Water Reuse

© 2023 The Authors

Water Reuse Vol 13 No 3, 329 doi: 10.2166/wrd.2023.029

Effects of adsorption and filtration processes on greywater microbiological contamination and the potential human health risk reduction

Md Maruf Mortula [a,*], Kazi Parvez Fattah[a], Fatima Iqbal[a,b] and Zahid Khan[a]

[a] Department of Civil Engineering, American University of Sharjah, Sharjah, United Arab Emirates
[b] Currently at Jacobs solutions, Sheikh Zayed Road, City Tower 2, Dubai, United Arab Emirates
*Corresponding author. E-mail: mmortula@aus.edu

MMM, 0000-0003-4974-2833

ABSTRACT

Recycling treated greywater (GW) for onsite, non-potable applications can reduce the potable water demand typically used for non-potable purposes. The conventional methods for GW treatment are limited in their ability to remove wide-ranging pollutants in ways that are inexpensive and use low energy. For this reason, effective and low-cost onsite treatment options are in demand. This study examines the effectiveness of sand filtration (SF), granulated blast furnace slag (GBFS), and activated carbon (AC) in the treatment of GW from a residential apartment building in Sharjah, United Arab Emirates. The study relies on four different pilot-scale experimental setups to investigate the effectiveness of SF, AC, and GBFS in treating microorganisms from GW and evaluate the microbial risk reduction using these treatment processes. A quantitative microbial risk assessment (QMRA) approach is used for risk assessment. Results show that GBFS achieves a higher reduction of total coliform (TC) (0.54–2.05 log removal) and fecal coliform (FC) (1.96–2.30 log removal) than AC. SF improves reduction by 0.13–3.39 log removal and 1.11–3.68 log removal for TC and FC, respectively. The study also reveals substantial FC and *Escherichia coli* risk reduction by SF, AC and GBFS.

Key words: activated carbon, adsorption, granular blast furnace slag, greywater recycling, human risk reduction, microbial reduction

HIGHLIGHTS

- The study examined the role of filtration and adsorption processes in removing microorganisms and eventual risk reduction from greywater.
- Filtration was effective in removing TC, FC, and *Escherichia coli*.
- Both AC and GBFS were effective in reducing all the microorganisms.
- GBFS was better in TC and FC removal than AC.
- All the treatment processes were capable of substantially reducing risk.

1. INTRODUCTION

Water scarcity is a significant issue impacting economic development, environmental sustainability, and quality of life. The sustainable usage of existing water resources is key to addressing the challenge of rising water demand. Black water originating from water closets contains significant amounts of pollutants in comparison to greywater (GW), which is defined as urban wastewater that excludes any contribution from water closets and includes wastewater (WW) from baths, showers, hand bowls, laundry, dishwashers, and kitchen sinks (Eriksson *et al.* 2002). Some literature (Al-Jayyousi 2004) excludes the contribution of kitchen WW from other GW streams. Given that approximately 50–80% of the WW is categorized as GW, one of the many strategies for water management can be recycling of GW for non-potable purposes. Onsite GW reuse after treatment can potentially mitigate water shortage issues.

Although GW reuse is critical, there are concerns about reclaimed water quality, human health perceptions, and economic considerations. The GW quality can vary with the source, geographical location, lifestyles, traditions, installations, level of occupancy, demographics, and chemical usage (Eriksson *et al.* 2002; Al-Jayyousi 2004). Additionally, biological and chemical degradation during storage and transportation can further alter the GW quality. In GW, total coliforms (TC) range from 101

to 107 CFU/100 mL (Eriksson *et al.* 2002). The concentration of coliforms significantly depends on the demographic distribution of the individuals. High concentrations of total and fecal coliforms have been estimated in the GW generated from a household comprising of young children with values of 3.2×10^5 and 1.5×10^3 CFU/100 mL, respectively, in comparison to 80 and 6 CFU/100 mL for a household without children (Rose *et al.* 1991). Commonly, microbiological contaminations are measured using *Escherichia coli*, TC, and fecal coliform (FC) indicators. *E. coli* is considered as the most common pathogenic indicator of pollution (Odonkor & Ampofo 2013). Also, the presence of *E. coli* is often associated with the potential presence of other pathogenic organisms, including *Salmonella* spp. or hepatitis A virus (Brüssow *et al.* 2004; Truchado *et al.* 2016). These bacteria are induced in the GW from washing hands after using latrine, washing infant's contaminated clothes including diapers, and washing food waste from the plates and utensils. The presence of these different microbiological contaminants makes the raw GW reuse undesirable. This often remains a key barrier to a wide-scale implementation of GW reuse. The treatment of GW has the potential to address the concerns of microbiological and other concerns of GW reuse.

GW treatment can be carried out onsite or at a separate, centralized location within a community. However, the treatment system should be sustainable, and the reclaimed water quality should meet the standard reuse guideline for health and safety. Among numerous available water treatment technologies, adsorption using solid materials is considered to be the simplest and most effective process (Crini *et al.* 2019). Activated carbon (AC) is an extensively and widely used adsorbent material for industrial-scale water treatment. However, although AC is a preferred adsorbent, its widespread application is limited due to it being quite expensive (Crini *et al.* 2019). Alternative materials, like ground blast furnace slag (GBFS), can act as a viable option (Zannerni *et al.* 2020). Even though the utilization of GBFS for the extraction of heavy metals and phosphorus from aqueous solution and wastewater has been documented (Mortula *et al.* 2007), no studies have evaluated the use of GBFS in the treatment of microorganisms. Apart from adsorption, there are other technologies used previously for GW treatment. An experimental study was carried out to assess the use of coagulation and flocculation on shower GW in the United Kingdom (Pidou *et al.* 2008). Both ferric sulfate and aluminum sulfate displayed comparable TC and *E. coli* removal efficiencies of more than 99.9%. A study by Sinno *et al.* (2022) found that electrocoagulation was capable of substantially removing turbidity, total organic carbon, chemical oxygen demand and grease. One of the studies evaluated the recycled vertical flow bioreactor by treating synthetic GW (Gross *et al.* 2007). The experimental results demonstrated that the *E. coli* reduced from 501,188 to 1.25 CFU/dL. Overall, the removal efficiency of filtration and membrane bioreactor treatment systems is 100% for *E. coli* and 99.9% for TC (Ghaitidak & Yadav 2013).

GW reuse is always associated with concerns of potential contamination and microbial regrowth and eventual spread of diseases. Different researchers have investigated and assessed the impact of GW reuse on human health. Untreated GW reuse often poses an unacceptable risk to human health from wide-ranging levels of exposure from disposal to reuse. Nunez *et al.* (2014) evaluated the risk of human health in Argentina due to untreated GW disposal practices. The study observed the presence of indicator organisms in the disposal sites and identified that the filtration process removes the microorganisms in significant numbers. Neto *et al.* (2018) investigated the risk of treated GW usage and observed toilet flushing exposure to be the highest microbial risk. Kusumawardhana *et al.* (2021) used a quantitative microbial risk assessment (QMRA) approach for conducting mathematical health risk estimation. The study investigated both rainwater and untreated GW for the risk posed by *Staphylococcus aureus*, *E. coli*, *Salmonella* spp., and *Giardia lamblia*, among others. They reported the necessity of appropriate GW treatment for acceptable risk. Shi *et al.* (2018) also used the QMRA approach for the risk estimation of treated GW. The study focused on the exposure of treated GW pollutants during toilet flushing and food-crop irrigation. Treatment using microfiltration was capable of ensuring an acceptable risk level for toilet flushing. Blanky *et al.* (2017) used the QMRA approach to evaluate the risk for toilet flushing and gardening from *Legionella* only. The researchers did not find significant risk reduction from the use of treated (using constructed wetland) and chlorinated GW for garden irrigation and toilet flushing.

Despite the obvious need, there have been limited studies on the treatment of GW for reuse purposes. Also, even though some of the individual treatment processes were investigated in terms of their efficiency of pollutant removal, the adsorption process was not examined for removing microbial contamination and the eventual risk. The primary goal of this paper is to investigate the effectiveness of microbial inactivation and risk reduction using AC, GBFS and sand filtration (SF) in treating GW. Pilot-scale experimental setup was used to treat GW generated in a large apartment complex. The QMRA approach was used to conduct the risk estimation for FC and *E. coli*.

2. METHODOLOGY

2.1. Study area

For this study, a 30-floor residential tower (Al Jawad Building) located in the Al Nahda area in Sharjah was chosen. The building was chosen as it already has an operational large-scale onsite GW treatment facility, which treats residential GW generated from baths and toilet sinks. The system proposed in this study was set up from the existing treatment system using a combination of different bypass pipes.

2.2. Adsorbent and filter media

To meet the objective of this study, three materials (AC, GBFS, and filter media (silica sand)) were used. AC and silica sand were used from Jacobi Carbons and National Factory for Processing and Treating Minerals, respectively. Table 1 shows the properties of AC, GBFS and silica sand.

2.3. GW collection system

While the commode flush water (black water) directly discharges into the sewer system, the water generated from the bathroom sink and bath discharges into a separate pumping system installed only for the collection of GW from these two sources. During the days when the volume of generated GW is higher than the treatment system's capacity, the excess GW overflows into the sewer system without treatment.

2.4. Experimental setup

The treatment process involved the collection of GW in an aerated underground storage tank. Two pumps (one duty and one standby) were used to lift the water into different treatment process designs. The filtration and adsorption columns had sizes of 137 mm in height and 33 mm in diameter and were operated with a maximum flow of 1 m^3/h at a pressure of 2.5–3.0 bar. Sodium hypochlorite was used for disinfection to maintain a residual chlorine level of 1 mg/L in all the setups. All the treatment designs were operated at a constant hydraulic loading rate (HLR) of 11.7 $m^3/m^2/h$ (or 280.8 $m^3/m^2/day$). The HLR was chosen in line with the existing GW treatment system within the apartment complex. Filtration and adsorption systems were back washed every day for 10 min at a flow rate of 2.5 m^3/h using effluent from the media.

Four different process designs were examined to accomplish the objectives of this study. Each of the designs was operated for 1 week but not simultaneously. The first design used alum pretreatment, followed by filtration, AC adsorption and chlorination (Figure 1(a)). A total of three sampling points (SP) were identified to evaluate the effectiveness of the different treatment processes. SP1 samples indicate the influent pollutant concentrations, SP2 samples represent the effluent concentrations from filtration and AC adsorbent, and SP3 samples represent effluent concentration after the disinfection process. The second design was similar to the first design, except that GBFS replaced the AC adsorption column (Figure 1(b)). This design was intended to assess the differences between AC and GBFS columns.

The third design relied on AC adsorbent for the treatment of the GW without any pretreatment using filtration (Figure 1(c)). Hence, the GW influent is directly fed into the AC column. Treatment also used a UV disinfection before the chlorination

Table 1 | Properties of the adsorbents and filter media

Properties	AC	GBFS	Silica Sand
Moisture content (%)	5	0.1–0.3	<0.05
Bulk density (kg/m^3)	450	1030–1098	>1500
Surface area (BET, m^2/g)	900	–	–
Specific gravity (g/cm^3)	–	2.89–2.90	>2.6
Effective size (mm)	0.4		0.43–0.85
Mean particle diameter (mm)	0.6		
Uniformity coefficient	1.5		<1.5
Calcium oxide/CaO (%)		40–42	
Silicon dioxide/SiO_2 (%)		32–34	99.9
Aluminum oxide/Al_2O_3 (%)		11–14	

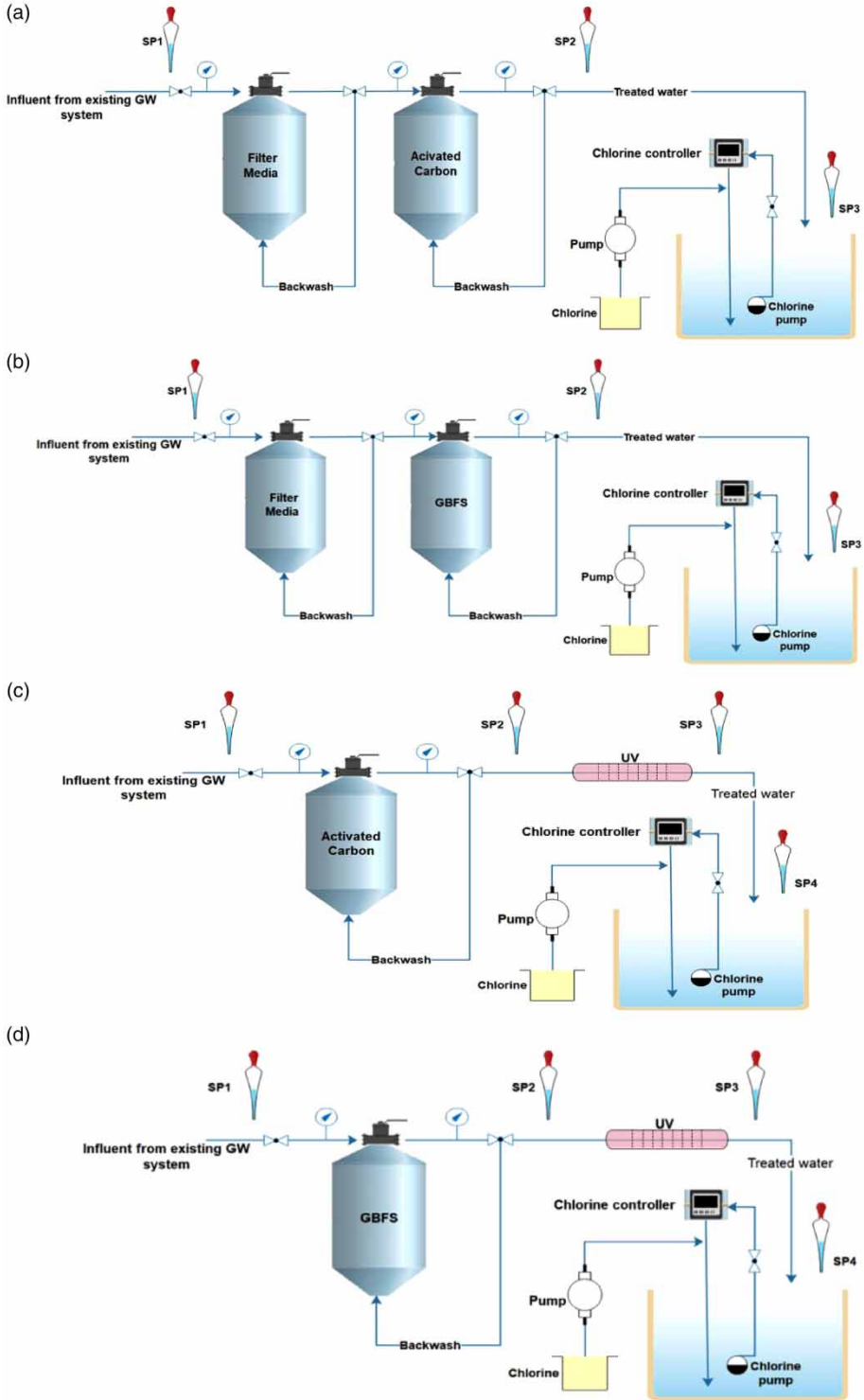

Figure 1 | GW treatment process designs. (a) Design 1, (b) Design 2, (c) Design 3, and (d) Design 4.

process. UV sterilizer with a treatment dose of 40 mJ/cm^2 was used in the treatment designs. The process design is similar to the first design except for the use of a filtration system, including alum pretreatment. The fourth design was similar to the previous (design 3), except GBFS was used in place of AC columns (Figure 1(d)). This was done mainly to determine the microbial removal efficiency of GBFS without any dependency on pretreatment.

2.5. Water quality analysis

Grab samples were collected every day at 6 pm (±1 h). Duplicate samples were collected from each sampling location to ensure the reliability and accuracy of the result. Sampling bottles for microbial analysis were sterile, to ensure there was no bacterial contamination, and sodium thiosulfate was used to neutralize the effect of any chlorine at the time of collection. For this reason, 50 mL sterile bottles with sodium thiosulfate were used. The sampling points were cleaned with 70% ethanol to ensure minimal bacterial contamination. All the samples for microbial analysis were tested on the day of collection.

Three types of microbial analysis including TC, FC, and *E. coli* were conducted in this study. Standard analytical techniques were adopted for analysis (APHA 2012). The presence of TC and FC is detected using the 'most probable number'. On the other hand, the presence of *E. coli* was detected using the 'spread plate method'. Table 2 shows the culture medium, technique, standard, and colony color used for assessing different microorganisms. Samples from each sampling location were collected three times a week and tested for microbial contamination.

The TC, FC, and *E. coli* analysis required the preparation of Petri dishes with a culture medium. For TC and FC culture media, a total of 3 L of culture medium was prepared. Each liter of culture medium was prepared by adding 33.12 g of TTC Tergitol-7 agar base in 1,000 mL of distilled water and heated to the boiling point to homogenously dissolve the medium. For *E. coli* analysis, a total of 2 L of culture medium (TBX agar) was prepared by suspending 36.6 g of TBX agar in each liter of distilled water. The prepared agar solution was autoclaved at 15 lbs pressure (121 °C) in a sterilized bottle and cooled to 45–50 °C. The solution (15 mL) was then poured into petri dishes and covered the circumference of the dish.

Since the GW samples were highly contaminated with bacteria, dilution of the samples was necessary. An ideal result is achieved when the number of colonies lies within the range of 20–80 colonies per membrane; however, the number of colonies should not be more than 200 (APHA 2012). The dilution factor varied (from no dilution to 1,010) at different sampling points depending on the microbial contamination of the sample.

The experimental procedure for both TC and FC was identical. The diluted samples were filtered through a cellulose acetate grid marked membrane with a uniform pore diameter of 0.45 μm. The bacteria-filled membrane was then placed on the Petri dishes with a culture medium avoiding any air pockets. The Petri dish was closed and kept in the incubator at 36 °C for 24 h (±1 h) and 44 °C for 24 h (±1 h) for TC and FC growth, respectively. For *E. coli* bacteria, the spread plate technique was applied. A 0.1 mL sample (without any dilution) was pipetted onto the Petri dish with TBX agar culture medium. A sterilized 'delta- shaped' spreader was used to evenly spread the sample over the Petri dish with culture medium. The Petri dishes were sealed with a lid and incubated at 44 °C for 24 h (±1 h). Once the incubation was done, the number of colonies was counted. Samples having more than 200 colonies were rejected and retested. The bacterial count is generally reported in terms of colony forming units (CFU). The bacterial colonies were counted using a colony counter. Based on the standard method 9222.B (APHA 2012), Equation (1) was used for calculating the number of TC/FC colonies per 100 mL, whereas Equation (2) was used for calculating the number of *E. coli* colonies per 100 mL in the tested sample.

$$\text{TC or FC (CFU)}/100\text{ mL} = \left[\frac{\text{Number of brick red colonies}}{\text{Volume of filtered sample}}\right] \times \text{Diltuion factor} \times 100 \quad (1)$$

$$\textit{E. coli}\text{ per CFU}/100\,\text{mL} = \left[\frac{\text{Number of blue-green colonies}}{\text{Volume of filtered sample}}\right] \times 100 \quad (2)$$

Table 2 | Culture medium, technique, standard, and colony assessment for bacterial count

Bacteria	Culture medium	Culturing technique	Standard	Colony
Total coliform	Triphenyl Tetrazolium Chloride (TTC) Tergitol-7 Agar	Membrane filtration on the medium; incubation at 36 °C for 24 h (±1 h)	ISO 9308-1	Brick red with yellow central halo
Fecal coliform	Triphenyl Tetrazolium Chloride (TTC) Tergitol-7 Agar	Membrane filtration on the medium; incubation at 44 °C for 24 h (±1 h)	ISO 9308-1	Brick red with yellow central halo
E. coli	Tryptone bile X-glucuronide (TBX) Agar	Spread 0.5 mL sample on medium; incubation at 44 °C for 24 h (±1 h)	ISO 16649-2	Blue-green

2.6. Health risk estimation

Microbial enumerations are critical for risk estimation. Even though *E. coli* is the most specific type of organism used for indicating pathogenic organisms, many wastewater treatment plants around the world only monitor TC and FC. In this study, the QMRA model was used for both FC and *E. coli*, as they are known as indicator organisms.

The quantitative microbial risk assessment (QMRA) guideline (Haas *et al.* 2014; Shi *et al.* 2018; Kusumawardhana *et al.* 2021) was used in the study. Among the different microorganisms tested in this study, *E. coli* was chosen as the most targeted to the pathogenic microorganisms. However, it is rare to find the exact number of pathogenic *E. coli* in GW systems. A pathogenic ratio is introduced to estimate the concentration of pathogenic *E. coli* using Equation (3).

$$C_P = C \times R_p \tag{3}$$

where C_P is the estimated concentration of pathogenic *E. coli* in domestic GW (CFU/100 mL), and R_p is the pathogenic ratio from *E. coli* to pathogenic *E. coli* (unitless). The pathogenic ratio is the proportion of *E. coli* with toxin genes to all *E. coli* (O'Toole *et al.* 2012). The value of R_p in different GW was estimated in the same study to be in the range of 0.028–1, with 1 being a conservative estimate of 100% pathogenic *E. coli*. In this study, the same range was used. Since approximately 8% of FC is pathogenic in nature, a R_p value of 0.08 was used for FC (Haas *et al.* 2014).

The pathway of exposure is considered to be the inhalation of splashed GW aerosols, similar to the one conducted during toilet flushing (Kusumawardhana *et al.* 2021). Pathogenic *E. coli* causing gastrointestinal infection is considered and it is assumed that all aerosols reaching human noses reach gastrointestinal tracts, representing the worst-case scenario. The dose was estimated using Equation (4).

$$\text{Dose}_P = C_p \cdot \text{PC} \cdot \text{IR} \cdot T \cdot F_{RA} \cdot \text{RR} \cdot N \tag{4}$$

where Dose_P is the daily dose (CFU), C_p is the concentration of pathogens in water (CFU/L), PC is the partitioning coefficient (L/m^3), IR is the inhalation rate (m^3/min), T is the duration of exposure event (min), F_{RA} is the fraction of respirable aerosol, RR is the retention rate, and N is the number of events per day. Typical parameters used in this study are presented in Table 3.

Two commonly used dose-response models (exponential and beta-Poisson) were used to develop the connection between the exposure level of the pathogen and the probability of adverse effects. The beta-Poisson model was used to estimate the infection risk, P_{inf}, using Equation (5).

$$P_{inf} = 1 - \left[1 + \text{Dose}_p \frac{2^{(1/\alpha)} - 1}{N_{50}}\right]^{-\alpha} \tag{5}$$

where Dose_p represents the dose of pathogenic *E. coli* of pathogenic FC inhaled (CFU), and α and N_{50} are the best-fit parameters of the model. The exponential model was used to estimate the illness risk, P_{ill}, using Equation (6).

$$P_{ill} = 1 - \exp(-k \times \text{Dose}) \tag{6}$$

Table 3 | Exposure parameter values

Variable	Value	Unit	Reference
Partitioning coefficient of aerosol for toilet flushing	2.3×10^{-5}	L/m^3	Blanky *et al.* (2017)
Inhalation rate of aerosol	0.013	L/m^3	Dean & Mitchell (2020)
Respirable fraction of aerosol	0.997 (uniform)		
Retention rate of aerosol	0.58 (uniform)		Hamilton *et al.* (2017)
Duration of toilet flushing	5	min	Hamilton *et al.* (2017)
Flush frequency	8	days	Shi *et al.* (2018)

where k is the best-fit parameter of the model that represents the pathogenicity of pathogenic *E. coli* or pathogenic FC. For the toilet flushing scenario, a frequency of eight times a day was applied to represent the worst-case scenario of a healthy human. The model parameters used in this study are shown in Table 4.

The annual infection and illness risk can be calculated based on the equations from Federigi *et al.* (2019) (Equations (7) and (8))

$$P_{\text{inf. annual}} = 1 - (1 - P_{\text{inf}})^{365.F} \tag{7}$$

$$P_{\text{ill.annual}} = 1 - (1 - P_{\text{ill}})^{365.F} \tag{8}$$

where $P_{\text{inf.annual}}$ and $P_{\text{ill.annual}}$ are the estimated annual infection risk and annual illness risk, respectively, and F refers to the frequency of toilet flushing per day (eight times used in this study).

3. RESULTS AND DISCUSSION

3.1. Total coliform

Table 5 presents mean influent TC for all the treatment process designs. The TC ranged from 1.0×10^{12} to 1.8×10^{14} CFU/100 mL. The values are higher than those in previous studies (1.2×10^{3}–8.2×10^{8} CFU/100 mL) (Masi *et al.* 2010). The high values may be due to, but not limited to, hand washing activity after toilet use, washing of babies or diapers of babies in the bath or sink, age distribution and number of household members, traces of urine, dead skin, sweating from body, and usage pattern and season (Blanky *et al.* 2015).

3.1.1. TC reduction by filtration and adsorbents

Figure 2 shows the overall log reduction achieved by all the treatment process designs. The log reductions by SF + AC and SF + GBFS were in the range of 0.4–1.7 and 3.9–4.8 log CFU/100 mL, respectively. With the SF, GBFS demonstrated higher TC removal efficiency than AC. Although the influent received by SF + GBFS had a higher TC concentration in contrast to SF + AC, TC reduction was still higher for SF + GBFS. The presence of a filtration system demonstrated higher TC removal than those designs without the filtration for both AC and GBFS systems. It can be explained by the enhanced straining process from both the filtration and adsorbents, as explained in the previous literature (Williams *et al.* 2007). Two different types of materials in filtration and adsorption columns might have provided additional advantages in removing TC through a

Table 4 | Exposure parameter values

Parameters for dose response model	Value	Reference
α	0.155	DuPont *et al.* (1971)
N_{50}	2.11×10^{6}	
k	1.22×10^{-8}	

Table 5 | Influent concentrations of different treatment designs

	TC			FC			*E. coli*		
	Mean CFU/100 mL			Mean CFU/100 mL			Mean CFU/100 mL		
Design	Day 1	Day 4	Day 7	Day 1	Day 4	Day 7	Day 1	Day 4	Day 7
1	2.3×10^{13}	2.4×10^{13}	1.3×10^{13}	2.5×10^{7}	8.0×10^{8}	5.2×10^{8}	5.5×10^{3}	4.0×10^{3}	4.0×10^{3}
2	4.5×10^{13}	1.3×10^{13}	2.1×10^{13}	3.5×10^{7}	1.2×10^{8}	1.0×10^{7}	4.0×10^{3}	6.5×10^{3}	1.2×10^{4}
3	1.5×10^{12}	2.0×10^{12}	1.0×10^{12}	3.0×10^{7}	9.0×10^{7}	7.0×10^{7}	9.0×10^{3}	1.6×10^{4}	1.4×10^{4}
4	1.1×10^{14}	2.2×10^{13}	4.0×10^{12}	2.0×10^{7}	8.0×10^{7}	1.4×10^{8}	2.0×10^{3}	8.0×10^{3}	1.4×10^{4}

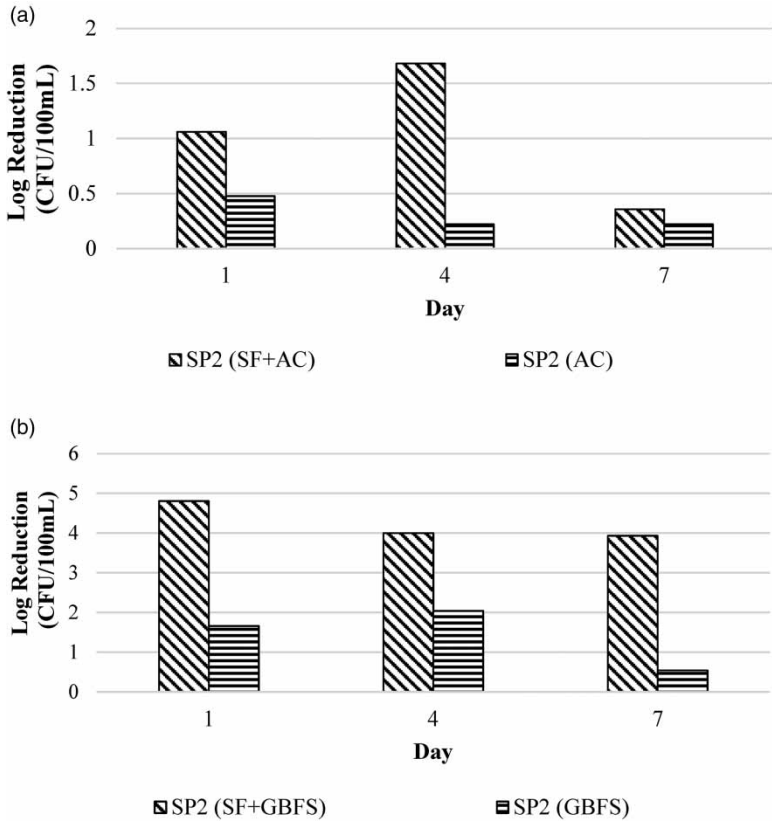

Figure 2 | TC log reduction comparison between (a) SF + AC (Design 1) and AC alone (Design 3) and (b) SF + GBFS (Design 2) and GBFS alone (Design 4).

collection of many different types of microorganisms. The efficiency of the double filtration system can also be attributed to the small particle sizes of the media and the highly active surface of the adsorbents.

3.1.2. TC reduction efficacy of adsorbents

Results demonstrated that the GBFS achieved significantly higher log reductions for TC than AC (Figure 2). GBFS reduced TC in the range of 0.5–2.1 log CFU/100 mL, compared to 0.2–0.5 log CFU/100 mL for AC. GBFS demonstrated higher TC removal of 97.8, 99.1, and 71.3% on days 1, 2, and 3, respectively, compared to 40.0, 40.0, and 66.7% on days 1, 2, and 3, respectively, for AC. This is consistent with the findings from TC removal with the filtration process (designs 1 and 2). The results are also consistent with the previous literature, where AC is capable of limited TC removal (Hijnen *et al.* 2010). The authors identified that the removal efficiencies are high for native coliform species. The presence of surface charge did not seem to have a significant impact on the TC reduction efficiency. The low (up to 0.5) log reductions from AC can be attributed to the fact that only the physical straining process has been effective and other surface-based processes did not contribute much to the TC removal mechanism. TC is typically made up of wide-ranging microorganisms, and AC was not capable of removing diverse groups of microorganisms well. Regarding the GBFS, the results are consistent with the literature, where 2 log reductions of TC were achieved using blast furnace slag (Abdolahnejad *et al.* 2017). However, the study mostly operated in slow medium filtration as opposed to rapid filtration in this study. For this reason, the result in this study is not likely to be due to schmutz deck media, which is typical for slow sand filters. It is possible that the presence of heavy metals in the blast furnace slag inactivated wide-ranging microorganisms. While it was not part of the current study, future investigations are planned to determine the impact of GBFS composition on the inactivation of different microorganisms.

3.1.3. TC reduction efficacy of filtration

The study revealed the role of filtration in enhancing TC reduction. Figure 2(a) compares the TC log reduction achieved by both SF + AC and AC alone, whereas Figure 2(b) exhibits the TC log reduction calculated for SF + GBFS and GBFS alone.

While the TC reduction ranged between 0.2 and 0.5 log CFU/100 mL with AC alone design, this reduction was improved to 0.4–1.7 log CFU/100 mL with the inclusion of a filtration system. A similar pattern was noticed with GBFS too. The TC removal with GBFS varied between 0.5 and 2.0 log CFU/100 mL, eventually improving to 3.9–4.8 log CFU/100 mL with the addition of the filtration system. The results (0.1–3.4 log CFU/100 mL) are consistent with the previous studies that ranged between 0.6 and 4 log CFU/100 mL (Li *et al.* 2012). Although the previous literature by Li *et al.* (2012) identified the processes to be improved straining process with time, quick ripening, and heterogeneity in the media (Williams *et al.* 2007), however, the reduced TC removal efficiencies toward the end of 7 days were noticeable in this study. The reduction in filter efficiency is a common phenomenon as the filter media becomes coated with flocs, plugging the voids between the filter grains, thereby reducing the straining capacity and adsorption ability of the media. Also, during the experiments on the 7th day, it was far from the last backwash conducted on the media. This indicated that the filter media were saturated.

3.2. Fecal coliform

Table 5 presents mean FC concentrations for GW influents for all the different designs. The FC varied in the range of 7.0–8.9 log CFU/100 mL. This is consistent and a bit higher than the previous studies that determined up to 6 log CFU/100 mL (Masi *et al.* 2010). This high value could be due to the presence of a nursery facility in the building, increasing the possibility of fecal contamination due to the washing of babies or diapers in the bath or sink.

3.2.1. FC reduction efficacy using filtration and adsorption

Figure 3 presents the overall log reduction achieved by both the treatment designs for days 1, ,4 and 7. The results indicate that with the filtration, GBFS showed higher performance efficiency by displaying lower FC counts (1.7–2.2 log CFU/100 mL) and higher removal efficacy (5.3–5.9 log CFU/100 mL) than filtration and AC treatment as FC counts (4.1–4.3 log CFU/100 mL)

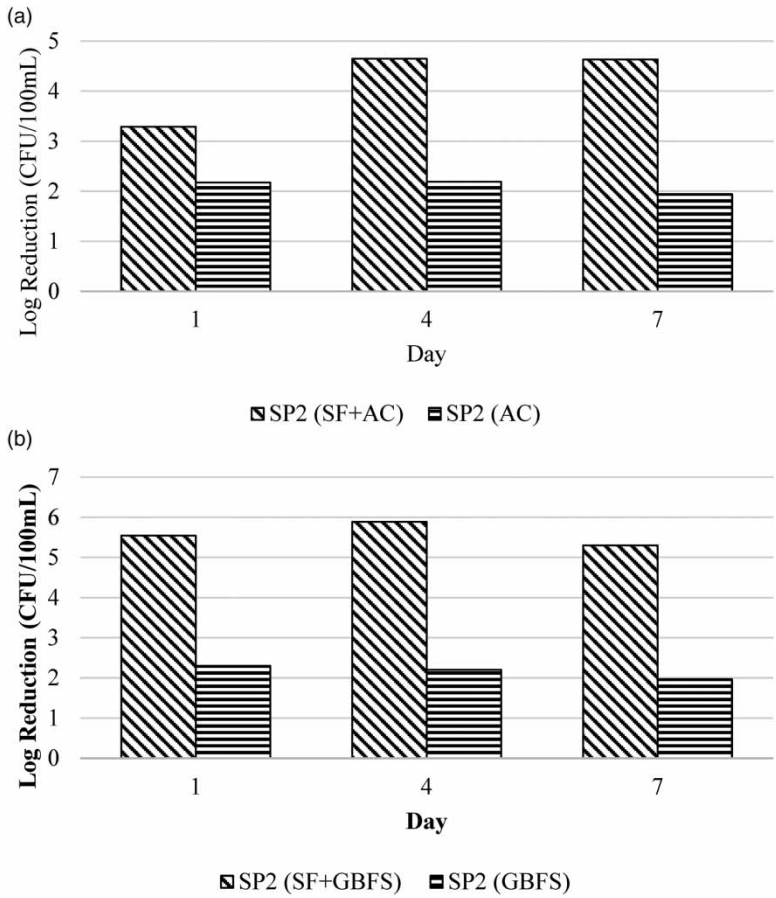

Figure 3 | Mean FC log reduction (a) SF + AC and AC alone and (b) SF + GBFS and GBFS alone.

and lower removal efficacy (99.95–99.99%) post-treatment. Since the experimental approach, including the sampling method, filtration media (sand filter), sampling time, sampling location, and HRT and backwash rate, was identical for both the designs, it can be concluded that GBFS have higher FC reduction efficiency compared to AC. Also, this finding coincides with the results achieved for TC, whereby filtration and GBFS demonstrated higher TC removal efficiency than AC. It is understandable as TC is a representation of a group of bacteria including FC. The performance is consistent with previous studies on FC removal efficiencies of other GW treatment systems (rotating biological contactors, membrane bioreactors, and filtration) (Friedler *et al.* 2011). The FC log reductions for both designs 1 and 2 were substantially higher than the TC log reductions. The results indicate that a combination of filtration and adsorption is capable of removing fecal coliforms better than other types of coliform organisms that are part of TC. Similar to TC, the combined (filtration and adsorption) GW treatment was capable of achieving higher FC log reductions than the adsorbents alone (designs 3 and 4). The results are logical since both filtration and adsorption were operated in a depth filtration mode.

3.2.2. FC reduction efficacy of adsorbents

Figure 3 illustrates the FC removal efficacy achieved by both AC and GBFS. Results demonstrated that AC without any filtration achieved a reduction of 2.2, 2.2 and 1.9 log CFU/100 mL for days 1, 4, and 7, respectively. For a similar design, GBFS successfully removed 2.3, 2.2 and 2.0 log CFU/100 mL on days 1, 4, and 7, respectively. The performance of the treatment system used in this study was better than a previous study on GW treatment using an aeration process followed by AC adsorption, which achieved the FC removal of approximately 45–56.7% (Radhi & Borghei 2017). GBFS was capable of removing FC better than AC. It is consistent with the findings from FC removal with the filtration process (designs 1 and 2). As reported earlier, AC has limited ability to perform on coliform organisms. However, the FC log reductions for AC were better than TC log reductions. It is consistent with the notion that the physical straining process is more suitable for FC than other coliform organisms. The FC log reductions of GBFS in both the designs (with and without filtration) were also higher than those for TC log reductions. The high FC log reductions for GBFS compared to AC may be due to the presence of heavy metals and metal oxides in the blast furnace slag that can cause FC to inactivate and eventually be removed. The adsorptive capacity of GBFS in fixed bed column experiments is better than that of AC, indicating the reason for the improved FC reduction (Mortula & Gagnon 2007).

3.2.3. FC reduction efficacy of filtration

Similar to TC, results demonstrated that filtration significantly improved the FC removal efficiency of the system. While the reduction ranged between 1.9 and 2.2 log CFU/100 mL with the AC alone, this reduction range was improved to 3.3–4.6 log CFU/100 mL with filtration and AC. Similarly, the use of GBFS alone achieved FC removal efficiencies of 2.0–2.3 log CFU/100 mL, which was improved to 5.3–5.9 log CFU/100 mL for the system with filtration and GBFS. Hence, the overall removal efficiency of the filtration process can be attributed to 1.3–2.5 and 3.3–3.6 log CFU/100 mL when combined with AC and GBFS, respectively. A similar reduction in FC removal efficiency (62.5–100%) has also been reported in several studies conducted on SF (Radhi & Borghei 2017; Sharath *et al.* 2017). Since the process used in this study was rapid SF, biological layers are highly unlikely to remove microorganisms, commonly observed in slow sand filters. FC log reductions using filtration are higher than TC log reductions. It is consistent with other adsorbents in that filtration can remove FC better than other coliform organisms. Unlike TC log reductions, there was no consistent decrease in FC removal on the 7th day. As was the case for decreased efficiency due to clogging, the impact was not that considerable for FC removal performance.

3.3. *Escherichia coli*

Table 5 also provides the mean *E. coli* numbers as CFU/100 mL quantified for GW influent corresponding to all the designs. The influent *E. coli* in this study ranged 3.7–4.2 log CFU/100 mL, and these are consistent with previous literature on GW (Atanasova *et al.* 2017).

3.3.1. *E. coli* reduction using filtration and adsorption

There were no *E. coli* bacteria observed in the effluents of both the treatment processes (filtration and adsorption) investigated in this study. It indicates that the efficiencies of *E. coli* removal using filtration combined with AC and GBFS were observed to be at 100% (Table 6). However, due to zero *E. coli* numbers in the effluents, the study could not examine the efficiencies of the filtration process alone. There were no studies examining both the filtration and adsorption processes together. Several studies have reported achieving 100% *E. coli* removal efficacy using filtration (Ghaitidak & Yadav 2013).

Table 6 | Mean *E. coli* values for SF + AC and SF + GBFS at SP1 (influent) and SP2 (post filtration/adsorption)

	Mean *E. coli* (CFU/100 mL)			Mean *E. coli* (CFU/100 mL)		
	Day 1	Day 4	Day 7	Day 1	Day 4	Day 7
Configuration	SP1			SP2		
1 (SF + AC)	5.50×10^3	4.00×10^3	4.00×10^3	0	0	0
2 (SF + GBFS)	4.00×10^3	6.50×10^3	1.20×10^4	0	0	0
3 (AC)	9.00×10^3	1.60×10^4	1.40×10^4	0	0	0
4 (GBFS)	2.00×10^3	8.00×10^3	1.35×10^4	0	0	0

However, some other studies reported that the *E. coli* removal efficiency of filtration ranged from 95.0 to 98.5% (Sharath *et al.* 2017). As observed, the influent concentrations were in the range of 10^3–10^4. For this reason, 3–5 log reductions of *E. coli* would be sufficient to remove the microorganism completely. It indicates that prior treatment using adsorption and/or filtration was capable of reducing *E. coli* completely. *E. coli* log reductions are similar to the FC log reductions and higher than the TC log reductions for the same experiments. It can be indicative of effective removal of *E. coli* using fixed bed column experiments and FC removal is predominantly due to *E. coli* removal. As there was not much characterization carried out in this study, it is difficult to make a conclusive statement. However, it could be due to the high proportions of *E. coli* in the FC.

3.3.2. *E. coli* reduction efficacy of adsorbents

The effluents of AC and GBFS (designs 3 and 4) treatments had zero *E. coli* (Table 6). It is noticeable and significant that despite variations in the influent *E. coli* numbers, all the setups achieved 100% *E. coli* removal. The adsorbents alone displayed excellent *E. coli* removal efficiency without prefiltration. Previous studies indicated that the presence of titanium in

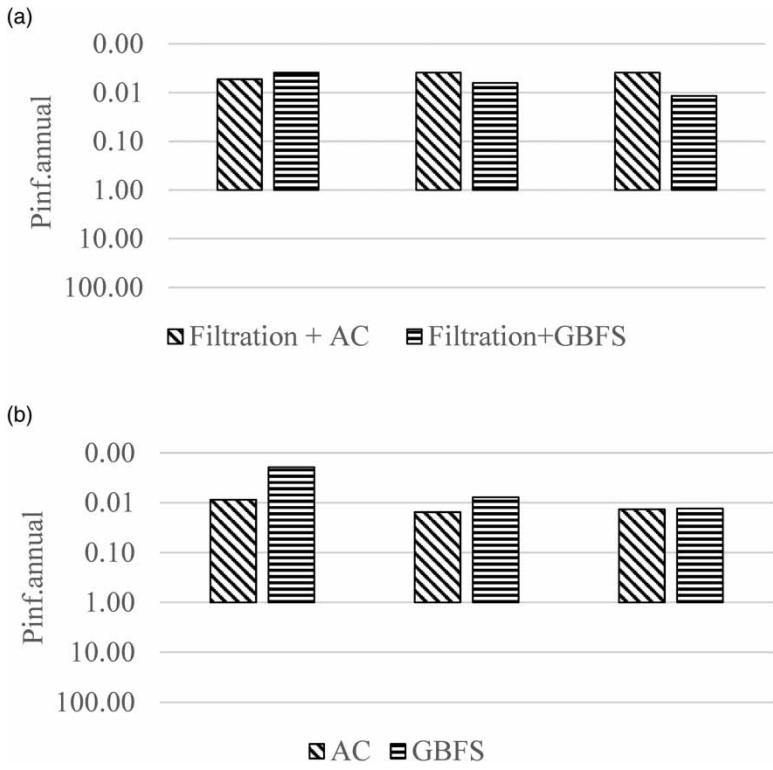

Figure 4 | Probability of annual infection for influents in all the designs at days 1, 4, and 7.

blast furnace slag was capable of inhibiting *E. coli* growth (Westholm *et al.* 2010; Ang *et al.* 2013). Westholm *et al.* (2010) observed 6 log reductions (higher than the present study) of *E. coli* to achieve complete removal. However, the process of *E. coli* removal using GBFS is not well explored. However, the high percentage of heavy metal in GBFS may have enough toxicity against *E. coli*. *E. coli* removal using AC was not that significant in another study (Hijnen *et al.* 2010). However, they could not identify AC providing significant barriers during water treatment. However, the study was focused on drinking water treatment, not identical or similar to the GW treatment in this study. Another study observed 2 log reductions of *E. coli* using unmodified AC (Pal *et al.* 2006). However, the differences in results in this study can be attributed to the type of AC. Depending upon the formation, there can be many different types of AC. So, there can always be some discrepancy in comparing the results.

3.4. Human risk reduction

3.4.1. *E. coli* risk reduction efficacy

The annual infection risk as estimated on the influent of the treatment processes demonstrated a risk level, much higher than 10^{-4} considered acceptable (Figure 4). It indicates that the GW needs to be treated prior to reuse. The annual illness risk was also estimated in the range of 10^{-5}–10^{-7}. It indicates a low annual illness risk. The variability shown in Figure 4 is due to the influent variability of the GW and is independent of the treatment. GW treatment due to all four designs using filtration and adsorption was capable of removing all *E. coli* (Table 6). Since *E. coli* is only an indicator organism, it is still possible that other pathogenic bacteria are present in the water. Even though *E. coli* is the most common pathogenic organism in the GW, a no-risk scenario for *E. coli* may not necessarily be applicable for all the pathogenic microorganisms.

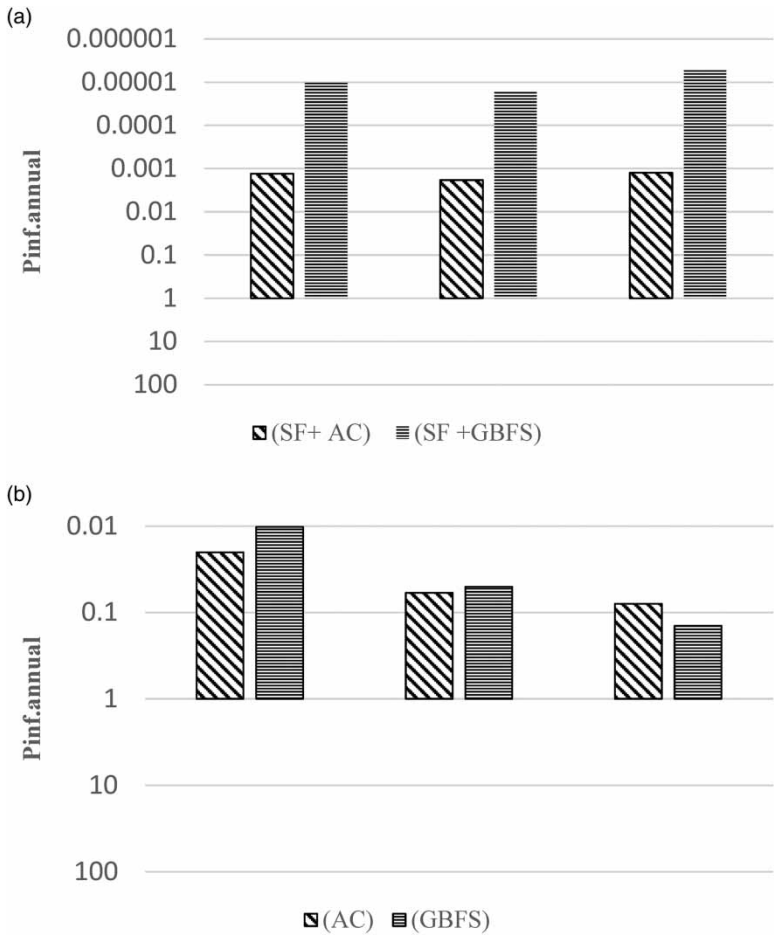

Figure 5 | Probability of annual infection for treated effluents in all the designs at days 1, 4, and 7.

3.4.2. FC risk reduction efficacy

The annual infection risk estimated on the treated effluent of all four designs is presented in Figure 5. The results from the effluent from treatment design 1 (SF + AC) showed a much higher risk than the acceptable limit of 10^{-4}. On the contrary, the use of GBFS in place of AC was capable of reducing the risk to less than the acceptable limit. It is in line with the previous discussion that GBFS is a better adsorbent than AC in removing microbial pollutants. However, as observed in the results from designs 3 and 4, adsorbents (AC, GBFS) alone were not capable of reducing the microbial risk to an acceptable level. The use of the filtration process was capable of reducing the risk significantly compared to the treatment designs without the filtration process. Since the treatment process design included disinfection at the end, the risk can eventually be mitigated. Disinfection in the experimental design was capable of reducing the FC concentration to zero, indicating acceptable risk. Only 8% of the FC was considered as pathogenic and eventually used in the risk assessment. So, even though the FC risk cannot be independently managed by filtration and adsorption, the use of disinfection can address the risk as the FC drops to zero. In addition, as FC is an indicator organism, other potential pathogenic viruses, cysts, and bacteria present and a multi-barrier approach can eventually ensure the resiliency of the risk reduction approaches.

The annual illness risk based on the exponential model as estimated on the treated effluent of all four designs is presented in Figure 6. The results from the effluent from treatment design 1 (SF + AC) and design 2 (SF + GBFS) showed a much lower risk than the acceptable limit of 10^{-4}. This model estimated a much lower risk compared to the beta-Poisson model. However, the use of GBFS was more successful in reducing the risk compared to AC. The results from designs 3 and 4 also showed

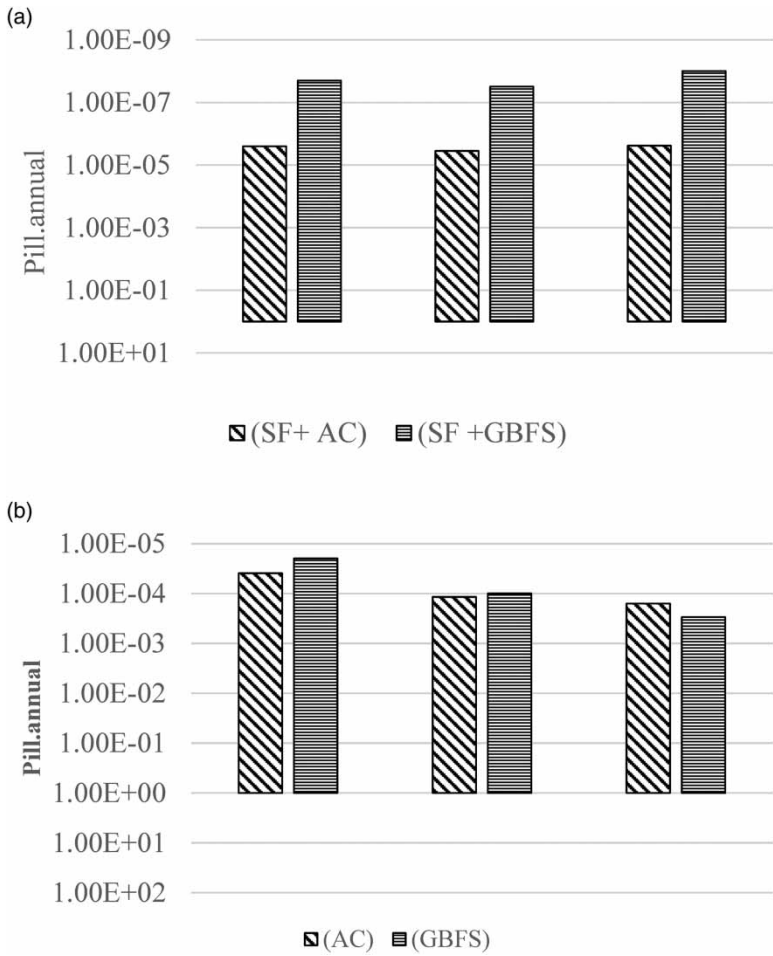

Figure 6 | Probability of annual illness for treated effluents in all the designs at days 1, 4, and 7.

that adsorbents (AC, GBFS) alone were not capable of reducing the microbial risk as low as they are with the filtration system. The use of the filtration process was capable of reducing the risk significantly compared to the treatment designs without the filtration process. Even though the results on the first day showed acceptable risk below 10^{-4}, with time, their efficiencies decreased, and they exceeded the acceptable limit. However, as discussed before, the use of disinfection will be capable of reducing the risk to an acceptable level.

4. CONCLUSIONS

The study examined the effect of filtration and adsorption processes for treating microorganisms from GW using large-scale experiments. The efficacy of SF combined with AC and GBFS in the reduction of TC, FC, and *E. coli* was evaluated. The study used the QMRA approach to evaluate the risk (beta-Poisson and exponential models) reduction due to GW treatment using adsorption and/or filtration. The TC removal achieved by SF + AC and SF + GBFS was in the range of 0.4–1.7 and 3.9–4.8 log CFU/100 mL, respectively. Likewise, SF + GBFS showed higher FC removal efficacy (5.3–5.9 log CFU/100 mL) than SF + AC (4.1–4.3 log CFU/100 mL). The filtration process was also capable of removing TC substantially. Adsorption treatment without a prior filtration process demonstrated high log reductions, however, lower than that with the filtration process. The AC and GBFS were very efficient in the reduction of TC. GBFS displayed higher percentage removal efficiency of 97.8, 99.1, and 71.3% on days 1, 4, and 7, respectively, than AC treatment of 40.0, 40.0, and 66.7% for the same days. AC achieved a reduction of FC by 2.17, 2.18, and 1.94 log CFU/100 mL for days 1, 4, and 7, respectively. GBFS successfully removed 2.30, 2.20, and 1.96 log CFU/100 mL on days 1, 4, and 7, respectively. The filtration process was capable of achieving substantial FC log reductions. GBFS also demonstrated higher FC log reductions than TC log reductions. FC log reductions were higher in the treatment designs than TC log reductions. It is indicated that fixed bed column filtration systems were more effective in removing FC than other coliform organisms. All the treatment designs were capable of completely removing all the *E. coli* bacteria. No *E. coli* strains were detected for any of the configurations after either adsorption alone or with filtration. The study also concludes that GBFS (with or without the inclusion of a sand filter) achieved higher microbial removal than AC. Lastly, the risk reduction achieved by AC and GBFS through exposure to FC and *E. coli* was substantial and often reaches acceptable limits. The $P_{\text{inf.annual}}$ results demonstrated that GBFS alone had produced effluent quality with lower health risk in comparison to AC. However, the addition of filtration was capable of improving risk reduction substantially. As the process design suggests, the use of chlorination/UV disinfection was capable of removing the microbial contamination completely, virtually eliminating the risk of *E. coli* contamination from toilet flushing.

ACKNOWLEDGEMENTS

The authors would like to acknowledge the Enhanced Faculty Research Grant (EFRG2-SRC-CEN-32 and FRG20-M-E44) and Open Access Program (OAPCEN-1410-E00177) in the American University of Sharjah for supporting the study. This paper represents the opinions of the authors and does not mean to represent the position or opinions of the American University of Sharjah.

DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

CONFLICT OF INTEREST

The authors declare there is no conflict.

REFERENCES

Abdolahnejad, A., Jafari, N., Ebrahimi, A., Mohammadi, A. & Farrokhzadeh, H. 2017 Removal of arsenic and coliform bacteria by modified sand filter with slag and zeolite from drinking water. *Health Scope* **6** (3), e15170. https://doi.org/10.5812/jhealthscope.15170.

Al-jayyousi, O. 2004 Greywater reuse: knowledge management for sustainability. *Desalination* **167** (1), 27–37.

Ang, T., He, Y., Xiangxin, X. & Yong, L. 2013 Antibacterial ceramin fabricated by the Ti-bearing blast furnace slag. In: Marquis, F. (ed.) *Proceedings of the 8th Pacific Rim International Congress on Advanced Materials and Processing* (F. Marquis, ed.). Springer, Cham. https://dx.doi.org/10.1007/978-3-319-48764-9_204.

APHA 2012 *Standard Methods For the Examination of Water and Wastewater*, 22nd edn. American Public Health Association (APHA), Washington, DC, USA.

Atanasova, N., Dalmau, M., Coman, J., Poch, M., Rodriguez-Roda, I. & Buttiglieri, G. 2017 Optimized MBR for greywater reuse systems in hotel facilities. *Journal of Environmental Management* **193**, 503–511.

Blanky, M., Martinez, S., Halpern, M. & Friedlier, E. 2015 *Legionella pneumophila*: from potable water to treated greywater; quantification and removal during treatment. *Science of the Total Environment* **533**, 557–565.

Blanky, M., Sharaby, Y., Martinez, S. R., Halpern, M. & Friedler, E. 2017 Greywater reuse – assessment of the health risk induced by *Legionella pneumophila*. *Water Research* **125**, 410–417.

Brüssow, H., Canchaya, C. & Hardt, W. D. 2004 Phages and the evolution of bacterial pathogens: from genomic rearrangements to lysogenic conversion. *Microbiology and Molecular Biology Reviews* **68**, 560–602. doi:10.1128/MMBR.68.3.560-602.

Crini, G., Lichtfouse, E., Wilson, L. D. & Morin-Crini, N. 2019 Conventional and non-conventional adsorbents for wastewater treatment. *Environmental Chemistry Letters* **17**, 195–213.

Dean, K. & Mitchell, J. 2020 Reverse QMRA for *Pseudomonas aeruginosa* in premise plumbing to inform risk management. *Journal of Environmental Management* **146**, 04019120.

Dupont, H. L., Formal, S. B., Hornick, R. B., Snyder, M. J., Libonati, J. P., Sheahan, D. C., Labrec, E. H. & Kalas, J. P. 1971 Pathogenesis of *Escherichia coli* diarrhea. *The New England Journal of Medicine* **285** (1), 1–9.

Eriksson, E., Auffarth, K., Henze, M. & Ledin, A. 2002 Characteristics of grey wastewater. *Urban Water* **4** (1), 85–104.

Federigi, I., Verani, M., Donzelli, G., Cioni, L. & Carducci, A. 2019 The application of quantitative microbial risk assessment to natural recreational waters: a review. *Marine Pollution Bulletin* **144**, 334–350.

Friedler, E., Yardeni, A., Gilboa, Y. & Alfiya, Y. 2011 Disinfection of greywater effluent and regrowth potential of selected bacteria. *Water Science and Technology* **63** (5), 931–940.

Ghaitidak, D. & Yadav, K. 2013 Characteristics and treatment of greywater – a review. *Environmental Science and Pollution Research* **20** (5), 2795–2809.

Gross, A., Kaplan, D. & Baker, K. 2007 Removal of chemical and microbiological contaminant from domestic greywater using a recycled vertical flow bioreactor (RVFB). *Ecological Engineering* **31**, 107–114.

Haas, C. N., Bose, J. B. & Gerba, C. P. 2014 *Quantitative Microbial Risk Assessment*. John Wiley & Sons, Inc. https://doi.org/10.1002/9781118910030.ch8.

Hamilton, K. A., Ahmed, W., Toze, S. & Haas, C. N. 2017 Human health risks for *Legionella* and *Mycobacterium avium* complex (MAC) from potable and non-potable uses of roof-harvested rainwater. *Water Research* **119**, 288–303.

Hijnen, W. A. M., Suylen, G. M. H., Bahlman, J. A., Brouwer-Hanzens, A. & Medema, G. J. 2010 GAC adsorption filters as barriers for viruses, bacteria and protozoan (oo)cysts in water treatment. *Water Research* **44** (4), 1224–1234.

Kusumawardhana, A., Zlatanovic, L., Bosch, A. & Hoek, J. P. V. D. 2021 Microbiological health risk assessment of water conservation strategies: a case study in Amsterdam. *International Journal of Environmental Research and Public Health* **18**, 2595.

Li, Y., Yu, J., Liu, Z. & Ma, T. 2012 Estimation and modeling of direct rapid sand filtration for total fecal coliform removal from secondary clarifier effluents. *Water Science and Technology* **65** (9), 1615–1623.

Masi, F., El Hamouri, B., Abdel Shafi, H., Baban, A., Ghrabi, A. & Regelsberger, M. 2010 Treatment of segregated black/grey domestic wastewater using constructed wetlands in the Mediterranean basin: the zero-m experience. *Water Science and Technology* **61**, 97–105.

Mortula, M. & Gagnon, G. A. 2007 Phosphorus adsorption and oven dried alum residual solids in fixed bed column experiments. *Journal of Environmental Engineering and Science* **6** (6), 623–628.

Mortula, M., Gibbons, M. & Gagnon, G. A. 2007 Phosphorus adsorption by naturally-occuring materials and industrial by-products. *Journal of Environmental Engineering and Science* **6** (2), 157–164.

Neto, H. A. S., Cohim, E. H. B., Sipert, S., Leao, A. S. & Cordeiro, T. S. 2018 Quantitative microbial risk assessment (QMRA) for domestic non-potable reuse of greywater: a case study for a Brazilian household. *International Journal of Development Research* **8** (3), 19527–19533.

Nunez, L., Molinary, C., Paz, M. & Tornello, C. 2014 Health risk assessments in greywater in Buenos Aires state, Argentina. *Revista Internacional de Contaminacion Ambiental* **30** (4), 341–350.

Odonkor, S. T. & Ampofo, J. K. 2013 *Escherichia coli* as an indicator of bacteriological quality of water: an overview. *Microbiological Research* **4**, 5–11. doi:10.4081/mr.2013.e2.

O'Toole, J., Sinclair, M., Malawaraarachchi, M., Hamilton, A., Barker, S. F. & Leder, K. 2012 Microbial quality assessment of household greywater. *Water Research* **45** (13), 4301–4313.

Pal, S., Joardar, J. & Song, J. M. 2006 Removal of *E.coli* from water using surface-modified activated carbon filter media and its performance over an extended use. *Environmental Science and Technology* **40** (19), 6091–6097.

Pidou, M., Avery, L., Stephenson, T., Jeffrey, P., Parsons, S. A., Liu, S., Memon, F. A. & Jefferson, B. 2008 Chemical solutions for greywater recycling. *Chemosphere* **71** (1), 147–155.

Radhi, A. A. & Borghei, M. 2017 Effect of aeration then granular activated carbon on removal efficiency of TOC, COD and coliform, fecal coliform for 'Sorkheh Hesar Canal' water. *International Journal of Computation and Applied Science* **3** (2), 201–206.

Rose, J. B., Sun, G. & Gerba, C. 1991 Microbial quality and persistence of enteric pathogens in graywater from various household sources. *Water Research* **25** (1), 37–42.

Sharath, D., Tekle, A. B. & Sushma, K. 2017 Design of sand filter unit for surface water treatment in Gubre City, Snnpr, and Ethiopia. *Journal Industrial Pollution Control* **33** (2), 1120–1127.

Shi, K., Wang, C. & Jiang, S. C. 2018 Quantitative microbial risk assessment of greywater on-site reuse. *Science of the Total Environment* **635**, 1507–1519.

Sinno, S., Tatan, B., Singer, M. N., Elkersh, K. & Fattah, K. 2022 Recycling of carwash greywater through electrocoagulation treatment. In: *Proceedings of the International Conference on Sustainable Environment and Urban Infrastructure*, February 21–24, 2022, Dubai, UAE.

Truchado, P., Lopez-Galvez, F., Gil, M. I., Pedrero-Salcedo, F., Alarcón, J. J. & Allende, A. 2016 Suitability of different *Escherichia coli* enumeration techniques to assess the microbial quality of different irrigation water sources. *Food Microbiology* **58**, 29–35. doi:10.1016/j.fm.2016.03.006.

Westholm, L. J., Drizo, A. & Renman, G. 2010 The use of blast furnace and electric arc furnace steel slag in water pollution control. In: *Proceedings of the 6th European Slag Conference of EUROSLAG*.

Williams, G. J., Sheikh, B., Holden, R. B., Kouretas, T. J. & Nelson, K. L. 2007 The impact of increased loading rate on granular media, rapid depth filtration of wastewater. *Water Research* **41** (19), 4535–4545.

Zannerni, G. M., Fattah, K. P. & Al-Tamimi, A. K. 2020 Ambient-cured geopolymer concrete with single alkali activator. *Sustainable Materials and Technologies* **23**, 1–9.

First received 31 March 2023; accepted in revised form 9 July 2023. Available online 19 July 2023