inlet design affects the degree of solid re-suspension in stored rainwater. Magyar et al. (2011) found that a side inlet resulted in reduced solid re-suspension than a centrally positioned inlet.

Rainwater quality is better at the top of a tank than the bottom, as suspended solids, heavy metals (e.g. Al, Fe, Pb, and Zn) (Magyar et al., 2007) and faecal coliforms tend to settle at the bottom of the tank as sludge (Coombes et al., 2000b; Amin et al., 2013). Thus, the withdrawal tap should be placed at least 0.5 m above the bottom of the tank to prevent accidental sludge withdrawal (Coombes et al., 2000a; Huston et al., 2012). The tap can be placed closer to the bottom of the rainwater tank to reduce dead volume if a floating filter is used to extract rainwater (Leggett et al., 2001). In this instance, regular tank de-sludging and maintenance should be carried out to prevent accumulation of heavy metals in tank sediment.

3.1.4. Additional treatment

Prior to reuse, rainwater may undergo additional filtration and disinfection treatment in order to lower health risks from rainwater reuse. Section 4.2 discusses additional filtration options while Section 4.5 discusses disinfection systems for both rainwater harvesting and greywater recycling systems in detail.

3.2. Greywater recycling system components

3.2.1. Collection system

A greywater recycling system consists of three major components: (1) a collection system with a coarse filter to remove gross solids (e.g. hair, lint); (2) a storage tank to equalise volumes of incoming greywater; and (3) a mandatory treatment train. Fig. 1(b) illustrates a generic greywater recycling system treating greywater from the washing machine, washbasins, and showers/baths. A mains water top-up system with a level controller is used to ensure that the greywater tank does not run dry when supplying toilet flushing or irrigation activities. Although Fig. 1(b) shows a configuration where mixed greywater is collected, greywater recycling systems can choose to recycle only light or dark greywater, resulting in different treatment trains.

3.2.2. Storage tank

Greywater enters the storage tank after it has been collected and coarsely filtered. Unlike rainwater harvesting systems, where storage tanks may appreciably contribute to the pollutant load in rainwater, greywater recycling studies focus more on the treatment portion, and are less concerned with material selection. Storage tanks for raw greywater, however, should be sized so that untreated greywater is not stored for more than 24 h (Dixon et al., 1999; Liu et al., 2010), as malodorous compounds and exponential coliform growth have been reported in greywater stored beyond 24 h (Dixon et al., 2000). Stored greywater from the tank should be treated prior to reuse, and potential treatment trains are discussed in Sections 4.2–4.5.

4. Treatment trains for hybrid systems

4.1. Water quality standards

Both rainwater and greywater should not be reused without treatment due to the adverse health risks involved. The treatment train selected for hybrid rainwater-greywater systems should be 'fit-for-purpose', or tailored to the end-use. Non-potable reuse applications have varying water quality specifications which depend on their end-use. Table 4 summarises several non-potable water quality guidelines and standards for treated rainwater and greywater. Data in Table 4 indicates large variation between guidelines. There is currently no enforceable international water reuse guideline for reclaimed water, thus hampering rainwater and greywater reuse (Li et al., 2009). Li et al. (2009) proposed two different water reuse standards based on unrestricted and restricted end-uses. Unrestricted irrigation is irrigation of crops that might be eaten uncooked, whereas restricted irrigation is irrigation of cereal, industrial, and fodder crops. Unrestricted end-uses have lower water quality requirements than restricted end-uses.

4.2. Physical treatment

Physical treatment options for hybrid rainwater-greywater systems range from filtration by granular media and membranes or sedimentation. Physical treatment is often used as pre-treatment prior to chemical or biological treatment, or as a polishing step prior to disinfection as TSS removal is known to improve the efficacy of downstream disinfection processes (Winward et al., 2008b).

Ahammed and Meera (2010) treated rainwater with two rapid sand filters: one coated with Fe hydroxide and the other with both Mn oxide and Fe hydroxide. Both filters produced treated effluents with lower turbidity, total coliforms, Zn, and Pb concentrations than an uncoated sand filter. The higher removal efficiencies were attributed to a net positive charge on the Fe hydroxide filter and a net negative charge on the dual filter. Rapid sand filters are capable of high throughput, but require frequent backwashing and exhibit low coliform removal. A low-cost, low throughput alternative to rapid sand filters is the slow sand filter, which is effective at removing coliforms. A slow sand filter and chlorination step treating rainwater collected from an airport reduced total coliforms and *E. coli* by 4 and 3 $\log_{10}$ units, although breakthroughs in TSS, turbidity and COD were observed (Moreira Neto et al., 2012). Paper, fibre and fabric filters may be utilised in place of granular media, although they are insufficient as standalone treatment units: a lignocellulosic fibre filter modified with Al oxide removed 22% of total nitrogen, 32% of total phosphorous and 33% of turbidity from rainwater influent (Kim et al., 2007).

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Table 4
International urban water reuse standards (adapted from U.S. Environmental Protection Agency (EPA) (2004))[m = mandatory; g = guideline].

	Faecal coliforms (max CFU/100 mL)	Total coliforms (max CFU/100 mL)	BOD_5 (max mg/L)	Turbidity (max NTU)	TSS (maxvmg/L)	DO (min (% sat.)	pH	Cl_2 residual (min mg/L)
Australia (NSW)	1	4	20	2				
Arizona	1			1			4.5–9.0	
California		2.2		2				
China (GB/T 18920–2002; irrigation) (m)		3	20	10		1	6.0–9.0	0.2
China (GB/T 18920–2002; toilet flushing) (m)		3	10	5		1	6.0–9.0	0.2
Cyprus	50		10		10			
European Commission bathing water (m)	2000	10,000		1		80–120	6.0–9.0	
France	1000							
Germany (g)	100	500	20	1 - 2 (m)	30	80–120	6.0–9.0	
Japan (m)	10	10	10	5			6.0–9.0	
Israel		12 (80%)	15		15	0.5		0.5
Kuwait (crops not eaten raw)		10,000	10		10			1
Kuwait (crops eaten raw)		100	10		10			1
Malaysia (Class IIB - recreational water) (m)	400	5000	3	50	50	5–7	6.0–9.0	
Malaysia (Class IV - irrigation) (m)	5000	50,000	12		300	<3	5.0–9.0	
Oman (11A)	200		15		15		6.0–9.0	
Oman (11B)	1000		20		30		6.0–9.0	
Singapore (toilet flushing) (g)		10	5	2			6.0–9.0	0.5–2.0
South Africa	0							
Spain (Canary Islands)		2.2	10	2	3		6.5–8.4	1
Texas (m)	75 (m)		5	3				
Tunisia			30		30	7	6.5–8.5	
UAE		100	10		10			
UK Bathing Water Criteria	2000 (m)	10,000 (m)		1		80–120	6.0–9.0	
US EPA (g)	14 for any sample, 0 in 90%		10	2			6.0–9.0	1
WHO (lawn irrigation) (m)	1000							

The simplest greywater systems consist of only filtration and disinfection (Brewer et al., 2001; March et al., 2004; Friedler et al., 2006). However, simple systems do not produce treated effluents of good quality. Treated laundry greywater from a nylon sock filter, sedimentation and chlorination treatment train (March et al., 2004) contained organics and solids concentrations exceeding the proposed greywater reuse standard (Li et al., 2009). Similarly, a slanted soil system treating kitchen greywater (Itayama et al., 2006) also exceeded the proposed greywater reuse standard on solids and organics content (Li et al., 2009).

Membranes, on the other hand, are more expensive than granular media filters, but produce better quality effluent. Li et al. (2008) found that a submerged direct ultrafiltration (UF) membrane filtration module removed 83% of influent TOC, although soluble ammonia and phosphate were eluted in the permeate. Similarly, Šostar-Turk et al. (2005) utilised a UF and reverse osmosis (RO) membrane to remove 56% and 98% of BOD from laundry greywater. Although the treated effluent from RO membrane in Šostar-Turk et al. (2005)'s study met the proposed greywater reuse standard (Li et al., 2009), the treated effluent from UF membrane did not, as a result of the pore size differences. Similarly, Ramon et al. (2004) used a direct nanofiltration (NF) membrane to achieve organics and suspended solid removals of 93% and 100%. Smaller membrane pore sizes therefore result in effluent of better quality in exchange for higher transmembrane pressure, fouling frequency and energy expenses (Ghunmi et al., 2011; Nghiem et al., 2006).

These studies show that coarse filtration cannot reduce physical, chemical, and microbiological parameters of influent greywater to non-potable reuse standards. On the other hand, membrane filtration (MF, UF, and RO) demonstrated excellent TSS, turbidity, and pathogen removal despite the higher transmembrane pressure and fouling frequency. Thus, rapid sand filters coupled with disinfection or membrane filtration are recommended for polishing effluents from hybrid rainwater-greywater systems due to the fouling frequency.

4.3. Chemical treatment

Coagulation, ion exchange, and photocatalytic oxidation are common chemical treatment processes reported in greywater treatment studies, whereas rainwater does not require chemical treatment. These processes are effective at removing suspended solids, organics, and surfactants (Li et al., 2009). Drawbacks of coagulation include the production of waste primary sludge which must subsequently be disposed of (Ghunmi et al., 2011), whereas ion exchange resins must be regularly regenerated. Lin et al. (2005) utilised a combined electrocoagulation-disinfection process to treat light greywater. Influent BOD_5, COD, TSS, and turbidity were reduced from 23 mg/L, 55 mg/L, 29 mg/L, and 43 NTU to 9 mg/L, 22 mg/L, 9 mg/L, and 3.6 NTU. Moreover, combined coagulation systems remove more organics than coagulation-only systems. Pidou et al. (2008) combined chemical coagulation and magnetic ion exchange resins (MIEX) to treat mixed greywater. Their study showed that the optimum doses for alum (24 mg/L) and ferric (44 mg/L) coagulants reduced to 5 mg/L each when MIEX and coagulation were combined. Similarly, Šostar-Turk et al. (2005) found that coagulation alone achieved 36% COD removal, but combined coagulation with sand filtration and GAC adsorption achieved 93% COD removal. Friedler and Alfiya (2010) utilised $FeCl_3$ and sand filtration to achieve COD and TSS removals of 65% and 94%. Antonopoulou et al. (2013) used $FeCl_3$ and $Al_2(SO_4)_3.14H_2O$ to coagulate mixed greywater and found that $Al_2(SO_4)_3.14H_2O$ provided the highest removal efficiencies of 81% COD and 79% TSS for coagulation alone. When coagulation was combined with sand filtration and GAC, total COD and TSS removals were 96% and 97%.

There has been a recent rise in studies using photocatalytic oxidation to treat greywater. Pidou et al. (2009) utilised a hybrid-membrane photocatalytic process with TiO_2 as the photocatalyst to treat both synthetic and real greywater. They found that shower gel, conditioner and real greywater formed an organo-TiO_2 complex which fouled their membrane. The organo-TiO_2 complex was broken down with additional UV irradiation, suggesting that hybrid photocatalytic processes must be well designed to ensure complete irradiation of TiO_2. Similarly, Sanchez et al. (2010) found that TiO_2 removes both DOC and anionic surfactants from greywater, although DOC removal efficiency did not increase proportionally with increased catalyst dosage. The study found that DOC removal was highest (65%) for light greywater, whereas dark greywater had low DOC removals (44%). Thus, the study recommended using TiO_2 only for light greywater with DOC <100 mg/L. Chong et al. (2015) used TiO_2 to treat RB5 dye in synthetic and real greywater. Their study noted the presence of organics such as COD and BOD_5 resulted in decreased removal efficiency of RB5 dye. These studies show photocatalytic technologies may not be suitable to treat dark greywater with high organics and solids contents, as both will adversely affect the penetration of UV light to the TiO_2 surface, thus limiting the efficiency of photocatalytic treatment.

These studies show that chemical treatment reduced organics, turbidity, and TSS in light greywater with low organic strength to non-potable reuse standards, although chemical treatment has limited efficacy in mixed or dark greywater with medium to high organic strengths. Chemical treatments are thus recommended only for treatment of light greywater. Effluent from chemical treatment processes will require further polishing via sand filtration coupled with disinfection or membrane filtration in order to reduce total coliforms to non-potable reuse guidelines.

4.4. Biological treatment

Greywater treatment by biological means has been reviewed extensively by Li et al. (2009) and Ghunmi et al. (2011). Biological treatments may be aerobic or anaerobic, and range from rotating biological contactors (RBC) (Nolde, 2000; Friedler et al., 2005), sequencing batch reactors (SBR) (Hernández Leal et al., 2010), upflow anaerobic sludge blanket (UASB) (Elmitwalli and Otterpohl, 2007; Hernández Leal et al., 2010), constructed wetlands (Gross et al., 2007) to membrane bioreactors (MBR) (Liu et al., 2005; Merz et al., 2007).

Aerobic RBCs and SBRs effectively remove organics and inorganics from light greywater. Nolde (2000) treated light greywater with a pilot-scale sedimentation, RBC, and UV disinfection process. Influent BOD_7 values were reduced from 50 to 250 mg/L to below 5 mg/L by the RBC step alone. Friedler et al. (2005) treated light greywater with a pilot-scale RBC, sand filtration and chlorine disinfection process. Prior to the RBC, light greywater was filtered through a 1 mm square mesh to remove gross solids and hair. Influent TSS, turbidity, COD, and BOD were reduced from 43 mg/L, 33 NTU, 158 mg/L, 159 mg/L to effluent values of 16 mg/L, 1.9 NTU, 46 mg/L, and 6.6 mg/L. The final effluent met Li et al. (2009)'s proposed standard for unrestricted reuse. Furthermore, Eriksson et al. (2009) reported a pilot-scale greywater plant with a sedimentation, RBC, sand filtration, and UV disinfection process. The RBC alone removed 84% COD, 97% BOD, and 94% TOC in influent greywater. Similarly, Hernández Leal et al. (2010) utilised an SBR to treat high-strength mixed greywater with sludge retention times and hydraulic retention times of 15 days and 11.7 h. The SBR reduced influent COD, total nitrogen, total phosphorous, and surfactants from 833 mg/L, 41.2 mg/L, 6.6 mg/L and 43.5 mg/L to effluent values of 82 mg/L, 31 mg/L, 4.4 mg/L, and 1.4 mg/L 97% of anionic surfactants in greywater was removed by the aerobic SBR.

MBRs combine biodegradation with membrane filtration, and feature higher pollutant removals than RBCs or SBRs. MBRs are known to produce high quality effluent capable of meeting non-potable reuse standards. Liu et al. (2005) treated low strength light greywater with a submerged MBR. The configuration reduced influent COD values from 130 to 322 mg/L to 18 mg/L, BOD_5 values from 99 to 221 mg/L to less than 5 mg/L, and anionic surfactants from 3.5 to 8.9 mg/L to less than 0.5 mg/L in the effluent. Merz et al. (2007) treated greywater with low organic strength from a sports and leisure club with a submerged MBR. Influent values of turbidity (29 NTU), COD (109 mg/L), BOD_5 (59 mg/L), TKN (15.2 mg/L), total phosphorous (1.6 mg/L), faecal coliforms (1.4×10^5 CFU/100 mL), and surfactants (299 μg/L) were respectively reduced to 0.5 NTU, 15 mg/L, 4 mg/L, 5.7 mg/L, 1.3 mg/L, 68 CFU/100 mL, and 10 μg/L in the permeate. Bani-Melhem and Smith (2012) treated greywater with a submerged MBR. Influent COD, NH_3-N, and turbidity values were reduced from 356 mg/L, 2.47 mg/L, and 80 FTU to 45 mg/L, 0.26 mg/L, and 3 FTU. The final effluent contained no TSS or faecal coliforms. These studies showcase the effectiveness of the MBR at organics, solids, and pathogen removal.

Anaerobic treatment has several advantages over aerobic treatment, which are: lower sludge production, no energy required for aeration, and the methane produced acts as an additional energy source (Ghunmi et al., 2011). Anaerobic treatment, however, removes less pollutants than aerobic treatment. Hernández Leal et al. (2010) found that a UASB operated with a HRT of 12 h at 32 °C removed 51% of influent COD and 24% of anionic surfactants. Elmitwalli and Otterpohl (2007) held an UASB at 30 °C at hydraulic retention times between 6 and 16 h. The UASB removed 52–64% of influent COD, 22–30% of total nitrogen, and 15–21% of total phosphorous. Furthermore, combined anaerobic-aerobic treatment processes do not feature significantly higher pollutant removals than aerobic-only systems. Hernández Leal et al. (2010) operated an anaerobic-aerobic UASB-SBR combination at 32 °C at a HRT of 12 h, and found that COD removals were similar to an aerobic SBR operated at a HRT of 12 h at 30 °C.

Based on the literature, anaerobic processes are not recommended for hybrid rainwater-greywater systems due to their poor organics and surfactants removal relative to aerobic processes, despite the advantages of methane production (Ghunmi et al., 2011). Aerobic processes are extremely effective at organics and solids removal, and MBRs have the added benefit of pathogen removal. RBCs and SBRs combined with a post disinfection step or standalone MBRs are recommended to treat mixed or dark greywater with high organic strength to ensure that non-potable reuse guidelines are met. Light greywater lacks the macronutrients (N, P) and trace nutrients (Zn, Cu) necessary to stimulate and sustain

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growth of microorganisms in activated sludge (Jefferson et al., 2001), and thus biological treatment is not recommended for light greywater due to the limited treatment efficiency. RBCs and SBRs have similar capital and operational costs (Ghunmi et al., 2011), and are recommended options for aerobic biological treatment. MBRs are good alternatives to RBCs and SBRs, although MBRs are significantly more expensive: Friedler and Hadari (2006) found that RBCs are economically feasible once a building reaches 7 storeys or 28 flats, whereas MBRs are economically feasible only at 38 storeys or 152 flats.

4.5. Disinfection

Disinfection is a mandatory treatment step which eliminates microorganisms. Chlorination is the most common form of disinfection as it is inexpensive, widely available, and unlike bromine, Ag or ozone, leaves residuals preventing microbial regrowth after disinfection. Chlorine has been utilised to disinfect both rainwater (Moreira Neto et al., 2012) and greywater (March et al., 2004; Friedler et al., 2011). However, chlorination produces toxic DBPs, bringing a need to find alternative disinfectants.

Hydrogen peroxide (H_2O_2) has been utilised to disinfect light greywater. Teh et al. (2015) found that 1 mL/L of H_2O_2 prevented bacterial regrowth for up to 3 days. Ronen et al. (2010) found no significant cost differences between application of chlorine and H_2O_2, and found that 99% of faecal coliforms were inactivated by 125 mg/L of H_2O_2 with a contact time of 35 min. H_2O_2 does not form DBPs, unlike chlorine.

Ozone is a chemical-free alternative to chlorine and is one of the strongest and most efficient disinfectants. A substantial drawback of using ozone is the lack of residuals preventing microbial regrowth. Kim et al. (2005) assessed the performance of 1 μm and 5 μm submerged metal membranes combined with ozonation to treat rainwater. Ozone injection prevented excessive pore blockage, hence significantly decreasing the rate of transmembrane pressure build up. Similarly, Oh et al. (2015) utilised ozone to disinfect filtered light greywater in a pilot-scale system, where ozone recirculation successfully removed all total coliforms. Although ozone application has high operating costs, ozone is suitable for hybrid rainwater-greywater systems as both rainwater and greywater contain chlorine-resistant *Cryptosporidium* oocysts. Indeed 1 mg/L of ozone for 5 min was enough to inactivate *Cryptosporidium* oocysts (Korich et al., 1990). A subsequent chlorination step may be applied in order to prevent microbial regrowth and lengthen the storage time of effluent in the hybrid rainwater-greywater system.

UV irradiation with a UV lamp eradicated *E. coli* and total coliforms from raw rainwater in a novel filtration-adsorption-disinfection system (Naddeo et al., 2013). An alternative to UV lamps is solar irradiation. Amin and Han (2009) improved the efficiency of solar disinfection (SODIS) by creating a solar collector (SOCO-DIS) to focus sunlight onto rainwater. SOCO-DIS achieved higher water temperatures than SODIS alone, and a collector incline angle of 37° increased microbial inactivation by 10%. Moreover, Gilboa and Friedler (2008) removed 2 log_{10} units of HPC, faecal coliforms, *Pseudomonas aeruginosa* and *Staphylococcus aureus* from biologically treated greywater with varying UV doses ranging from 69 to 439 mW s/cm^2. They found that the most UV-resistant microbe in greywater was faecal coliforms followed by HPC, *Pseudomonas aeruginosa,* and *Staphylococcus aureus*, suggesting that high doses of UV irradiation is required to ensure complete disinfection. An advantage of UV irradiation is that no DBPs are generated, unlike ozone and chlorine, and no dosing pump apparatus is required (Gilboa and Friedler, 2008). Similarly, Couto et al. (2015) eradicated all *E. coli* from greywater collected in an airport at an intensity of 113 mW/cm^2. Faecal coliforms, *Pseudomonas aeruginosa*, and *Staphylococcus aureus* all did not exhibit regrowth 6 h after UV doses was applied (Friedler et al., 2011). UV irradiation is therefore a cost-effective disinfection for hybrid rainwater-greywater systems, although prior removal of suspended solids is necessary to maximise pathogen removal (Winward et al., 2008a).

Boiling or pasteurisation inactivates viral, parasitic and bacterial pathogens at the expense of high energy consumption. However, inactivation at sub-boiling temperatures is also possible. Spinks et al. (2006a) found that the critical temperature range for microbial inactivation in raw rainwater is 55–65 °C for *Enterococcus faecalis, E. coli*, *Pseudomonas aeruginosa, Salmonella typhimurium, Serratia marcescens, Klebsiella pneumonia, Aeromonas hydrophila* and *Shigella sonnei*. The most thermotolerant microbe was *Enterococcus faecalis*, which required 7–19 s at the same temperature range for 1 log_{10} unit removal, whereas all other species required less than 6 s. The study suggested that rainwater might be safely added to hot water systems in cold countries. This treatment step may not be feasible for hybrid systems due to the high energy requirements.

Costly and unusual alternatives include disinfection by essential oils and silver (Ag). Winward et al. (2008c) found that 468 mg/L of origanum oil was sufficient to reduce total coliforms in greywater by 2 log_{10} units, and prevented coliform regrowth for up to 14 days. However, the high concentrations of origanum oil required currently limits its use as a disinfectant when upscaling systems. In contrast, an average Ag dose of 9.9 μg/L was sufficient to ensure 1.5 log_{10} and 2 log_{10} removal units for *E. coli* and total coliforms in filtered rainwater (Adler et al., 2013). However, using Ag as a disinfectant may result in Ag leaching into rainwater (Bielefeldt et al., 2009). Ag nanoparticles are cytotoxic and may cause argyria, which is a permanent skin discolouration (Wijnhoven et al., 2009). Hence, the health risks and environmental effects associated with Ag must be considered prior to future applications.

Both ozone and UV disinfection systems are recommended to treat rainwater and greywater in hybrid systems. Ozone disinfection is recommended as both rainwater and greywater contain ozone-resistant *Cryptosporidium* and *Giardia*, whereas UV disinfection is recommended due to its effectiveness in pathogen removal and its cost-effectiveness (Friedler et al., 2011). A chlorination step after ozone and UV is recommended to prevent microbial regrowth, as HPC bacteria regrew within 6 h after a UV dose of 147–439 mJ/cm^2 (Friedler et al., 2011). On the other hand, boiling, origanum oil and silver are not recommended disinfection processes for hybrid systems due to their high operating costs.

5. Recommended hybrid system configurations

A review of the literature revealed three major hybrid system configurations. Fig. 2 illustrates all three configurations with recommended treatment trains. All three designs are subject to highly variable flows and pollutant loads from both rainwater (Sazakli et al., 2007) and greywater (Eriksson et al., 2009), and thus an equalisation basin or storage tank is required for each design in order to normalise both rainwater and greywater influent prior to treatment.

Design 1 features a hybrid system where rainwater and greywater are treated separately. This system has been reviewed by Li et al. (2010) and simulated by Zhang et al. (2010) and Peterson (2016). Heavy metals and pathogens in untreated rainwater can be reduced with careful design of the catchment and storage tank detailed above. As such, rainwater requires a first-flush diversion, storage, and disinfection step to meet unrestricted non-potable reuse standards (Fig. 2). Light greywater should be treated via a chemical treatment step, and polished by sand filtration/disinfection or membrane filtration (Fig. 2). In contrast, mixed and dark

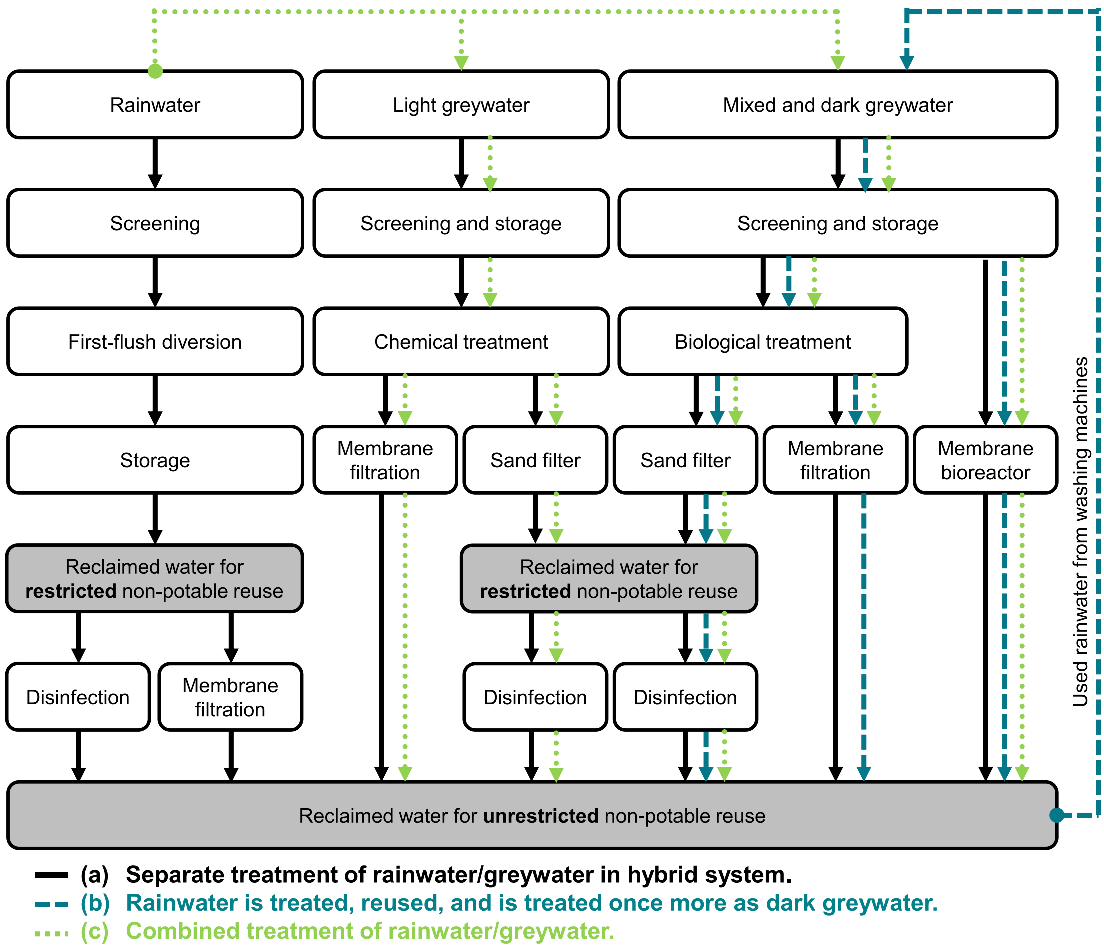

Fig. 2. Proposed hybrid rainwater-greywater system designs.

greywater with high organic strength should be treated biologically, and polished by sand filtration/disinfection or membrane filtration. Treated rainwater can supplement washing machines, toilet flushing, irrigation, and air conditioning water demands, whereas treated greywater can augment toilet flushing and irrigation water demands.

Design 2 features an integrated hybrid system where rainwater is reused once for non-potable uses (e.g. washing machine), before it is regenerated as dark greywater for toilet flushing and irrigation. Design 2 has been implemented in Australia (Muthukumaran et al., 2011) and Brazil (Ghisi and Mengotti De Oliveira, 2007; Ghisi and Ferreira, 2007). Rainwater must be filtered and disinfected in a similar fashion to Design 1. Subsequently, mixed and dark greywater would be treated as stipulated by Fig. 2. Treated rainwater would supplement washing machines, whereas treated greywater would augment toilet flushing, irrigation, and landscaping water demand. This system would be most effective in countries with high rainfall, as rainwater is fully utilised.

Design 3 combines untreated rainwater, greywater, and mains water top-up into a single tank prior to treatment. An alternative to Design 3 would be to treat rainwater and greywater separately before combining both streams. This has the advantage of diluting treated greywater with rainwater to help meet unrestricted non-potable reuse standards, as rainwater contains less organics, solids, and pathogens than greywater (Table 1). In Design 3, a mixture of rainwater and light greywater should be treated as light greywater, whereas rainwater mixed with dark greywater is treated as dark greywater in order to minimise health risks (Fig. 2). Treated rainwater-greywater may hence supplement toilet flushing and irrigation. Design 3 has been simulated by Dixon et al. (1999), and implemented in Austria (Weissenbacher and Müllegger, 2009), Brazil (de Gois et al., 2015), China (Zhang et al., 2009), and the UK (Birks et al., 2004). A concern in Design 3 is how mixing rainwater with greywater and mains water affects the overall removal efficiencies of treatment systems. Birks et al. (2004) treated a rainwater, greywater, and groundwater mixture with a RO, UF, and chlorination process and successfully produced high quality effluent that was free of *E. coli*, total coliforms, and faecal enterococci. Furthermore, Kim et al. (2007) demonstrated that greywater fouled their metal membrane more rapidly than the rainwater-greywater mixture, showing that dilution of greywater with rainwater may enhance the lifespan of membrane and granular filters. Moreover, the acidity of rainwater is negated by neutral greywater if both are combined, hence lowering corrosion rates. However, high pathogen concentrations may result from mixing rainwater with greywater, resulting in a need for higher disinfectant doses and hence, higher operating costs. Additional research in this area is recommended to investigate whether rainwater should be mixed with greywater, and a cost comparison of these three different options will determine the option with the highest return-on-investment (ROI).

6. Conclusions

Rainwater and greywater are both highly variable in quality and quantity, and an equalisation basin is proposed to normalise influent into a hybrid rainwater-greywater system. Both rainwater and greywater should not be reused without treatment in order to minimise health risks from pathogens. The most frequently reported pathogens in rainwater are *Aeromonas* followed by *Salmonella*; in greywater, *Pseudomonas* followed closely by *Staphylococcus.* Three flow configurations for hybrid systems have been identified: (a) separate treatment of rainwater and greywater; (b) rainwater reused for washing machines prior to reuse as greywater; and (c) combined treatment of rainwater and greywater. In all three designs, rainwater may be treated via first-flush diversion and disinfection alone. If rainwater is combined with greywater, then the resulting mixture should be treated as greywater. Greywater, however, must undergo further physical, chemical, and biological treatment. Aside from membrane filtration, physical treatment demonstrates poor removal of organics and solids. Membrane filtration is thus recommended to polish effluents from a chemical or biological treatment step. Chemical treatment effectively removes solids, organics, and surfactants in light greywater, although they perform poorly for both mixed and dark greywaters with high organic strength. Aerobic biological treatment is recommended for treating mixed and dark greywaters due to efficacy in removing organics, solids, and surfactants. Anaerobic biological treatment is not recommended due to its inferior organics and surfactants removal relative to aerobic processes. A combination of ozone or UV with chlorine is recommended to eradicate chlorine-resistant *Cryptosporidium* oocysts from hybrid rainwater-greywater systems and to prevent microbial regrowth. Further research on the applicability of hybrid rainwater-greywater systems, environmental benefits, and cost differences between each design configuration, as well as data on the local quantity and quality of rainwater and greywater will be necessary in order to facilitate widespread adoption of hybrid rainwater-greywater systems.

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