Comparing the performance of biosand filters operated with multiday residence periods

Candice Young-Rojanschi and Chandra Madramootoo

ABSTRACT

Biosand filter (BSF) users are instructed to operate their filters with residence times of 1–48 hours. However, studies to date have only tested residence periods of 6–36 hours, with no studies extending to 48 hours, or investigating whether BSF performance decreases beyond 48 hours. The goal of this study was to compare *Escherichia coli* removal in filters operated with one-, two-, and three-day residence periods. Nine laboratory-scale column filters were operated in parallel, with three replicates for each of the three residence periods, over 84 days. Filters were fed with lake water supplemented with *E. coli*. Influent, effluent, and control samples were tested for a range of parameters including *E. coli* colony forming units; turbidity; pH; electrical conductivity; dissolved oxygen (DO); and nitrogen as ammonia, nitrate, and nitrite. Hydraulic conductivity and DO of pore water were measured for different sand depths within the filters every six days. The study found no significant difference in *E. coli* removal by extending the residence period to three days. However, water treated in filters with increased residence periods had lower DO concentrations and increased nitrite levels.

Key words | biosand filter (BSF), household drinking water treatment, point-of-use (POU), slow sand filtration (SSF)

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INTRODUCTION

Biosand filters (BSFs) are considered to be one of the most promising household water treatment technologies presently available (Sobsey *et al*. 2008). Laboratory trials have demonstrated the effectiveness of BSFs at removing >5 $\log_{10}$ Giardia cysts, 3.7 $\log_{10}$ Cryptosporidium oocysts (Palmateer *et al*. 1999), 1.9 $\log_{10}$ *Escherichia coli* colony forming units (cfu), and an average of 0.5 $\log_{10}$ bacteriophage plaque forming units (pfu) (Elliott *et al*. 2008). Randomized controlled field trials in Cambodia (Stauber *et al*. 2012), Kenya (Tiwari *et al*. 2009), and the Dominican Republic (Stauber *et al*. 2009) have shown significant reductions in diarrhoeal disease associated with BSF use.

BSFs are biologically active granular media filters that are operated intermittently, and thus do not require continuous pumping. The filter is designed to operate such that a batch, or dose, of influent water is added all at once, creating a pressure head that drives flow. The new dose enters the interstitial spaces of the media, displacing water from the previous dose. As filtration progresses, the pressure head decreases. The time between doses is the residence period. The filter outlet is constructed such that the media remains saturated during the residence period. During the first weeks of operation the filter ripens as the schmutzdecke, or filter cake, develops and the filter's capacity to remove microorganisms improves.

BSFs in field trials show significantly improved water quality, but *E. coli* reductions are smaller and more variable than in laboratory trials (Vanderzwaag *et al*. 2009; Fiore *et al*. 2010). Operational factors, including the frequency of dosing, were suggested as a cause of high variability in *E. coli* removal (ranging from 0% to 99.7%) found during a field test of BSFs in the Dominican Republic (Stauber *et al*. 2006).

It is recommended to operate BSFs by dosing between one and four times per day, with a minimum residence

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period of one hour and a maximum of 48 hours (Centre for Affordable Water and Sanitation Technology (CAWST) 2012). However, a study on BSF use in Guyana found that households operated their filters with a mean residence period of three days (Young-Rojanschi 2013).

Two laboratory studies in the peer-reviewed literature investigated the impact of residence periods on BSF removal of bacterial indicators. Baumgartner *et al.* (2007) compared a BSF operated with 12- and 36-hour residence periods, and found significantly better total coliform removal with 12-hour operation. However, their study was limited in that it used a single filter which had been ripened for 30 days with 12-hour dosing. It was then operated under the different experimental conditions and the filter effluent sampled for total coliforms.

Jenkins *et al.* (2011) compared 5- and 16-hour residence periods, with improved fecal coliform removal occurring with the longer residence period. However, their 'long' residence period was still shorter than one day.

No studies in the literature extend residence periods to two days or beyond, as were seen in the Guyana study. It has been predicted that residence periods longer than 48 hours will lead to nutrient depletion and starvation of the biolayer (CAWST 2012).

The goal of this study was to compare BSFs operated with residence periods of one, two, and three days for *E. coli* removal, dissolved oxygen (DO) profiles, and evidence of nitrification.

MATERIALS AND METHODS

Experimental design

A randomized complete block design with three treatments was used for this experiment. The treatments were one-, two-, or three-day residence times. The three blocks were based on location within the laboratory to account for any variation in environmental conditions. Each of the three blocks included each of the three treatments. The experiment ran for 84 days from November 2011 to February 2012. Laboratory-scale column filters were constructed rather than full-scale BSFs to enable running the nine units in parallel (Elliott *et al.* 2011).

Filter design

The filters for this experiment were designed to model a CAWST V10 filter (CAWST 2012) with 55 cm media depth and a 5 cm standing head (Figure 1). The diffuser plate was installed 2 cm above the standing head. They were constructed of 10 cm diameter transparent acrylic tubing which was covered in black plastic between doses and measurements in order to prevent algal growth (Young-Rojanschi & Madramootoo 2013). As with the V10 filter, the dosing volume was set to be equivalent to the pore volume of the sand, in this case 1.8 L. The total volume of water remaining

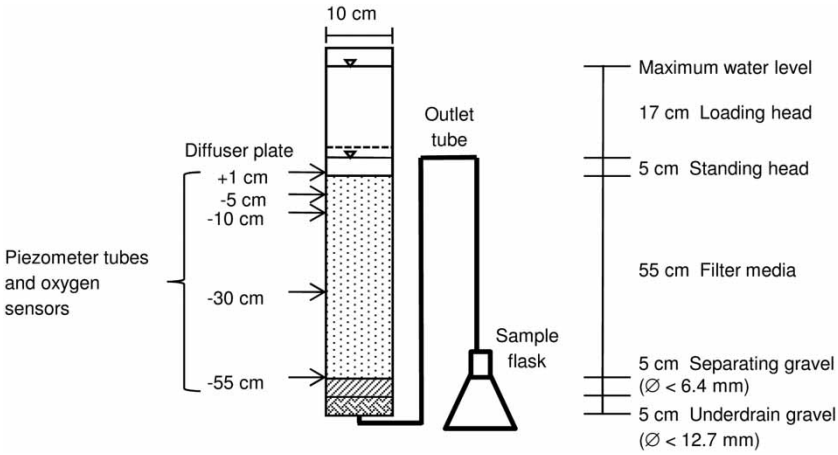

Figure 1 | Schematic of experiment.

in the filter during the residence period including the standing head, sand pore volume, underdrain pore volume, and outlet tube, was approximately 2.5 L. Filters had DO sensors and piezometer tubes installed at locations corresponding to 1 cm above the sand surface, and 5, 10, 30, and 55 cm below.

The filter media was locally purchased sand. The media had a porosity of 0.42, maximum diameter of 0.7 mm, effective diameter (d_{10}) of 0.17 mm, and a uniformity coefficient of 2.06. The two underdrain layers were composed of locally purchased crushed gravel that was washed and sieved so that the fine top layer had a diameter range of 0.7–6.4 mm and the coarse bottom layer had a diameter range of 6.4–12.7 mm.

To prevent air binding, the filters were partially filled with water before adding the media to the desired depths. Distilled water was boiled to decrease the saturation point of air in water and so allow much of the air to be released, as air bubbles would be a problem for the piezometer tubes in these filters. The water was then cooled to room temperature and slowly introduced through the outlet tube.

Influent water

Water was initially collected daily from Lac St Louis, Quebec. It was stored overnight to bring it to room temperature. After the lake ice thickness increased beyond 20 cm, water was collected weekly and stored frozen at an outdoor location until two days before dosing, when it was brought indoors to thaw and reach room temperature. Influent temperature was $19.9 \pm 0.9\ ^{\circ}C$.

Influent was supplemented with *E. coli* strain B (ATCC#11303) each day before dosing. Each week, new cultures of *E. coli* were grown in tryptic soy broth to log phase (5 hours at 37 °C). Growth curve experiments performed before the onset of the experiment established that the *E. coli* concentration at this point would be approximately 3×10^8 colony forming units (cfu) per millilitre. The cultures were serially diluted with phosphate buffered saline, divided into daily aliquots, and refrigerated at 4 °C. The average *E. coli* concentration in the filter influent, over 46 sample days spread throughout the experiment, was $\log_{10}$ 2.1 cfu/mL with a standard deviation of $\log_{10}$ 1.2 cfu/mL (Table 1).

Table 1 | Dosing schedule for treatment groups, where day 0 corresponds to day six of the previous cycle

Day of cycle	One-day group	Two-day group	Three-day group
0	X	X	X
1	X	–	–
2	X	X	–
3	X	–	X
4	X	X	–
5	X	–	–
6	X	X	X

X indicates a dosing event, – indicates dosing did not occur.

Dosing

Each dose was poured into the filter up to a maximum 17 cm loading head. It was not possible to add the full 1.8 L dose at once. In a full-sized biosand filter, the upper reservoir is wider than the sand surface, allowing for a larger dosing volume without increasing the loading head. In the case of the columns used in this experiment, however, the diameter of the column was the same throughout, and so adding the full dose to the upper reservoir would have resulted in a higher loading head (and thus higher flow rates) than exist in full-sized BSFs. The remaining portion of the dose was added after the loading head had decreased by approximately 3 cm.

Measurements and sampling

Filter dosing followed a six-day cycle in order to accommodate the three residence periods (Table 1). All treatment groups were operated consistently throughout the experiment (i.e., the one-day filters were always dosed every day, and the three-day filters every three days). These results cannot be extrapolated to the case where a filter that was matured under one set of conditions is then operated under a different set of conditions.

Effluent samples were taken from all treatment groups on each cycle day six. Sterile effluent flasks were placed at the outlet tube of each column, and left in place for up to two hours after dosing to collect the effluent. At this point filtration was 80–100% complete (1.4–1.8 L filtered). Samples were mixed before analysis.

The influent samples for the treatment groups were taken from the dose prior to that of the effluent sample. For the one-day group, this corresponded to cycle day five. For the two-day group this was cycle day four, and for the three-day group this was cycle day three (Table 1). Five supplementary test days were added in which all filters received the same influent water, and were then sampled at their following dose. There was no difference in results between the two types of sampling (different influent but same test day versus same influent but different test day) and so the results were pooled. Control samples were collected periodically through the experiment. Influent water was set aside in a loosely-capped, sterile, glass media bottle, wrapped in black plastic, and kept beside the filters on the laboratory bench for one, two, or three days. The samples were analysed with their corresponding filters.

Instantaneous flow rate, q, was analysed by taking the derivative of the fitted curve of time versus water level for the top piezometer tube, which was located above the media surface. Hydraulic conductivity, K, was analysed by taking the slope of the linear fitted curve of the head difference between the two piezometers of interest divided by the distance between them, versus q, following Darcy's law.

E. coli colony forming units were enumerated with m-coliblue24® broth (Hach Company, Loveland, CO) using the membrane filtration method (American Public Health Association (APHA) *et al.* 1998). Each sample was analysed at least twice, but usually three times, using different sample volumes or dilutions as appropriate. In effluent samples this resulted in a 100 mL sample, as well as a 1:10 and 1:100 dilution. When a plate yielded too many colonies, or not enough to count, it was discarded and the count from the other plate for the same sample was used. If no plate resulted in countable colonies, the data was recorded as missing.

Turbidity was measured using a Lamotte 2020e turbidimeter (Lamotte Company, Chesterton, MD). Temperature, electrical conductivity (EC), pH, and DO were measured using a YSI 556 multimeter (YSI Inc., Yellow Springs, OH). In-situ DO measurements were taken with RedEye® oxygen patches read with a Neofox fluorescence probe (Ocean Optics, Dunedin, FL) which had been calibrated using the YSI 556 multimeter. This is a non-intrusive method: readings were taken through the transparent acrylic filter wall. Nutrient samples were taken every six days from influent and filter effluent samples until day 60. Hach methods TNT 830, TNT 835, and 8507 were used with a DR-2800 Spectrophotometer to measure nitrogen as ammonia, nitrate, and nitrite respectively (Hach Company, Loveland, CO). Total nitrogen samples were acidified with hydrochloric acid and stored at 4 °C until the end of the experiment and measured using the perchlorate method (APHA *et al.* 1998).

Tracer tests

Two sets of tracer tests were performed on the filters. The first was to test columns for short-circuiting. The second was to assess the operational conditions of the experiment. In the first test, the standing head was decanted from the media surface before doses were applied (Elliott *et al.* 2008). The second test was under experimental conditions, with a 5 cm standing head and a 1.8 L dose volume. In each case, after at least 1.7 L of water had filtered (corresponding to a loading head below one cm), the next dose was applied.

Filters were rinsed several times with distilled water before testing. The tracer was a solution of 200 mg/L NaCl in distilled water. Grab samples were taken from filter effluent every 100 mL and measured for EC as a proxy for NaCl concentration (Elliott *et al.* 2008). The dose of tracer was followed by three doses of distilled water.

Statistical analysis

Statistical analysis was performed using generalized linear mixed modelling (GLMM) in the statistical software SAS 9.3 (SAS Institute Inc., Cary, NC). GLMM was selected in order to control for possible environmental differences within the laboratory (blocking effect), and the lack of independence between measurements from the same filter (repeated measures). The covariate of interest was the assigned treatment group.

RESULTS AND DISCUSSION

Tracer tests

The first tracer tests for short-circuiting indicated that, like with full-sized BSFs, the laboratory filters functioned as

plug flow reactors (Elliott *et al.* 2008), with a Morrill dispersion index of 1.8.

Under experimental conditions, the test required four doses to complete (Figure 2). In the first dose, the tracer entered the standing head and upper media pores, with no tracer detected in the effluent. Forty-nine percent of the cumulative tracer volume had exited the filter by the end of the second dose, with 96% by the end of the third. The final 4% exited the filter in the fourth dose, as is illustrated by the dashed 'cumulative tracer fraction' line. After the 84 days of the experiment, the tracer tests showed a slight shift in the curve. The results were 0% of tracer exiting in the first dose, 54% in the second, 97% by the third, with the remainder exiting in the fourth. The difference in the cumulative tracer fractions after the second dose (54% versus 49%) was statistically significant ($p < 0.000$). This suggests that there had been some settling of media and removal of fine material during the experiment. These fractions were common to all treatment groups, with a standard deviation of less than 1%, indicating that the nine filters were very similar in construction, media placement, and pore volume.

Because the short-circuiting tests indicated plug flow within the media, it was assumed that most mixing occurred within the standing head. The first 0.4 L of water exiting the filter during the second dose did not contain tracer. This indicates that each new dose does not completely displace the pore water in the filter. The 0.4 L can be accounted for by 0.02 L of water in the outlet tube, 0.28 L in the gravel underdrain, and 0.1 L in the bottom of the sand profile, corresponding to the media depth of 52–55 cm from the surface. This included the bottom piezometer tube and DO sensor.

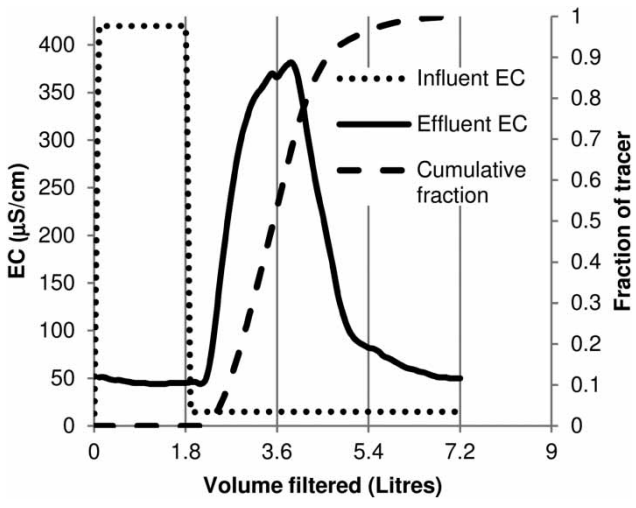

Figure 2 | Tracer test results from a representative filter (Filter 9).

DO profile

Figure 3 presents the results from 12 DO profile experiments on each of the three filters per treatment group. There was no DO sensor at the water surface (5 cm above sand surface), so in Figure 3 it was assumed that, at one hour after dosing, the DO level would be the same as the influent DO (average of 10.6 mg/L). The DO levels in the influent water, especially late in the experiment, were supersaturated in this study. Outdoor temperatures ranged from a daytime high of 14 °C in November to an overnight low of −24 °C in January. Saturation levels for DO are higher at lower temperatures and the DO did not fully drop to saturation before dosing, even after sitting at room temperature for two days. It was assumed that by one day after dosing the DO level at the water surface would have reduced to saturation levels, which is illustrated by the +5 cm data point in Figure 3 for days one, two, and three.

At one hour after dosing, the DO at 1 cm above the sand surface was measured to be similar to that of saturation, which was calculated to be 8.84 mg/L for the laboratory conditions (Figure 3). It was assumed that the drop in DO between the 1 cm and −30 cm sensors primarily occurred due to the filtration step. This is because at one hour after dosing filtration was generally not yet complete. On average, this came to a drop of 1 mg/L of DO. This decline in DO, however, was not linear. Even at only one hour after dosing, the −5 and −10 cm sensors often had DO levels below that of the −30 cm sensor. This was likely related to declining flow rates. Water at the deeper sensor had passed through the schmutzdecke of the filter quickly, then slowly through the deeper media. On the other hand, water which was at the higher sensors at one hour after dosing had begun the filtering process later in the cycle, remaining in the filter reservoir before moving slowly through only the schmutzdecke. The −55 cm sensor was not considered here because, as the tracer test indicated

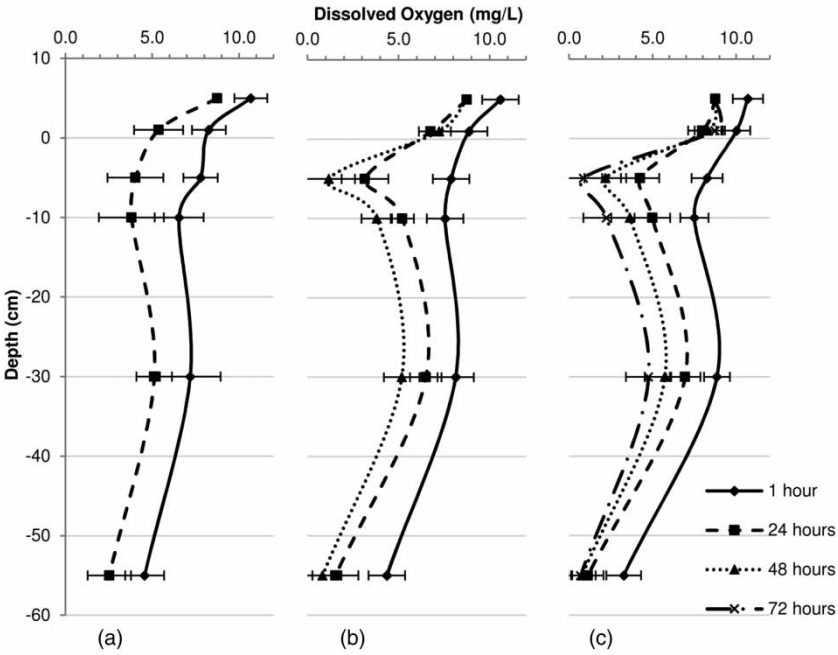

Figure 3 | DO profiles for (a) one-day, (b) two-day, (c) three-day filters.

(Figure 2), it corresponded to water that was not introduced in the same dosing cycle.

For all treatment groups, the DO profiles (Figure 3) confirmed that oxygen was consumed throughout the entire media profile over time. This is illustrated by the complete shifting of the curve from one hour after dosing to one day after dosing. Note that DO measurements within the filter were also lower than the DO measured in the control samples. The larger decrease in DO at the −5 cm suggests that the highest rate of oxygen consumption did occur at the top of the filter. A physical limitation to the filters in this experiment was that there was no sensor immediately below the sand surface at the schmutzdecke. It is expected that the DO consumption peak would have occurred between the sand surface and the −5 cm sensor. The schmutzdecke is the zone within a slow sand filter in which the highest level of biological activity occurs (Unger & Collins 2008).

The rate of oxygen consumption at different depths differed by treatment group. At one hour after dosing for all depths other than 55 cm, the one-day treatment group had the lowest DO, showing that more oxygen had been consumed during the filtration and/or the early residence period in those filters. The more frequent dosing in the one-day group would have ensured an abundance of oxygen and nutrients, which are essential to the growth of aerobic microorganisms. The longer residence periods, on the other hand, may have resulted in low oxygen zones within the filters, which would limit the growth of aerobic microorganisms. Although the mean DO remained above 0 mg/L for all treatment groups at all depths, for both the two- and three-day filters the −5 and −55 cm sensors were frequently at 0 mg/L, as can be seen by the error bars in Figure 3.

The influent water in this experiment had higher DO than is typical in tropical environments. At 30 °C, for example, the saturation DO level for water is only 7.54 mg/L. In conditions with lower influent DO, some regions of the filter may be anaerobic earlier. In a previous study the schmutzdecke was anaerobic after only 24 hours (Young-Rojanschi & Madramootoo 2013).

Water quality

Paired *t*-tests were used to compare control samples to their corresponding influent sample. Water in the control samples had decreasing DO, and increasing Nitrogen as NO_2^- and NH_4^+, with time. *E. coli* concentrations also decreased with time after the first day, with marginal reductions on

Table 2 | Influent and control sample water quality parameters

Parameter	Influent	One-day control	*p*	Two-day control	*p*	Three-day control	*p*
E.coli, $\log_{10}$ cfu/mL	2.1 ± 1.2 (46)	2.1 ± 1.1 (8)	0.851	1.4 ± 1.8 (8)	0.051	0.0 ± 1.3 (7)	0.034
Turbidity, NTU	3.4 ± 1.5 (65)	4.0 ± 2.0 (7)	0.666	3.3 ± 2.0 (9)	0.050	2.9 ± 1.9 (8)	0.895
DO, mg/L	10.6 ± 0.9 (63)	8.5 ± 0.3 (6)	0.001	8.06 ± 0.3 (9)	0.000	7.0 ± 0.7 (7)	0.000
EC, µS/cm	117 ± 23 (65)	135 ± 5 (6)	0.388	122 ± 18 (9)	0.183	121 ± 17 (8)	0.216
pH	7.4 ± 0.2 (63)	7.5 ± 0.1 (7)	0.784	7.6 ± 0.2 (9)	0.155	7.5 ± 0.2 (7)	0.427
N_{org}, µg/L	342 ± 124 (18)	–	–	–	–	–	–
NO_3^-N, µg/L	459 ± 102 (25)	476 ± 113 (6)	0.656	463 ± 105 (7)	0.684	413 ± 79 (7)	0.283
NO_2^-N, µg/L	6 ± 3 (28)	9 ± 3 (7)	0.038	10 ± 4 (8)	0.016	12 ± 5 (8)	0.002
NO_4^-N, µg/L	93 ± 21 (28)	151 ± 14 (7)	0.000	176 ± 34 (8)	0.000	181 ± 67 (8)	0.000

p values test for a significant difference between the control sample and influent. Mean ± standard deviation (number of samples).

Table 3 | Filter effluent quality of one-, two-, and three-day residence periods. *p* values test for significance that one of the treatment groups is different from the other two

Parameter	One-day residence period	Two-day residence period	Three-day residence period	*p* value
E.coli (Log_{10} reduction)	1.8 ± 0.8 (38)	1.9 ± 0.8 (41)	1.8 ± 0.9 (37)	0.235
Turbidity, NTU	2.7 ± 0.7 (48)	2.2 ± 1.0 (47)	1.3 ± 0.4 (47)	< 0.001
DO, mg/L	7.2 ± 0.5 (48)	7.1 ± 0.3 (47)	6.7 ± 0.4 (47)	< 0.001
EC, µS/cm	155 ± 21 (48)	157 ± 19 (47)	163 ± 16 (47)	< 0.001
pH	8.0 ± 0.1 (48)	8.0 ± 0.1 (47)	8.0 ± 0.1 (47)	0.600
N_{org}, µg/L	149 ± 69 (18)	154 ± 94 (18)	161 ± 88 (17)	0.109
NO_3^-N, µg/L	630 ± 69 (18)	611 ± 58 (18)	543 ± 56 (17)	< 0.001
NO_2^-N, µg/L	3 ± 2 (18)	6 ± 5 (18)	14 ± 10 (17)	< 0.001
NH_4^+N, µg/L	11 ± 5 (18)	10 ± 5 (18)	10 ± 5 (17)	0.559

Mean ± standard deviation (number of samples).

the second day ($p = 0.051$), and significant reductions by the third ($p = 0.034$) as compared to the influent (Table 2).

There was no significant difference in *E. coli* removal between treatment groups (Table 3) suggesting that operating filters with residence periods of up to three days is not detrimental to filter performance. This may have been due, in part, to natural die-off. The three-day control samples had significantly less *E. coli* than the influent water (Table 2) even without the benefit of filtration. Another possibility is that the three-day residence period was short enough that the microbial communities in the filter had not yet depleted the nutrients available in the feed water, as had previously been assumed to happen (CAWST 2012). As filter ripening was noted in all filters, and the DO profiles indicated the likelihood of active aerobic microbial communities throughout all filters, this seems probable.

There were, however, significant differences in effluent quality in terms of turbidity, DO, EC, NO_3^--N, and NO_2^--N. The difference in effluent DO between treatments corresponds to the differences in the DO profile illustrated in Figure 3. However, effluent DO levels were higher than DO levels noted within the filter column. This was likely due to aeration occurring at the filter outlet and within the collection flask, which was open to the atmosphere. All treatment groups exhibited increased EC and pH as compared to the influent. When noted in previous studies, this phenomena has been attributed to calcium carbonate leaching from the concrete filter box (Murphy *et al*. 2010a). As the filters in this study were constructed of acrylic rather than

concrete, the effect is more likely due to leaching from the filter media rather than the filter walls. Water with a longer contact period with the media (i.e., longer residence period) had a greater EC increase.

After ripening, nitrification appears to have occurred in all filters, with decreased NH_4^+ concentrations in effluent as compared to the influent and no significant difference between treatment groups. Denitrification also appears to have occurred, with decreasing total nitrogen and increasing NO_3^- in all filters. This is consistent with the findings of Murphy *et al.* (2010b). Murphy *et al.* (2010b) hypothesize that nitrification occurs within the schmutzdecke during filtration, while denitrification occurs within the media depth during the residence period. This was based on the assumption that the majority of DO in influent water is consumed by the schmutzdecke during filtration, leaving the media column nearly anaerobic for the entire residence period. The present experiment questions that assumption for most situations. As was noted earlier, the DO levels of water within the deeper sand column had only decreased by 1 mg/L as compared to the standing head, leaving the entire filter aerobic. The pore water in the bottom of the sand column corresponding to the −55 cm sensor is excluded from this observation as it belonged to a different dose. The decreasing DO levels at all media depths during the residence period suggest the activity of aerobic microorganisms in the media. This is consistent with Elliott *et al.* (2011) attributing bacteriophage reduction at depth in BSFs to aerobic microbial processes. The regions most likely to develop anaerobic zones, and which thus may have been the regions in which denitrification occurred, were the top few centimetres of sand below the schmutzdecke, and the bottom of the media where some portion of water was remaining from previous filtration cycles. The remainder of the filter, which retained aerobic conditions, is likely where nitrification occurred.

Denitrification appeared to play an increased role in the treatment groups with longer residence periods, with significantly lower NO_3^--N remaining in three-day filters, and significantly higher NO_3^--N in one-day filters (Table 3). The difference in nitrite levels was also significant, with three-day filters at 14 μg/L compared to one-day filters at 3 μg/L. Denitrifying microbes prefer oxygen-poor environments, which were more likely to occur in the two- and three-day filters (Table 3). In this experiment, nitrate and nitrite concentrations for all filters remained well below the World Health Organization (WHO) guideline values of 11 mg/L NO_3^--N and 0.9 mg/L NO_2^--N (WHO 2011). However, influent total nitrogen in this experiment was very low (<1 mg/L). In cases where influent nitrogen levels are high, as with Murphy *et al.* (2010b), it would be important to consider the noted effect of higher nitrite levels in filter effluent when residence periods are extended.

Hydraulic loading rate

The maximum hydraulic loading rate in the filters, q_{max}, at the onset of the experiment was 0.72 ± 0.06 m/h. This is much higher than the 0.2 m/h maximum flow rate recommended for conventional slow sand filters (Crittenden *et al.* 2005), but lower than the 1.1 m/h noted elsewhere for BSFs (Elliott *et al.* 2006). CAWST (2012) recommends a maximum flow rate of 0.4 m/h for BSFs, which is much lower than what was seen in this experiment. The filter media used in this experiment followed CAWST recommendations, as did media depth and maximum loading head, so the difference in q_{max} values was likely due primarily to differences in methodology. CAWST (2012) developed its methodology to be easily performed by project implementers and community health workers in households in low-resource settings. They measure the volume of filter effluent in the first 60 seconds after filter dosing. As the surface area of the filter is known, it is possible to transform this volume into a velocity. This method is effective for meeting its specific purpose, which is to provide a form of quality control on filter media and installation. However, it is not the most accurate method of determining the instantaneous q_{max} for the purpose of calculating hydraulic conductivity or considering possible scouring. In practice, the filter requires some moments after dosing to reach q_{max}, and by one minute after dosing the q has already significantly declined from its instantaneous maximum value, which would lead to volumetric methods underestimating q_{max}. In the case of the present experiment it was possible to directly time the descent of the loading head by observation in the top piezometer tube, plot the results, and fit a curve to the results knowing that

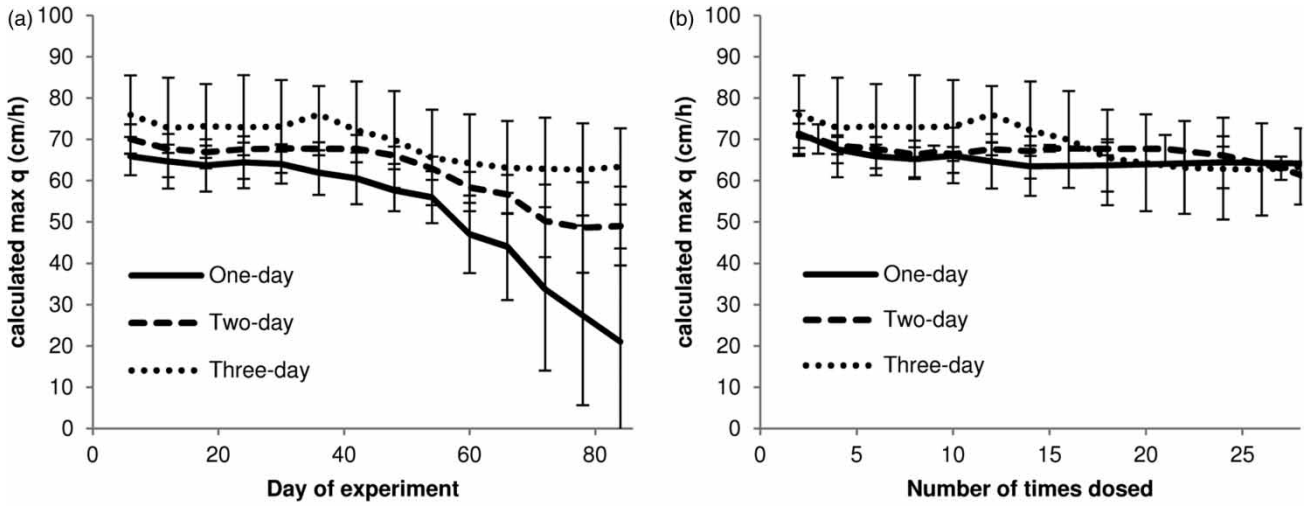

Figure 4 | Hydraulic conductivity by (a) day of experiment and (b) number of times dosed.

flow through the column would follow Darcy's Law. The derivative of this equation allowed for a calculation of the instantaneous q when the loading head was 17 cm. Flow rate is a significant parameter for BSF performance, with higher flow rates resulting in reduced performance (Jenkins *et al.* 2011). It is not possible to directly compare the q_{max} from this experiment with the CAWST recommendations because the measurement methodologies were so different, but if the flow rates in this experiment were higher, then the results of this study may underestimate the performance of a full-sized V10 filter.

The q_{max} declined through the experiment for all treatment groups as the filters began to clog (Figure 4). The decline was largest for filters in the one-day group when the filters were compared by day of the experiment (Figure 4(a)), but was similar for all treatment groups when compared instead by the number of times the filter had been dosed (Figure 4(b)).

The hydraulic conductivity (K) of the media was highest towards the bottom of the filters (8.8×10^{-5} m/s) and lower towards the top (3.6×10^{-5} m/s), even before filter ripening. This was likely related to the installation method, and would also occur in full-scale field BSFs. During installation, it was observed through the transparent acrylic column walls that fine particles would remain in suspension while coarser particles would quickly settle. This resulted in some layering of the media which could be observed, and may have led to a gradient of particle size through the filter depth.

Influence of the laboratory environment

A temperature gradient existed in the laboratory, with the average effluent temperatures from Block 1 filters being 0.4 °C warmer than those of Block 3 filters. Location within the laboratory had a small but statistically significant effect on *E. coli* removal, turbidity, pH, and NO_3^--N. Block 1 had higher *E. coli* removal ($p = 0.013$) as compared to block 2, but also higher effluent turbidity ($p = 0.024$). Block 1 filter effluent had significantly lower pH

Table 4 | Effluent water quality by block

Parameter	Block 1	Block 2	Block 3
E.coli, $\log_{10}$ removal	2.1 ± 0.9	1.7 ± 0.8	1.9 ± 0.9
Turbidity, NTU	2.1 ± 1.0	1.7 ± 1.0	1.9 ± 1.3
DO, mg/L	6.9 ± 0.5	7.0 ± 0.6	7.2 ± 0.6
EC, µS/cm	152 ± 19	154 ± 19	147 ± 23
pH	7.9 ± 0.1	8.0 ± 0.1	8.0 ± 0.2
NO_3^-N, µg/L	608 ± 81	607 ± 59	572 ± 69
NO_2^-N, µg/L	6 ± 7	7 ± 5	10 ± 10
NH_4^+N, µg/L	10 ± 5	10 ± 4	11 ± 5

Mean ± standard deviation.

than both blocks 2 and 3 ($p < 0.001$ for each). Block 3 had significantly lower NO_3^--N than Blocks 1 ($p = 0.012$) and 2 ($p = 0.022$).

The significant impact of laboratory location was unexpected, as all filters were placed in a row on the same laboratory bench, for which conditions would vary only slightly. However, the sensitivity of other biologically active filters, such as conventional slow sand filters, to temperature has been well documented (Jabur 2006; Petry-Hansen *et al.* 2006; Unger & Collins 2008). Block 1 was nearest to the laboratory refrigerator, which may have acted as a heat source, and furthest from the door. Although the difference in performance reached statistical significance, the magnitude of the effect was small (Table 4), as might be expected with a small but consistent temperature difference.

CONCLUSIONS

- *E. coli* removal was not significantly impacted by increasing the length of the residence period in the BSF from one day to two or three days.
- Longer residence periods lead to reduced DO in filter effluent and throughout the filter profile.
- Longer residence periods lead to increased nitrite concentrations in filter effluent.
- When dosing volumes are equivalent to sand pore volume (approximately 70% of total water volume in the filter), approximately half of the filter effluent corresponds to influent from the previous dose while the remaining originates from earlier doses. This may lead to anaerobic conditions developing at the bottom of the filter media profile.
- Further studies are necessary to understand the role of temperature on filter performance.

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