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Feasibility of a semi-batch vertical-flow wetland for onsite residential graywater treatment

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ABSTRACT

The technical feasibility of graywater treatment, via a semi-batch vertical flow wetland (SB-VFW), was evaluated at a single family home for aboveground water reuse. Graywater treatment was achieved to a quality level that was well within the regulatory compliance limits (e.g., with respect to biochemical oxygen demand (BOD), total suspended solids (TSS) and turbidity) for above ground reuse. System performance over the course of a five-month field study demonstrated that the necessary treatment time was typically ~3 h for a 300 L batch. The SB-VFW operated in a stable manner owing to the use of a coconut soil substitute that has high hydraulic conductivity and also enabled avoidance of clogging due to excess biofilm growth. The latter was made possible through the use of a bio-based laundry (BBL) detergent which had a lower organic content relative to conventional laundry detergents. The system was able to return to normal operation even after detergent shock loads. Results of the present study are encouraging and suggest that residential level graywater treatment is technically feasible and can thus potentially expand the range of possible water reuse applications.

1. Introduction

Graywater is considered a viable alternative nonpotable water resource for onsite treatment and reuse in residential homes in urban (Zhang et al., 2009) and rural (Al-Jayyousi, 2002) communities. For example, homeowners in California are allowed to collect and treat graywater from bathroom and laundry machines for aboveground nonpotable use (Yu et al., 2013), which account for 32% and 26%, respectively, of the total indoor water consumption (DeOreo, 2011). Raw graywater can vary in quality (Table 1) particularly with respect to TSS, turbidity and BOD_5. Regulatory requirements for graywater nonpotable reuse (subsurface, aboveground or indoor) vary regionally and among different countries (Table 2). For example, treated graywater with high turbidity (<10 NTU) can be reused for aboveground nonpotable water use purposes, including indoor toilet flushing in the United Kingdom, but the turbidity limit for the same type of water uses is <2 NTU in California (Table 2). Among the different regions/countries that regulate graywater reuse, California has one of the strictest set of requirements ((Yu et al., 2013; Boyjoo et al., 2013), Table 2). These graywater reuse requirements are designed to safeguard public health and minimize odors and other nuisances due to prolonged storage of raw graywater (Dixon et al., 2000). Various commercial graywater treatment systems using biological treatment processes (NSW Health Dept, 2011) are available but they are expensive (>$5000 for a single family system (EMRC, 2011)). Thus, there have been efforts to develop low cost and low maintenance treatment systems using constructed wetlands (Hoffmann et al., 2011).

Previous studies (Table 3) have demonstrated that constructed wetlands are effective in removing BOD_5, total suspended solids (TSS) (Niemczynowicz and Fittschen, 1997; Shrestha et al., 2001) and turbidity to levels that meet strict aboveground nonpotable graywater reuse requirements such as in the USA and Australia (Table 2). However, the use of conventional constructed wetlands is limited given the need for a large deployment area and long treatment time (Kadlec and Wallace, 2009). A viable alternative to conventional constructed wetland for treatment of graywater is the use of a compact semi-batch vertical-flow wetland (SB-VFW) with recirculation (VFWR) (Lemon et al., 1996; Gross et al., 2007; IWA, 2000). The SB-VFW has a smaller footprint and requires shorter treatment time compared to horizontal-flow wetland designs (Table 3).

A typical SB-VFW consists of an upper soil layer and may contain an underlying support layer of gravel, rock or trickling filter media (Lemon et al., 1996; Gross et al., 2007; IWA, 2000). It is noted

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Table 1
Average reported water quality for graywater originating from laundry and bathroom sources.

Parameters	Graywater sources	
	Laundry + bathroom	Bathroom
Turbidity, NTU	24 ± 8	58 ± 26.2
TDS, mg/L	410*	388*
TSS, mg/L	68 ± 88	47 ± 16
pH	8 ± 2	7.4 ± 0.5
BOD, mg/L	185 ± 73	99 ± 73
COD, mg/L	366 ± 173	221 ± 160
TOC, mg/L	–	38 ± 12.7
NH_4–N, mg/L	1*	3.8 ± 4.3
NO_3–N, mg/L	24 ± 33	2.1 ± 2.6
TN, mg/L	31 ± 27	7.5 ± 2.5
PO_4–P, mg/L	15 ± 23	0.5
TP, mg/L	11*	–
E. coli, CFU/100 mL	–	4×10^{6}*
Fecal, CFU/100 mL	4×10^{4}*	$8 \times 10^5 \pm 2 \times 10^5$
Total coliform, CFU/100 mL	$4 \times 10^7 \pm 5 \times 10^7$	$1 \times 10^4 \pm 1 \times 10^5$
References	(Surendran and Wheatley, 1998; Andersen et al., 2002; Sostar-Turk et al., 2005; Friedler, 2004; Friedler et al., 2006; Gilboa and Friedler, 2008)	(Surendran and Wheatley, 1998; Friedler et al., 2006; Gilboa and Friedler, 2008; Nolde, 2000; Friedler et al., 2005; Lamine et al., 2012; Hourlier et al., 2010; Ronen et al., 2010; Pidou et al., 2008; Zhang et al., 2009)

Note: * indicates only one cited literature source reported the data.

Table 2
Water quality criteria for aboveground nonpotable graywater reuse in United Kingdom, United States of America and Australia.

Standards	Type of reuse	Treatment level equivalent	Water quality criteria
United Kingdom U.K. Environment Agency (2011)	Sprinkler; Car washing; Toilet flushing; Garden watering; Pressure washing; Washing machine use		<10 NTU of turbidity; pH 5–9.5; <2 mg/L of residual chlorine; <0.5 mg/L of residual chlorine for non-spray garden watering; Undetectable residual bromine levels for all spray application and non-spray garden watering
California, USA California Building Standards Commission, (2010)	Aboveground nonpotable reuse	Disinfected tertiary	Turbidity: 2 NTU (avg.); 5 NTU (max) Total coliform: 2.2 MPN/100 mL (avg.), 23MPN/100 mL (max in 30 days)
Wisconsin, USA Wisconsin Department of Commerce (2015)	Surface irrigation except food crops; clothes and vehicle washing; air conditioning; soil compaction; dust control; washing aggregate; making concrete	Disinfected tertiary	pH 6–9; $\leq$10 mg/L of BOD_5; $\leq$5 mg/L of TSS; 1.0–10 mg/L of residual chlorine
	Toilet and urinal flushing	Disinfected primary with filtration	pH 6–9; $\leq$200 mg/L of BOD_5; $\leq$5 mg/L TSS; 0.1–4.0 mg/L of residual chlorine
NSF/ANSI 350R	Restricted indoor and unrestricted outdoor water reuse	Residential capacity $\leq$ 1500 GPD	pH 6–9; $CBOD_5$: $\leq$10 mg/L of (avg.), 25 mg/L (max.); TSS: $\leq$10 mg/L (avg.); 30 mg/L (max.) Turbidity: 5 NTU (avg.); 10 NTU (max.) *E. coli*: 14 MPN/100 mL (mean); 240 MPN/100 mL (max.) $0.5 \leq$ storage vessel disinfection ≤ 2.5 mg/L
New South Wales, Australia NSW Health Dept (2011)	Toilet flushing; Cold water supply to washing machines; Garden irrigation with local approval	Disinfected Secondary	<20 mg/L BOD_5; <20 mg/L TSS; <10 CFU/100 mL fecal coliforms
Western Australia, Australia DOH WA, (2010)	Toilet flushing Cold water supply to washing machines; Irrigation	Disinfected Tertiary	<10 mg/L of BOD; <10 mg/L of TSS; <1 MPN/100 mL of *E. coli* <1 PFU/100 mL of coliphages; <1 CFU/100 mL of clostridia;
Victoria, Australia EPA Victoria (2013)	Toilet flushing; Cold water supply to washing machines; Surface irrigation, Sub-surface irrigation	Disinfected Tertiary	<10 mg/L of BOD; <10 mg/L of TSS; <10 CFU/100 mL of fecal coliforms;

Table 3
Wetland designs and their reported treatment performance for treatment of graywater.

Country	Graywater sources	Wetland type	Media	Recirculation	Mode of operation	Treatment capacity, m^3/day	Wetland area, m^2	Treatment capacity per area, m^3/m^2day	HRT, days	Turbidity, % removal (NTU)	Organics, % removal (mg/L)	TSS, % removal (mg/L)	Total coliform, % removed (/100 mL)
Sweden (Niemczyn and Fittschen, 1997)	Sinks, kitchen, shower, washing machine	HF	Soil, not specified	No	Continuous flow	40	600	0.07	4	–	BOD_7: 95.8% (<5); COD: 87% (46)	–	–
Sweden (Schonborn et al., 1997)	Sinks, kitchen, shower, washing machine, urinal	VF-HF	Fine grained sand, iron armament pieces, sandy and stony loam	No	Continuous flow	40	38	0.04	–	–	BOD_5: 96% (5 mg/L); COD: 91% (27)	–	–
Nepal (Shreshtha et. al, 2001)	Bathroom, shower, washing machines and kitchen sinks	VF	Coarse sand	No	Continuous flow	0.5	6	0.08	–	–	BOD_5: 97% (5.2)	97% (2.6)	–
Costa Rica (Dallas and Ho, 2005)	Mixed, not specified	HF	PET	No	Lab batch	0.01	0.38	0.03	4.2	–	BOD_5: 92% (13)	–	Fecal coliform 99.99% (2.1×10^3 CFU)
	Mixed, not specified	HF	Crushed rocks	No	Lab batch	0.01	0.38	0.03	2.5	–	BOD_5: 88% (18)	–	Fecal coliform 99% (2.6×10^5 CFU)
Costa Rica (Dallas et al., 2004)	Mixed, not specified	HF	Crushed rocks	No	Continuous flow	2.5	30	0.08	7.90	98% (2)	BOD_5: 99% (3)	–	Fecal coliform: 99.999% (16CFU)
Israel (Gross et al., 2007 *Sklarz et al., 2009)	Mixed, not specified	VF	Peat, soil and randomly packed plastic media or tuff	Yes	Semi-batch	0.45	0.9	0.5	0.3-0.5	87%* (6*)	BOD_5: 100% (0.7); or COD: 81% (157)	98% (3)	Fecal coliform: 99% (2×10^5 CFU)
Uganda (Katukiza et al., 2014)	Bathroom, shower, laundry and kitchen sinks	VF	Crushed lava rock	No	Batch, single pass	0.22	0.2	1.1	0.06/10 L-batch	–	BOD_5: 72% (390); COD: 90% (198); TOC: 69% (277)	93% (80)	99.9% (3.38×10^4 CFU)
Brazil (Paulo et al., 2013)	Laundry, bathroom	HF-VF	Coarse and fine gravel	No	Continuous flow	0.7	7.8	0.09	0.8	95% (13)	BOD_5: 95% (22)	92% (9.6)	–

Note: Values in brackets are reported average effluent concentration values. HF—a horizontal flow wetland; VF—vertical flow wetlands; HRT—hydraulic retention time.

that the use of siliceous soil in SP-VFWs and SB-VFWs has met with limitations due to relatively low hydraulic conductivity and its tendency to clog (Platzer and Mauch, 1997; Zhao et al., 2009). Clogging in SP-VFWs and SB-VFWs is typically exacerbated by excessive biofilm growth stimulated by high organic loading (Platzer and Mauch, 1997; Leverenz et al., 2009). Severe clogging in VFWs can also lead to anaerobic condition in the soil layer (Platzer and Mauch, 1997) and environmental nuisances such as odor (Cooper, 2009). Thus, management of clogging in SB-VFWs is required to ensure their long term treatment performance (Cooper, 2009). Another challenge in graywater treatment via SB-VFW is that fine silt and clay may leach from the siliceous soil, which in turn can lead to high effluent turbidity (2–16 NTU) (Sklarz et al., 2009) exceeding regulatory requirements for aboveground reuse (Table 1) and impairing disinfection effectiveness (Winward et al., 2008a,b). Thus, an alternative design that prevents leaching of soil material would be important for SB-VFW use.

SP-VFWs and SB-VFWs are different from horizontal-flow wetlands since they can nitrify ammonium to nitrite and nitrate. However, SP-VFW do not denitrify nitrite and nitrate to nitrogen gas due to the wetland's aerobic environment (Kadlec and Wallace, 2009). On the other hand, nitrification is limited but denitrification is common in horizontal-flow wetlands due to the limited aerobic conditions (Kadlec and Wallace, 2009). Thus, use of a SP-VFW followed by a horizontal-flow wetland is common in order to achieve complete nitrogen removal (Cooper, 2009; Zhang et al., 2014; Leverenz et al., 2010). For example, nitrification was reported in a SB-VFW with the reported effluent nitrate–nitrogen at 8.6 mg/L from an influent level of 3 mg/L (Gross et al., 2007). Although denitrification in SB-VFWs has not been reported in graywater treatment, denitrification has been documented in SP-VFWs for treatment of domestic wastewater (Laber et al., 1997). It was shown that treatment of domestic wastewater with two SP-VFWs in-series enabled denitrification through addition of a carbon source to the influent of the second SP-VFW (Laber et al., 1997). Given the above, it is plausible that nitrification–denitrification could also occur in SB-VFW however such a behavior has not been previously demonstrated.

In the present work, a graywater SB-VFW treatment was developed and a five-month field study was undertaken in order to optimize the treatment approach with the following specific objectives: (a) demonstrate the feasibility of achieving water quality (e.g., BOD5, TOC and turbidity) for permitted aboveground nonpotable reuse (Table 2), (b) decrease the required batch treatment time using a reasonably small treatment system, (c) eliminate leaching of siliceous soil material by using coconut coir soil substitute that has high hydraulic conductivity, (d) assess the potential for simultaneous nitrification and denitrification, and (e) evaluate treatment effectiveness when using a bio-based laundry detergent with the goal of reducing graywater organic content and avoiding excessive biofilm growth.

2. Materials and methods

2.1. Materials and graywater sources

The performance of the graywater treatment system (described in Section 2.2) was evaluated for treatment of bathroom, showers, handwashing sinks and laundry waters. Two types of detergents were evaluated for clothes washing: (a) a United States Department of Agriculture certified bio-based laundry (BBL) detergent, (b) a conventional non-bio-based laundry (non-BBL) detergent. The BBL detergent was reported to be less toxic to graywater effluent microbial community compared to the non-BBL detergent (Pickering et al., 2013). The non-BBL detergent contained 22 organic compounds; while the BBL detergent contained 16 ingredients, seven of which were organic compounds and nine were inorganic compounds. The total organic carbon (TOC) content for the BBL and non-BBL detergents were determined (Section 2.3) to be ~72 mg-TOC/g-detergent and ~150 mg-TOC/g-detergent, respectively.

2.2. Graywater treatment system

Onsite residential graywater treatment and reuse was evaluated over a five-month study with an automated SB-VFW system (Fig. 1) deployed in a single-family home. The wetland (top) and reservoir (bottom) units had dimensions of 1.1 m (L) × 0.7 m (W) × 0.7 m (H). The wetland unit consisted of a bottom plastic container, with large holes bored through the bottom, for housing a coconut coir soil substitute layer (the "soil" layer) placed above a layer of cross-flow media (CFM). The soil layer, which was compartmentalized using fabric containers (Smart Pots, Oklahoma City, OK), served as the medium for soil microbial community and for plant growth. A number of ordinary garden plants, along with wetland plant species (*Carex spissa* and *Phyla nodiflora*), were planted in the wetland. Household garden plant species included *Aeonium purpureum* and *Crassula ovate, Equisetum hyemale, Nasturtium, Narcissus impatiens,Anigozanthos*, and *Colocasia.* The viability of the plants was assessed based on their overall appearance, e.g., leaf greenness and ability to produce flowers for flowering plants. CFM installed below the soil layer had a specific surface area of 226 m^2/m^3 (AccuPac Cross Flow Media, Brentwood Industries, Reading, PA). The CFM provided structural support for the soil layer with its open channel configuration enabling significant aeration while avoiding channel clogging. Dripping water from the wetland unit was captured by the reservoir below (the bottom reservoir), from which water was recirculated to the wetland during the treatment period.

Graywater, which was recirculated from the bottom reservoir, was distributed over the wetland using a gravity-driven flow distribution system. The distribution system consisted of partially filled enclosed conduits with holes at the bottom (installed just above the soil layer), which were arranged to facilitate even flow distribution over the wetland. Uniformity of the flow distribution was verified by determining the local flow rate at the different

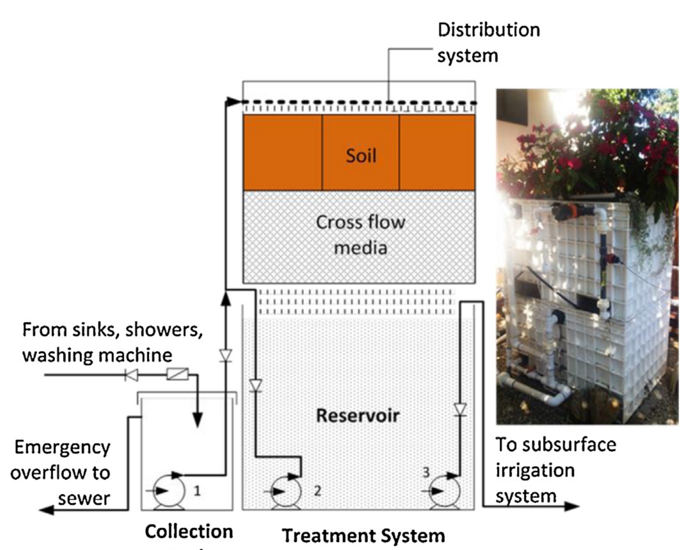

Fig. 1. Schematic diagram of the graywater treatment system. The upper wetland unit consists of a soil layer and a cross-flow media (CFM). Graywater intake (pump 1), recirculation (pump 2) and discharge (pump 3) are controlled by three different submerged pumps.

locations beneath the wetland unit. The enclosed conduits also served to avoid direct human contact with graywater and reduce water loss due to evaporation. The recirculation flow rate was measured by an inline paddlewheel flow meter (GF Signet P51530-P0, El Monte, CA) connected to a powered transmitter/totalizer (GF Signet 3-8550-2, El Monte, CA).

Automated system operation was enabled by a low-cost irrigation controller for operation of the system pumps for water inflow to the SB-VFW, recirculation within the SB-VFW system and water outflow for reuse (Fig. 1). Graywater was pumped from the collection tank to the SB-VFW system using a submerged sump pump having a maximum pumping flow delivery of up to 3400 L/h (Little Giant 5-ASP, Fort Wayne, IN) that was installed inside the collection tank. A small fountain pump (100 W) (Laguna Maxflow 2400, Mansfield, MA) submerged in the bottom SB-VFW reservoir was used for graywater recirculation. A third sump pump (Little Giant 5-ASP, Fort Wayne, IN) in the bottom reservoir served for pumping the treated effluent for irrigation. In principle, a single pump could be used for both recirculation and for outflow (e.g., irrigation); however, for the purpose of the present study, two pumps were used to simplify the operational control.

The SB-VFW system was designed to operate in a semi-batch mode given the intermittent generation of graywater. A collection tank of 300 L storage capacity was used for collecting and storing raw graywater between treatment cycles. Raw graywater entered the collection tank by gravity feed and was first filtered through a nylon filter fitted over the inlet pipe in order to protect the sump pump from being damaged by hair and other stringy substances. Once filled, raw graywater was pumped from the collection tank to the wetland feed distribution system with graywater recirculating (through the wetland unit) until the treated graywater effluent discharge.

2.3. Analytical methods

The SB-VFW system treatment effectiveness was evaluated based on water quality analysis of the influent and treated effluent. Grab samples were collected directly from the mid-water column of the collection tank (influent) and the reservoir of the SB-VFW system (effluent) using two auto-samplers (Hach Sigma MAX900, Loveland, CO). For each experiment, influent samples were collected just prior to pumping the graywater into the SB-VFW system, Grab effluent samples from the SB-VFW system were collected for evaluation over a 24 h period at times of 0, 0.5, 1, 2, 3, 8 and 24 h. Collected samples were kept in a cooler (<4 °C) before transporting back to the laboratory for water quality analysis. It is noted that all the reported water quality data are provided as average values over the different operational periods unless specified otherwise.

Water quality of the treated graywater was evaluated with respect to the SB-VFW system's ability to remove organics (TOC, five-day biochemical oxygen demand (BOD_5), and chemical oxygen demand (COD)), TSS and turbidity to levels that would meet the aboveground nonpotable water reuse requirements (Table 1). System performance was also assessed with respect to removal of ammonium–nitrogen (NH_4–N) and nitrate–nitrogen (NO_3–N). The study consisted of two phases. Phase I focused on evaluating the long-term treatment performance of the SB-VFW system for graywater that contained BBL detergent from clothes washing. Additionally, detergent shockload tests were performed in two consecutive days to evaluate the impact of sudden increase in organics and turbidity on treatment performance. On the first day of the detergent shockload test, 160 g of BBL detergent was added into the 300 L graywater collection tank containing the raw graywater, followed by manual mechanical mixing prior to taking a grab sample for analysis. The raw graywater was then pumped into the treatment system and the recirculation commenced. On the next day, 80 g of BBL detergent was added into 300 L charge of raw graywater and a grab sample was collected for analysis. Effluent samples were collected at 0, 0.5, 1, 2, 3, 8 and 24 h and analyzed for turbidity and organics for each day of the detergent shockload tests. During Phase II, non-BBL detergent was used by the residents. Water sampling and laboratory analysis protocols in Phase II were identical to those used in Phase I. Detergent shockload was not conducted during Phase II because of the rapid deterioration of treatment performance during the first month after commencing with the treatment of non-BBL graywater.

Total organic carbon (TOC) and dissolved organic carbon (DOC) were determined using Standard Methods 5310C (Aurora 1030D TOC Analyzer, OI Analytical, College Station, TX). Samples for DOC analysis were prepared by water samples filtration through 0.8 μm filters (MCE with modified acrylic housing, Millipore Millex Sterile Syringe Filters, EMD Millipore, Billerica, MA). In order to determine the biodegradable portion of the TOC in the influent and effluent streams, five-day seeded TOC tests (Khan et al., 1998) were conducted. For both cases, 300 mL volume samples were placed in amber bottles and kept in the dark at room temperature (22 °C). In the five-day test, the influent was seeded with 1 mL of the effluent. It is noted that the effluent was not added to the influent sample in the non-seeded biodegradable TOC test, and the effluent samples were not seeded. All samples were aerated daily (during the test period). The difference in TOC before and after the above tests represented the level of non-biodegradable organics.

Turbidity (USEPA Method 180.1; 2020 Portable Turbidity Meter, LaMotte, Chestertown, ML) was used as the primary monitoring parameter to assess reduction in suspended particles/colloidal matter. Other monitored water quality parameters included pH, conductivity (Oakton CON 11 Economy Meter, Vernon Hills, IL), temperature, total inorganic carbon (Standard Methods 5310C; Aurora 1030D TOC Analyzer, OI Analytical, College Station, TX), particle size (AccuSizer 780, Santa Barbara, CA), ammonium (Hach Method 10205, Loveland, CO), nitrate (Hach Method 10206, Loveland, CO), phosphate (Hach Methods 10209 and 10210, Loveland, CO), and dissolved oxygen (Dissolve Oxygen Tracer PocketTester, LaMotte, Chestertown, ML). TOC and DOC were the primary parameters for monitoring organic concentrations. BOD_5 and COD were calculated by establishing the relationships between TOC and COD (COD:TOC ratio) and BOD_5 and COD (BOD:COD ratio) using three batches of samples. COD was calculated from the TOC using the established influent COD:TOC ratio of 4.40, and the effluent (after 3 h treatment) COD:TOC ratio of 2.22 based on previous observations (Yu et al., 2014). The estimated COD was then used for calculating BOD_5. Influent and effluent (after 3 h treatment) BOD_5:COD ratios were 0.31 and 0.16, respectively.

Biofilm growth on the CFM surfaces was evaluated using clear polyvinyl film sampling coupons that were attached to the bottom of the wetland unit. These coupons were sampled after three months of treating BBL graywater, and after a month of treating non-BBL graywater. The biofilm sample coupons were stained using gram stain (Gram Stain Advanced, Hardy Diagnostics, Santa Maria, CA), air dried, and cover-slipped in Prolong Gold (Life Technologies, Carlsbad, CA) before imaging on an Zeiss Imager D1 microscope with Axiocam ICc 1 color camera, and processed using the Axiovision Software (Zeiss Microscopy GmbH, Oberkochen, Germany).

3. Results and discussions

3.1. Influent and effluent water quality

Raw graywater that contained laundry water generated from clothes washing using the BBL ("BBL graywater") or non-BBL

Table 4
Water quality of influent (raw) and treated (effluent) BBL and non-BBL graywater.

	BBL graywater		Non-BBL graywater	
	Raw graywater[a]	Effluent after 3 h[a] treatment	Raw graywater[b]	Effluent after 24 h treatment[b]
pH	6.8	6.6	6.8	6.6 ± 0.1
TDS, mg/L	337 ± 28	347 ± 26	295 ± 7	313 ± 12
DO, mg/L	1.6 ± 0.4	4.9 ± 1.3	2.5 ± 1.8	7.0 ± 1.3
Temp, C	21.4 ± 1.2	22.0 ± 1.0	20.1 ± 2.6	17.4 ± 2.6
Turbidity, NTU	21.6 ± 12.7	0.3 ± 0.3	29.8 ± 13.1	0.5 ± 0.1
TSS, mg/L	23.5 ± 9.3	<1	35.3 ± 1.3	<1
TOC, mg/L	31.5 ± 8.9	6.2 ± 1.0	101.2 ± 22.0	8.0 ± 1.1
Total inorganic carbon (TIC), mg/L	29.4 ± 5.4	30.5 ± 6.3	21.1 ± 6.6	20.0 ± 8.0
DOC, mg/L	24.6 ± 7.7	5.6 ± 0.9	87.8 ± 17.5	7.7 ± 1.3
bDOC, mg/L	20.4 ± 7.7	1.4 ± 0.8	80.1 ± 15.5	2.5 ± 1.1
DIC, mg/L	28.6 ± 4.6	27.4 ± 5.7	19.0 ± 6.3	18.2 ± 7.8
Particulate TOC, mg/L	7.7 ± 2.7	0.5 ± 0.3	13.4 ± 4.6	0.4 ± 0.3
COD[a], mg/L	128 ± 46	6.0 ± 4.7	490 ± 112	17.4 ± 5.7
BOD[b], mg/L	40 ± 14	<1	152 ± 36	2.8 ± 2.2
NH_4–N, mg/L	5.30 ± 1.95	1.70 ± 0.82	2.24 ± 0.39	0.10 ± 0.02
NO_3–N, mg/L	0.12 ± 0.05	2.47 ± 1.52	0.21 ± 0.03	0.13 ± 0.03
PO_4–P, mg/L	3.89 ± 0.73	2.44 ± 0.24	4.00 ± 0.80	2.13 ± 0.47

[c]Average for five batches. Note: Samples from two separate treatment cycles (i.e., days) were collected twice weekly and once per week when treating BBL and non-BBL graywater, respectively.
[a] COD was calculated from the TOC using an influent COD:TOC ratio of 4.40, an effluent COD:TOC ratio of 2.22 based on three selected observations from three separate treatment cycles (or days) collected in the same week (Section 2.3). Influent and effluent BOD:COD ratios were 0.31 and 0.16, respectively. *N*—number treatment cycles (i.e., days).
[b] Average for 25 batches.

("non-BBL graywater") detergents had organics, total suspended solids (TSS) and turbidity concentrations exceeding the water quality requirements for aboveground water reuse (Table 2). Continuous field operation of the SB-VFW system for the BBL and non-BBL graywater over 3.5 and 1.5 months, respectively, demonstrated that average treated water quality (Table 4) complied with the aboveground graywater reuse standards (Table 2). Significant turbidity removal was achieved with average effluent turbidity of 0.3 NTU for BBL graywater (relative to ~22 NTU for the raw BBL graywater) after 3 h of treatment, and 0.1 NTU for non-BBL graywater (relative to ~30 NTU for the raw non-BBL graywater) after 24 h of treatment. Such effluent levels were well below even the most stringent turbidity requirement of <2 NTU for aboveground graywater reuse in California (Table 1).

The BBL and non-BBL graywater effluent had low BOD_5 of 0.5 mg/L and 2.8 mg/L (representing reduction of >97% relative to both influent types), respectively, and TSS of <1 mg/L for both effluents (representing reduction of >97% and >95% relative to the influent graywater); the above treatment performance was also well within the BOD_5 and TSS requirements for aboveground graywater reuse in Australia and Wisconsin (Table 1). Raw BBL and non-BBL graywater pH was 6.8 (relative to tap water pH of 6.4) and decreased to 6.6 after 24 h of treatment. The slightly higher raw graywater pH was likely due to the use of detergents containing alkalis (e.g., sodium bicarbonate and sodium citrate in BBL detergent, and monoetholamine citrate in non-BBL detergent),

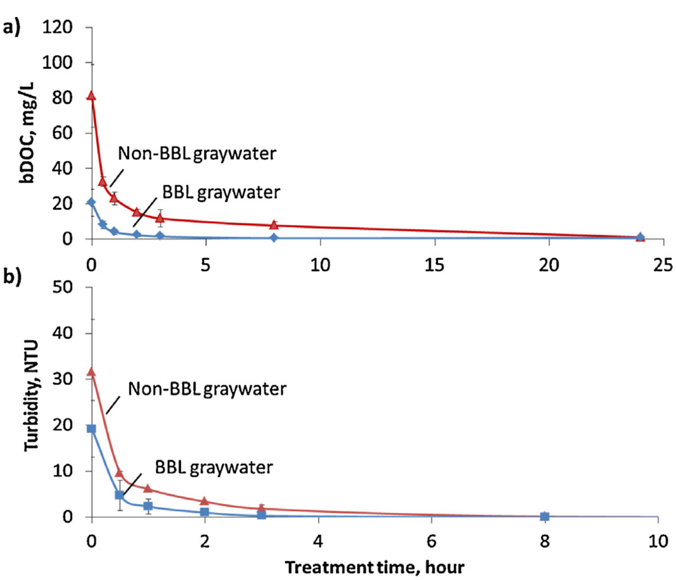

Fig. 2. Average bDOC (a) and turbidity (b) removal of BBL (25 treatment cycles) and non-BBL (four treatment cycles) graywater indicates that the SB-VFW system was effective in removing both turbidity and bDOC. Error bars represents one standard deviation of the data set.

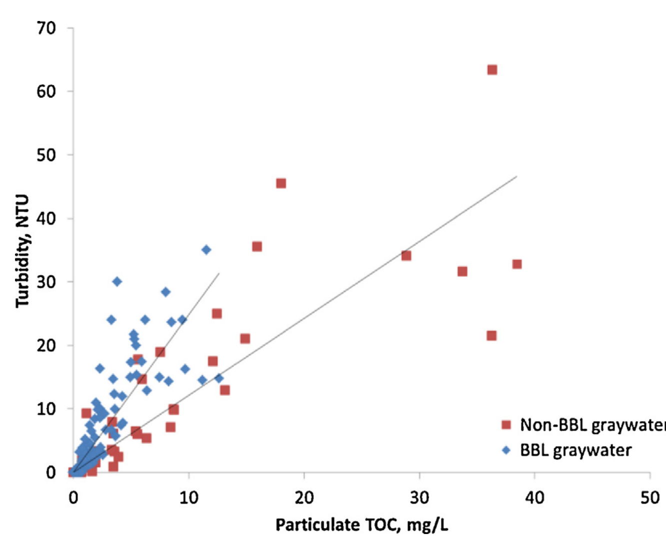

Fig. 3. Correlation of particulate TOC (i.e., total TOC—DOC) with graywater turbidity suggesting that graywater turbidity was possibly of mostly organic origin. The data set comprised of 25 and 4 of 24 h treatment cycles of BBL and non-BBL graywater batches, respectively, collected over a period of five-months. The R^2 values for the BBL and non-BBL graywater linear correlations were 0.72 and 0.67, respectively.

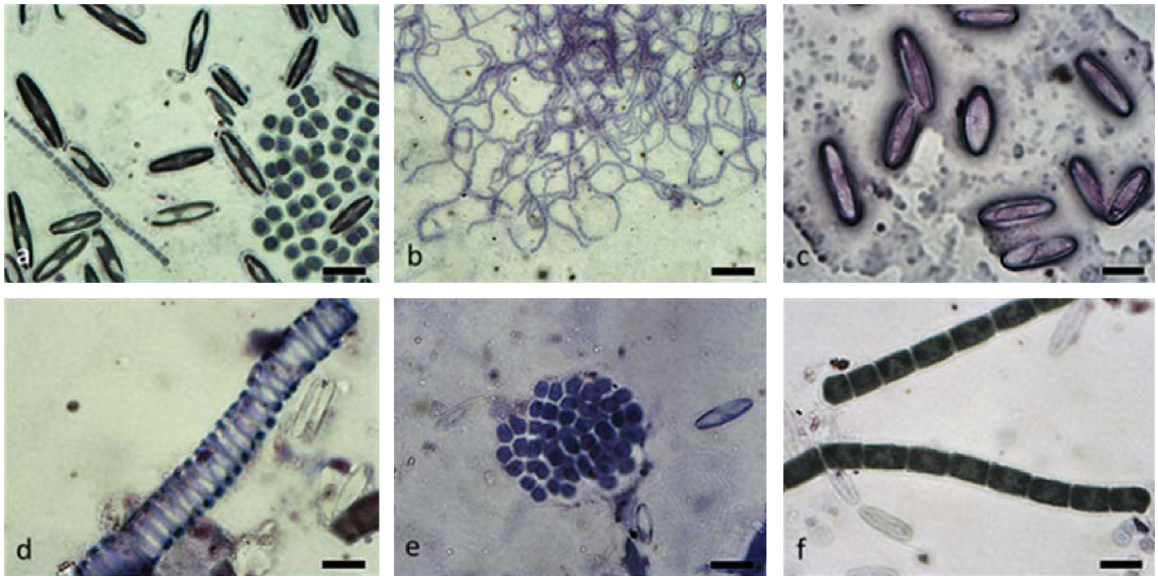

Fig. 4. Biofilm images (from the plastic media) during BBL graywater treatment. (a) Protozoa and algae, (b) fungi, (c) protozoa, bacteria, and algae, (d) algae, (e) algae cluster, (f) algae. The presence of protozoa suggests aerobic environment in the plastic media. Scale bars represent 10 μm.

which increases surface charge of fabric and soil particles to enhance cleaning performance (Bajpai and Tyagi, 2007). The slight reduction of graywater effluent pH after treatment could be due to alkalis reduction due to biodegradation of organic alkalis, such as the citrate, or other physio-biochemical reactions that neutralize inorganic alkalis such as sodium bicarbonate and carbonate. The raw BBL and non-BBL graywater TDS levels were 337 mg/L (53% higher than tap water) and 295 mg/L (34% above tap water) and increased slightly (by 3% and 6%, respectively) after treatment (Table 4). Such slight increase in TDS post treatment, although within experimental variability, could also be the result of dissolution of divalent salt precipitate (e.g., calcium carbonate in the raw graywater) during recirculation in which the pH decreased relative to the raw graywater. It is also noted that the non-BBL graywater part of the study was commenced after 3.5 months operation with the BBL graywater. Therefore, it is possible that TDS of the non-BBL treated graywater effluent was skewed to higher level than the influent due to slow release of sodium and/or calcium salts that may have accumulated in the soil matrix. Irrespective of the above, it is stressed that the treated effluent TDS for both types of graywater (Table 4) was well below the maximum recommended level (i.e., 1000 mg/L for irrigation (USEPA, 2012)).

The treatment time required for reaching the target effluent quality for aboveground nonpotable water reuse (Table 2; Fig. 2) for BBL and non-BBL graywater were 3 and 24 h, respectively. The above performance difference was attributed to the lower TOC level in the raw BBL graywater (∼32 mg/L) with 22% of the TOC in particulate form. The biodegradable dissolved organic carbon (bDOC) in the raw BBL graywater was at ∼20.4 mg/L bDOC, which was ∼64% of the total TOC, and declined by 93% after 3 h of treatment. On the other hand, the raw non-BBL graywater TOC was 101 mg/L with 14% of the TOC in particulate form, while the remaining being in dissolved form. Raw non-BBL graywater bDOC was 80 mg/L (∼80% of the TOC), and was reduced by 99% after 24 h of treatment (Fig. 2a). High TOC in the raw non-BBL graywater was attributed to both its high TOC content (150 mg TOC/g-detergent relative to 71 mg TOC/g-detergent for the BBL detergent) and its higher (recommended) detergent dosage. The manufacturer recommended laundry dosage for the non-BBL detergent are ∼44 g-detergent/normal-load and ∼64 g/heavy-load as compared to ∼19 g-detergent/normal-load and ∼38 g-detergent/heavy-load for BBL detergent, which led to the higher non-BBL graywater TOC.

The raw graywater turbidity correlated linearly with the total organic carbon associated with particulate organics (i.e., TOC minus DOC, represented as particulate TOC) as shown in Fig. 3. The SB-VFW was effective for turbidity removal of up to 99% (from 30 NTU to 0.3 NTU) within 3 h of treatment. Coincidentally, the time required for 93% bDOC removal for BBL-graywater was also achieved within about 3 h of treatment. A similar level of turbidity removal (∼98%) was achieved for the non-BBL graywater within ∼8 h (to 0.5 NTU, Fig. 2b), along with 90% bDOC (to 8 mg/L bDOC) over the same period with further removal to 97% bDOC (to ∼2.5 mg/L) at 24 h.

The bDOC degradation kinetics for the non-BBL relative to the BBL graywater for the SB-VFW system can be analyzed using a simple model that considers the hydraulic retention time in the wetland, wetland bed depth, wetland area and removal rate constant. In the model formulation, the wetland SB-VFW is approximated as a plug-flow reactor with negligible dispersion, the reservoir below the wetland is approximated as a mixed tank reactor, and bDOC removal (by the wetland) is described by first-order kinetics. Accordingly, the bDOC level can be expressed as:

$$C_{\text{eff}} = C_{\text{i,gr}} \times \exp\left[-\frac{t}{\theta}\left[1 - \exp\left(-k\frac{A\varepsilon L}{Q}\right)\right]\right] \quad (1)$$

where C_{eff} is the effluent bDOC (mg/L), $C_{\text{i,gr}}$ is the influent bDOC, θ is hydraulic retention time in the wetland (i.e., the ratio V_R/Q, $\theta = 0.53$ h); V_R is the reservoir volume ($V_R = 0.3$ m^3; L wetland bed

Table 5
Effluent bDOC and turbidity achieved by the wetland with and without the use of cross flow media.

Parameters	Treatment time, hours	With CFM[a]	Without CFM[b]
bDOC, mg/L	3	1.3 ± 0.8	1.3 ± 1.2
	8	0.3 ± 0.2	0.2 ± 0.2
Turbidity, NTU	3	0.3 ± 0.1	0.2 ± 0.1
	8	<0.1	<0.1

[a] 10 graywater batches treated over a period of five weeks.
[b] 3 graywater batches treated during the same one week period.

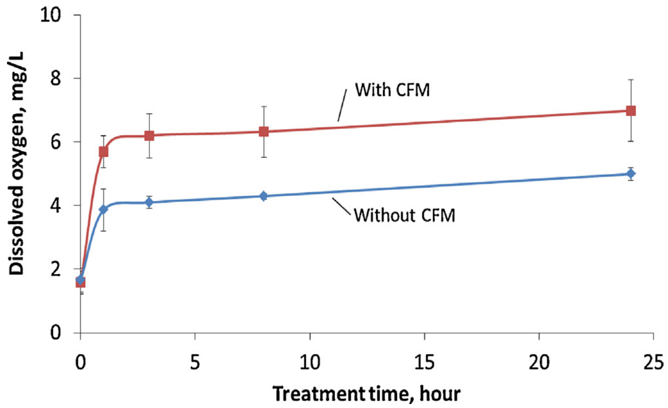

Fig. 5. Dissolved oxygen level achieved for graywater treatment with and without the CFM over 4 and 2 weeks of treatment, respectively.

thickness (L, $L = 0.12$ m); A is the wetland area ($A = 0.68\ m^2$), Q is the graywater recirculation flow rate ($Q = 0.58\ m^3/h$), ε is the wetland porosity ($\varepsilon = 0.9$), k is the removal rate constant (h^{-1}) and t is time (h). The bDOC removal rate constants for BBL and non-BBL graywater being $4.0\ h^{-1}$ and $1.5\ h^{-1}$, respectively, were determined from fitting Eq. (1) to the bDOC data (Fig. 2). It is noted that the faster bDOC removal kinetics for BBL graywater could be, in part, due to the greater biomass availability in the SB-VFW in comparison to that which could be achieved with the BBL graywater of lower content of degradable organics. The above results and improved treated effluent quality for BBL graywater, relative to non-BBL graywater, suggest that use of BBL detergent would be preferable when considering residential graywater treatment and reuse.

3.2. VFWR system characterization and long-term treatment of BBL graywater

The SB-VFW wetland unit acted as a packed-bed filter that captured particulate matter and provided surface area for biofilm growth. Thin slimy ("biofilm") layers was observed on the CFM surface and plastic coupon samples just underneath the soil layer. Microscope biofilm images revealed a rich collection of diverse organisms (Fig. 4). The role of the CFM section on bDOC and turbidity removal was evaluated by removing it from the SB-VFW system for 14 days. The effluent bDOC levels after 3 h of BBL graywater treatment with and without the CFM, were both ~1.3 mg/L (Table 5). Similarly, effluent turbidity after 3 h of

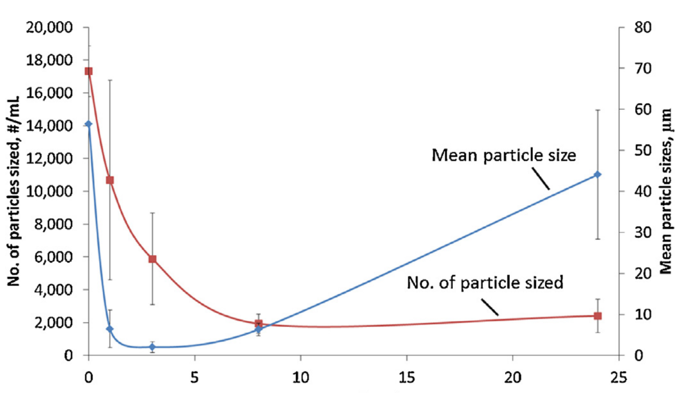

Fig. 6. Effluent mean particle size and particle number concentrations during treatment of BBL graywater (one-week operation).

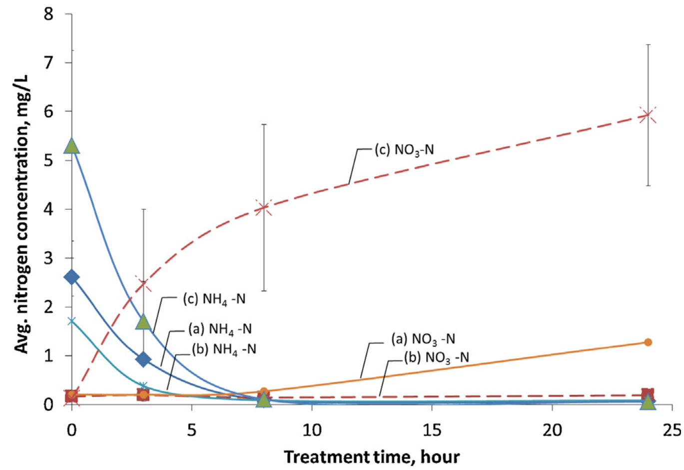

Fig. 7. Change in NH_4–N and NO_3–N levels for: (a) treatment of BBL graywater batch spiked with BBL detergent (i.e., average over two shockload events; Section 2.3), (b) treatment of a new BBL graywater batch after discharge of detergents spiked graywater treated in (a), and (c) normal BBL graywater treatment operation (three weeks average) prior to laundry detergent shockload events.

treatment was ~0.3 NTU and 0.2 NTU with and without the CFM, respectively (Table 5). Turbidity and bDOC removal in the above tests indicated that the CFM did not affect organics or turbidity removals. The CFM, however, enabled aeration enhancement during treatment (Fig. 5) as evident by the higher DO level (~6 mg/L) relative to the case without the CFM (~4 mg/L). High DO (e.g., >6 mg/L) has been reported to enhance biodegradation of organics (Chapman et al., 1976) as well as nitrification (>5 mg/L DO) (Stenstrom and Poduska, 1980). Furthermore, higher DO concentration allowed for greater oxygen diffusion into the biofilm, thereby enhancing organic degradation by the biofilm (Costerton et al., 1995).

Particle size analysis revealed that the soil layer provided effective removal of particles from non-BBL graywater. The SB-VFW system was effective in removing large particles and achieved

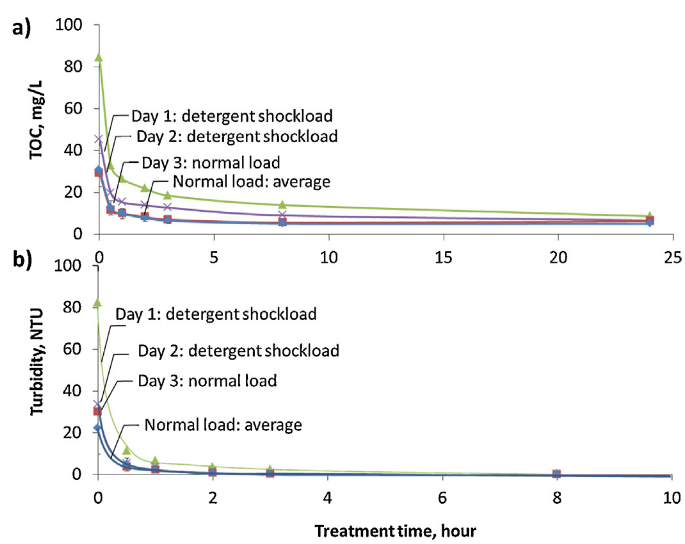

Fig. 8. Removal of (a) TOC and (b) turbidity by the treatment system during BBL detergent shockload tests (Day 1: detergent shockload (160 g BBL detergent added into 300 L raw BBL graywater) and Day 2: detergent shockload (80 g BBL detergent added into 300 L raw BBL graywater), immediately after detergent shockload tests (Day 3: normal load) and the average normal load water quality.

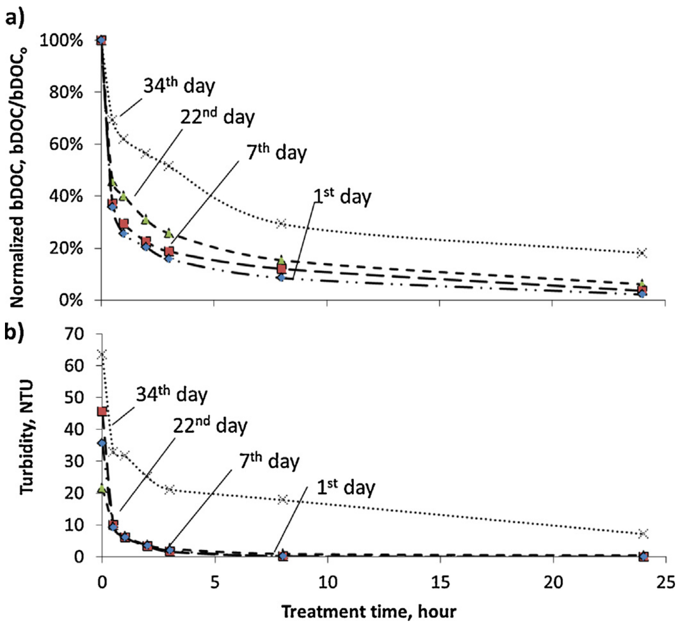

Fig. 9. (a) bDOC, and (b) turbidity treatment performance change during the one month of treating non-BBL graywater.

~88% size reduction (i.e., from 56 μm to 6 μm) within the first hour of treatment (Fig. 6). Further reduction of particle size to 3.8 μm and 67% reduction of effluent particle concentration (down to ~5,800 particles/mL) was achieved after 3 h of treatment (Fig. 6). Over the same treatment period, the effluent bDOC of ~1 mg/L was attained indicating that a limited source of organics was available for the microorganisms in the SB-VFW system. It has been shown that low organic content stimulates extracellular polymer substance (EPS) excretion by microorganisms, which would promote bioflocculation to form larger suspended particles in the water phase (Sheng et al., 2010). The presence of larger aggregates, as a result of bioflocculation, is indeed suggested by the suspended matter size increase (e.g., from 3.8 μm to 44 μm) and particle concentration decrease (e.g., from ~5800 particles/mL to 2000 particles/mL after 24 h of treatment), which are consistent with other biological treatment systems (Leu et al., 2012).

The aerobic environment in the SB-VFW allowed for nitrification to take place, which was indicated by ammonium–nitrogen (NH_4–N) removal and nitrate–nitrogen (NO_3–N) production during the 24 h treatment cycle (Fig. 7). For example, the raw BBL graywater contained 5.2 ± 2.0 mg/L NH_4–N, which was reduced to 0.1 ± 0.1 mg/L after 8 h of treatment. Simultaneously, NO_3–N was undetected in the influent, but increased to ~2.5 ± 1.5 mg/L and ~5.9 ± 1.4 mg/L after three and 24 h of treatment, respectively (Fig. 7). The continual increase of NO_3–N indicates that denitrification did not take place. However, denitrification was observed only at higher organic loading during two BBL detergent shockload events where TOC was increased from the normal average of 32 ± 8.9 mg/L to 84 ± 1.5 mg/L and 45 ± 0.9 mg/L, with NO_3–N remaining <0.1 mg/L during those two 24 h treatment cycles. Nitrification also took place during the detergent shockloading events with NH_4–N being reduced from 2.6 mg/L to <0.1 mg/L within the first 3 h and maintaining at the same above level over the 24 h treatment period. Nitrification–denitrification during the

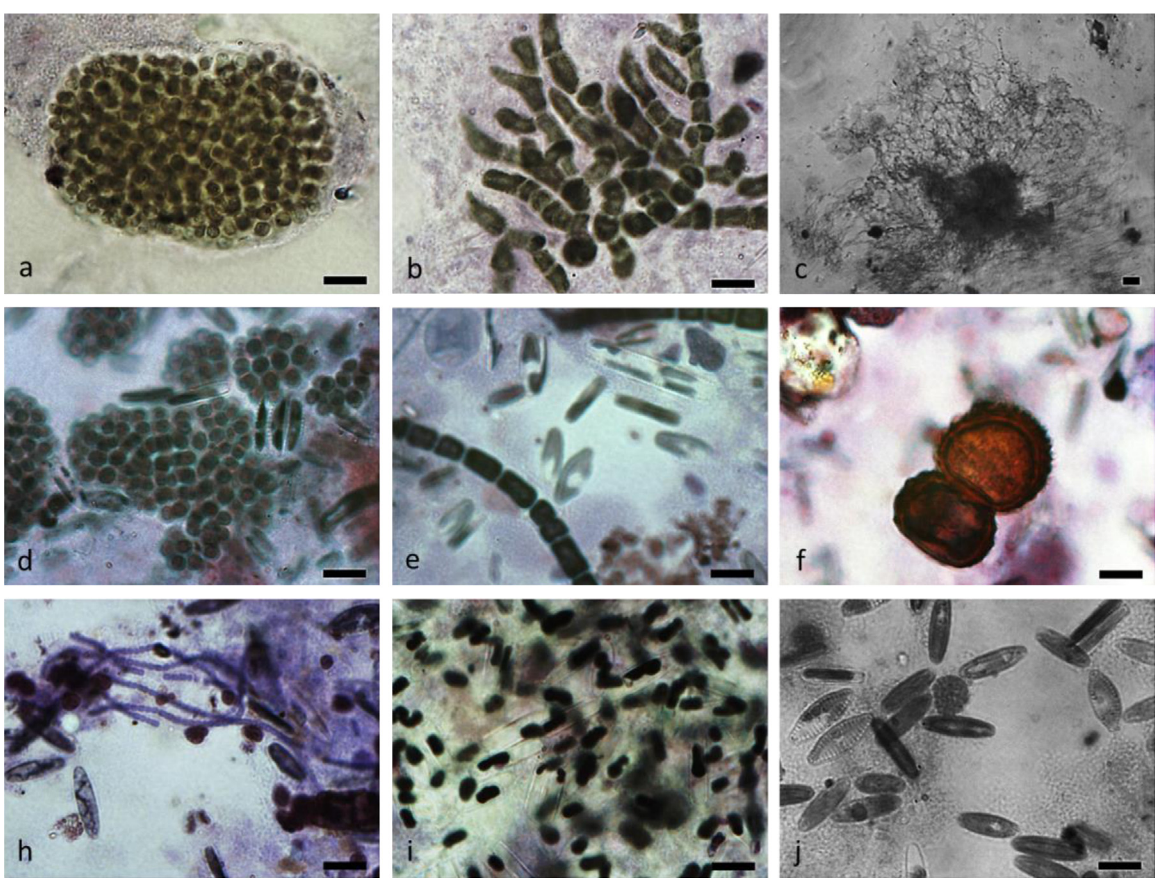

Fig. 10. Biofilm images of samples collected from the plastic media when the treatment system was treating non-BBL graywater. (a) a cluster of algae, (b) algae, (c) fungi, (d) clusters of algae and protozoa, (e) algae and protozoa, (f) eukaryotic microorganisms, (h) algae and protozoa, (i) protozoa, (j) protozoa and algae. Protozoa were found in large number in the biofilm indicating that there was plentiful of bacteria for protozoa to forage on. Scale bars are 10 μm.

detergent shockloading tests suggests that aerobic and anoxic environments co-existed in the soil layer of the SB-VFW system while sufficient carbon source and an anoxic environment enabled denitrification.

The SB-VFW system remained effective with respect to turbidity removal but TOC removal required longer treatment time than usual during two sequential detergent shockloading tests. On the first day of the detergent shockload test, the SB-VFW system achieved ∼61% TOC removal from raw BBL graywater (84 mg/L TOC) within the first half-hour (Fig. 8a). Turbidity removal of 86% was achieved (from 82 to 11 NTU) for the same treatment period (Fig. 8b); removal to the level of 99.6% and below detection was attained after 8 and 24 h of treatment, respectively (Fig. 8b). On the second day of the detergent shockload test, the SB-VFW system produced effluent with turbidity of <1 NTU (0.8 NTU) after 3 h of treatment, and TOC of 6.8 mg/L after 24 h of treatment (Fig. 8). SB-VFW treatment of a new BBL graywater batch immediately after the two-day detergent shockload tests was again at a high treatment efficiency (Fig. 8, e.g., ∼0.3 mg/L bDOC and 0.2 NTU turbidity) with 3 h of treatment. The above observations suggest that the SB-VFW is resilience to detergent shockloads and thus indicates that the system can handle fluctuation in TOC and turbidity levels.

3.3. Organic loading and clogging of the VFW system

The SB-VFW system operated in a stable manner, without clogging during the three month of daily BBL graywater treatment at a recirculation flow rate of 9.5 L/min (or 46 turnovers/24 h). During the operation of the system, the soil hydraulic conductivity was invariant (∼1.5 cm/min). High organic content is known to enhance biomass growth and may also affect the hydraulic conductivity of the soil layer (Zhao et al., 2009). Accordingly, in order to further evaluate the importance of controlling organic loadings in the raw graywater when using the SB-VFW system, the residents were asked to switch from using the non-BBL detergent to the BBL detergent, which contained a lower level of organics (by a factor of ∼2.5 times). After the switch from treating BBL graywater to non-BBL graywater, the hydraulic conductivity of the SB-VFW system remained (for nearly 22 days; Fig. 9a) at about 1.5 cm/min as during the previous 3.5 months of treating BBL graywater. The treatment system achieved 66% bDOC removal (e.g., from 90 mg/L to 32 mg/L bDOC) within the first 30 min of treatment (Fig. 9a) and further removed up to 84% after 3 h of treatment. The effluent bDOC was at 2 mg/L, achieving 98% removal after 24 h (Fig. 9a). Progressive reduction in bDOC removal was observed when the non-BBL graywater was treated for extended period, only attaining 82% of bDOC removal (32 mg/L) (Fig. 9a) for a 24 h batch after 34 days of operation. Turbidity removal by the SB-VFW remained effective, during the first 22 days of treating non-BBL graywater (attaining treated graywater turbidity of <1 NTU and below detection limit after eight and 24 h of treatment, respectively (Fig. 9b)). Progressive wetland soil clogging eventually resulted in significant loss of hydraulic conductivity, and by day 34, water overflow from the wetland into the bottom reservoir resulted in marked increase in effluent turbidity and bDOC. The above system behavior is consistent with other studies in which it was reported that soil clogging can be a concern at high graywater loading of organics and turbidity, as well as with detergents of high organic content (Platzer and Mauch, 1997; Zhao et al., 2009).

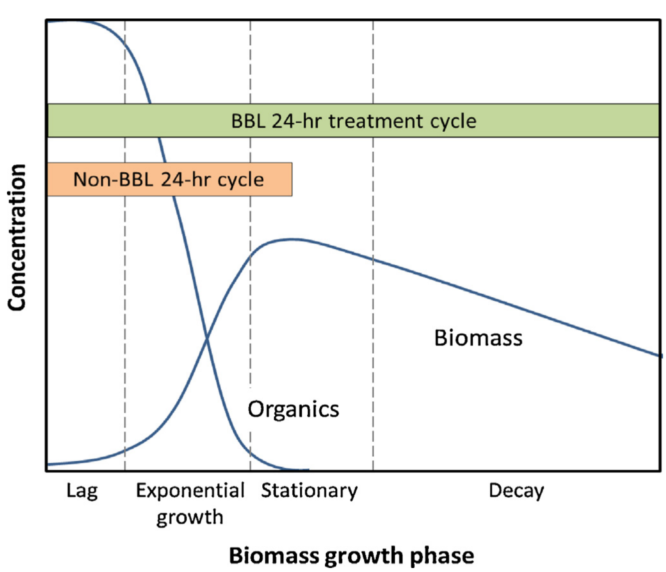

Fig. 11. Biomass growth phase diagram for the SB-VFW system adapted based on the growth phases in a typical biological batch reactor (Metcalf et al., 2004). The cycle of non-BBL graywater is shown to only overlap with the lag, exponential growth and stationary phases; while the non-BBL graywater treatment cycle is shown to overlap with the above three phases in addition to the decay phase.

Reduction of soil hydraulic conductivity after treating non-BBL graywater for over a month was likely due to excess biomass growth stimulated by the organic rich non-BBL graywater. Increase biomass was evident as a biofilm growth (∼2 mm) on the CFM surfaces (Section 3.2) was observed when treating non-BBL graywater. In contrast, a thinner biofilm layer of <1 mm (Fig. 5) was observed when treating BBL graywater. The thick biofilm encountered when treating the organic-rich non-BBL graywater suggests that this type of graywater was capable of supporting greater biomass growth (Fig. 10). Excessive biomass growth can lead to reduction in soil porosity and thus its hydraulic conductivity, and eventual clogging in the SP-VFWs soil (Zhao et al., 2009).

The SB-VFW system is a batch treatment process, and thus the relationship between biomass growth and organic concentration in the SB-VFW system can be conceptualized by a typical biological batch reactor biomass growth behavior as depicted in Fig. 11. In a typical biological batch reactor, biomass growth occurs in the presence of organics while biomass reduction takes place when undergoing endogenous decay under low organic conditions (Fig. 11) (Metcalf et al., 2004). In the present system, when BBL graywater was treated, the bDOC of the recirculating water was already at ∼1 mg/L after 3 h of treatment, which means that water of low organic content was recirculated in the SB-VFW system for approximately 21 hours during the 24 h treatment cycle (Fig. 2a). Recirculation of water of low organic concentration beyond 3 h after complete removal of organics allowed for endogenous biomass decay (Fig. 11). When non-BBL graywater was treated, however, the bDOC concentration remained relatively high (reduced to ∼2 mg/L bDOC after 24 h of treatment) even for a 24 h of treatment cycle (Fig. 2a). Therefore, it is reasonable to expect that the biomass growth was only at the stationary phase when a fresh batch of raw graywater recharged into the SB-VFW system.

In order to illustrate the importance of endogenous decay, a hypothetical treatment cycle of the SB-VFW system, for the BBL and non-BBL, is presented in Fig. 11. Results of the present study suggest that in the non-BBL treatment cycle, there was relatively abundance of organics at the end of the 24 h treatment cycle, which permitted the biomass to undergo the lag, exponential growth and the stationary phases before the next treatment cycle began. An absence of an endogenous decay phase in the non-BBL graywater treatment cycle would mean that the biomass would continue growing and accumulating after each treatment cycle. The above could explain the gradual clogging of the wetland (soil layer). The resulting hydraulic conductivity reduction would then necessitate lowering of the recirculation flow rate to avoid excessive ponding and possible overflowing of water from the top wetland section (Fig. 1). In contrast, in the treatment of BBL graywater, the organics were significantly depleted for a given batch within 3 h of treatment with bDOC reduced to ~1 mg/L (Fig. 2). As a result, the biomass is likely to have undergone an extended endogenous decay phase during the remaining 24 h treatment cycle. It is argued here that the endogenous biomass decay phase is crucial for maintaining a stable biomass (i.e., prevent excessive growth) in the soil layer. The above arguments and higher performance of the SB-VFW for treatment of BBL relative to non-BBL graywater suggest that the use of detergents that are lean in organics would be more suitable if one's goal is to effectively treat graywater for aboveground onsite reuse. For graywater that is high in organics, a thicker soil layer or longer treatment time would be necessary in order to promote sufficient endogenous biomass decay to avoid overgrowth of biomass that would then clog the soil layer. However, the above options would result in higher system footprint and possibly increased odor and other nuisance due to longer raw graywater storage time, as well as larger storage volume. On the other hand, for treatment of graywater with low organic concentrations, a smaller treatment system could also be devised to provide sufficient treatment.

4. Conclusions

The feasibility of residential graywater treatment via a semi-batch vertical flow wetland (SB-VFW) with recirculation was evaluated in a 5-month field study demonstrated the following:

1) Treatment of graywater, from showers, sinks and laundry, in the SB-VFW which contained a coconut coir soil substitute was achieved to a level that meets the strict water quality requirements (with respect to BOD_5, TOC and turbidity) for above ground nonpotable reuse.
2) Treatment time of about 3 h was typically sufficient for batches of up to ~300 L in a small footprint SB-VFW system.
3) System performance was stable when a bio-based laundry detergent was utilized given its lower organic content relative to typical non-bio-based detergents. Clogging of the wetland soil medium due to biofilm overgrowth was avoided when treating graywater of low organic loading (e.g., when utilizing a bio-based detergent) owing to the formation of a stable biofilm exists in the wetland. However, at higher organic loading rates or with detergents of higher organic content, clogging may be a concern.

Results of the present study suggest that there is merit in exploring the use of compact SB-VFW systems for increasing graywater reuse in urban environments, as well as possibly for deployment as a polishing process for certain industrial water treatment applications.

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