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Investigation on the effects of design and operational variables on the efficacy of biosand filters

by

Julie A. Napotnik

Presented to the Graduate and Research Committee

of Lehigh University

in Candidacy for the Degree of

Doctor of Philosophy

in

Environmental Engineering

Lehigh University

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Approved and recommended for acceptance as a dissertation in partial fulfillment of the requirements for the degree of Doctor of Philosophy.

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Investigation on the effects of design and operational variables on the efficacy of biosand filters

Table of Contents

List of Figures

List of Tables

Abstract

The following paper reports on the efforts made to assist in the overall implementation of one specific household water treatment (HWT) for improving water quality for people in developing countries, biosand filters (BSFs). It is recognized that BSFs are not applicable for every situation or community. When BSFs were first developed for household applications, the minimum sand bed depth was determined to be 50 cm, based on existing Canadian regulations for water treatment through large-scale, high-capacity sand filters. We questioned this basic assumption, and investigated whether smaller, lighter, and cheaper BSFs (with a shorter sand bed depth) are as effective as the traditional large, concrete filter. The overall project objective was to assess the efficacy, effectiveness, and acceptability of a smaller biosand filter, both in the laboratory and in the field, with the overall goal of demonstrating successful performance and acceptability of the smaller BSFs to reduce implementation costs, allowing more households to be reached. Hopefully, the results presented herein will provide additional insight and quantified data on the operational considerations and removal capabilities of various types of full-scale BSFs to aid in the justification and support for future implementation efforts.

In section one, the background and scope of the problem of water access and quality in developing countries is reviewed, including a brief overview of several household water treatment technologies that are currently used. The introduction, section two, provides a detailed description of the biosand filter and the experimental setup that

was the focus of the laboratory research. Sections three through six contain the manuscript style descriptions of the four studies conducted, including the results and conclusions. The last and final section, section seven, is a summary of conclusions including findings and lessons learned gained in from the execution and evaluation of this research.

The research conducted and reported herein tested the general hypothesis that biosand filtration can be effective on a smaller, cheaper scale than currently practiced with the concrete BSF. In particular, we investigated how the efficacy of the CAWST BSF compared to smaller bucket-sized BSFs with respect to removal of turbidity, total coliforms, *E. coli*, MS2 coliphage, and *Cryptosporidium parvum* oocysts from raw drinking water supplies. Specifically, the research attempted to answer the following questions regarding BSF performance:

(1) Are the removal efficiencies of smaller BSFs significantly different from the concrete BSF?

(2) Is removal efficiency impacted by the turbidity of the source water?

(3) To what extent do slight disturbances affect the performance of the bucket BSFs?

(4) Can the BSF be modified (i.e., by the addition of rusty nails in the diffuser basin) to significantly improve the removal of viruses in the BSF?

(5) How is the removal efficiency impacted by the length of the pause period?

(6) If smaller sized BSFs can offer an acceptable level of removal (based on the laboratory results), how will a smaller BSF perform in the field and will it be acceptable to end-users?

Four separate studies (Sections 3.0 – 7.0 and summarized below), were conducted to answer the questions outlined above.

Effect of sand bed depth and media age on bacteria and turbidity removal

The main objective of the first study was to build several full-scale BSFs, simulate real-world usage conditions, and assess the long-term efficacy (9-month study period) for particulate and bacteria removal. Four replicates of three different filter designs were built: the traditional concrete BSF, and two scaled-down versions that use a 5-gal and 2-gal bucket, respectively, as the casing material. The major difference among the three BSF designs was the depth of the sand layer: approximately 54, 15, and 10 cm for the concrete, 5-gal bucket, and 2-gal bucket BSFs, respectively. This study investigated (1) how the efficacy of the CAWST (Centre for Affordable Water and Sanitation Technology version 10) BSF performed with respect to removal of turbidity and *E. coli* from raw drinking water supplies, (2) whether biosand filtration could be effective with scaled-down 5-gal and 2-gal bucket BSFs, (3) the effects of low and high turbidity feed water on filter performance and maintenance, and (4) the effects of filter maintenance (i.e., cleaning) on filter performance.

All bucket-sized filters, and two of the concrete filters, had hydraulic loading rates (HLRs) in the range of 0.2-0.3 $m^3/(m^2*hr)$ for the majority of the testing period. The smaller sand bed depths in the bucket-sized filters did not impact filter performance with respect to turbidity and *E. coli* removal or the effluent levels of turbidity and *E. coli*. All filters produced effluents with a mean turbidity of <0.6 NTU. In addition, 78%, 74%, and 72% percent of effluent samples for the concrete, 5-gal, and 2-gal filters, respectively, had *E. coli* concentrations <1 CFU/100 mL.

Based on the data collected in this study, the CAWST v10 concrete filter was able to achieve 98.1 – 98.4% turbidity removal and 3.8 – 4.0 log *E. coli* removal. The scaled-down BSFs, constructed in 5-gal (15cm bed depth) and 2-gal (10cm bed depth) buckets, were shown to be as effective (p-values >0.05) as the CAWST v10 concrete (54cm bed depth) configuration for both turbidity and *E. coli* removal. Alternating the influent turbidity between periods of high and low turbidity (~50 and ~5 NTU, respectively) did not influence either turbidity removal or *E. coli* removal. Periodic filter maintenance (i.e., cleaning the top of the sand bed) exhibited no correlation to either removal values or effluent levels of either *E. coli* or turbidity ($p<0.05$ and $|r|<0.4$). The smaller bucket-sized filters were found to be a viable alternative to the concrete BSFs for the removal of bacteria and turbidity from drinking water.

Transport effects on hydraulic loading rate and removal performance

BSFs designed using smaller and/or lighter casing material can result in reduced logistical requirements and implementation costs. However, the increased portability of a smaller, lighter design presents a potential negative consequence: the ability to move the installed/operational filter by the homeowner and potentially disturb the system. This study investigated the effects of moving and agitation on filter performance, using mature BSFs which had been in use for over nine months prior to the move. Data were analyzed for four replicate filters of three different filter types: the traditional concrete BSF and two plastic bucket (5-gal and 2-gal, respectively) BSFs.

Filters were moved approximately 1 km and monitored for hydraulic loading rates (HLRs) and *E. coli* removal for eight weeks following the move. Moving the filters

resulted in reduced HLRs, likely due to sand compaction, but *E. coli* removal remained high ($\log_{10}$ removal $\geq$ 2.8 for all sizes) and increased significantly as compared to data collected prior to the move. The resulting operational implications of moving BSFs are discussed.

Influence of sand depth and pause period on microbial removal in traditional and modified BSFs

The results of the first study showed that small biosand filters (sand bed depths of 10-15 cm) were effective at removing bacteria and turbidity. However, the impact of shorter bed depths on removal rates for smaller, sub-micron particles (such as viruses), as well as the impact of shorter pause periods on filter performance, remained unknown. For the third study, biosand filters with three different sand bed depths were modified with the addition of iron nails in the diffuser basin and evaluated for bacterial, protozoal, and virus removal over six different pause periods (1, 3, 6, 12, 24, and 72 hours).

The BSF configurations tested proved effective at removing the microbial contaminants over a range of pause periods. Removal of bacteria and protozoan cysts for all filter types and sizes ranged from 3 $\log_{10}$ to 4 $\log_{10}$. The addition of nails resulted in significantly better bacteria removal for all filter sizes, while only the smallest filters exhibited significantly better protozoan removal with the addition of nails. Virus removal for all filter types and sizes ranged from <1 $\log_{10}$ to 6 $\log_{10}$. Both the pause period and filter type (size/configuration) influenced virus removal, and the addition of nails to the filter significantly improved virus removal at the shorter pause periods.

Field evaluation of plastic-cased filters in Nicaragua

The fourth study was a field investigation to assess 1) the effectiveness of plastic-cased BSFs for improving water quality, 2) user acceptability and use, and 3) operational performance of the units. Two types of household BSFs were built, installed, and monitored over a three month period in four rural communities near San Juan del Sur, specifically a large filter made from (10in diameter) PVC pipe and a small filter made from a 5-gallon plastic bucket. The filters were designed based on the proportions of the CAWST v10 concrete BSF, that is there were proportionally designed with respect to filter media layers (i.e., sand, rock, and gravel) with the major differences between the types being the sand bed depths and reservoir volumes, which were 54cm and 15cm, and 12L and 3.6L for the large (PVC pipe) and small (5-gal) filters, respectively.

From the results of this study, the 5-gal bucket and PVC BSFs performed similarly with respect to *E. coli* removal. After approximately 6 months of use, the median log reduction values (LRVs) for the bucket and PVC BSFs were 1.73 and 0.95, respectively.

1.0 Household Water Treatment Processes and Technologies

As of 2012, the Joint Monitoring Programme (managed by the World Health Organization (WHO) and the United Nations Children's Fund (UNICEF)) estimated that approximately 800 million people in the world do not have access to an improved source, and that figure increases by hundreds of millions more for those without sustainable access to safe water (WHO/UNICEF 2012). It is important to clarify that an "improved drinking water source" only indicates an improvement in access and does not guarantee that the water is safe to drink. Infrastructure alone, e.g., a community groundwater well, community pipe/tap system, or household taps, would be considered an improved source even when no treatment is performed.

The greatest gains have been made in providing access to peoples within urban areas of the developing world, while those in rural regions represent over 80% of the 800 million still in need (WHO/UNICEF 2012). Community-based water systems, e.g., piped water systems and community wells, are not always feasible or the most appropriate solution, especially in rural areas. Since household water treatment and safe storage (HWTS) has been shown to reduce the number of diarrheal episodes by between 35% and 39% (WHO/UNICEF 2000), making HWTS options more accessible and affordable has the potential to significantly improve the quality of life for those in both rural and urban settings.

In terms of direct risk, a lack of clean drinking water supplies has been shown to lead to an increased incidence of deaths from water-borne disease, especially among poor

communities (Pruss 2002). Furthermore, there is a "growing sense that health is linked inexorably to socio-economic development" (NIC 2000). In 2005 a report on the threat to human health due to disease, identified that over a dozen countries in Africa over the last 20 years had per capita declines in income while population growth increased, thus "where population growth has been most rapid there has been little economic growth to accompany it" (Pirages 2005) and so developing countries bear the greatest burden associated with waterborne disease. Over 2 billion cases of diarrhea (WHO/UNICEF 2012) associated with unsafe water occur yearly, primarily caused by unsafe drinking water and inadequate sanitation and hygiene. Providing safe, reliable, piped water to every household is would undoubtedly yield optimal health gains; however, it is not always feasible. Community involvement and education and training on the principles, proper operation, and maintenance of any technology/system are critical. In addition, the socio-economic and cultural differences between communities within a country, including urban and rural settings will require different approaches and potentially different solutions. Since not all communities are at a state of development that can support community-based systems, treating water at the household level in these instances offers a sustainable alternative to providing safe drinking water for many under-developed communities.

As part of the Millennium Development Goals, the WHO supports incremental improvements in unsafe water supplies to accelerate the heath gains associated with safe drinking water (WHO 2011). One such interim improvement is household water treatment and safe storage (HWTS) to prevent contamination of water during collection,

transport, and use in the home. A growing body of evidence demonstrates that the use of HWTS technologies, such as chlorine tablets or filters, improves the microbiological quality of household water and reduces the burden of diarrheal disease in users (Clasen et al. 2007, Waddington et al. 2009, Fewtrell et al. 2005).

The potential treatment options used at the household level are based on same processes that are used for community based systems. The primary processes that govern water treatment can be categorized into three general classes 1) sedimentation, 2) filtration and 3) disinfection.

Sedimentation is a physical process where the settling of suspended particles in water happens due to the gravitational force acting on the particle. Sedimentation can be happen naturally, where particles are large enough that they settle out on their own, or can be a combination of physical/chemical processes enhanced through the addition of a chemical, or coagulant. Coagulation targets suspended solids that are too small to settle out by gravity within a reasonable timeframe (typically range of 0.001 – 1um). Suspended particles can be organic matter that impart color and/or turbidity and can also be microorganisms. Most colloids have net negative surface charge (organics and microbes) results in repulsion of particles that coupled with their small size cause them to remain in suspension. Coagulants are chemicals that are used to neutralize the surface charge of the suspended particles by adsorbing to surface, reducing the negative charge and repulsive forces, to allow them to aggregate and form larger particles, flocs. The most common coagulants in community-based treatment systems are aluminum and iron salts (e.g., aluminum sulfate: Al2(SO4)3, ferric sulfate: Fe(SO4)3, and ferric chloride:

FeCl3). Coagulation is the chemical treatment where the colloids are destabilized; flocculation is the physical process, the gentle mixing action required to induce the formation of the larger flocs. Flocculation is followed by a period of sedimentation, where the flocs will undergo gravitational settling and settle out.

Filtration often follows coagulation/flocculation to help remove flocs. Filtration is primarily a physical removal process where particles are separated out based on physical properties. Water flows through a filter is and removed based on size exclusion of the water flow channel, can be through porous media (e.g., sand, activated carbon) or can be a membrane. Often in water treatment porous media filters are used, most popular media types are sand, activated carbon, or a combination of the two. For sand filters, suspended particles are removed within the pore spaces between sand grains by straining, interception, impaction, settling, and Brownian diffusion. Some adsorption of particles, especially microbial constituents, happens. For activated carbon filters, size exclusion mechanisms within the pore spaces are enhanced by the ability to remove dissolved species through absorption. After a time, filters will clog or foul and the trapped particles will need to be removed to continue use.

Disinfection of drinking water is often the final step in water treatment where by waters are rendered safe from pathogens, either by killing or inactivating microbes. It is a chemical process, most often using chlorine (chlorine gas, sodium hypochlorite or calcium hypochlorite) to oxidize and effectively kill/inactivate any microorganisms. Chlorine oxidizes microbial enzymes and inhibits essential metabolic processes. The major advantage of chlorine is its ability to leave a residual disinfection concentration in

the water supply. Residual free chlorine is the available chlorine left in the water after a specified contact period, which can further disinfect any newly introduced biological contamination. Ozone is a more powerful oxidant and is more effective against cysts and virus than chlorine but it offers no residual (Reynolds & Richards 1996). Disinfection treatments are influenced by the cleanliness of the water; other contaminants, especially colloidal organic material, will react with the chlorine making less available to react with organisms. The major advantage of chlorine is its ability to leave a residual disinfection concentration later in the water supply distribution system. Residual free chlorine is the available chlorine left in the water after a specified contact period, which can further disinfect any newly introduced biological contamination.

All three of the general treatment processes have been scaled-down and used to improve drinking water quality in developing countries. The following sections outline several examples for each process.

1.1 Coagulation/Flocculation/Sedimentation

Natural sedimentation is often routinely employed in developing countries in response to limited access to water sources as opposed to a conscious effort to improve the quality of the water. People in the developing world travel can travel over an hour one-way to collect the water needed for throughout the day. Natural sedimentation will occur in the storage containers. While natural sedimentation can improve the aesthetics of the water, as previously mentioned it is the unsettable microbial contaminants that are typically of most concern for households in developing countries. Several

coagulation/flocculation regimes to enhance contaminant removal are found in developing countries; two currently-used alternatives are outlined below.

1.1.1 Moringa seeds

The seed kernels of the *Moringa oleifera* tree are a natural coagulant when dried and crushed into powder. The trees are native to northern India and are reportedly now grown throughout the tropics, especially in Africa. The powerful coagulation capability of the seeds is attributed to the large quantities of low molecular weight, water-soluble proteins that carry a net positive charge in solution. As with synthetic polymers or mineral-based coagulants, the proteins interact with the negatively charged particulates in the water and enhancing their ability to form flocs and settle out of solution. The general dosage recommendation is one shelled seed (~200mg) is used to treat 1L of very turbid water (Lea 2010). One of the most important benefits of this technology is that it is made from locally-available materials, sustainable, and offers the potential for scale-up and economic benefit.

1.1.2 PUR packet

The PUR packet is a mineral based treatment produced by P&G (Procter & Gamble, Cincinnati, Ohio). The packet contains iron sulfate and calcium hypochlorite: a coagulant and a disinfectant. For this technology, users are instructed to add packet the packet to 10L (approximately one 5-gal bucket) of water, rapidly mix for 5 minutes, let stand until no further settling visible, filter through a cloth, and finally let the water sit for approximately 20 minutes to disinfect. Pur packets have been tested both in the

laboratory and in the field that yielded >5 $\log_{10}$, 2 $\log_{10}$ and 1 $\log_{10}$ removals for waterborne bacteria, viruses, and protozoa, respectively (Souter 2003). The PUR packet was developed in conjunction with and is subsequently often recommended by the World Health Organization (WHO), especially for disaster response activities. The primary disadvantage of this technology is that it is manufactured by P&G and that poses greater risk to disturb the supply chain from production point to water treatment location.

1.2 Filtration

While filtration through cloth filters is often recommended after coagulation/filtration, it is typically not efficient enough on its own to substantially improve drinking water quality. Two household water filtration options that are currently-used as stand-alone treatments are outlined below.

1.2.1 Ceramic pot filters

Clay pot filters are designed to sit inside a 5-gal bucket, i.e., a clean water receptacle, with a spigot installed at the bottom. Water is added to the inside of the pot and filters through pore spaces in the clay. The filtered water is collected in the bucket. The clay pots are made with varying amounts and types of burnable materials (e.g, sawdust, coffee husks, and rice husks). When the clay pots are fired in a kiln the burnable materials are effectively removed leaving a matrix of pore spaces in the filter. Primary removal mechanism, as with all filters is size exclusion and adsorption onto the filter media. Some organizations add silver nitrate to the filters, either before or after the firing process, to act as a bacteriocide.

Ceramic pot filters have been shown to be effective at removing some levels of bacteria and high levels of turbidity and are produced locally. The main disadvantage is the relatively slow flow rates (as low as 0.25L/hr). In addition, these filters require periodic maintenance, users must scrub the inside of the filter, and this offers the potential for contamination of the outside (clean) region of the filter and increased handling leads to greater potential for damage (brittle terra cotta clay plots). (Lantagne et al 2010).

1.2.2 Biosand Filtration

BSFs are small scale, intermittently-operated slow sand filters traditional housed in a concrete casing with sand as the primary filter media. The contaminated water is poured into the top of the filter and the water flows through layers of sand and rock. The BSF purifies water through a combination of both mechanical and biological mechanisms, including exclusion, adsorption, predation, and natural die-off.

The placement of the outlet tubing is situated so that a standing water layer above the sand is maintained inside the filter. This supernatant layer supports the development and maintenance of a biologically active region at the sand surface, termed the schmutzdecke, which reportedly enhances microbial removal. The supernatant layer keeps the filter media saturated and allows for oxygen diffusion between charges enabling a biologically active region in the sand media. Within this region, microorganisms that are trapped by or adsorbed onto the sand grains can consume bacteria and other pathogens present in the water. In addition, microbes are also subject

to natural die-off due to inherently short life spans, nutrient scarcity, and/or non-optimal temperature. Additional details on BSF operation are presented in Section 2.0 and the results of BSF performance testing are presented in Sections 3.0 – 6.0.

1.3 Disinfection

1.3.1 Safe water system (SWS)

The safe water system is actually a three-step methodology for improving overall water and sanitation conditions: 1) treatment of water with dilute sodium hypochlorite; 2) utilization of a clean, safe storage container; and 3) education on proper hygiene and sanitation. The SWS was developed and is promoted by the Pan American Health Organization and the US Centers for Disease Control (CDC). This review will focus only on the water treatment aspect of the regime, where the disinfectant is added to the water. The standard solution is a 1.25% sodium hypochlorite solution. Users are instructed to use either a single (1 cap full) or double dose (2 cap fulls) at 1.875 and 3.75 mg/L sodium hypochlorite, respectively. The SWS is relatively easy to administer, is effective at low suspended solids concentrations, is relatively inexpensive and offers a residual disinfection concentration in the water.

The primary disadvantage of this treatment is that end users do not care to drink water that tastes and smells like chlorine. Therefore, the SWS is recommended to be used in conjunction with a pretreatment (e.g., cloth filtration, settling/decanting, or filtration) to remove the majority of the suspended particles thus reducing the chlorine demand. The

reduced turbidity often reduces the requirement to only one dose, or 1 cap full, which is often acceptable for drinking by the end-users.

Pretreatment in conjunction with 1 dose of SWS was shown to maintain the CDC recommended 0.2mg/L of free chlorine after 24 hour thereby effectively offering residual disinfection in water storage containers (Lantagne 2008, Koltarz 2009). Some have suggested that the dilute bleach solution will degrade rapidly, quoting half lives on the order of weeks to months; however, testing performed by Koltarz et al in 2010 showed that pH-stabilized solutions kept out of sunlight maintained concentration for minimum of 12 months over a range of temperatures (Lantagne et al. 2011).

1.3.2 Solar water Disinfection (SODIS) Method

The SOlar water DISinfection (SODIS) Method is a simple procedure where clear plastic bottles are filled with untreated water and are exposed to sunlight for 6 hours. If the water is turbid, pretreatment is recommended as suspended solids will block the infiltration radiation and reduce effectiveness of the treatment. The simply technology leverages the fact that ultraviolet (UV-A) light in the wavelength range 320-400nm is a natural germicide. The UV-A light causes severe damage to the DNA of the micro-organisms, thus disabling it from replication. At the UV-A level of radiation, the effect is most potent for bacteria, then viruses and is less effective for cysts. (Eawag, the Swiss Federal Institute of Aquatic Sciences and Technology). This technology is widely accepted by end-users based on the ease of implementation and no requirement of additional materials. Clear, plastic water bottles are ubiquitous and most often are already

at the household, thus no additional supplies are required. Increased turbidity and variable cloud cover can reduce effectiveness, but under normal operating conditions 3 $\log_{10}$ removals of bacteria and 3-4 $\log_{10}$ removals for Polio and Hepatitis viruses were reported (EAAWG 2012).

2.0 Introduction

The focus of the research effort summarized herein was on biosand filtration for household water treatment. The following section provides a short history of the BSF, the operating principles, and introduces the experimental setup for the studies conducted.

The BSF has been in use for years in communities around the world providing those without access to a community-based water source a means for treating water at the household level. However, even for communities that utilize the technology, field studies (Augilar 2009) have shown that the filters are often deployed in areas where the primary wage earning population often resides away from the primary residence for long periods of time, often during planting and harvesting seasons. For these communities that rely on BSF at their primary residence, workers often have no water treatment options during the most critical production periods. For others, the manufacturing cost and the difficulty of transporting the cumbersome concrete casing from the production site can eliminate the technology as a viable option.

The main objectives of the research were to build several full-scaled BSFs, simulate real-world usage conditions as much as possible, and test and document the efficacy for particulate and microbial removal. Three separate laboratory studies were performed on the filters: (1) turbidity and bacteria removal of full-scale filters, (2) effect of filter transport on performance, and (3) pause period and iron oxide effects on microbial removal; and a fourth study was conducted in Nicaragua to assess the efficacy and acceptance of the smaller filters by end-users in the field.

2.1 Traditional concrete BSF design

The traditional concrete BSF, designed in the 1990s (Manz 2007 & 2008), is a combination of technologies currently employed in community-based treatment systems: a traditional slow sand filter (SSF) and a biological contactor. A detailed comparison of BSF and conventional SSF design parameters is available (Elliott et al., 2006).

As depicted in **Figure 1**, the traditional BSF design (Manz 2007 & 2008) is an intermittently-operated SSF where a concrete container 0.3m x 0.3m x 0.9m: w x d x h), is used to enclose the filter media, layers of sand and gravel with five distinct regions of the filter are 1) the influent reservoir, air space above the filter media where the untreated water or charge water is introduced to the system and which allows for diffusion of oxygen to the water, and includes a diffuser basin, minimizes disturbance of the sand layer as a new charge of water moves from the reservoir to the filter area; 2) the supernatant, a constant standing water layer (5-10 cm = 2-4 inches) that supports a biologically active region at the sand surface; 3) schmutzdecke, or the biologically active region which develops at the sand surface; 4) the fine sand, the primary filter media; and 5) the rock layer, consists of coarse sand and gravel that supports the sand and promotes plug flow.

Figure 1: Cross-sectional view of the traditional concrete BSF.

The BSF purifies water through a combination of both mechanical and biological mechanisms, including exclusion, adsorption, predation, and natural die-off. Contaminants can be removed from the water column via size exclusion resulting from the limited pore space between the sand grains or by adsorption onto the surface of the sand grains or other adsorbed particles. The supernatant layer keeps the filter media saturated and allows for oxygen diffusion between charges enabling a biologically active region in the sand media. Within this region, microorganisms that are trapped by or adsorbed onto the sand grains can consume bacteria and other pathogens present in the water. In addition, microbes are also subject to natural die-off due to inherently short life spans, nutrient scarcity, and/or non-optimal temperature.

The BSF has two modes of operation: the run and the pause period. During a run, untreated water is poured into the reservoir and passes through the filter. The water introduces oxygen, nutrients and microbes to the system. Trapped and adsorbed contaminants will clog the pore openings; over time the flow rate will decrease. The pause period is the time between charges when there is no flow through the filter and the water is sitting stagnant in the filter. During the pause period, oxygen diffuses through the air in the reservoir through the supernatant layer to schmutzdecke, predation and natural die-off of microbes occur, organic matter is oxidized, and partial unplugging of the pores by motile cells and Brownian diffusion.

Previous studies, both laboratory and in the field, have yielded unique insights on the dominant filter removal mechanisms and end-user needs and requirements for sustained use in the real world. Previous research has shown that performance efficacy is

highly dependent upon several factors, namely (1) filter ripening over weeks of operation, (2) the daily volume charged to the filter, (3) the pause time between charge volumes, (4) influent water quality and (5) type of sand media (Elliott et al. 2008, Hijnen et al. 2004, Baumgartner et al. 2007, and Stauber et al. 2006).

Furthermore previous test results (Baumgartner et al. 2007, Elliot et al. 2008, Jenkins et al. 2011) demonstrated that contaminant removal is enhanced for water that is allowed reside in the sand bed for a pause period as compared to a continuous flow of water through the filter with no residence time. Elliot et al (2008) found that performance was maximized when less than one pore volume (18.3-L in the filter design studied) was charged to the filter per day and this has important implications for filter design and operation. Based on these results, the most efficient filter design would have a reservoir volume (or charge volume) that would equal the pore space of the fine sand bed media. The traditional BSF design had a reservoir volume of 18.5 L versus a pore volume of 8.9 L, which resulted in 9.6 L of the feed water passing through the fine sand area during the current charge and thus had minimal contact time with the sand. In 2008, the Centre for Affordable Water and Sanitation Technologies (CAWST), modified the traditional BSF design (**Figure 2**) by repositioning the water outlet and increasing the volume of sand so the two volumes, reservoir and sand pore space, were equal (CAWST v10).

The CAWST v10 design ensures that all feed water spends at least one pause period in the sand bed prior to collection and use thus maximizing the opportunity for contaminant removal by biological treatment and adsorption within the sand bed. CAWST maintained the overall dimensions of the concrete mold so that those

communities and organizations already manufacturing filters could simply modify their existing mold. The resulting effect was a BSF with a charge volume reduced to 12 L.

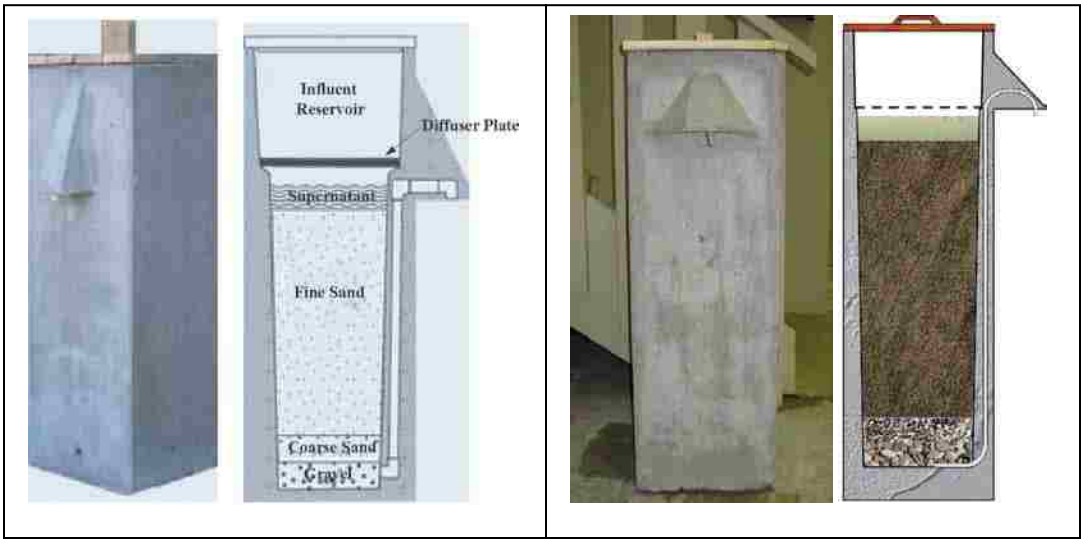

Figure 2. Comparison of the original concrete BSF (left panel) to the CAWST version 10 (right panel). Outside view of the BSF showing the spout from which treated water is collected and a cross-sectional view of the BSF showing the internal components. Left panel: photo and schematic adapted from http://www.cleanwaterforhaiti.org/. Right panel: Photo and schematic courtesy of CAWST.

2.2 Experimental Setup

Four replicates of three different filter designs were built in the laboratory; the CAWST version 10 concrete BSF, and two scaled-down versions that used a 5-gal and 2-gal bucket as the casing material (**Figure 3**). The smaller BSFs were designed using the same operating principle as the CAWST version 10, specifically that the influent reservoir volume, or charge volume, equals the pore volume of the filter media, or sand layer. The major difference among the three BSF designs is the depth of the sand layer:

approximately 54 cm for the concrete BSF, 15 cm for the 5-gal bucket BSF, and 10 cm for the 2-gal bucket BSF. The reduced sand bed layer equates to filter charge volumes (and influent reservoirs) of 12L for the concrete, 3.6L for the 5-gal bucket, and 1.5L for the 2-gal bucket.

Figure 3. Photograph of laboratory setup. The four concrete BSFs located on the left, the four 5-gallon bucket BSFs are located in the center, and the four 2-gallon bucket BSFs are located on the right. Photo Credit: J. Napotnik

Influent, or charge, water consisted of either unaltered spring water or non-chlorinated tap water augmented with creek water and sediments from Monocacy Creek (Bethlehem, PA), an unimpacted surface water (no wastewater discharges) and tributary to the Lehigh River. Creek water/sediments were collected in 20L carboys and held at room temperature; maximum hold time was one week.

2.3 Statistical Analysis

The Kruskal-Wallis test is a non-parametric test for a difference in central location (median) between two or more independent samples (i.e., filters) measured on an

ordinal or continuous scale with similar distributions (normalcy not required). Null hypothesis is that the samples are from the same population, and the p-value is the probability of rejecting the null hypothesis when it is, in fact, true. A significant p-value ($p<0.05$) implies that at least two samples have different medians, or are from different populations.

If multiple hypotheses are tested (i.e., if multiple sample sets are compared simultaneously), the chance of rejecting at least one null hypothesis is increased. To control for this overall type I error for multiple comparisons, the conservative Bonferroni method (Sheskin 2003, Conover 1999) was used, which is equivalent to performing t-tests on each pair of groups. In this study for multiple comparisons, if the null hypothesis was true ($p>0.05$) and the populations were all statistically the same, then only the p-value is reported here. When the comparison of multiple samples resulted in the rejection of the null hypothesis ($p<0.05$), indicating a significant difference among populations, the overall p-value is reported along with the significant p-values between individual sample sets.

In addition, Pearson correlation tests were performed to identify significant correlations (p-value <0.05) between variables. The strength of the relationship was inferred based on the resultant correlation coefficient, r; the closer the coefficient is to 1 or -1, the stronger the correlation, where 1 and -1 are strong positive and negative correlations, respectively. A significant, strong correlation was defined as having both $p<0.05$ and $|r|\geq 0.4$.

All statistical analyses were performed with the Analyse-It add-in (Analyse-It Software, Ltd., Leeds, England) for Microsoft Excel. Some results are presented as outlier boxplots where the whiskers extend to the furthest observations within ±1.5 times the interquartile range (IQR) of the first and third quartile (Q1 and Q3, respectively); near outliers are observations within 1.5-3 times the IQR of Q1 and Q3 and marked by a "+"; and, far outliers are observations greater than 3 times the IQR of Q1 and Q3 and are marked by a "*". **Figure 4** identifies and defines the various components of the boxplot utilized hereafter.

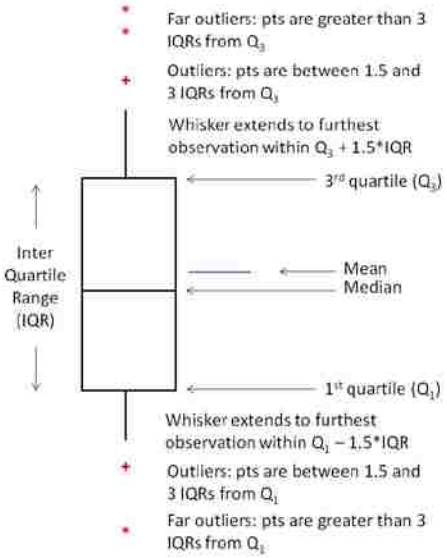

Figure 4. Boxplot definition

3.0 Effect of Sand Bed Depth and Media Age on Bacteria and Turbidity Removal

The main objective of the study was to build several full-scale BSFs, simulate real-world usage conditions, and assess the long-term efficacy (9-month study period) for particulate and bacteria removal. Four replicates of three different filter designs were built: the traditional concrete BSF, and two scaled-down versions that use a 5-gal and 2-gal bucket, respectively, as the casing material. The major difference among the three BSF designs was the depth of the sand layer: approximately 54, 15, and 10 cm for the concrete, 5-gal bucket, and 2-gal bucket BSFs, respectively.

This study investigated (1) how the efficacy of the CAWST (Centre for Affordable Water and Sanitation Technology version 10) BSF performed with respect to removal of turbidity and *E. coli* from raw drinking water supplies, (2) whether biosand filtration could be effective with scaled-down 5-gal and 2-gal bucket BSFs, (3) the effects of low and high turbidity feed water on filter performance and maintenance, and (4) the effects of filter maintenance (i.e., cleaning) on filter performance.

All bucket-sized filters, and two of the concrete filters, had hydraulic loading rates (HLRs) in the range of 0.2-0.3 $m^3/(m^2*hr)$ for the majority of the testing period. The smaller sand bed depths in the bucket-sized filters did not impact filter performance with respect to turbidity and *E. coli* removal or the effluent levels of turbidity and *E. coli*. All filters produced effluents with a mean turbidity of <0.6 NTU. In addition, 78%, 74%, and

72% percent of effluent samples for the concrete, 5-gal, and 2-gal filters, respectively, had *E. coli* concentrations <1 CFU/100 mL.

Based on the data collected in this study, the CAWST v10 concrete filter was able to achieve 98.1 – 98.4% turbidity removal and 3.8 – 4.0 log *E. coli* removal. The scaled-down BSFs, constructed in 5-gal (15cm bed depth) and 2-gal (10cm bed depth) buckets, were shown to be as effective (p-values >0.05) as the CAWST v10 concrete (54cm bed depth) configuration for both turbidity and *E. coli* removal. Alternating the influent turbidity between periods of high and low turbidity (~50 and ~5 NTU, respectively) did not influence either turbidity removal or *E. coli* removal. Periodic filter maintenance (i.e., cleaning the top of the sand bed) exhibited no correlation to either removal values or effluent levels of either *E. coli* or turbidity ($p<0.05$ and $|r|<0.4$). The smaller bucket-sized filters were found to be a viable alternative to the concrete BSFs for the removal of bacteria and turbidity from drinking water.

3.1 Introduction

As of 2012, over 400,000 BSFs have been implemented in households in over 60 countries, serving more than 2.5 million people (CAWST 2012b). The BSF has been successful in reducing the incidence of diarrheal disease (Clasen et al. 2007; Sobsey 2002) and will continue to help meet the safe water Millennium Development Goal (WHO/UNICEF 2005). The traditional concrete BSF produces high quality drinking water, is durable, and is easy to use and maintain. The filters have a manufacturing cost (materials and labor) ranging from $15-60 USD (CAWST 2012b; CDC 2012); however,

depending on the country, additional costs for fuel, electricity, edcuation/training, etc. can drive the cost to $70-100 (Activewater 2009; The Water Project 2013). The initial installation cost in one development program is estimated at $50, for which 36% ($18) is attributed to transportation and education (CDC/USAID 2008). While there are no other costs for consumables or maintenance, the BSF can still be too costly for some of the poorest households in the developing world. In addition, the size (0.3 m x 0.3 m x 0.9 m, w x d x h) and weight (250 lbs) of the concrete filter make it cumbersome and difficult to transport beyond the initial installation site.

This study tested the hypothesis that biosand filtration can be effective with smaller, lighter and less expensive units in order to more sustainably meet the needs of a larger global market. Making BSF technology more accessible to a broader population will reduce the incidence of waterborne diarrheal disease, increase the productivity and earning capacity of the average household, and help households and communities break the cycle of sickness and poverty which currently plagues billions of people worldwide.

This study investigated (1) the efficacy of the Centre for Affordable Water and Sanitation Technology (CAWST) version 10 (v10) BSF with respect to removal of turbidity and *E. coli* from raw drinking water supplies, (2) whether biosand filtration could be effective with scaled-down 5-gal and 2-gal bucket BSFs, (3) the effects of low and high turbidity feed water on filter performance and maintenance, and (4) the effects of filter maintenance (i.e., cleaning) on filter performance.

3.2 Materials & Methods

3.2.1 Bacterial Growth and Enumeration

Freeze-dried *Escherichia coli* ATCC® 11775™ (Manassa, VA) were rehydrated and propagated for 24 hrs at 35°C in Luria-Bertani (LB) broth (BD Diagnostic Systems, Sparks, MD). Stock solutions in LB broth were prepared from isolated colonies grown on LB agar plates (BD Diagnostic Systems Sparks, MD) for 24 hrs at 35°C and stored at -80°C with 10% glycerol. Prior to experimentation, clonal plates were made from thawed stock solution, and a single colony was propagated in a volume of LB broth sufficient to spike the entire influent volume for one test day to minimize any genetic variation in the bacterial community. The target spike concentration for the influent was 1E6 CFU/100 mL.

The *E. coli* concentration of the inoculated broth was estimated via the optical density at 600nm obtained by a DR-4000 spectrophotometer (HACH Company, Loveland, CO) and an experimentally determined standard curve. Inoculated broth concentration was confirmed by direct plate counts on LB agar plates. Filter influent and effluent samples were analyzed via Standard Methods 9222 for membrane filtration for members of the coliform group (Rice et al. 2012). The average filter influent concentration was 2.8E3 CFU/100 ml (max = 1.8E4, min = 7.44, s.d. = 5E3).

All samples/dilutions were performed in triplicate and resultant colony counts were averaged. Serial dilutions were prepared using dilution water (Buffered Dilution Water Pillows, Hach, Loveland, CO). Samples were vacuum-filtered through a 47-mm,

0.45-µm pore size cellulose ester membrane filter. Prior to filtering each set of dilutions for a given sample, 100 mL of ultrapure water (Milli-Q, Millipore Corp., Billerica, MA) were filtered as a negative control to check for possible contamination. Following sample filtration, filters were placed in a culture dish that contained a sterile pad and 2 mL of m-ColiBlue 24® broth and incubated at 35 ± 0.5°C for 24 hrs.

For each plate, colony counts were recorded as colony forming units (CFUs), and the resulting concentration (CFU/100 mL) was calculated based on the total volume of original sample filtered. For all samples, at least one plate yielded a statistically valid number of *E. coli* CFUs (i.e., 30-100 CFUs); instances when more than a single dilution plate yielded statistically valid counts, the resulting concentrations were averaged together. Instances where all plates yielded no CFUs, the detection limit (1CFU/total volume analyzed) was used as the effluent concentration for the subsequent calculation of removal efficiency.

Filter effluent concentrations (CFU/100 ml) were classified by potential human health risk associated with *E. coli* concentrations. The five risk levels are based on WHO guidelines (WHO 1997) and are defined as follows: 1) acceptable or within conformance: 0 - <1 CFU/100 ml, 2) low risk: 1 - <10 CFUs/100 ml, 3) moderate risk: 10 - <100 CFUs/100 ml, 4) high risk: 100 - <1000 CFUs/100 ml, and 5) very high risk: ≥1000 CFUs/100 ml.

3.2.2 Water Quality Parameters

Chemical and physical water quality data were collected for each sample (influent

and filter effluents) to monitor any changes in nutrient levels that could potentially influence filtration efficacy and biolayer development. Each assay was performed in triplicate and the results were averaged. Individual stock solutions of water quality standards were of analytical or reagent grade (Fisher Scientific, Pittsburgh, PA). Working standards were prepared by diluting stock solutions with laboratory grade, Milli-Q water (Millipore, Billerica, MA), as required per protocol. **Table 1** outlines the methods, detectable ranges and detection limits for each parameter. In addition on days when microbial testing was performed, turbidity was measured using a Hach Turbidimeter Model 2100P.

Table 1. Water quality parameters, test method, detectable range, and detection limit.

Parameter	**Test Method**	**Detectable Range**	**Detection Limit**
Alkalinity	Digital Titrator, Hach model 16900, using Sulfuric Acid; Hach Method 8203	10-160 mg/L as CaCO3	10 mg/L
Hardness	Digital Titrator, Hach model 16900, using EDTA; Hach Method 8204	100-400 mg/L	10 mg/L
Organic Carbon, total	Hach Method 10129 and Test 'N Tube™ Vials for low range for the DR/4000 Spectrophotometer	0 to 20.0 mg/L C	0.3 mg/L C
Nitrogen, total	Hach Method 10071 Persulfate Digestion Method using Test 'N Tube™ Vials for the DR/4000 Spectrophotometer	0 to 25.0 mg/L N	2 mg/L N
Phosphorus, reactive (orthophosphate)	Hach Method 8048 PhosVer 3 (Ascorbic Acid) Method using Powder Pillows for the DR/4000 Spectrophotometer	0 to 2.500 mg/L PO_4^{3-}	0.045 mg/L PO_4^{3-}
Manganese, total	Hach Method 8149 PAN Method using Powder Pillows for the DR/4000 Spectrophotometer	0 – 0.700 mg/L	0.005 mg/L
Iron, total	Hach Method 8008 FerroVer Method using Powder Pillows for the DR/4000 Spectrophotometer	0 – 3.000 mg/L	0.008 mg/L

3.2.3 Hydraulic Loading Rate

The flow rate from each filter was measured directly after adding a full reservoir volume to the filter (effectively the peak flow rate of the filter since the hydraulic head was at its maximum). Flow rates were measured using a graduated cylinder and stop watch. From this peak flow rate, the peak hydraulic loading rate (HLR) was calculated (**EQN 1**) to normalize the data to the sand surface area of each filter. The area of the top of the fine sand layer for the concrete, 5-gallon bucket, and 2-gallon bucket filters was 0.059, 0.059, and 0.039 m^2, respectively.

$$HLR = Q/A \qquad \textbf{(EQN 1)}$$

where HLR = hydraulic loading rate ($m^3/(m^2 \cdot hr)$)

Q = flow rate (m^3/hr)

A = area (m^2)

For example, the maximum recommended flow rate for the CAWST v10 concrete filters is 0.4 L/min (CAWST 2012a); conversion of flow rate to HLR, based on eqn. 1, is as follows:

$$HLR = [0.4L/min * 60min/hr * 1m^3/1000L] / 0.06\ m^2 = 0.4\ m^3/(m^2*hr) = 0.4\ m/hr$$

Filter flow rates were monitored to identify when filters required cleaning; cleaning was performed when flow rates decreased to approximately less than half the initial clean bed value. Minimum flow rates impact end user acceptability, as users will

discard or discontinue use of a filter that takes too long to filter (CAWST 2012a). Users are directed to clean the filter when they feel it has become too slow; thus our designation of half the initial flow rate is a subjective minimum. To clean the filters, the top 0.5 – 1 cm of the sand bed, where the schmutzdecke develops, was disturbed to suspend trapped particles and biofilm material from the top of the sand layer into the supernatant. The resultant dirty supernatant was discarded and replaced with clean, non-chlorinated water. This cleaning process was repeated until the supernatant water was visibly clear. Since effective filter performance has been attributed to a well-developed schmutzdecke (Elliott et al. 2008; Palmateer et al. 1999), we evaluated the potential effects of filter cleaning (i.e., schmutzdecke disturbance) on filter performance. The term "schmutzdecke age" was used for this analysis and is the number of days since the most recent cleaning.

3.2.4 Experimental Setup

Four replicates of three different filter designs were built in the laboratory: the CAWST v10 concrete BSF, and two scaled-down versions that used a 5-gal and 2-gal bucket, respectively, as the casing material. The smaller BSFs were designed using the same operating principle as the CAWST v10, specifically the influent reservoir volume, or charge volume, equalled the pore volume of the filter media (in this case, the sand layer). Schematics of the three filter designs are presented in **Figure 5**. The major difference among the three BSF designs was the depth of the sand layer: approximately 54 cm for the concrete BSF, 15 cm for the 5-gal bucket BSF, and 10 cm for the 2-gal bucket BSF. The sand bed depths equated to filter charge volumes (and influent reservoir

volumes) of 12L for the concrete BSF, 3.6L for the 5-gal bucket BSF, and 1.5L for the 2-gal bucket BSF.

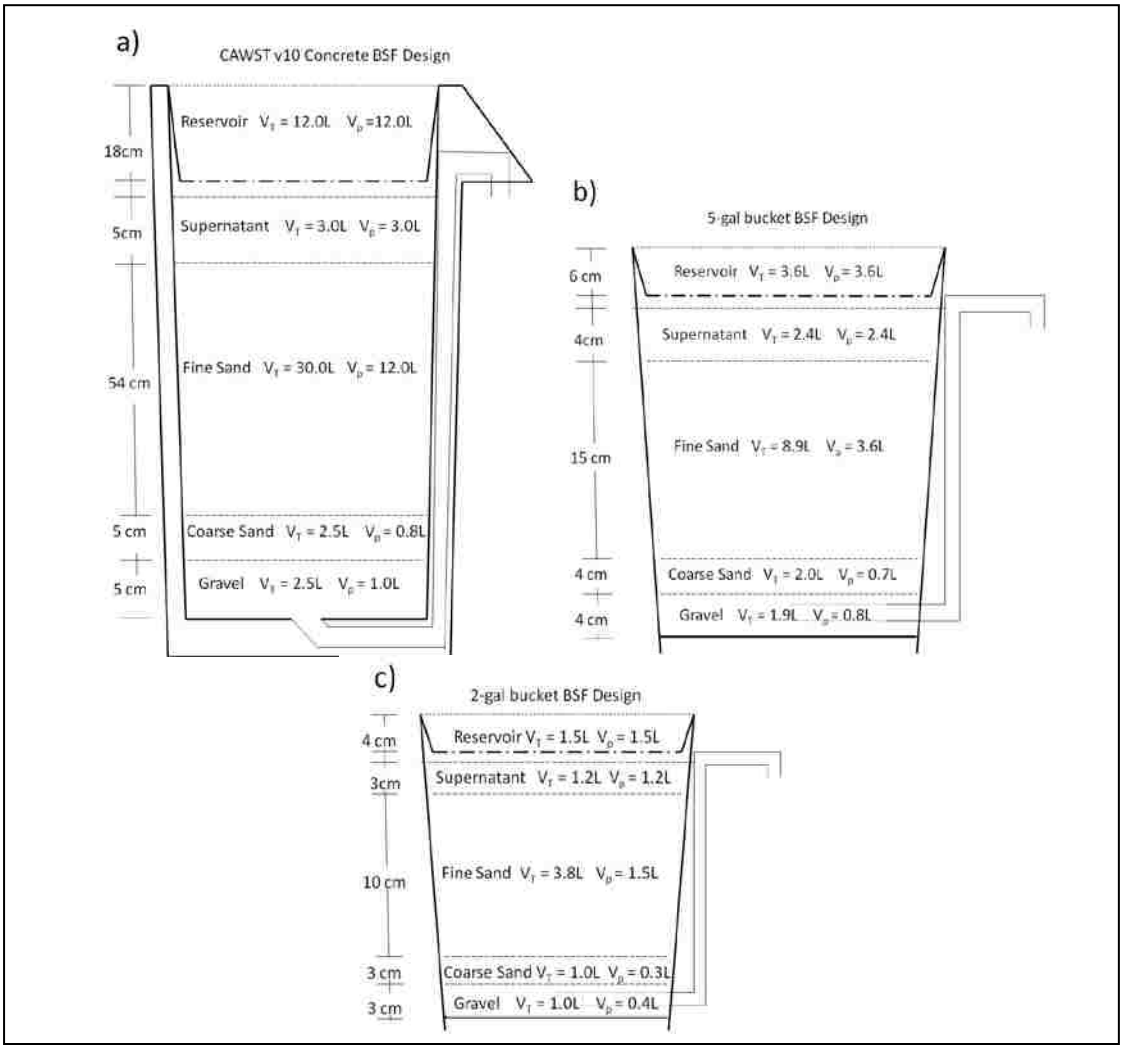

Figure 5. Schematics of the filter design for the a) concrete, b) 5-gal bucket, and c) 2-gal bucket casings (not to scale), highlighting the differences in depth, total volume (V_T) and pore volume (V_P) for the filter regions.

The performance of biological treatment processes have inherent variability associated with fluctuations in media type, raw water characteristics, temperature, and operator attention (Metcalf & Eddy, 2002). As a small-scale biological treatment process,

BSFs are also vulnerable to these variables. By convention, BSFs use sand as the primary media type. Crushed rock is the recommended source type as river and beach sands often contain salts and organic material and can therefore become a potential source of contamination (CAWST 2012). However, Duke and Mazumder (2009) showed that if the sand is properly prepared, i.e., washed and sized, that there was no significant difference in filter performance between crushed rock and beach sand.

The production of BSFs in the field is often a group or community event. Sieving and washing of the sand media is a very large component and time consuming activity of the overall production. As long as the proper sized screens are used, the sieving of the sand is straight forward and multiple operators will have minimal effects on the resulting end product. Conversely, washing the sand is highly operator dependent and can have a significant impact on the final product as it is directly responsible for the resulting particle size distribution of the sand media.

The proper size distribution of the sand was achieved in the lab using the field "jar test" which involves suspending the sand grains in clean water and then visually estimating the settling rate of the sand after the suspension is vigorously shaken (CAWST 2012a). It is a simple, yet extremely effective method. Based on the jar test, the sand media (all purpose sand, Green Pond Nursery, Bethlehem, PA) was washed twice to remove the smaller sand grains. A composite sample was collected during the installation of each filter and a complete sieve analysis was completed on each composite. The sand was characterized by the effective size and uniformity coefficient, where *the effective size* (d_{10}) is the size of the sieve (in mm) through which 10% of the sample of sand by weight

will pass and the uniformity coefficient (d_{60}/d_{10}) describes the distribution of particle sizes and is defined as the ratio of the sieve size (in mm) through which 60% of the sample will pass, to the effective size of the sand. **Table 2** reports the resulting d_{10}, d_{60}, and uniformity coefficient of each filter. All sand was within the recommended range for effective size (0.15-0.20mm) and uniformity coefficient (<2.5) (CAWST 2012a). In addition, there was no significant difference in the uniformity coefficients for the different filter sizes (p=0.9574).

Table 2. Particle size distribution parameters of the sand media for the four replicate filters of each size.

	Concrete				5-gal bucket				2-gal bucket			
d_{10} (mm)	0.18	0.18	0.18	0.17	0.18	0.18	0.18	0.19	0.16	0.17	0.16	0.17
d_{60} (mm)	0.33	0.3	0.42	0.32	0.32	0.33	0.32	0.43	0.29	0.34	0.31	0.3
U*	1.83	1.67	2.33	1.88	1.78	1.83	1.78	2.26	1.81	2.00	1.94	1.76

*Uniformity Coefficient, $U = d_{60}/d_{10}$

The filters were challenged for 9 months to test the effect of influent water quality on filter performance. Influent water was charged to the filters three times per day with a three-hour pause period, or idle time, between fills. The concrete filters held 12L per fill, equating to 36L/day/filter; 3.6L for a 5-gal (B) filter equating to 10.8L/day/filter; and 1.5L for a 2-gal (A) filter equating to 4.5L/day/filter. A large spike tank (120 L) was utilized to prepare a single influent batch for all 12 filters (total of 68.4 L required for a single fill of all filters).

Influent water was compared with filter effluents to assess the efficacy of each filter type in removing turbidity and *E. coli*. In an attempt to simulate real world conditions, the turbidity of the influent water was fluctuated between high and low levels, approximately 50 NTU and 5 NTU, respectively. Biosand filtration is not recommended

for extremely turbid waters since very turbid water will increase the particle loading per fill and the required frequency of filter cleaning, thus increasing the likelihood of filter abandonment by the user. CAWST recommends that for cases of high turbidity (> 50 NTU), waters allowed to settle naturally (typically within 1 hour) will most often result in a maximum turbidity of approximately 50NTU (CAWST 2012a). Influent water consisted of unaltered spring water augmented with Monocacy Creek (Bethlehem, PA) water/sediments to obtain the desired turbidity level.

3.2.5 Statistical Analysis

The Kruskall-Wallis test with Bonferroni type I error protection was performed to determine whether there was a significant difference in performance i) across filters of the same type (i.e., four replicates of each size), and (ii) across the three filter types (i.e., concrete, 5-gal, and 2-gal BSFs). In addition, Pearson correlation tests were performed to identify significant correlations (p-value <0.05) between variables. A significant, strong correlation was defined as having both $p<0.05$ and $|r|\geq 0.4$. All statistical analyses were performed with the Analyse-It add-in (Analyse-It Software, Ltd., Leeds, England) for Microsoft Excel.

3.3 Results

3.3.1 Water Quality

The nutrient requirements for developing and sustaining biological activity in the filter will be dependent upon a number of parameters, such as the composition of the

microbial community, daily biomass production rate, geographical location, and time of year. However, in general, microbial communities require, at minimum, adequate levels of carbon, nitrogen, and phosphorus (a.k.a. the macro nutrients). Because the influent water turbidity was variable, testing was performed to ensure nutrients were present to support the biological growth in filters.

All filters produced effluents with similar TOC concentrations ($p=0.4242$) with resultant averages of 7.7, 7.9, and 9.4 for the concrete, 5-gal, and 2-gal filters, respectively (see **Figure 6**). The TOC that remains in the effluent will primarily be dissolved as turbidity levels were consistently <1 NTU (see Section 3.3.3). High levels of TOC in the effluent can lead to offensive odors and tastes and if waters are chlorinated after filtration.

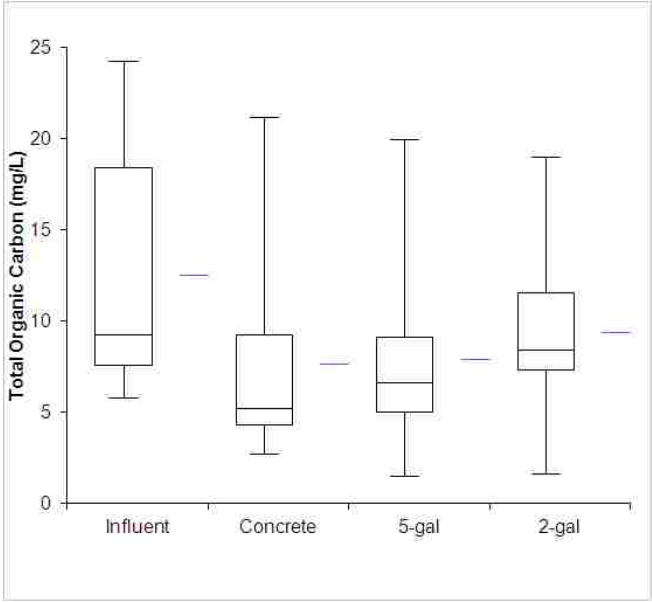

Figure 6. Total organic carbon concentration (mg/L) for the influent and filter effluents.

Evaluation of the normalized TOC concentrations for each filter type (**Figure 7**) suggest that there may be a normal cycling of the biofilm community, based on the peaks seen around test day 100 and 250. The unspiked influent water contained total organic carbon TOC levels ranging from 5.8-24.2 mg/L with an average of 12.5mg/L; these values are within the typical range for surface waters of 1 to 20 mg/L (Bouwer and Crowe 1988) and are comparable to other laboratory studies that used pasteurized primary effluent for filter influents, range of 7.5 to 12.6 mg/L (Elliot et al. 2008).

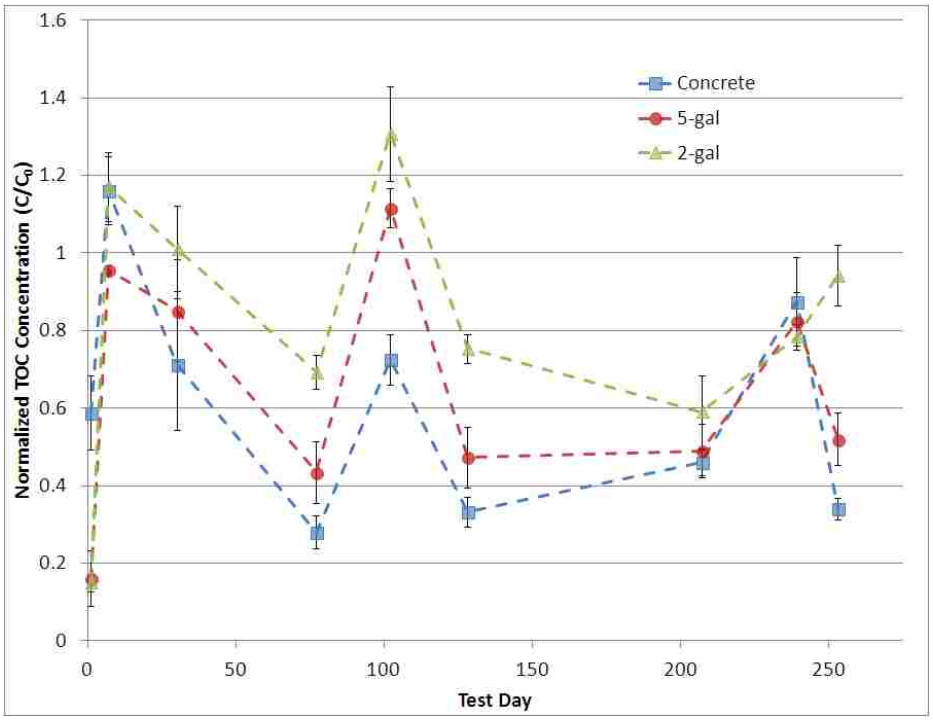

Figure 7. Normalized total organic carbon (TOC) concentrations (mg/L) by filter type.

The total nitrogen of the influent water averaged 6.4mg/L and ranged from 1 to 14 mg/L(Figure 10); the effluent concentrations averaged 4.4, 4.4, and 4.8 for the concrete, 5-gal, and 2-gal filters, respectively (**Figure 8**). Naturally occurring levels of nitrogen in surface water

will vary substantially, the high levels observed in the influent (≥6 mg/L) is not uncommon for a watershed with a large amount of agricultural land use (Mueller and Spahr, 2005), such as is the case for the Monocacy Creek.

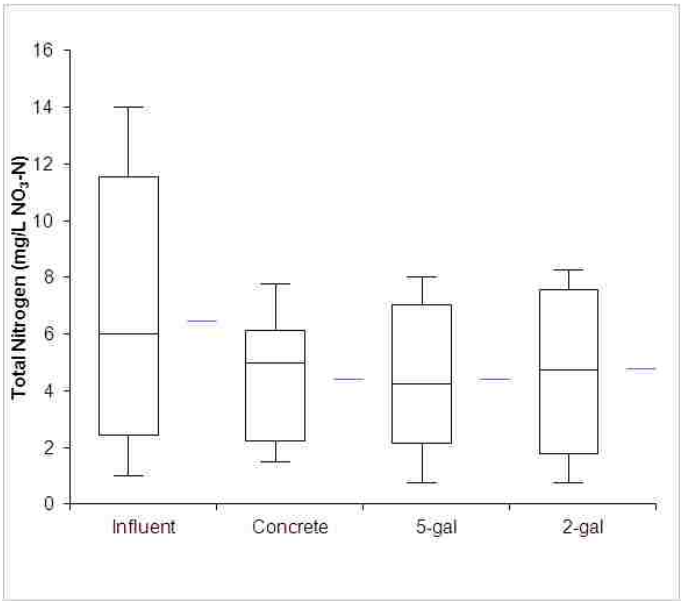

Figure 8. Total nitrogen concentration (mg/L as NO_3-N) for the influent and filter effluents.

The orthophosphate (dissolved phosphorus) concentration of the influent water ranged from 0.07 to 0.4 mg/L with an average of 0.2mg/L, and the average filter effluents were 0.09, 0.06, and 0.06 for the concrete, 5-gal bucket and 2-gal bucket filters respectively (**Figure 9**). The phosphorus levels of the influent water were consistently in excess of reported natural background levels for surface waters, which are reportedly less than 0.03 mg/L and can range between 0.005 to 0.05 mg/L (Mueller and Spahr, 2005). Again high levels of phosphorus are not uncommon in agricultural watersheds.

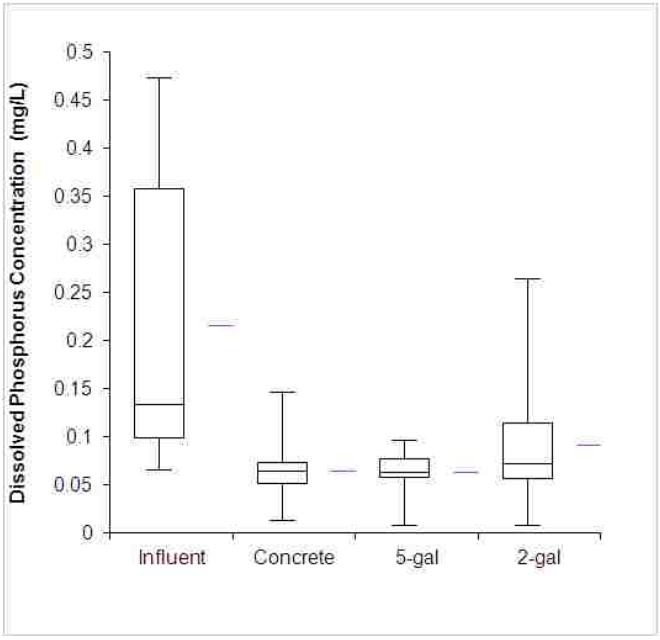

Figure 9. Total dissolved phosphorus (orthophosphate) concentration (mg/L as PO_4^{3-}) for the influent and filter effluents.

On average the pH of the influent was 7.5 and was significantly different between the concrete and bucket filters ($p=0.0043$), with averages of 8.1, 7.7, and 7.7 for the concrete, 5-gal and 2-gal bucket filters, respectively (shown in **Figure 10).** The concrete filters exhibit the most variability across filters of the same type and initially produced some water with high (>9) pH. These outliers were observed on Day 1 of testing is attributed to leaching of calcium carbonate from the concrete casing material. Elevated pH, ≥ 8.0 was observed for the first 35 days of operation after which time the effluents of the concrete filters lowered to an average of 7.7 (**Figure 11**), the same as seen for the bucket filter effluents.

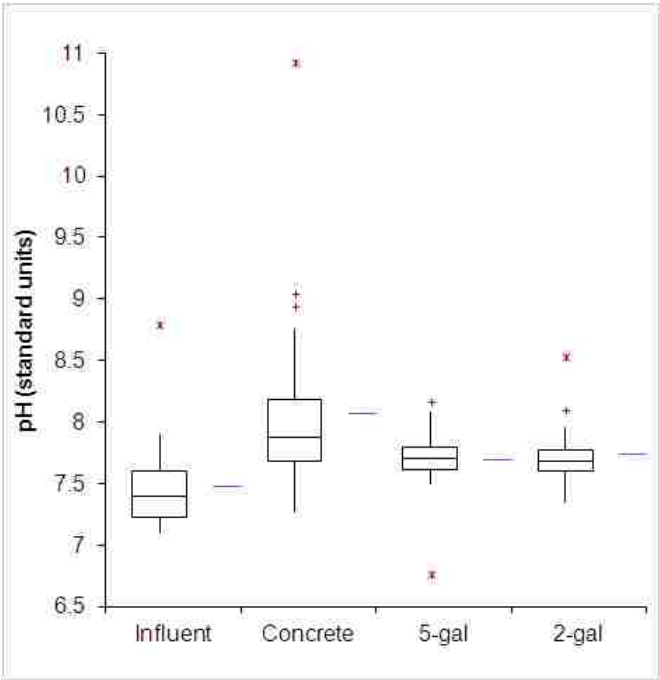

Figure 10. pH values for the influent and filter effluents.

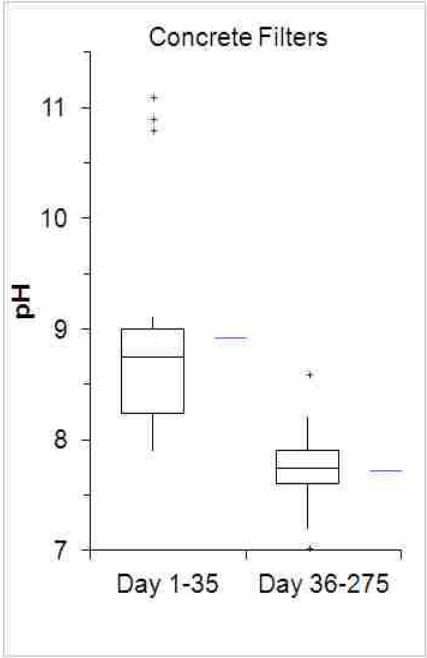

Figure 11. Change in pH values for concrete filters over time.

In comparison, high alkalinity values were only detected in the concrete filter effluents on test day 1 (outliers on **Figure 12**). The overall averages for alkalinity were 41.0, 43.3, 45.0 and 45.0 for the influent, concrete, 5-gal, and 2-gal filters, respectively. The alkalinity of the influent averaged 41.0 mg/L and ranged from 30-58 mg/L, which is comparable to other the waters of previous studies that ranged from 15 to 50 mg/L (Elliot et al 2008, Unger and Collins 2008). There was no significant difference (p=0.3893) in the alkalinity of the influent water and filter effluents. The hardness of the influent water ranged from 247 to 492 mg/L (as $CaCO_3$), typical of very hard waters (> 181 mg/L). The average hardness of the filter effluents 306, 313, and 295 mg/L (as $CaCO_3$) and were not significantly different from one another (p=0.9751) (**Figure 13**).

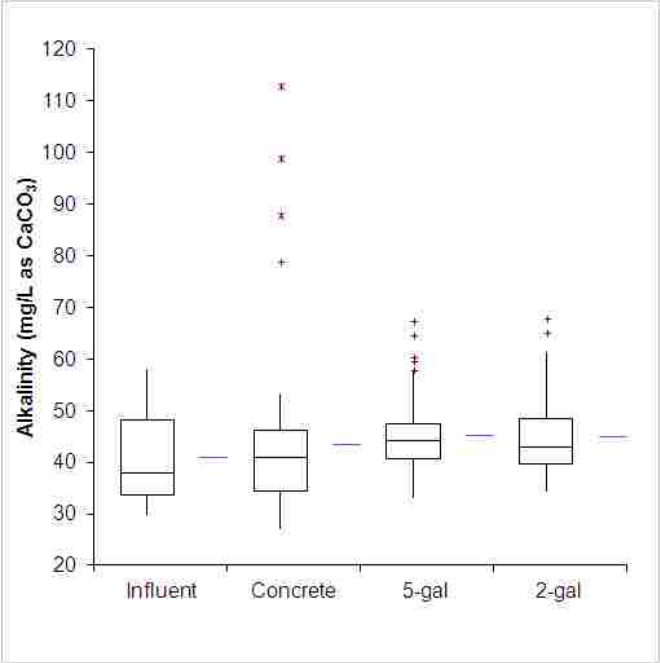

Figure 12. Alkalinity (mg/L) of the influent and filter effluents.

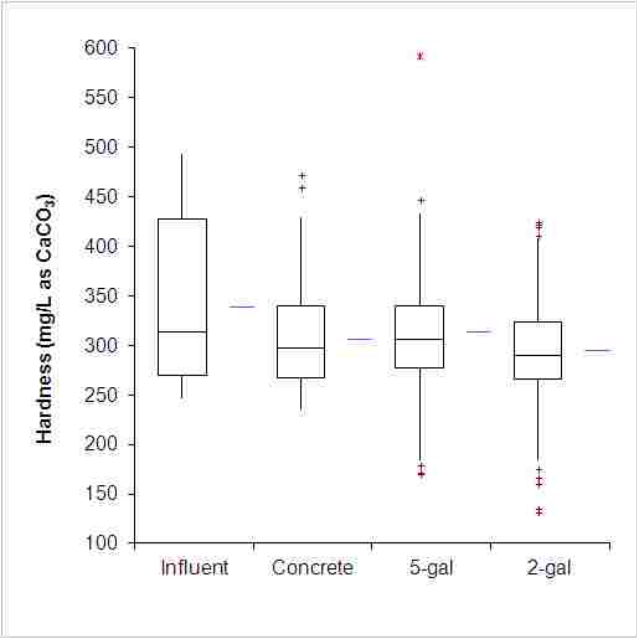

Figure 13. Hardness (mg/L) values for the influent and filter effluents.

Manganese is a commonly encountered contaminant and mostly considered an aesthetic issue, as it will impart a grey color to clothing or food (e.g., rice) when it is present in high enough concentrations. Manganese concentrations were elevated in all filter effluents (**Figure 14**) and were significantly higher than the influent concentration ($p<0.0001$).

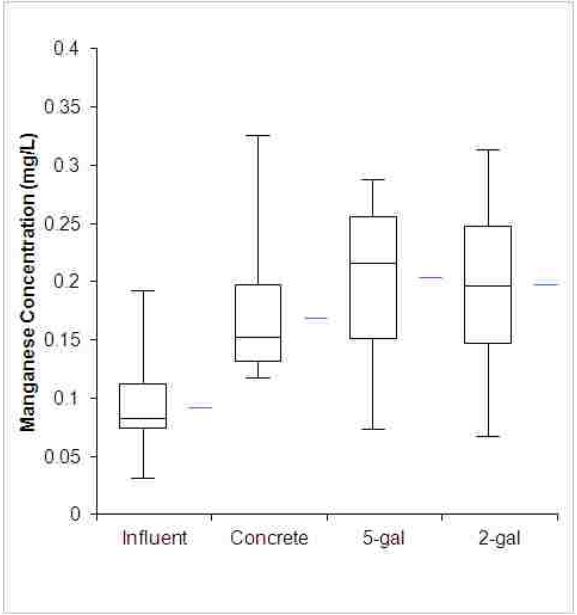

Figure 14. Manganese concentration (mg/L) for the influent and filter effluents.

3.3.2 Hydraulic Loading Rates

The HLR of all filters decreased over time as expected and attributed to fouling of the filter. The HLRs of the concrete filters decreased more rapidly than for the other filters. The HLRs of the concrete filters had reduced to half the initial starting value after approximately 30 days of operation, whereas the HLRs of the bucket filters were ≥75% of their initial values (**Figure 15.a**). After the first cleaning (test day 30), the concrete filters diverged into two groups ($p<0.001$) based on HLR; both sets of concrete filters continued to exhibit the same trend of decreasing HLRs until cleaned, at which time the HLRs increased in magnitude by 0.08-0.1 $m^3/(m^2*hr)$. Around test day 207, the two groups reconverged, and all four concrete filters operated at approximately the same HLR until the end of the study (test day 273). The cause for this divergence and reconvergence was not identified. In effort to minimize bias, cleaning was performed by the same

technician and filter order was alternated; however, cleaning technique and user influence could still be a factor in the phenomenon.

The HLRs of the 5- and 2-gal bucket filters were statistically similarly with respect to HLR (p=0.1172 and p=0.4807, respectively) for the entire length of the study. Furthermore, a pair-wise comparison of the HLRs for all filters grouped all the bucket-sized filters and two of the concrete filters; the two concrete filters that had slower HLRs for the majority of the testing (**Figure 16**) were significantly different from the other ten filters ($p \leq 0.001$).

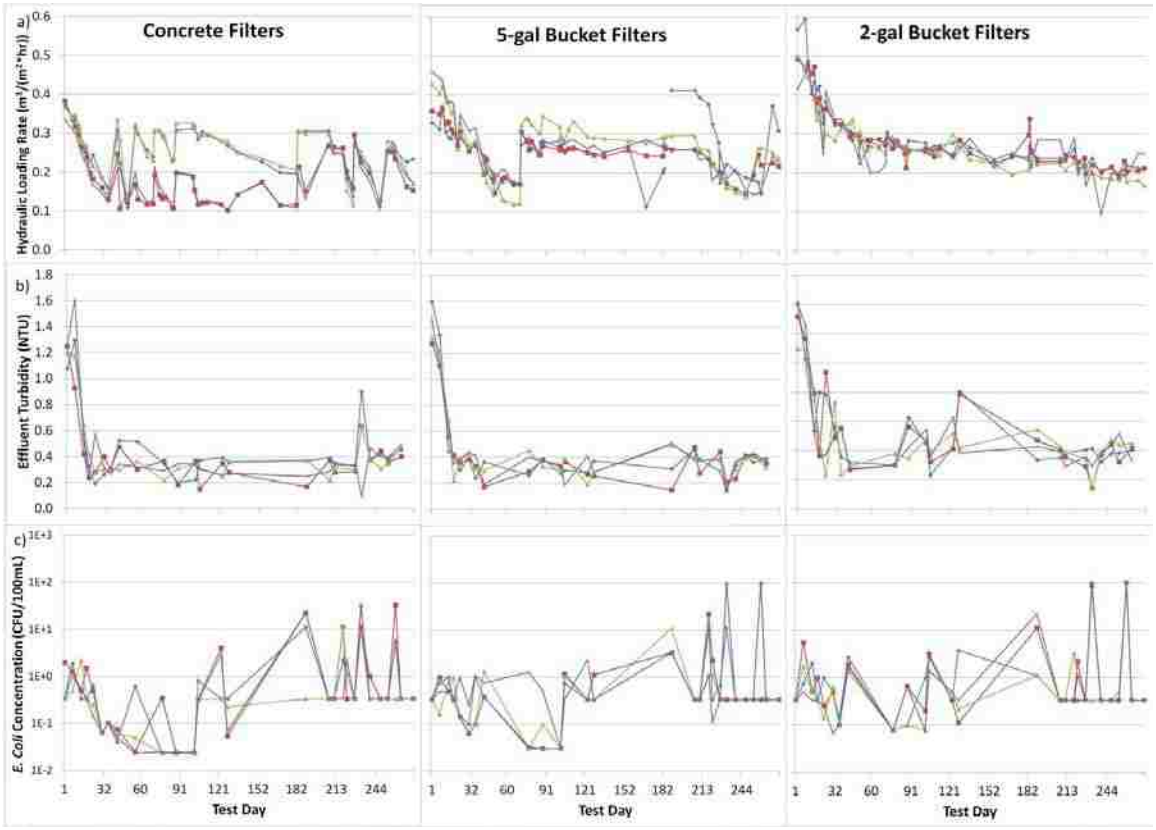

Figure 15. Filter performance by test day for a) hydraulic loading rate ($m^3/(m^2*hr)$), b) effluent turbidity level (NTU), and c) effluent *E. coli* concentration (CFU/100mL).

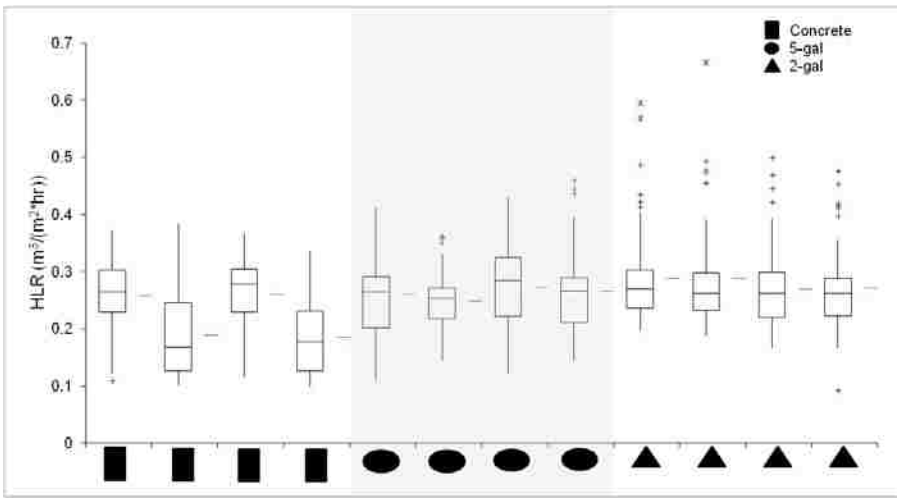

Figure 16. Hydraulic loading rates ($m^3/(m^2*hr)$) for each test filter.

3.3.3 Contaminant Removal Levels

Removal capabilities were similar for replicate filters (i.e., four filters of the same size) and for filters of different sizes (i.e., concrete, 5-gal, 2-gal). **Figure 17** displays the boxplots for a) $\log_{10}$ *E. coli* removal and b) percent turbidity removal for each filter and offers a visual summary of the similarity in the mean, median and distributions of the removal capabilities for each filter. **Table 3** provides the median removals and resultant p-values for comparison of replicate filters and all twelve filters. The geometric mean of the removals for the concrete, 5-gal, and 2-gal filters were 3.66, 3.59, and 3.34 for *E. coli* removal ($\log_{10}$) and 97.4, 97.4, and 96.8 percent for turbidity removal.

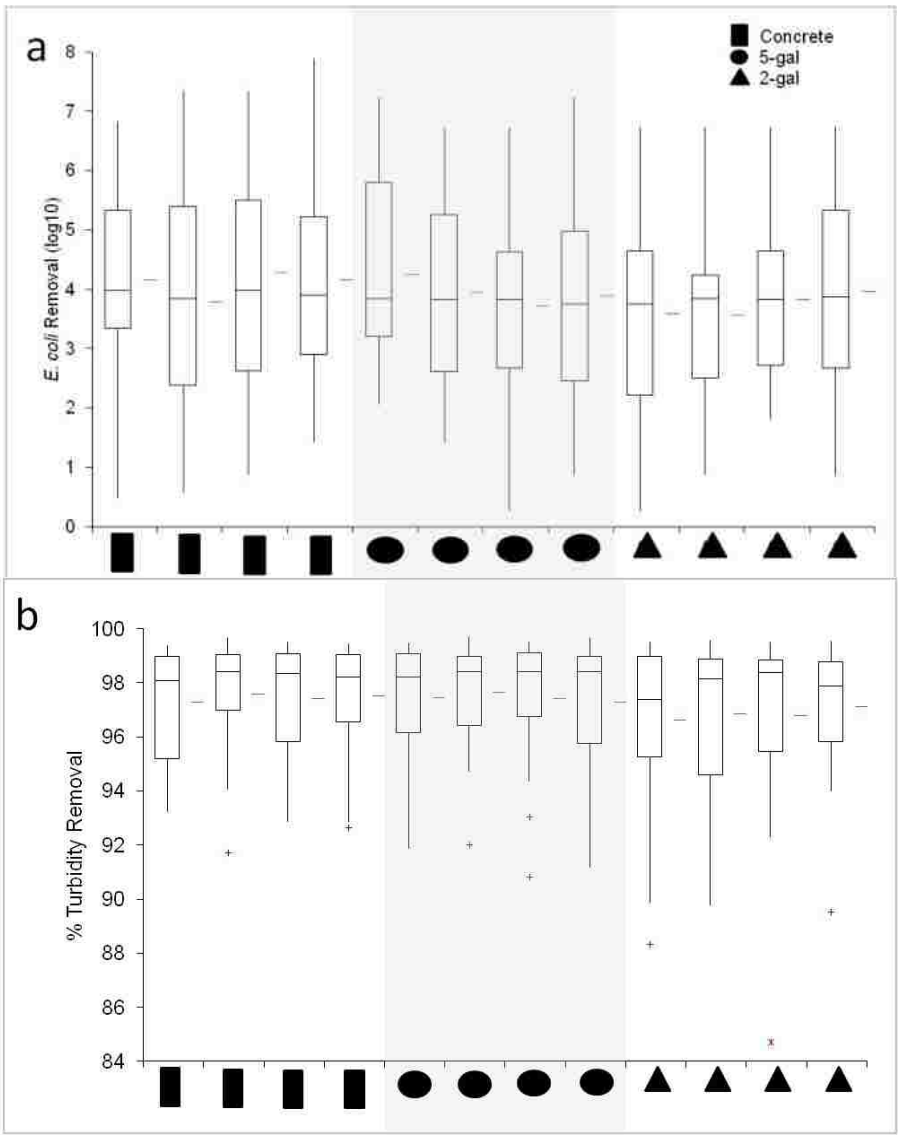

Figure 17. Performance of individual test filters for a) *E. coli* removal (log_{10}) and b) percent turbidity removal.

Table 3. Median *E. coli* and turbidity removals for each test filter for all test days with p-values for comparison of 1) replicate filters and 2) all twelve filters.

	E. coli Removal (log10)					% Turbidity Removal				
	Replicate Filters					Replicate Filters				
Filter Size	1	2	3	4	p-value	1	2	3	4	p-value
Concrete	4.0	3.8	4.0	3.9	0.9745	98.1	98.4	98.3	98.2	0.9651
5-gal	3.8	3.8	3.8	3.7	0.9620	98.2	98.4	98.4	98.4	0.9869
2-gal	3.7	3.8	3.8	3.9	0.9082	97.4	98.2	98.4	97.9	0.9943
All 12 filters					0.9758					0.9927

3.3.4 Effluent Levels

The effluent *E. coli* concentration for each filter size were evaluated with respect to the the WHO risk classifications; the resultant categories (i.e., size and classification) were subdivided by the total age of the filter (i.e., test day) (**Figure 18)**. For the concrete filters, all (100%) effluent waters tested were below the high risk classification for *E. coli*. For all three filter sizes, over ninety percent of all effluents tested were either within conformance of WHO guidelines or were low risk (<10 CFU/100 mL); specifically, the percentage of effluent samples within conformance (<1 CFU/100mL) were 78%, 74%, and 72% for the concrete, 5-gal, and 2-gal filters, respectively. During the first 30 days of use, all filters produced water either in conformance with the WHO guidelines of <1 CFU/100mL or was low risk at 1 - <10 CFU/100mL (**Figures 18 and 15c**).

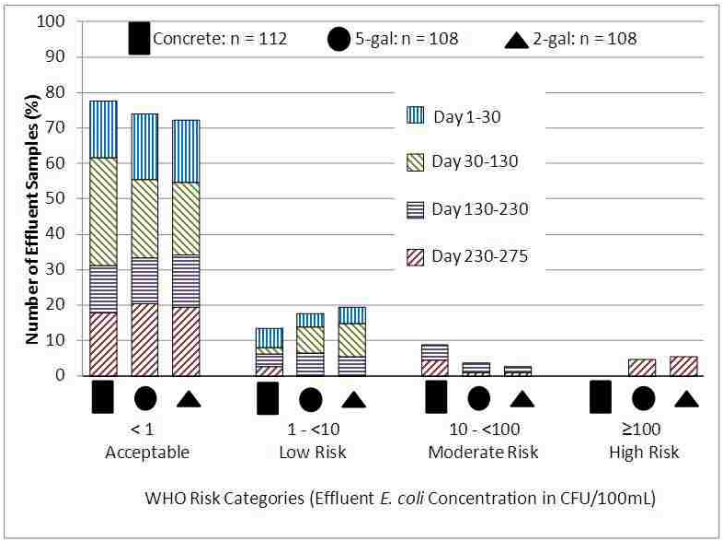

Figure 18. Percentage of effluent samples classified by WHO risk category for *E. coli* concentration (CFU/100mL) level. The four hazard classifications are i) <1: conformity with WHO guidelines, ii) 1 - <10: low risk, ii) 10 - <100: moderate risk, and iv) ≥100: high risk. The sample size (n) for each filter type was 112, 108 and 108 for the concrete, 5-gal, and 2-gal filters, respectively.

As shown in **Figure 18**, 4.6% (5/108) and 5.6% (6/108) of sample effluents for the 5- and 2-gal bucket filters, respectively, were classified as high risk with concentrations ≥ 100 CFU/ 100ml. All of these samples were from two test days, day 232 and day 259, when the *E. coli* concentrations were at the higher range of spiked influent values at 2.6E3 and 1.1E4 CFU/100mL, respectively. The influent (spiked *E. coli* plus creek water) averaged 2.8E3 CFU/100mL with a standard deviation of ±5E3 and ranged from 1.9E0 – 1.8E4 CFU/100mL (min – max). For test day 232, both the 5-gal and 2-gal filter effluents yielded similar results: two replicate filters produced effluents >100CFU/100mL, one was 10-100 CFU/100mL, and one was <1 CFU/ 100mL. In comparison on test day 259, three of the 5-gal filters were >100 CFU/ 100mL and one was <1 CFU/100mL; and all the 2-gal filter effluents were >100CFU/100mL. These high effluent concentrations did not indicate a breakthrough of bacteria as there were three additional test days between day 232 and day 259 where all four replicates of the 5-gal and 2-gal filters produced water <1 CFU/100mL. In addition, on the last two test days, 263 and 273, all effluents from the bucket-sized filters were <1CFU/100mL.

All twelve filters produced waters with similar turbidity levels (**Figure 19**, p=0.0724) and all effluent samples were less than 2NTU over the entire study period (**Figure 15b**). Furthermore, comparison of all filter turbidities by test day (**Figure 15b**) identified that all effluents were consistently below 1NTU after the first two weeks of operation (test day >14). **Table 4** lists the minimum, maximum, average and standard deviation of turbidity values for influent water and effluents from the filters. **Figure 19** shows representative samples of the influent and the filter effluents.

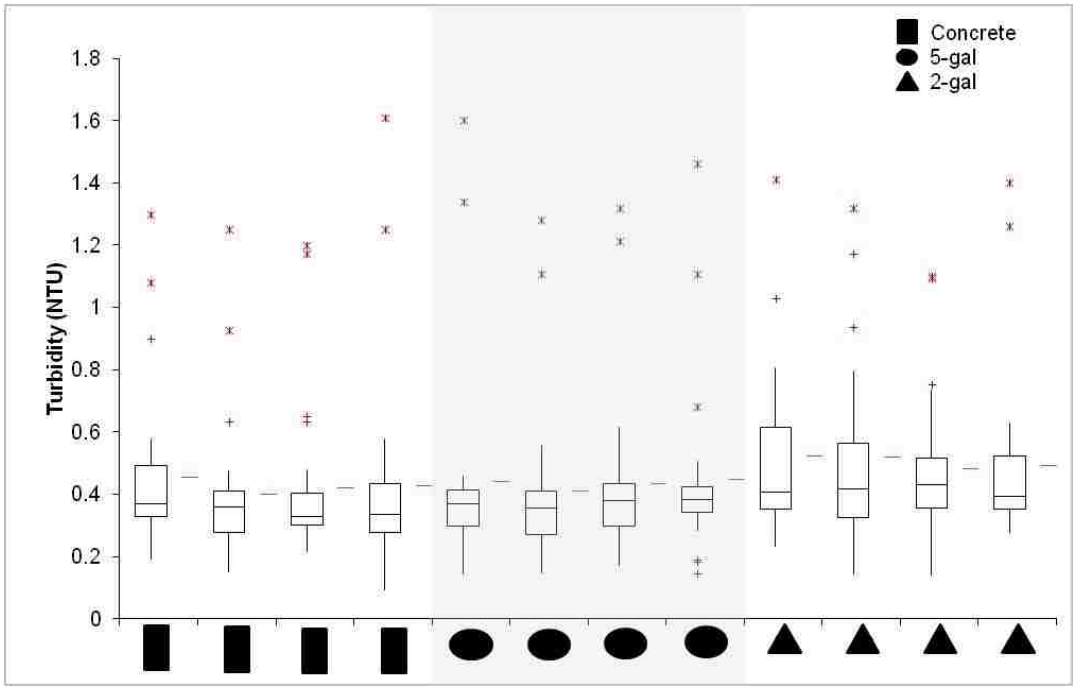

Figure 10. Effluent turbidity (NTU) for each test filter.

Table 4. Minimum, maximum, and average turbidity (NTU) for the influent and the effluents for each filter type.

	Min	Max	Avg	Std Dev
Influent	4.82	61.37	30.17	18.40
Concrete	0.15	1.61	0.43	0.29
5gal buckets	0.15	1.60	0.46	0.31
2gal buckets	0.23	1.41	0.53	0.28

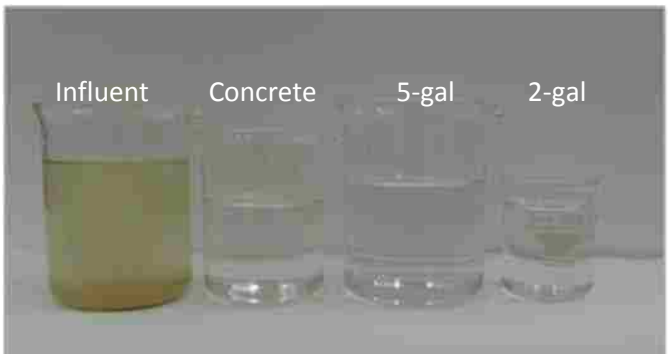

Figure 19. (from left to right): Influent turbidity of 48.5 NTU compared with effluents from the concrete, 5-gal and 2-gal filters, at 0.35, 0.38, and 0.36 NTU, respectively.

3.3.5 Bivariate Analyses

The data were analyzed to identify whether any correlations exist between variables, e.g., influence of influent turbidity on resulting effluent turbidity; influence of effluent turbidity on either *E. coli* concentration in the effluent or *E. coli* $\log_{10}$ removal; and influence of operating variables (e.g., HLR, total age, schmutzdecke age, and sand depth) on either removal, effluent level, and/or HLR. For example, the influent turbidity was compared to the corresponding effluent turbidities (**Figure 20**); however, no correlation was identified (p=0.0301, r=0.13). **Table 5** summaries all the parameters evaluated, identifying the variables, p-value, and Pearson correlation coefficient for each analysis.

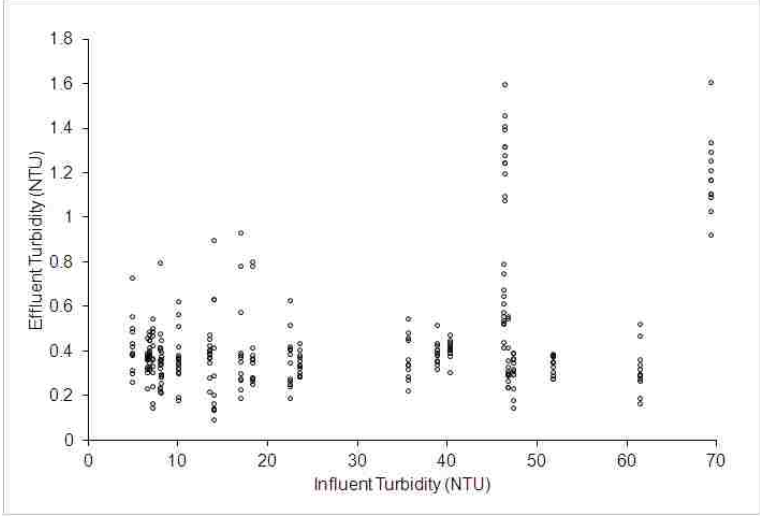

Figure 20. Influent turbidity vs. effluent turbidty levels (NTU) for all filters.

Table 5. Pearson correlation parameters used to identify significance of design and operating parameters on performance: effluent level, contaminant removal, and hydraulic loading rate (HLR). Bold values indicate a significant, strong correlation defined as p-value<0.05 and |r|≥0.4.

	Effluent *E. coli* Conc (CFU/100mL)		*E. coli* Removal (LRV)	
	p-value	**r**	**p-value**	**r**
Effluent Turbidity (NTU)	0.9140	0.01	0.1010	-0.16
% Turbidity Removal	0.4211	0.08	0.0117	0.15

	Effluent Turbidity (NTU)		Turbidity Removal (%)	
	p-value	**r**	**p-value**	**r**
InfluentTurbidity (NTU)	0.0301	0.13	**<0.0001**	**0.67**
HLR (m3/m2/hr)	**<0.0001**	**0.61**	0.1757	-0.08
Schmutzdecke age (days)	0.0108	-0.16	0.0540	0.12
Total age (days)	<0.0001	-0.26	0.0512	0.12
Sand depth (cm)	0.0006	-0.21	0.2032	0.08

	Effluent *E. coli* Conc (CFU/100mL)		*E. coli* Removal (LRV)	
	p-value	**r**	**p-value**	**r**
Influent *E. coli* Conc (CFU/100mL)	0.0190	0.13	**<0.0001**	**0.54**
HLR (m3/m2/hr)	0.4807	0.04	0.4006	-0.05
Schmutzdecke age (days)	0.6169	0.03	0.4571	0.05
Total age (days)	0.3755	-0.05	0.7426	0.02
Sand depth (cm)	0.0174	-0.13	0.0934	0.10

	HLR (m3/m2/hr)	
	p-value	**r**
Schmutzdecke age (days)	0.0138	-0.15
Total age (days)	**<0.0001**	**-0.41**
Sand depth (cm)	<0.0001	-0.31

As previously discussed, individual filters exhibited a large range of removal values for both *E. coli* and turbidity (**Figure 15**). *E. coli* $\log_{10}$ reduction values (LRVs) ranged from less than 1 to greater than 7. Turbidity removal also yielded a large variance with minimum and maximum percent removals at 84.7 and 99.7, respectively. The large

variance in contaminant removal was attributed to the fact that removal rates (**EQN 2 and 3**) are dependent on the influent concentrations (which themselves were quite variable): percent removal is calculated as the ratios of particles captured within the filter to those entering it, so lower influent concentrations mean there are fewer particles which can be captured within the filter and generally result in lower removal values.

$$Percent\ removal = \frac{C_{inf} - C_{eff}}{C_{inf}} * 100 \qquad \textbf{(EQN 2)}$$

$$log_{10}Removal = -log_{10}(\frac{C_{eff}}{C_{inf}}) \qquad \textbf{(EQN 3)}$$

where, C_{inf} = concentration or turbidity of the influent (CFU/100mL or NTU)

C_{eff} = concentration or turbidity of the effluent (CFU/100mL or NTU)

The Pearson coefficients of correlation (**Table 5**) suggest there is a moderately strong linear relationship between the removal value (log removal for *E. coli* and % removal for turbidity) and the influent level for *E. coli* concentration and turbidity ($p<0.001$ for both; $r = 0.54$ and 0.67 for *E. coli* and turbidity, respectively) . The scatter plots of these data sets are depicted in **Figure 21**. Transformation of both datasets into logarithms yield coefficients of determination of 0.6: i.e., approximately 60% of the total variation in removal can be explained by the log-linear relationship between influent level and removal. The upper points of **Figure 21b** indicate the maximum removal that can be reported based on the influent concentration and the detection limit for each test day. For example, an influent concentration of 7.4 CFU/100ml and a detection limit of

0.1 CFU/100ml (for a processed sample volume of 1000ml) resulted in a maximum removal of 1.9 (where, removal = - $\log_{10}(0.1/7.4)$).

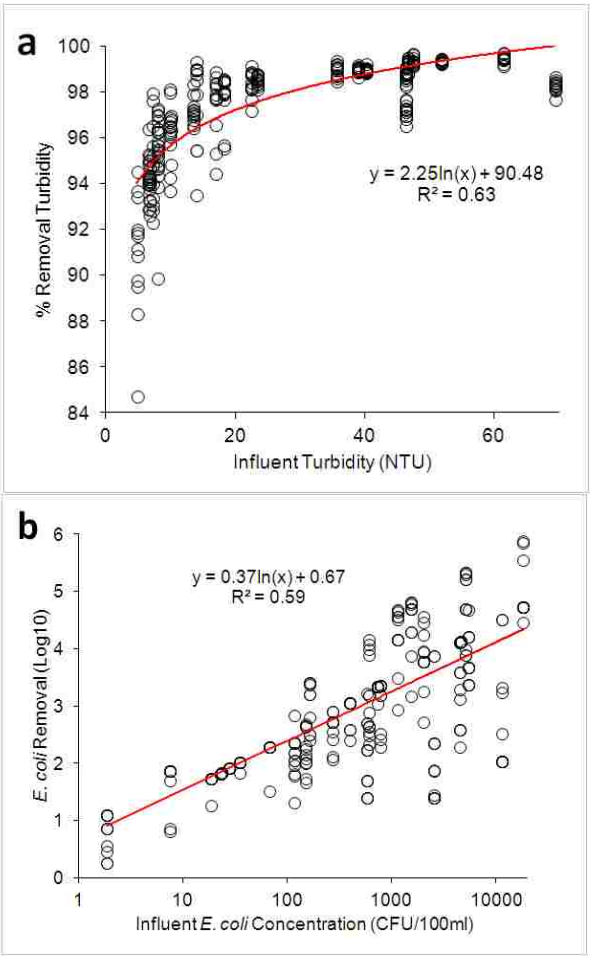

Figure 21. Contaminant removal as a function of influent level for a) turbidity (n=280) and b) *E. coli* (n=328), with trendline equations and coefficients of determination (R^2).

Because the influent levels showed a correlation to removal levels, the effect of the variability in the influent turbidity on *E. coli* removal levels was evaluated. Each *E. coli* removal data point was plotted against the number of days the influent had been at that turbidity level, either high (approximately 50 NTU) or low (approximately 5 NTU). The data were evaluated for all filters and each filter type separately (**Figure 22**). For all

filter types, the relationship was stronger for the first half of the study (test day ≤140 days); however, no significant correlation was identified.

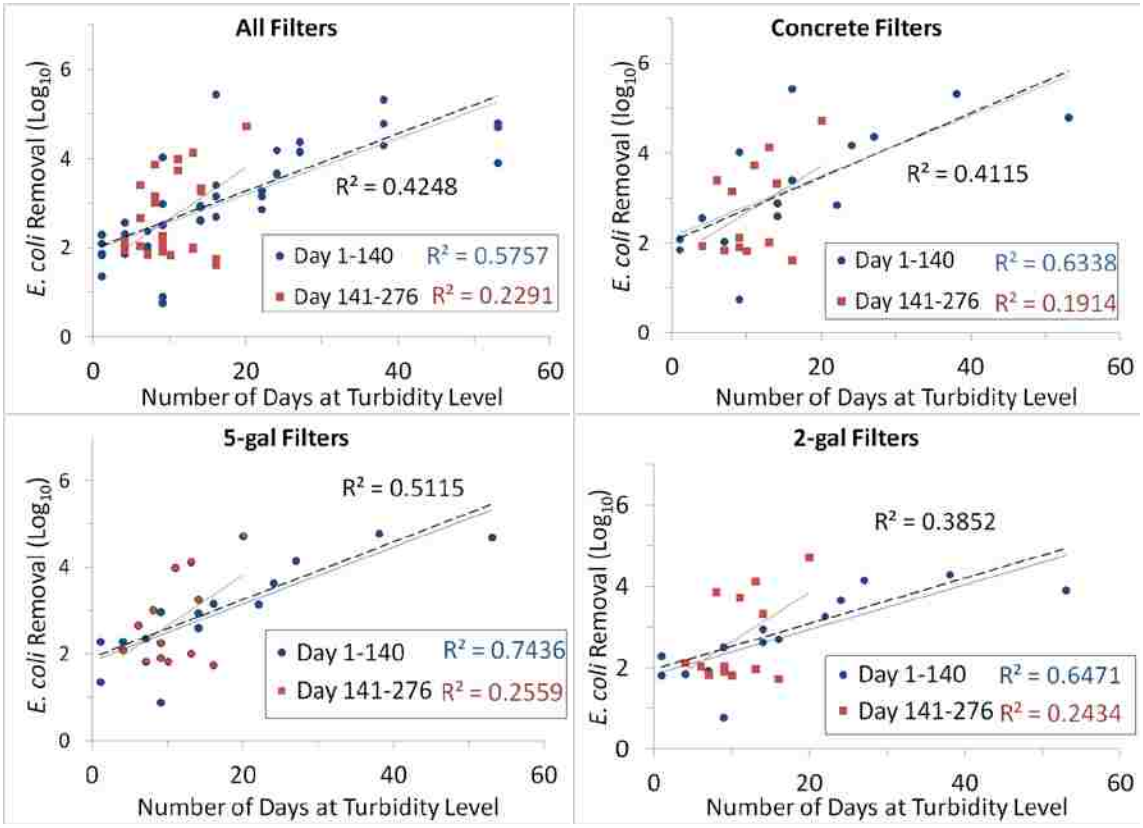

Figure 22. *E. coli* removal as a function of time at turbidity level for all filters and fore each filter type: concrete, 5-gal bucket, and 2-gal bucket filters. Blue circles represent early test days (1-129), red squares are for later test days (201-276), and the black trendline and correlation constant are for the complete data set.

Additional Pearson correlation tests (**Table 5**) were performed to in an attempt to account for the additional 40% variance in removal performance (**Figure 21**). As presented in **Table 5** and **Figure 23**, effluent turbidity exhibited a significant and strong positive correlation to filter HLR ($p<0.0001$ and $r=0.61$). However, further evaluation of the data showed that beyond the first two weeks of operation (removing all data points >1 NTU), the strength of correlation diminished ($p <0.0001$, $r = 0.25$, and $R^2 = -0.068$).

Evaluation of *E. coli* log reduction values (LRVs) and percent removal of turbidity as a function of HLR yielded no significant correlations (**Table 5**).

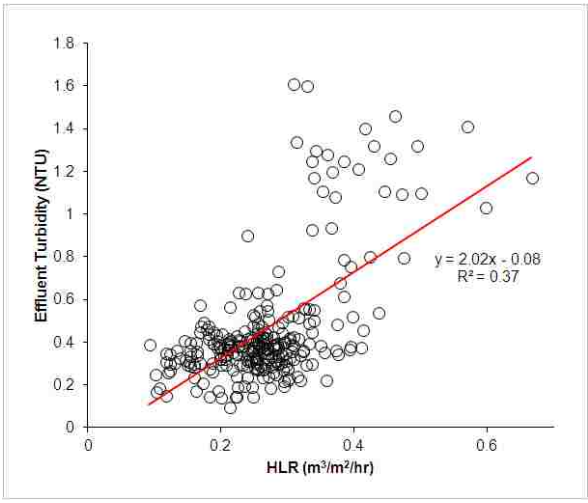

Figure 23. Effluent turbidity (NTU) as a function of hydraulic loading rate (HLR) ($m^3/(m^2*hr)$) (n=268).

As cleaning was performed as needed (determined by each filter's HLR), microbial challenge experiments were performed on the filters over a range of schmutzdecke ages, or at varying times since the most recent cleaning. **Table 6** identifies the number of microbial challenge experiments performed for each filter size by schmutzdecke age; Schmutzdecke age is defined as the time since the filter was last cleaned (not the time since the filter was first installed). For data evaluation the schmutzdecke age was grouped into four categories: 1) one week (1-8 days), 2) two weeks (9-19 days), 3) three to four weeks (20-32 days) and 4) more than four weeks (>32days). There was no noticeable increase in the effluent bacteria concentration and no noticeable reduction in the removal rate (i.e., either percent removal or LRVs) for filters that had been cleaned recently (within one week) versus several weeks. The data showed

a significant but weak (p=0.0138 and r=-0.15) negative correlation between the age of the schmutzdecke and the filter's HLR (see **Table 5, Figure 24**).

Table 6. Number of Microbial Challenge Experiments Performed for Various Schmutzdecke Age Groups

	Schmutzdecke Age			
	1 Week	2 Weeks	3-4 Weeks	> 4 Weeks
Days since last cleaning	(1-8 days)	(9-19 days)	(20-32 days)	(>32 days)
concrete	11	5	7	5
5-gal	7	6	6	8
2-gal	7	6	6	8

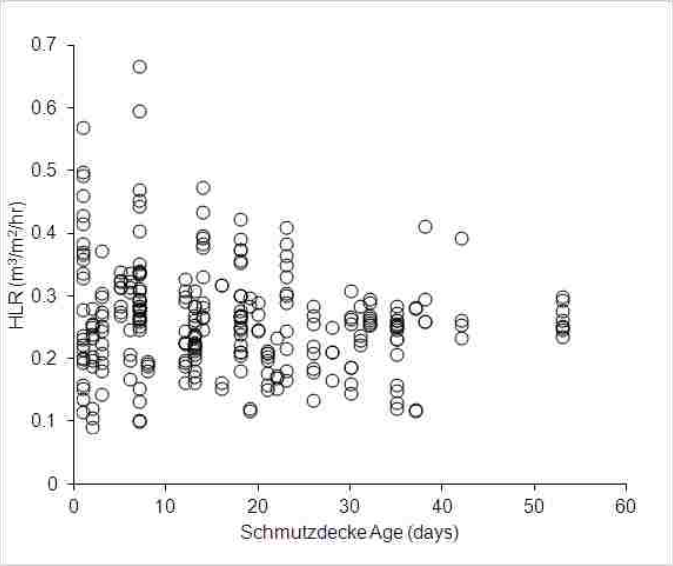

Figure 24. Hydraulic loading rate (HLR, $m^3/(m^2*hr)$) as a function of schmutzdecke age. Schmutzdecke age is defined as days since the last cleaning; relationship presents a significant (p=0.0138) but weak correlation (r=-0.15) to HLR.

3.4 Discussion

Some BSFs in the field have been in use for several years, yet most laboratory tests are conducted in a relatively short time frame and relatively little is known about long-term performance. Recent work investigating the long-term performance of virus removal in full-scale BSFs suggests that virus removal is enhanced as the filter media ages, or the total age of the filter increases (Bradley et al. 2011; Elliot et al. 2011); however, we are unaware of any other long-term study of bacteria and turbidity removal performance of full-scale filters conducted under laboratory conditions.

Influent and effluent waters were monitored for the macro nutrients (i.e., carbon, nitrogen, and phosphorus), as well as for pH, alkalinity and hardness. Testing confirmed the presence of the macro nutrients in the influent at levels that could support biofilm development in the filters (summary data presented in **Table 7**Carbon (as total organic carbon) was present at levels comparable to other laboratory BSF studies (Elliot et al. 2008 & 2011; Palmateer et al. 1999; Duke et al. 2009). Nitrogen and phosphorus concentrations were well above the limiting range of 0.1 – 0.3 mg/L (Metcalf & Eddy 2003). No extreme changes in any of water quality parameters were observed; the small fluctuations in the water quality parameters did not correlate with any reductions in bacteria removal. The high levels of nutrients, in particular nitrogen and phosphorus, in the influent could have enhanced the biological activity in the filter and been a factor in the high bacterial removal levels.

Table 7. Summary of Water Quality Values in the Influent Water

	Avg ± Stdev	Range (Min - Max)
Total Organic Carbon (mg/L)	12.5 ± 6.8	5.8 - 24.2
Total Nitrogen (mg/L)	6 ± 4.8	2 - 14
Phosphorus (mg/L PO_4^{3-})	0.21 ± 0.15	0.07 - 0.47
pH	7.5 ± 0.4	7.1 - 8.8
Alkalinity (mg/L)	41 ± 9.8	30 - 58
Hardness (mg/L)	339 ± 77.3	247 - 492

As shown in **Table 5**, there was no correlation between total age and *E. coli*, either $\log_{10}$ removal or effluent concentration. However, a weak negative correlation (r=-0.26) was observed between effluent turbidity and total age ($p<0.0001$). The HLR did show a significant and strong negative correlation with the total age of the filter ($p<0.0001$ and r=-0.41). This correlation exhibited the largest coefficient of determination when a logarithmic transformation of the data (i.e., logarithmic trendline) was performed (**Figure 25)** and exemplifies that, over time, the filter HLR will decrease and eventually level off.

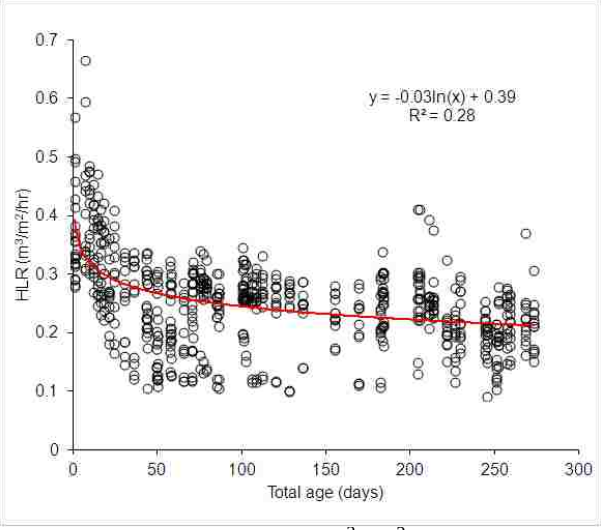

Figure 25. Hydraulic loading rate (HLR) ($m^3/(m^2*hr)$) as a function of total age of filter (days) (n=659).

As the majority of the larger particles are removed at the top of the filter by the schmutzdecke, cleaning the filters should restore the HLRs close to the initial values. When filters were cleaned, HLRs increased as expected and then subsequently decreased over time as turbid influent water continued to be charged to the filters. The different sized filters have different charge volumes but approximately the same sand surface area (0.059, 0.059, and 0.039 m^2 for the concrete, 5-gal, and 2-gal filters, respectively); thus, even with the same influent turbidity level and the same number of daily filter charges, the concrete filters were exposed to a greater daily load of particles. The flow rates of the concrete filters were observed to decrease faster, and as a result, the concrete filters required more frequent cleaning than the bucket sized filters. If the smaller filters were filled more frequently to obtain the same volume of treated water in a given day, it is likely that the flow rate reductions and requisite cleaning schedules would be more consistent with the concrete filters. It is important to note that even after the filters were cleaned (schmutzdecke age effectively reset to zero), all filter types continued to produce water with turbidity <1 NTU (**Figure 15b**). Over the course of the testing, 28 microbial challenge experiments were performed on the filters and neither cleaning schedule (i.e., schmutzdecke age) nor fluctuations in HLRs negatively impacted *E. coli* or turbidity removal (**Figures 26 and 27)**. This data confirms that to improve flow rate the filters should be cleaned by agitating the top layer of the schmutzdecke and decanting off the dirty supernatant.

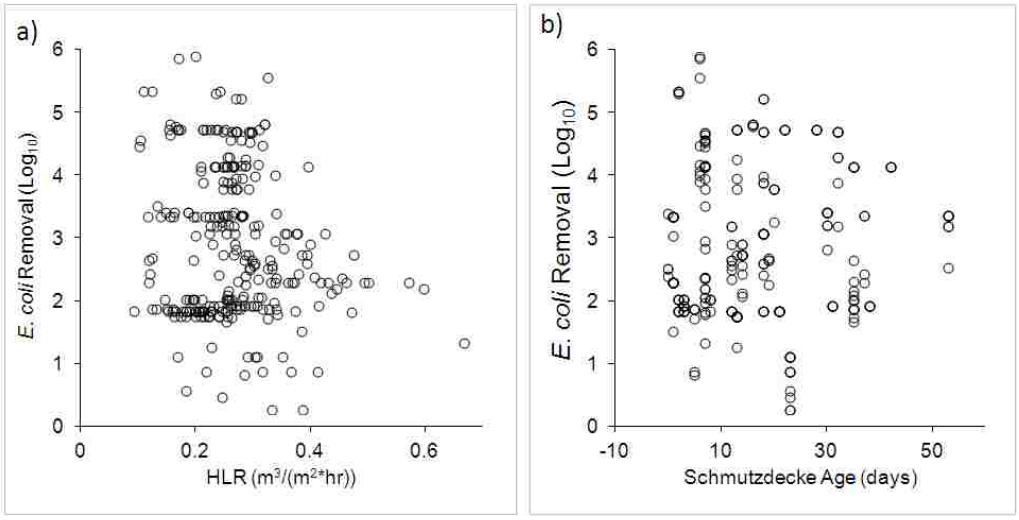

Figure 26. *E. coli* removal (Log10) as a function of a) hydraulic loading rate (HLR) ($m^3/(m^2*hr)$) (n=268) and b) schmutzdecke age (days) (n=268).

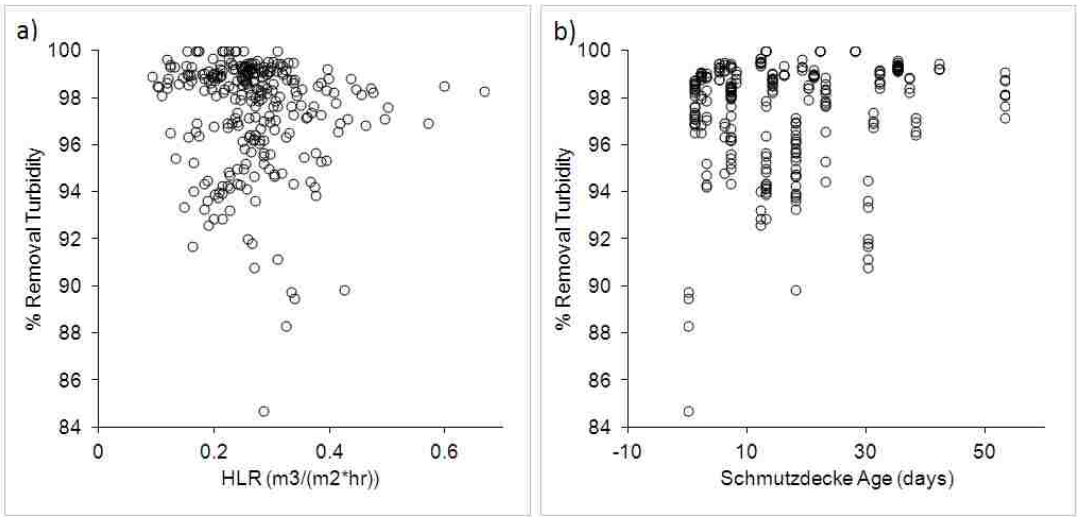

Figure 27. Percent Turbidity removal as a function of a) hydraulic loading rate (HLR) ($m^3/(m^2*hr)$) (n=268) and b) schmutzdecke age (days) (n=268).

The results from this study showed that the filters can be scaled down to yield similar HLRs and removal rates for *E. coli* and turbidity as compared to larger versions. With comparable surface areas of the sand layer across the filter sizes, reducing the sand bed depth and maintaining the reservoir volume to equal the pore space of the sand bed yielded similar HLRs for various filter sizes. All eight bucket-sized and two of the concrete filters had HLRs in the range of 0.2-0.3 $m^3/(m^2*hr)$ for the majority of the testing. Two of the concrete filters actually exhibited significantly slower HLRs for approximately six months of the testing, highlighting the variability of performance in the filters, as these filters were charged with the same waters and cleaned by the same technician.

The smaller sand bed depths in the bucket-sized filters did not impact filter performance with respect to turbidity and *E. coli* removal (**Figure 17**) nor the effluent levels of turbidity or *E.coli* (**Figure 15**). These findings are in agreement with previous research that has shown that sand bed depth did not impact the removal of coliforms (Bellamy 1985; Buzunis 1995). Bellamy proposed that sand bed depths could be reduced to ~40cm with no change in bacteria removal performance. Date presented here showed that a sand bed depth of 10cm was adequate for removal of turbidity and *E. coli*.

In general, other research on slow sand filters and microbial transport through saturated porous media has shown that it is not just sand depth but sand size distributions and flow rates that can influence removal (ASCE 1991; Hyusman & Verstraete 1993; Johnson & Logan 1996; Hermansson 1999; Hijnen et al. 2004). This study demonstrated that while the sand bed depth was reduced, reducing the overall adsorption capacity of

the filter, controlling the sand size distribution and HLR effectively provided the same level of removal efficiency as filters with deeper sand beds.

WHO guidelines state that drinking water should be less than or equal to 5 NTU, as turbidity in excess of 5 NTU may be unacceptable to users; furthermore, it is recommended that water be less than 1 NTU for chlorination to be effective (WHO 1997). All filter sizes tested in this study were in conformance with the WHO guidelines for turbidity, as all effluents produced a mean turbidity of <0.6 NTU. The mean turbidity level of the concrete filters was 0.4 NTU, which is consistent with past reports that concrete filters (CAWST version 9) routinely produced waters <1 NTU (Buzunis 1995; Duke et al. 2006; Elliott et al. 2008). Past reports have attributed lower filtered water turbidity over time to (1) enhanced particle straining due to biolayer formation; (2) improved depth filtration by slowing the filtration rate; and/or (3) altered surface properties of the filtration media (Elliott et al. 2008).

The removal levels from this study were linearally correlated with the influent level for both *E. coli* and turbidity, with 60% of the variance accounted for by these relationships. While HLR did not show a direct influence on bacteria or turbidity removal (p-values of 0.4006 and 0.1757, respectively; **Table 5**), effluent turbidity was correlated to HLR, which in turn was slightly correlated to total age of the filter. These relationships were not identified for *E. coli*, for either effluent level or removal.

The lack of correlation between filter performance and schmutzdecke cleaning suggests that mechanisms of removal below the surface of the sand (depths greater than

1- 2cm) are responsible to a greater extent for contaminant removal than the very top layer of the schmutzdecke that gets removed during cleaning. The data show that regular cleanings do not interfere with filter performance for sand bed depths ranging from 10 to 54 cm. These data suggest that, in addition to straining, biological activity and adsorption are important removal mechanisms at work within at least the top 10cm of the sand bed (because the smallest filters, with a bed depth of 10 cm, performed as well as the larger filters). A previous study showed that a supernatant layer of 12.5 cm supported a biologically active zone to a depth of 10 cm within the sand bed (Buzunis 1995). With supernatant depths of 4 cm and 3 cm for the 5- and 2-gal bucket filters, respectively, it is possible that the entire sand beds were biologically active, thus accounting for high bacteria removals even after schmutzdecke cleaning.

A three-to-four week ripening phase, reportedly required for the development of the biologically active region (CAWST 2012a; Elliot et al 2008), was not necessary during this study to achieve high bacterial removal as all effluent *E. coli* levels were consistently <10 CFU/100 ml during the first 30days of testing (**Figure 15**). The development and maturation of a biofilm layer will be dependent on the quality of the source water, including nutrient levels as well as types of colonizing microorganisms. The filters in this study did not need a long maturation period in order to effectively remove bacteria during the initial start-up phase. Varying the turbidity of the influent water between ~5 NTU and ~50 NTU did not induce any significant change in removal performance of either *E. coli* or turbidity.

3.5 Conclusions

Based on the data collected in this study, the CAWST v10 concrete filter was able to achieve 98.1 – 98.4% turbidity removal and 3.8 – 4.0 $\log_{10}$ *E. coli* removal. Scaled-down BSFs, constructed in 5-gal (15cm bed depth) and 2-gal (10cm bed depth) buckets, were shown to be as effective (p-values >0.05) as the CAWST v10 concrete (54cm bed depth) configuration for both turbidity and *E. coli* removal. Alternating the influent turbidity between periods of high and low turbidity (~50 and ~5 NTU, respectively) did not influence either turbidity removal or *E. coli* removal. Periodic filter maintenance (i.e., cleaning the top of the sand bed) exhibited no correlation to either removal values or effluent levels of either *E. coli* or turbidity ($p>0.05$ and $|r|<0.4$).

While providing similar water quality with respect to turbidity and *E. coli* as shown herein, it is important to identify that filters with shorter sand bed depths may result in a reduced removal capability for other constituents, such as nitrogenous compounds (Muhammad et al. 1996) and smaller sized microorganisms, such as viruses. In addition, this study evaluated the performance of the filters using the same number of charges (or fills) per day and the same pause period between charges. However, the smaller charge volume of the bucket-sized filters means that to meet the drinking water needs of a household, the smaller filters will likely need to be charged more often than the concrete filter, effectively reducing the pause period for the bucket filters. The effect of a shorter pause period on the performance of the smaller, bucket-sized filters needs to be investigated. However, comparison of the filters based on the same number of fills per

day shows that the smaller bucket-sized filters are a viable alternative to the concrete BSFs for the removal of bacteria and turbidity from drinking water.

4.0 Transport Effects on Hydraulic Loading Rate and Microbial Removal Performance

BSFs designed using smaller and/or lighter casing material can result in reduced logistical requirements and implementation costs. However, the increased portability of a smaller, lighter design presents a potential negative consequence: the ability to move the installed/operational filter by the homeowner and potentially disturb the system. This study investigated the effects of moving and agitation on filter performance, using mature BSFs which had been in use for over nine months prior to the move. Data were analyzed for four replicate filters of three different filter types: the traditional concrete BSF and two plastic bucket (5-gal and 2-gal, respectively) BSFs.

Filters were moved approximately 1 km and monitored for hydraulic loading rates (HLRs) and *E. coli* removal for eight weeks following the move. Moving the filters resulted in reduced HLRs, likely due to sand compaction, but *E. coli* removal remained high ($\log_{10}$ removal ≥ 2.8 for all sizes) and increased significantly as compared to data collected prior to the move. The resulting operational implications of moving BSFs are discussed.

4.1 Introduction

BSFs have been deployed in over 70 countries since the early 1990s (Manz et al. 1993), providing improved drinking water for rural populations without access to public treatment systems. Many BSF projects target communities in remote, rural areas with

limited or no improved roadways. In light of the logistical and economic challenges associated with transporting the traditional concrete casing and filter media (rock and sand) to these remote communities, some implementing organizations use plastic casing materials. In some cases, the dimensions of the casing are also reduced, subsequently reducing the requisite volumes of filter media that need to be transported.

The increased portability of a smaller, lighter BSF design presents a potential negative consequence: the ability to move the installed/operational filter by the homeowner and potentially disturb the system. Typically, implementing organizations recommend that installed filters should not be moved and that the sand and rock should be removed and replaced after relocation to minimize negative impacts on filter performance (CAWST 2012). With traditional concrete filters, relocation is typically not an issue since installed filters, with rock, sand, and water, weigh several hundred pounds. However, it is reasonable to anticipate that users may move installed filters over shorter distances (e.g., within a household or from one house to another), especially for either the smaller bucket-sized or plastic-cased filters. Transport of a full-sized, plastic-cased BSF (PVC pipe casing with 12 L reservoir volume) over an approximate distance of 200 ft (across the street) was observed during a field study in Nicaragua (**Figure 28**). Three additional PVC filters from this same field study were observed to be moved over larger distances ranging from 0.2 km to 1.2km. The increased potential for filter transport following installation, and the subsequent effects on performance, are potential concerns associated with either changing the filter casing material and/or reducing the overall filter

size. Until now, however, there was no supporting evidence of the effect of filter transport on performance.

Figure 28. Installed filter being transported across the street from one household to another.

This study investigated the effects of moving and agitation on filter performance. Following a nine-month contaminant removal study on twelve full-scale BSFs (four each of three different types: traditional concrete, 5-gal plastic bucket, and 2-gal plastic bucket), the filters were moved approximately 1 km to a new laboratory location. Although the moving distance was short, the size and weight of the filters required the use of hand carts and a moving truck. All efforts were made to minimize tilting and disruption of the filters, but some jostling could not be avoided. For eight weeks

following the move, the filters were monitored for hydraulic loading rates (HLRs) and *E. coli* removal.

4.2 Methods

4.2.1 Experimental Approach

Filter performance was monitored for an eight-week period following transport of the filters approximately 1 km to a new laboratory location. Filters were flushed ten times prior to testing. Four *E. coli* challenge experiments were performed, and HLRs were monitored weekly to identify filters that required cleaning (HLRs were also recorded after each cleaning). Results of the *E. coli* challenge experiments and the HLRs were compared to previous results obtained in a 9-month study conducted on the same filters at the original laboratory location. For all filters, the sand bed pore volumes equaled the filter charge volumes (and influent reservoir volumes) and were 12L for the concrete BSF, 3.6L for the 5-gal bucket BSF, and 1.5L for the 2-gal bucket BSF.

4.2.2 Bacterial Growth and Enumeration

Microbial analyses were performed in accordance with previously described procedures (Section 3.2.1). Samples, influent and effluents, were analyzed via membrane filtration for *E. coli* via Standard Method 9222 (Rice et al 2012). All samples (diluted or undiluted) were analyzed in triplicate. Following membrane filtration, membrane filters were placed in a culture dish that contained a sterile pad and 2 mL of m-ColiBlue 24® broth (Hach Company) and incubated at 35 ± 0.5°C for 24 hours. After the incubation period, membrane filters that yielded colonies with 10-100 colonies were considered

acceptable and counted. Concentrations were calculated according to the Standard Method 9222. If replicate filters of samples yielded acceptable colony counts, the resulting concentrations were averaged. For instances when all filters yielded zero colonies, the detection limit (1 CFU/total volume analyzed) was used as the effluent concentration for the subsequent calculation of removal efficiency (i.e., log reduction).

4.2.2 Hydraulic Loading Rate

Peak flow rates were measured at maximum hydraulic head and normalized to the peak HLR as previously described (Section 3.2.3). The pressure head was the same for each filter of the same type, specifically 18 cm, 6 cm, and 4 cm for the concrete, 5-gal bucket, and 2-gal bucket filters, respectively. Filters were filled with the same charge volume each time, i.e., 12 L, 3.6 L, and 1.5 L for the concrete, 5-gal bucket, and 2-gal bucket filters, respectively.

4.3 Results and Discussion

Prior to the move, the filters were filled three times per day for nine months, and the average post-cleaning HLRs dropped a total of 30.4%, 22.8%, and 18.0% over this time period for the concrete, 5-gal bucket, and 2-gal bucket filters, respectively (**Figure 29; Table 8**). These results show that the initial HLR of a newly installed filter is not regained even after cleaning; cleaning was performed 11, 10, and 9 times on the concrete, 5-gal bucket, and 2-gal bucket filters, respectively. It is reasonable to attribute the majority of this reduction in HLRs to the entrapment of particles within the pore spaces of the sand bed (i.e., filter clogging).

When a filter is installed, the sand is added to standing water within the filter casing to prevent air binding and short circuiting. During the first few runs of the filter (following installation), the flow of the water will induce sorting and some compaction of the sand particles. A 6-8% reduction in porosity (i.e., percent pore volume) was calculated from measuring the change in the height of the sand layer following the first three charges to filter post-installation. On average, the porosity of the sand bed during installation was approximately 45% and decreased to approximately 41% after three charge volumes. The particle settling and subsequent porosity reduction was observed by a reduction of the HLRs. Specifically, the filters HLRs reduced by 12-16% following the first three charge volumes.

For eight weeks following the move, HLRs were monitored multiple times per week and were observed to be substantially less than they had been prior. The post-move HLRs dictated cleaning filters almost every week; on average, cleaning was performed six out of the eight weeks for the concrete and 2-gal bucket filters, and seven out of the eight weeks for the 5-gal bucket filters. As depicted in **Figure 29**, the HLRs measured directly after cleaning following the move were significantly less than those from the original location (p-value <0.0001 for all three sizes comparing original location vs. post-move). Specifically, the move induced another 24-35% reduction in the HLRs (**Table 8**) corresponding to a total 41-48% reduction from the initial HLR observed at the original installation location. HLR reduction associated with filter transport is likely due to additional sand compaction and possibly some blocking of the outlet tube (some sand was visually observed in the outlet tubes during the deconstruction of the filters).

Compaction of the sand bed will result in reduced porosity, reduced pore velocities and increased frictional resistance which will reduce the HLR.

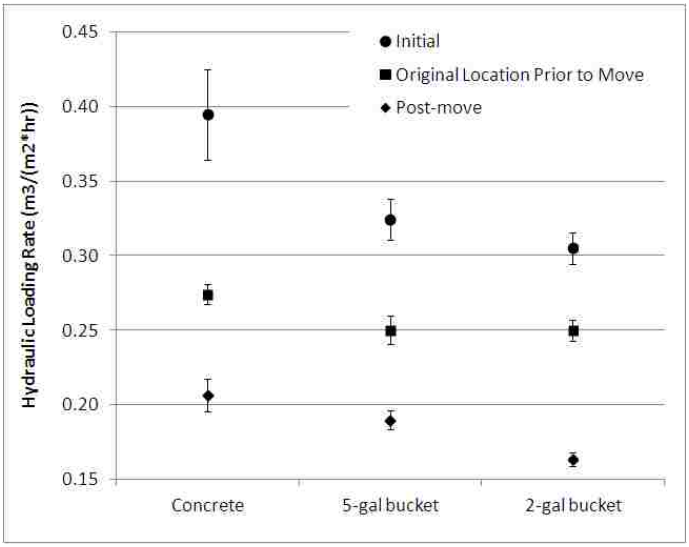

Figure 29. Hydraulic loading rates for newly installed clean filters (initial, n=4 for each filter type), cleaned filters after nine months of testing in the original test location prior to the move (n=44, 40, 36 for the concrete, 5-gal bucket and 2-gal bucket filters, respectively), and cleaned filters after the move (n=21, 24, 19 for the concrete, 5-gal bucket and 2-gal bucket filters, respectively). Error bars indicate the standard error.

Table 8. Percent Reductions in Hydraulic Loading Rates (HLRs) observed from normal use over time and from filter transport effects.

	Percent Reduction		
Comparison of HLRs	Concrete	5-gal	2-gal
Initial vs. original location[a]	30.4	22.8	18.0
Original location[a] vs. post-move[b]	24.8	24.0	34.5
Initial vs. post-move[b]	47.7	41.4	46.3

[a] average HLR of cleaned filters at the original location over nine months

[b] average HLR of cleaned filters after the move

During this study period, the filters were subjected to four microbial challenge experiments (four replicate filters yielded n=16). The results of these challenge experiments were compared to those performed in the previous nine-months at the original location (test day 1-275, Section 3.0) and showed a significant increase in log removal after the move (p-values of 0.0143, 0.0067, and 0.0392 for the concrete, 5-gal bucket, and 2-gal buckets, respectively). **Figure 30** displays the range of removals for each filter size over the five test periods, four from the original location and the fifth from the post-move. The additional compaction of the sand bed from the move would reduce pore volumes and thus increase the entrapment of *E. coli* cells via interception during the filter operation (i.e., the run) and via sedimentation when the filter was at rest (i.e., during the filter pause period).

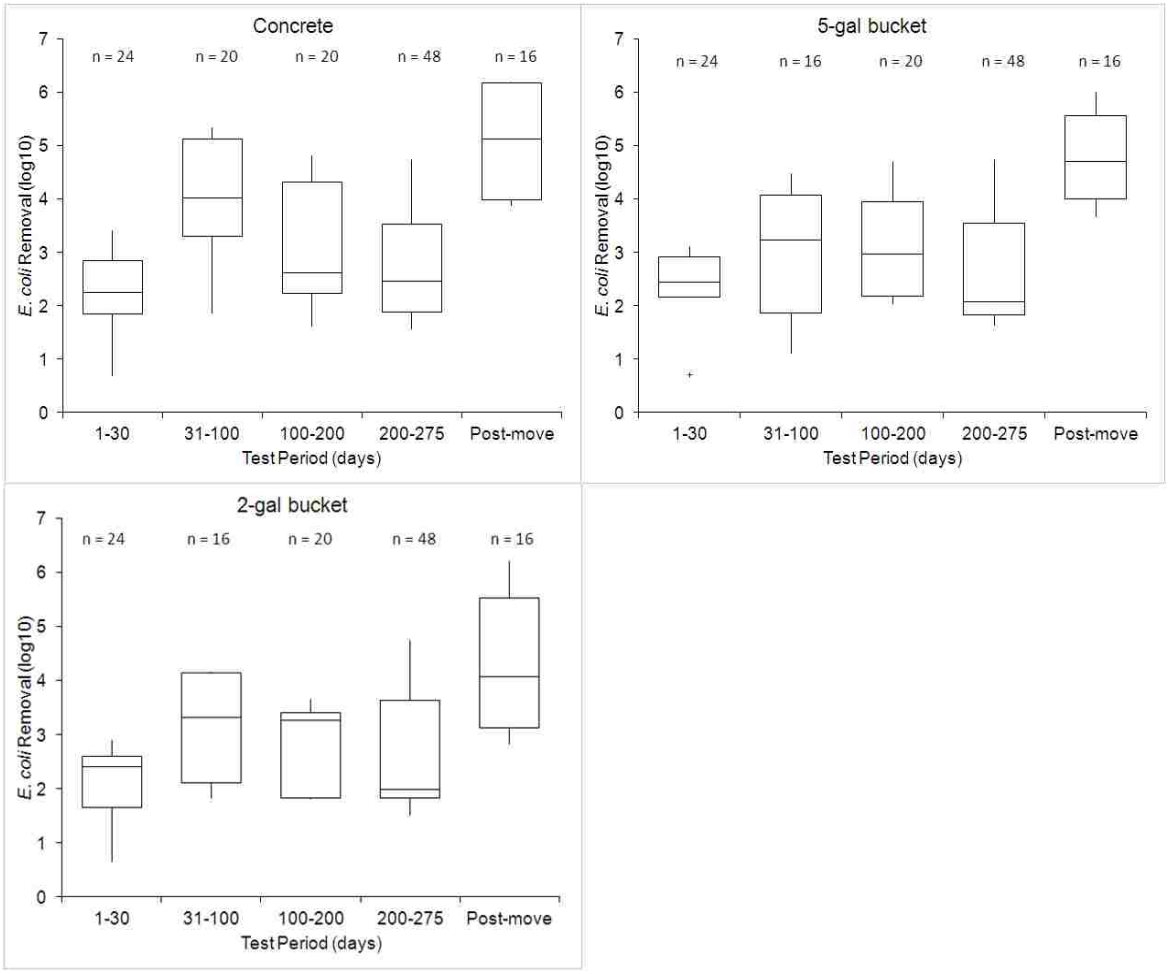

Figure 30. Range of *E. coli* removals (log_{10}) for filters before (test days 1-275) and after (post-move) transport to a new laboratory location. Boxes indicate the 25th, median, and 75th quartiles, respectively. Whiskers extend to furthest observations within 1.5 of the 25th and 75th quartiles, respectively, with any outliers, less than or greater than these values, identified by asterisks.

4.4 Conclusions

This study has shown that transporting filters over a moderate distance (0.5 mi) resulted in reduced HLRs, likely due to sand compaction, but *E. coli* removal remained high and was significantly improved. While it is likely that filters will slow over time as particles become trapped within the pore spaces of the sand media, the rate and magnitude of HLR decline is difficult to predict as it will be determined, in part, by the

turbidity and particle size distribution of the water charged to the filters. For this study, filters were charged three times per day following a three-hour pause period with turbid waters ranging from 5 to 50 NTU; long-term monitoring of replicate filters of various sizes showed that: 1) within the first year of use, the post-cleaning HLR can slow by 25% as compared to the initial HLR, and 2) an additional 25% reduction in HLR is a reasonable estimate if filters are transported after installation.

In conclusion, smaller filters may yield a greater potential for movement after installation by the end-user, but if adequate flushing of the filter is conducted post-move, this will not result in a risk to human health from a bacterial removal standpoint. In this study, filters were flushed with ten charge volumes following the move and then tested for bacterial removal capabilities. Following the move, filters exhibited greater bacterial removal capabilities and reduced HLRs associated with reduced porosity, increased frictional resistance and slower pore velocities. Thus, the greater risk appears to be in the potential for filter abandonment if the HLR drops to a level deemed unacceptable by the end-user. Proper education on the use and maintenance of any sized BSF is critical to sustained use and water quality improvement. Because the first charge volumes post-move were not tested and since transport of the filter has the potential to release previously trapped particles, the importance of post-move flushing of the filter and potential impacts to HLRs associated with filter transport should be incorporated into educational materials to set reasonable expectations among users and discourage behaviors which may reduce the value of the filters in the eyes of the intended beneficiaries.

Additional work is needed to evaluate the turbidity and resultant contaminant levels in the first charges volumes that follow filter transport. Additional testing may prove that a reduced number of fills is required to adequately flush the filter and produce quality water. In addition, this study only evaluated the resultant effects of a three-hour pause period on turbidity and *E. coli* removal levels. Future efforts should focus on evaluating other operational parameters (i.e., shorter and longer pause periods) and investigate the resultant removal capabilities for other pathogens.

5.0 Influence of Sand Depth and Pause Period on Microbial Removal in Traditional and Modified Filters

Previous work showed that small biosand filters (sand bed depth ≤10cm) are effective at removing bacteria and turbidity. However, the impact of shorter bed depths on removal rates for smaller, sub-micron particles (such as viruses), as well as the impact of shorter pause periods on filter performance, remained unknown. Biosand filters with three different sand bed depths were modified with the addition of iron nails in the diffuser basin and evaluated for bacterial, protozoal, and virus removal. Biosand filtration proved effective over a range of pause periods, and removal of bacteria and protozoan cysts for all filter types and sizes ranged from 3 $\log_{10}$ to 4 $\log_{10}$. The addition of nails resulted in significantly better bacteria removal for all filter sizes, while only the smallest filters exhibited significantly better protozoan removal with the addition of nails. Virus removal for all filter types and sizes ranged from <1 $\log_{10}$ to 6 $\log_{10}$. Both the pause period and filter type (size/configuration) influenced virus removal, and the addition of nails to the filter significantly improved virus removal at the shorter pause periods.

5.1 Introduction

The sand bed depth and filter charge volume are two critical design parameters that influence filter performance. Past laboratory studies (Baumgartner et al., 2007; Elliot et al., 2008; Jenkins et al., 2011) demonstrated that contaminant removal is enhanced for water that resides in the sand bed for a full pause period, as compared to water that flows

continuously through the filter with no residence time. Elliot et al. (2008) found that performance was maximized when less than one pore volume was charged to the filter per day.

In this study, we operated three different-sized BSFs to test microbial removal efficacy over a range of pause periods. The standard sized filter was the Centre for Affordable Water and Sanitation (CAWST) version 10 (v10) concrete BSF (with 54 cm sand bed depth and12 L charge volume). Two other filters were built using 5-gal and 2-gal buckets as casing material. The bucket filters were built using the same design principal as the CAWST v.10 BSF, specifically that the charge volume equaled the pore volume of the sand bed. Based on the dimensions of the buckets and the aforementioned design principal, the 5-gal and 2-gal bucket BSFs had sand bed depths of 15 cm and 10 cm, respectively, and charge volumes of 3.6 L and 1.5 L, respectively.

While the first study (Section 3.0) showed that the majority of turbidity and bacteria removal took place within the top 10-15 cm of traditional BSFs, it was not clear whether other microbial contaminants, especially viruses, would be as effectively removed in scaled-down BSFs. The small size (typically 0.005-0.3 μm) and negatively-charged surfaces of most viruses suggest that viruses may be more likely to pass through sand filters, especially if bed depths are reduced. In other studies, traditional concrete BSFs were modified with an iron source for the removal of microorganisms and naturally-occurring arsenic, with variable levels of success (Bradley et al., 2011; Chiew et al., 2009; Meng et al., 2001; Lukasik et al., 1999). In the presence of oxygen and water, iron readily corrodes to form a positively-charged iron oxide precipitate that binds

to negatively-charged water contaminants through electrostatic attraction; since the contaminant-laden iron oxide particles are readily captured within the filter by straining and adsorption, the quality of the treated water is improved. Thus, we hypothesized that the addition of an iron source to the scaled-down BSFs would enable virus removals comparable to those observed in traditional BSFs. For this study, small iron (non-galvanized) nails were added to the diffuser basins of the BSFs to test the hypothesis that virus removal can be enhanced in smaller-sized BSFs through the introduction of postively-charged iron oxide to the system. .

It was hypothesized that the nail-modified filters could also enhance the adsorption/adhesion of the negatively-charged viruses. In this way, the viruses would become attached to the iron oxide particles and adsorption to and/or interception by the sand grains would be enhanced. The basis of this hypothesis is the chemical and physical processes associated with iron hydroxide precipitation. Nails were added to the diffuser basin to introduce elemental iron into the system. While the complete chemistry for the formation of hydrolysis reactions and products is not fully understood (Metcalf & Eddy 2002) and was not the fouc of this investigation, in general the formation of ferric hydroxide is dependent on presence of dissolved oxygen. Thus in an aqueous, aerobic environment as is found in the influent waters for the filters, the iron nails were expected to readily oxidize to Fe(II). These Fe(II) species are relatively unstable and will oxidize quite rapidly to Fe (III) species. Fe (III) species are not very soluble at pH>5 and will readily form precipitates; the most common species in natural waters is the hydrated Fe(III) hydroxide (Fe2O3*H20), which is positively charged. The Fe(III) ions and

resulting hydroxides would neutralize the negative surface charge of the viruses, and would increase the effective particle size of the viruses and would enhance removal through the pore spaces in the sand bed.

For an end-user to obtain the same amount of treated water in a given day, the scaled-down BSF would need to be filled more frequently than the traditional BSF, resulting in shorter pause periods between fills. Thus, the objectives of this study were to investigate (1) whether six different BSF designs (three sand bed depths, each with and without iron nails) perform significantly differently with respect to bacteria, protozoa, and virus removal, (2) whether modifying the BSF with nails in the diffuser basin significantly improves microbial removal, and (3) the impact of pause period on the removal efficiencies of the six BSF designs. The filters were tested over six different pause periods (1, 3, 6, 12, 24, and 72 hours) at a targeted turbidity level of 50 NTU.

5.2 Materials & Methods

5.2.1 Experimental Setup

For this study, twelve full-scale BSFs were built and tested, including four replicate filters of three filter sizes (i.e., the traditional concrete BSF (CAWST v10), as well as two smaller versions constructed in a 5-gal bucket and 2-gal bucket, respectively). Two BSFs of each size were modified by adding iron nails (non-galvanized ¾" finishing nails, Code 1AC06, Tree Island Industries Ltd., Richmond, BC) to the diffuser basins, specifically 5 kg, 1.5 kg, and 0.625 kg were added to the concrete, 5-gal and 2-gal filters, respectively. The filters were tested at six different pause periods of 1, 3, 6, 12, 24, and

72 hours. Filters were ripened for one month prior to testing, and filters were charged twice per day during this ripening period. Filters were operated at each pause period for approximately six weeks; all filters were cleaned at the start of a new pause period run. The shorter pause periods were tested first and in the following order 6, 3, and 1 hr; the filters were charge three times per day during these pause period runs. The longer pause periods runs followed and were performed in order, specifically 12, 24, and 72 hr.

Influent water consisted of dechlorinated tap water augmented with local creek water and sediments (Monocacy Creek, Bethlehem, PA) to obtain a target turbidity level of 50 NTU and spiked with viruses, protozoa, and bacteria (as described below) on microbial removal test days. Microbial removal tests were performed in triplicate for bacteria and in duplicate for protozoa and viruses. Peak flow rates were measured at maximum hydraulic head and normalized to the peak HLR as previously described (Section 3.2.3). Hydraulic loading rates (HLRs) were monitored weekly to identify when cleaning needed to be performed; cleaning was performed when flow rates decreased to approximately half of the initial clean bed value.

5.2.2 Water Quality Parameters

On days when microbial tests were performed, 300ml aliquots of the influent and filter effluent samples were collected and analyzed for pH (standard units), turbidity (NTU) and conductivity (uS/cm). Turbidity was measured using a Hach Turbidimeter Model 2100P; pH and conductivity were recorded using an Oakton PC 510 bench meter. Alkalinity and hardness (both, as mg/L $CaCO_3$) were analyzed biweekly on days when

microbial testing was not performed; 500ml aliquots of influent and effluent samples were used to perform these water quality analyses. Alkalinity and hardness were measured using a Hach Digital Titrator (model 16900) via Hach Methods 8203 and 8204, respectively.

5.2.3 Viruses

Freeze-dried stock solutions of MS2 coliphage and *Escherichia coli* host (Strain 15597) were obtained from ATCC (Manassas, VA). MS2 coliphage was propagated in broth inoculated with host bacteria, and titer was determined using the double agar layer method (ATCC protocol for 15597-B1) to prepare stock solutions. The stock solution aliquots were combined with 15% glycerol (v/v) in cryogenic vials and stored at -80 °C (Adams 1959). We tested the effects of a single freeze/thaw cycle on virus stock concentration using the single agar layer method (Adams 1959), and these effects were taken into account when estimating the volume of freezer stock solution needed for experiments.

A new vial of frozen stock was used for each test run, and any remaining stock was discarded, eliminating any potential effects of multiple freeze-thaw cycles on the resultant titer. Using the estimated stock concentration, influent water was spiked to obtain a target concentration of 1E5 plaque forming units (PFUs)/ml. Thawed stock solutions were analyzed via the single agar layer method to confirm titer of the stock solution. In addition, 1-L aliquots of the spiked influent and resultant effluent samples

were collected, and four replicates of five dilutions for each sample were analyzed using the single agar layer method.

5.2.4 Protozoan Cysts

Cryptosporidium parvum oocysts were obtained from Waterborne Inc. (New Orleans, LA). *C. parvum* oocyst stock concentrations were confirmed via hemacytometer counts. Influent water was spiked to obtain a target concentration of 5E3 oocysts/ml. A 1-L aliquot of the influent and effluent, respectively, was processed by membrane filtration using 3-µm GE polycarbonate membrane filters (GE Healthcare, Pittsburgh, PA). Following membrane filtration, the filters were washed and eluted in PBS solution, and samples were concentrated by centrifugation as previously described (Oda et al., 2000; Wolyniak et al., 2009).

Concentrated samples were stained with MeriFluor Detection Reagent (Meridian Bioscience, Inc., Cincinnati, OH). After a 30-min contact time with the detection reagent, samples were centrifuged at 1300 rpm for 5 min, rinsed with wash buffer, centrifuged at 1300 rpm for 5 min, and resuspended in a final volume of approximately 50-100 µL. For each sample, the final volume measurement was recorded and the entire volume was then plated onto a single well of a MeriFluor pretreated slide and allowed to dry at room temperature (~22-25 °C). Per the manufacturer's protocol, the MeriFluor mounting media was added to the wells prior to fixing a cover slip.

C. parvum oocysts were enumerated via fluorescent microscopy using a Nikon epifluorescence microscope (Nikon, Inc., Melville, NY) with a FITC filter block (490-500 nm excitation, 510-530 nm emission). Each well was scanned using the 40X

objective, and oocysts were confirmed using the 100X objective to confirm apple-green fluorescence of ovoid objects 4 to 6 μm in diameter (EPA method 1622). A positive control, consisting of 1 L of ultrapure water spiked with 1E4 oocysts, was included each time the assay was performed. The positive control was kept on the bench top until the effluent samples were collected, and then all filter and control samples were processed collectively. All results were corrected for losses based on recovery numbers of the control sample for that specific test date.

5.2.5 Bacteria

Microbial analyses were performed in accordance with previously described procedures (Section 3.2.1). Samples, influent and effluents, were analyzed via membrane filtration for *E. coli* via Standard Method 9222 (Rice et al 2012). All samples (diluted or undiluted) were analyzed in triplicate. Following membrane filtration, membrane filters were placed in a culture dish that contained a sterile pad and 2 mL of m-ColiBlue 24® broth (Hach Company) and incubated at 35 ± 0.5°C for 24 hours. After the incubation period, membrane filters that yielded colonies with 10-100 colonies were considered acceptable and counted. Concentrations were calculated according to the Standard Method 9222. If replicate filters of samples yielded acceptable colony counts, the resulting concentrations were averaged. For instances when all filters yielded zero colonies, the detection limit (1 CFU/total volume analyzed) was used as the effluent concentration for the subsequent calculation of removal efficiency (i.e., log reduction).

5.2.5 Statistical Analysis

The Kruskall-Wallis test with Bonferroni error protection was performed to determine whether there was a significant difference in performance (i) across the various pause periods for each of the six filter types and (ii) between the two configurations (i.e., with and without nails) for filters of the same size operated at the same pause period. Pearson correlation tests were performed to identify any correlations between microbial removal and either pause period or sand depth. All statistical analyses were performed with the Analyze-It add-in (Analyze-It Software, Ltd., Leeds, England) for Microsoft Excel.

5.3 Results

5.3.1 Hydraulic Loading Rates

The HLRs for the filters with and without nails were similar for the concrete and 2-gal filters ($p>0.05$, **Figure 31, Table 9**). For the 5-gal filters, the filters with the nails were slightly lower than for the traditionally configured (no nails) replicates ($p=0.0001$). While the concrete HLRs for pause periods of 24- and 72-hr are higher than those for the other pause periods, this is attributed to the total number of fills that the filters received during each run. Each pause period test run took approximately two months to complete. Thus, during the longer pause periods, the filters were charged fewer times and were exposed to a lower loading of particles (via influent turbidity) than during the shorter pause period test runs.

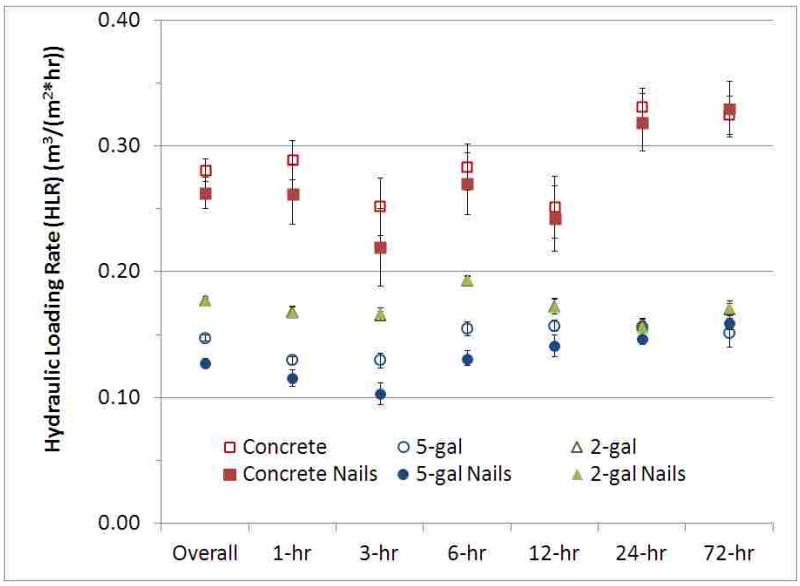

Figure 31. Median Hydraulic Loading Rates (HLRs) for all filter configurations for each pause period. Error bars represent the standard error (stdev/√n).

Table 9. Median and standard deviation values for Hydraulic Loading Rates (HLRs) for each pause period.

Median Hydraulic Loading Rate ($m^3/(m^2*hr)$)

Period	Concrete	Concrete - Nails	5-gal	5-gal - Nails	2-gal	2-gal - Nails
Overall	0.281	0.263	0.147	0.128	0.177	0.178
± Stdev	0.090	0.012	1.5	1.3	1.6	0.022
1-hr	0.289	0.262	0.130	0.116	0.168	0.168
± Stdev	0.084	0.11	0.49	0.43	0.63	0.011
3-hr	0.252	0.220	0.130	0.103	0.165	0.168
± Stdev	0.10	0.14	0.57	0.44	0.70	0.014
6-hr	0.283	0.270	0.155	0.131	0.193	0.194
± Stdev	0.096	0.15	0.96	0.81	1.2	0.023
12-hr	0.251	0.243	0.157	0.142	0.172	0.173
± Stdev	0.085	0.090	0.54	0.49	0.60	0.022
24-hr	0.332	0.319	0.157	0.147	0.158	0.156
± Stdev	0.042	0.065	0.44	0.42	0.45	0.018
72-hr	0.325	0.329	0.152	0.160	0.170	0.172
± Stdev	0.037	0.055	0.37	0.39	0.42	0.012

5.3.2 Water Quality

The iron concentration was monitored to evaluate the potential to degrade the water quality via the introduction of nails into the diffuser basin. The total

iron concentration was measured in the influent, the supernatant (i.e., the standing water layer above the sand media), and the filter effluents. The iron concentration in the supernatant was significantly greater than the influent ($p<0.0001$); whereas, the effluent iron concentrations were similar to the influent concentration ($p>0.05$) (**Figure 32**).

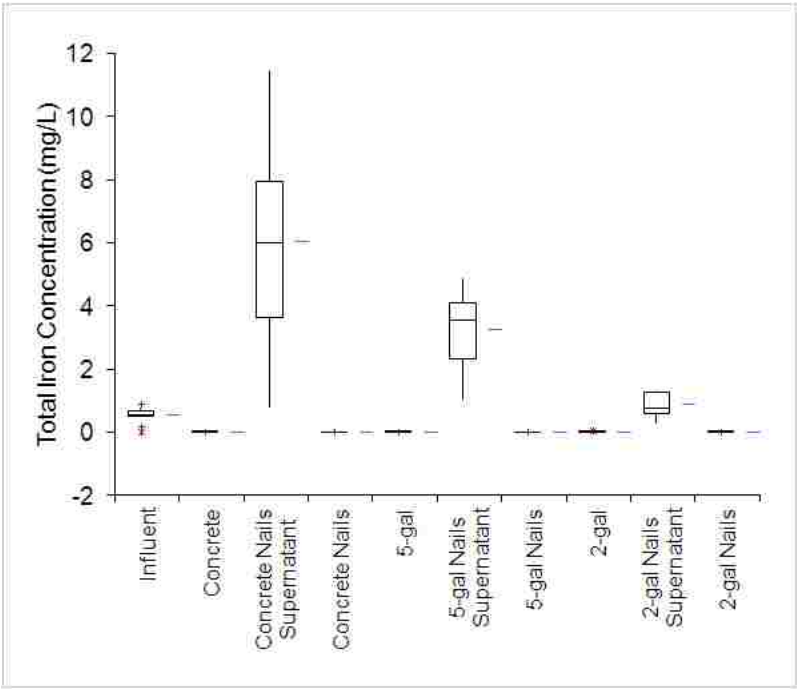

Figure 32. Total iron concentration (mg/L) in the influent, standing water layer (supernatant) of the nail configuration filters, and filter effluents.

The pH of the influent was near neutral for the entire study with an overall average of 7.3. The minimum and maximum average pHs from each test period run were 6.7 and 7.7, respectively. As has been observed previously (Section 3.0), the pH of the effluents from the concrete filters was slightly higher than

those from the bucket filters. However, there was no significant difference in the overall populations ($p>0.05$).

The turbidity of concrete effluents was consistently below 1 NTU on all test days for all pause period runs, resulting in overall (from all pause period runs) average turbidity levels of 0.64 and 0.49 NTU for the traditional and nail configurations, respectively. The turbidity of the effluents from the 5- and 2-gal bucket filters fluctuated more during this study than had been previously observed (Section 3.0) with several effluents yielding greater than 1 NTU; however, all filter effluents were consistently below 1.5 NTU for all bucket filters (both traditional and with nail configurations).

The conductivity of the influent on average ranged from 106 μS/cm to 183 μS/cm over the pause period runs with an overall average of 132 μS/cm for the entire study. The conductivities of the filter effluents were not significantly different to the influent for any of the pause period test runs ($p>0.05$). The average pH, conductivity, turbidity for each filter type by pause period is presented in **Table 10**.

The influent water had an average alkalinity of 27 mg/L ($CaCO_3$) and hardness of 342 mg/L ($CaCO_3$) over the entire test period. These values are within the range of values that was observed for water from Monocacy Creek in the previous study (Section 3.0). The filters yielded waters with similar ($p>0.05$) ranges of both alkalinity and hardness. The average values of the alkalinity and

hardness for the influent and filter effluents for each pause period test run are presented in **Table 11**.

Table 10. pH, conductivity (µS/cm), and turbidity (NTU) in the influent and filter effluents.

	Influent		Concrete		Concrete Nails		5-gal		5-gal Nails		2-gal		2-gal Nails	
	Avg	±Stdev	Avg	±Stdev	Avg	±Stdev	Avg	±Stdev	Avg	±Stdev	Avg	±Stdev	Avg	±Stdev
1-hr Pause Period														
pH	7.24	0.30	7.91	0.32	8.68	0.26	7.56	0.77	7.38	0.28	7.39	0.27	7.46	0.25
Conductivity (µS/cm)	105.57	38	129.28	26	125.15	34	149.80	47	133.80	11	131.68	28	128.25	29
Turbidity (NTU)	48.14	4.05	0.54	0.14	0.54	0.17	0.79	0.15	0.50	0.07	0.84	0.17	0.51	0.08
3-hr Pause Period														
pH	6.67	0.19	6.56	1.08	7.44	0.84	6.40	0.19	6.36	0.22	6.38	0.22	6.43	0.23
Conductivity (µS/cm)	136.20	12	142.68	14	132.33	15	143.54	12	143.43	11	145.93	12	146.90	11
Turbidity (NTU)	42.12	12.84	0.64	0.34	0.36	0.09	0.78	0.24	0.54	0.19	0.96	0.34	0.41	0.11
6-hr Pause Period														
pH	7.15	0.19	8.07	0.62	8.60	0.30	6.71	0.43	6.74	0.42	6.96	0.48	6.89	0.44
Conductivity (µS/cm)	114.13	31	127.66	38	117.18	30	116.47	36	106.68	35	111.32	35	109.34	37
Turbidity (NTU)	51.01	5.09	0.65	0.22	0.45	0.11	0.98	0.35	0.89	0.45	1.32	0.44	0.85	0.40
12-hr Pause Period														
pH	7.68	0.48	8.09	0.35	8.62	0.31	7.70	0.30	7.72	0.33	7.82	0.37	7.74	0.31
Conductivity (µS/cm)	182.57	67	138.13	15	133.49	16	145.19	15	137.86	13	144.31	16	142.25	17
Turbidity (NTU)	49.55	7.61	0.74	0.31	0.51	0.24	0.89	0.51	0.52	0.17	0.79	0.40	0.58	0.25
24-hr Pause Period														
pH	7.52	0.22	8.24	0.32	8.77	0.31	7.65	0.37	7.62	0.32	7.74	0.34	7.71	0.29
Conductivity (µS/cm)	128.76	32	129.27	20	129.98	17	133.01	24	131.34	21	133.93	25	135.95	25
Turbidity (NTU)	50.55	5.56	0.71	0.38	0.61	0.34	0.85	0.64	0.72	0.53	0.68	0.33	0.63	0.33
72-hr Pause Period														
pH	7.63	0.54	8.55	0.23	9.10	0.23	7.65	0.29	7.58	0.31	7.64	0.36	7.67	0.35
Conductivity (µS/cm)	125.87	42	131.12	11	143.89	16	132.74	11	134.39	11	128.66	32	128.40	31
Turbidity (NTU)	48.64	2.42	0.53	0.15	0.50	0.16	0.56	0.17	0.53	0.17	0.58	0.25	0.46	0.09

Table 11. Alkalinity and hardness (mg/L as $CaCO_3$) of the influent and filter effluents

Alkalinity (mg/L $CaCO_3$)

	Influent		Concrete		Concrete Nails		5-gal		5-gal Nails		2-gal		2-gal Nails	
Pause Period	Avg	±Stdev	Avg	±Stdev	Avg	±Stdev	Avg	±Stdev	Avg	±Stdev	Avg	±Stdev	Avg	±Stdev
1-hr	24	1	30	3	27	3	29	2	30	2	35	5	33	2
3-hr	31	1	35	2	30	3	36	4	35	3	35	3	36	3
6-hr	29	12	29	5	26	2	30	5	29	6	37	14	31	5
12-hr	19	1	24	1	25	1	25	1	25	1	23	2	24	2
24-hr	23	11	33	10	33	8	31	8	30	7	33	10	34	10
72-hr	33	13	28	0	30	3	30	2	35	4	44	1	41	1

Hardness (mg/L $CaCO_3$)

	Influent		Concrete		Concrete Nails		5-gal		5-gal Nails		2-gal		2-gal Nails	
Pause Period	Avg	±Stdev	Avg	±Stdev	Avg	±Stdev	Avg	±Stdev	Avg	±Stdev	Avg	±Stdev	Avg	±Stdev
1-hr	362	50	268	32	263	45	273	39	265	28	293	22	283	17
3-hr	310	14	330	24	328	43	335	42	323	22	338	10	335	40
6-hr	462	31	403	23	369	10	358	91	347	94	418	15	368	71
12-hr	257	21	234	15	233	51	226	82	223	14	203	17	215	32
24-hr	342	12	278	90	285	83	265	74	264	46	288	87	297	96
72-hr	319	28	327	28	360	71	285	71	294	41	275	22	275	78

5.3.3 Virus Removal

The MS2 bacteriophage removal rates were significantly different i) across the pause periods tested for a single filter configuration and ii) for the different filter configurations (i.e., with and without nails) for each filter size (i.e., p-value <0.0001 for the concrete, 5-gal and 2-gal buckets) (**Figure 33**, **Table 12** for p-values).

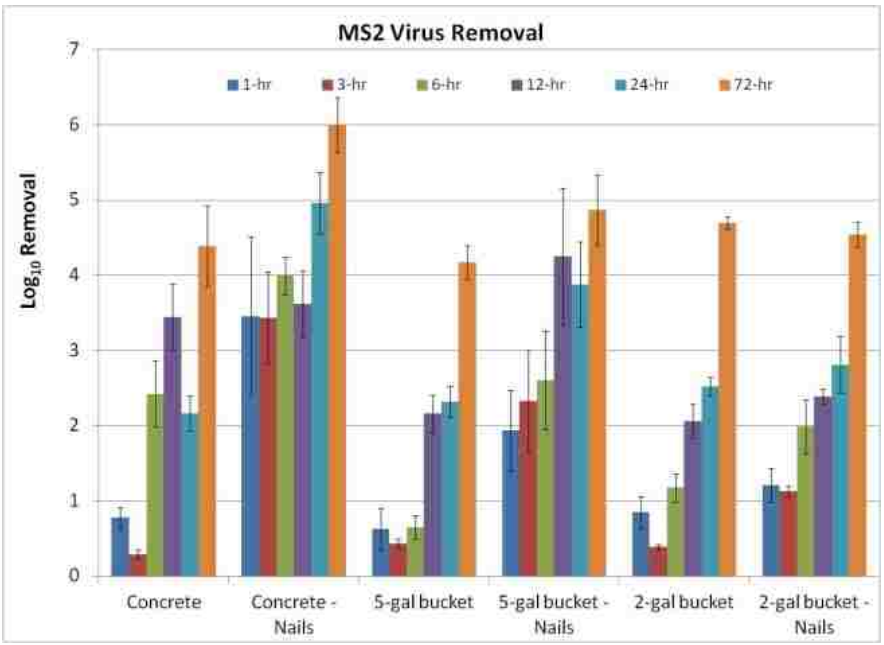

Figure 33. Median removal values for all filter configurations over six pause periods for MS2 bacteriophage. Each bar represents the median value of two trials performed on duplicate filters and error bars represent the standard error (stdev/√n).

Evaluation of the MS2 $\log_{10}$ removal values across each filter type (**Figure 34; Table 12**) showed similar filter performance across several pause periods (e.g., three performance groups were observed for the concrete BSF with no nails: filters operated at 1 and 3 hr pause periods had the lowest removal; filters operated at 6, 12, and 24 hr pause periods showed similar mid-range removal levels; and filters operated at a 72 hr pause period had the highest observed removal). This trend was confirmed by performing a K-W test with Bonferroni protection to identify a difference in the medians (n=4) of MS2 removals for the pause periods tested for each filter type (size and configuration). This trend was confirmed by the Kruskall-Wallis test on the median (n=4) MS2 removals for the pause periods tested for each filter type (**Table 8);** the overall p-values from the six

datasets were all <0.05, confirming a significant difference in the MS2 removal medians among pause periods of the same filter type. The performance groupings of the pause periods for each filter configuration were based on the p-values for each pairwise comparison (p-values not shown, but a performance group consisted of pause period comparisons with p values >0.05). For all three filter sizes without nails, three performance groupings were observed. The addition of nails to each filter size improved MS2 removal at the same pause period (**Figure 34; Table 12**).

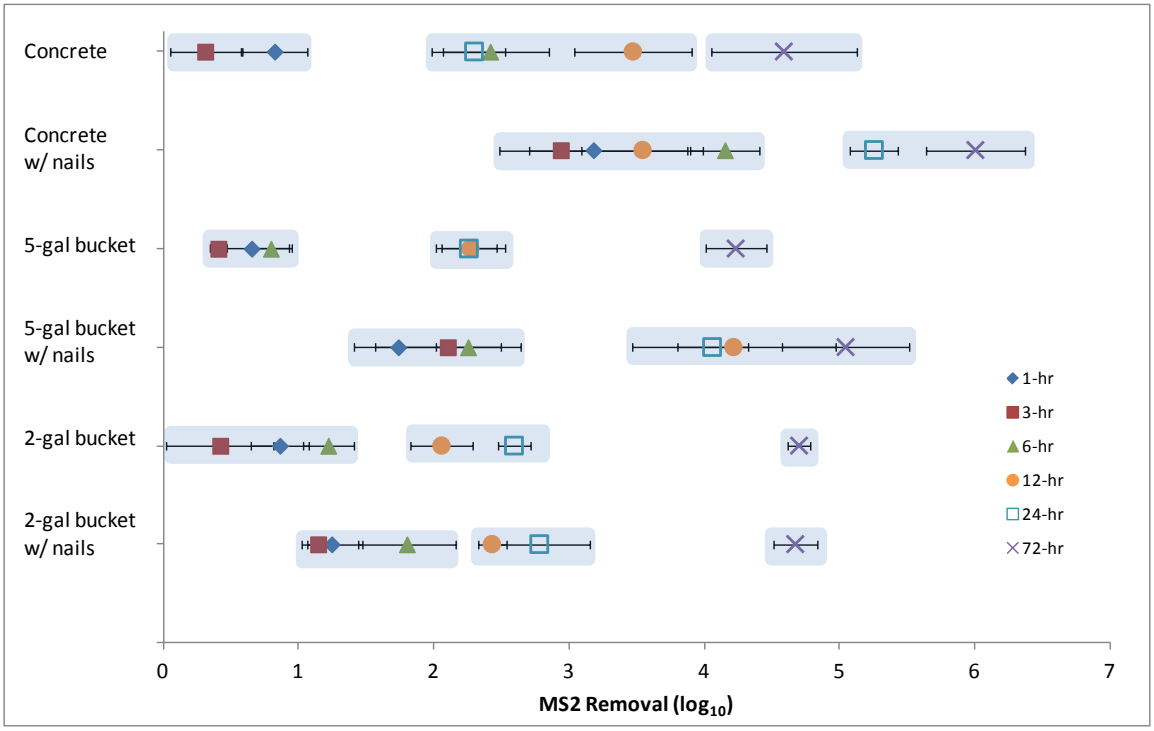

Figure 34. Median MS2 removal (log_{10}) for each filter type (n=4) for six different pause periods (i.e., 1-, 3-, 6-, 12-, 24-, and 72-hr pause periods). Error bars indicate the standard error (stdev/√n). Shading indicates performance groupings (i.e., statistically similar populations) based on individual K-W p-values from all pairwise comparisons.

Table 12. Median MS2 removal (log_{10}) for each filter configuration per pause period (n=4). The p-values apply to the overall comparison of median removals for the six pause periods tested for a given filter type.

	MS2 Removal (log_{10})[1, 3]						
	1-hr	3-hr	6-hr	12-hr	24-hr	72-hr	p-value[2]
Concrete	0.82	0.31	2.42	3.47	2.30	4.59	0.0013
Concrete w/ Nails	3.18	2.94	4.15	3.54	5.25	6.00	0.0036
5-gal bucket	0.65	0.41	0.79	2.27	2.26	4.23	0.0017
5-gal bucket w/Nails	1.74	2.10	2.25	4.21	4.06	5.04	0.0039
2-gal bucket	0.86	0.42	1.22	2.06	2.59	4.70	0.0007
2-gal bucket w/Nails	1.24	1.14	1.80	2.43	2.78	4.67	0.0021

[1] median values where n=4

[2] overall Kruskal-Wallis p-value for multiple comparison across pause periods for each filter type

[3] groupings are based on pairwise comparison of pause periods (e.g., 1-hr vs. 3-hr, 1-hr vs. 6-hr, etc.; p-values not shown)

All filter types exhibited a strong, statistically significant positive correlation between MS2 coliphage removal and pause period, i.e., removal increased as pause period increased (**Table 13; Figure 35**). The correlation was the strongest for the smallest filters without nails (**Table 13**). For all filter sizes, MS2 coliphage removal improved with the addition of nails, as evidenced by the upward shift of the data points in **Figure 35**.

Table 13. Pearson correlation parameters for MS2 removal as a function of pause period for each filter type

Filter Type/Configuration	r statistic[1]	p-value
Concrete	0.70	0.0001
Concrete with nails	0.76	<0.0001
5-gal bucket	0.91	<0.0001
5-gal bucket with nails	0.63	0.0010
2-gal bucket	0.95	<0.0001
2-gal bucket with nails	0.90	<0.0001

[1] with a confidence interval of 95%

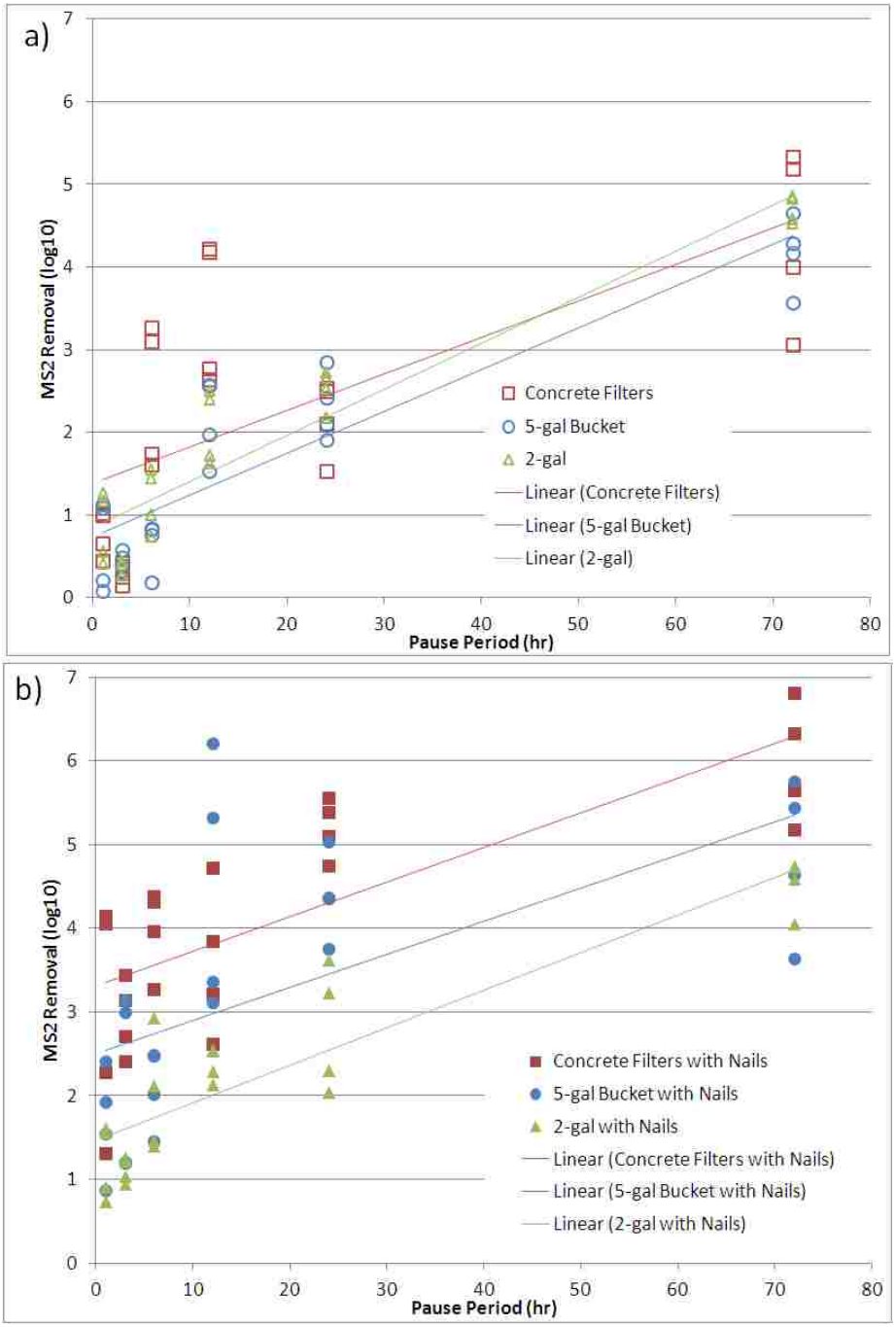

Figure 35. MS2 removal plotted as a function of pause period for each filter type, specifically for a) traditional (no nails) and b) modified (with nails), with trendlines for each filter size.

MS2 coliphage removal as a function of sand bed depth was evaluated to identify the pause periods for which there was a significant correlation (**Table 14; Figure 36**). For filters without nails (**Figure 36a**), a significant positive correlation was only observed at pause periods of 6 and12 hrs; MS2 coliphage removal at these pause periods was enhanced with increasing sand depth. For filters with nails (**Figure 36b**), MS2 coliphage removal increased with bed depth at all pause periods, and the positive correlation between virus removal and bed depth was significant at all pause periods except 12 hrs.

Table 14. Pearson correlation parameters used to identify significant relationship between sand bed depth and MS2 removal for two filter configurations (i.e., traditional and modified) over six pause periods (i.e., 1-, 3-, 6-, 12-, 24-, and 72-hrs). Bolded values indicate a significant, strong correlation defined as $|r| > 0.5$ and $p < 0.05$.

Filter Configuration	Pause period (hr)	r statistic	p-value
Traditional (no nails)	1	0.02	0.9420
	3	-0.47	0.1206
	6	**0.80**	**0.0033**
	12	**0.75**	**0.0046**
	24	-0.35	0.2587
	72	-0.07	0.8351
Modified with nails	1	**0.59**	**0.0444**
	3	**0.59**	**0.0420**
	6	**0.71**	**0.0091**
	12	0.17	0.5993
	24	**0.68**	**0.0153**
	72	**0.71**	**0.0099**

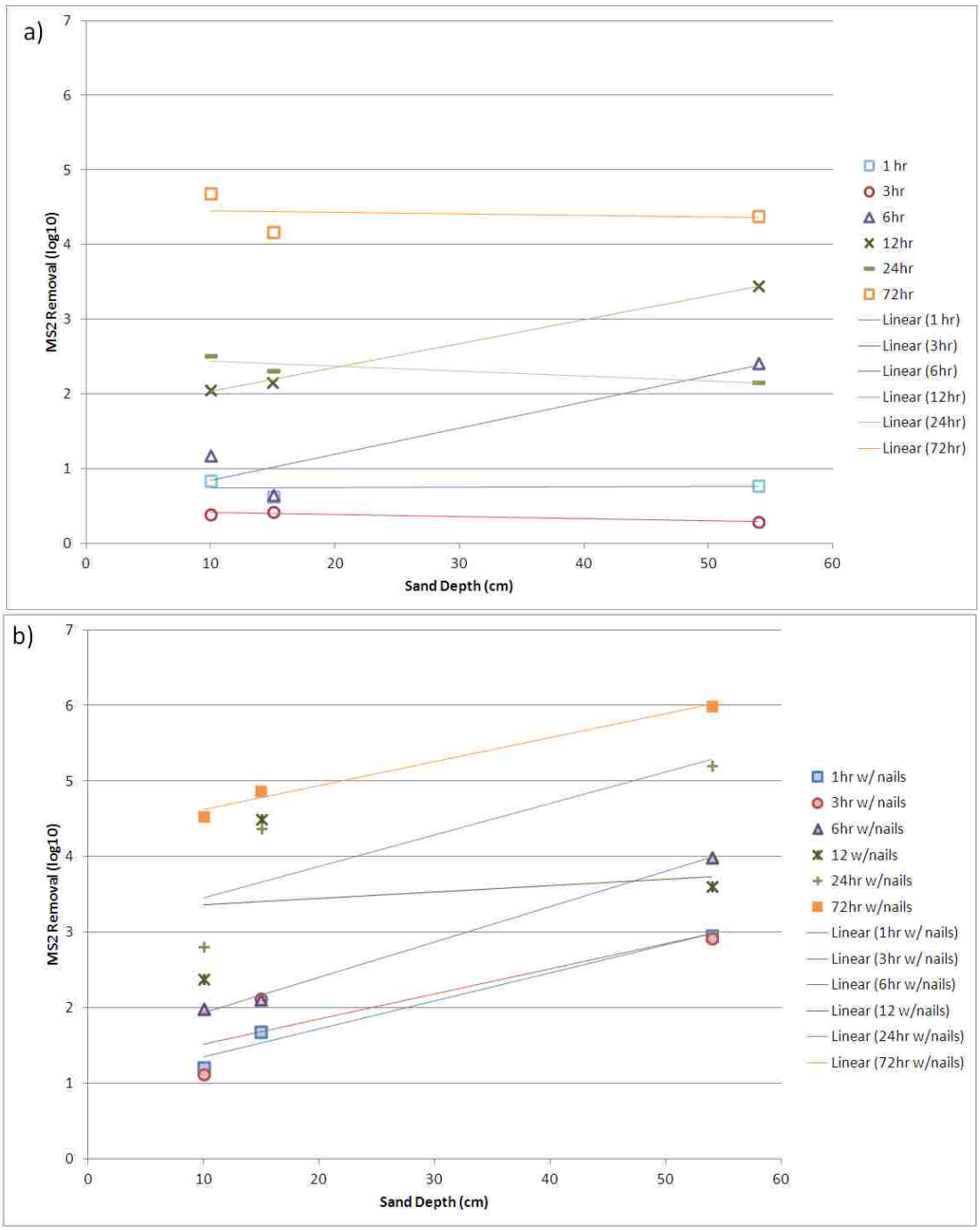

Figure 36. MS2 removal plotted as a function of sand bed depth for a) traditional (no nails) and b) modified (with nails) filters with trendlines for each pause period.

5.3.4 Protozoan Removal

No trends were observed between pause period and *C. parvum* removal across all filter types (**Figure 37**), and there was no significant difference among the median *C. parvum* removals across the pause periods tested for the 5-gal, 5-gal with nails, and 2-gal filters (**Table 15**). For the concrete filters, the *C. parvum* removals from the 1 hr pause period were significantly different from those obtained at the 3, 6, and 12 hr pause periods from the pairwise comparisons; for the concrete filters with nails, the 1 hr pause period removals were significantly different from the 72 hr removals. For the 2-gal filters with nails, two pairwise comparisons of *C. parvum* removals yielded significant p-values: 1 hr vs. 12 hr, and 12 hr vs. 24 hr. The geometric mean for all pause periods (n=24) for each filter type is presented in **Table 15**. Comparison of the datasets for each filter type/configuration showed (**Figure 38**) that for the concrete and 5-gal bucket size filters, adding the nails did not significantly improve *C. parvum* removal (p-value >0.05); whereas for the 2-gal bucket size filters, adding nails did significantly improve *C. parvum* removal (p-value = 0.0031).

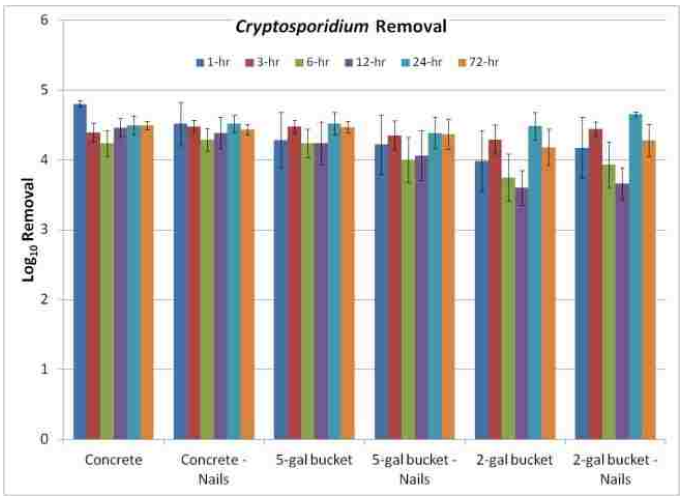

Figure 37. Median removal values for all filter configurations over six pause periods for *C. parvum*. Each bar represents the median value of two trials performed on duplicate filters and error bars represent the standard error (stdev/√n).

Table 15. Median *C. parvum* removal (log_{10}) for each filter configuration per pause period (n=4). Geometric mean and p-value comparison for all pause periods for each filter configuration (without and with nails).

	C. parvum Removal (log10)[1]								
	1-hr	3-hr	6-hr	12-hr	24-hr	72-hr	p-value[2]	Geo Mean[3]	p-value[4]
Concrete	4.80	4.14	4.25	4.26	4.65	4.53	0.0097	4.4	0.0964
Concrete - Nails	4.80	4.58	4.55	4.65	4.50	4.43	0.0451	4.6	
5-gal bucket	3.66	4.56	3.98	4.13	4.50	4.43	0.6884	4.2	0.0617
5-gal bucket - Nails	4.80	4.58	4.55	4.50	4.65	4.53	0.0666	4.6	
2-gal bucket	3.07	4.42	3.24	3.51	4.50	3.78	0.0654	3.7	0.0031
2-gal bucket - Nails	4.80	4.40	4.39	3.59	4.65	4.53	0.0143	4.4	

[1] median values where n=6

[2] overall Kruskal-Wallis p-value for multiple comparison across pause periods for each filter type

[3] geometric mean of removal across all pause periods, n=36

[4] Kruskal-Wallis p-value comparing configurations (i.e., with and without nails) for each filter size

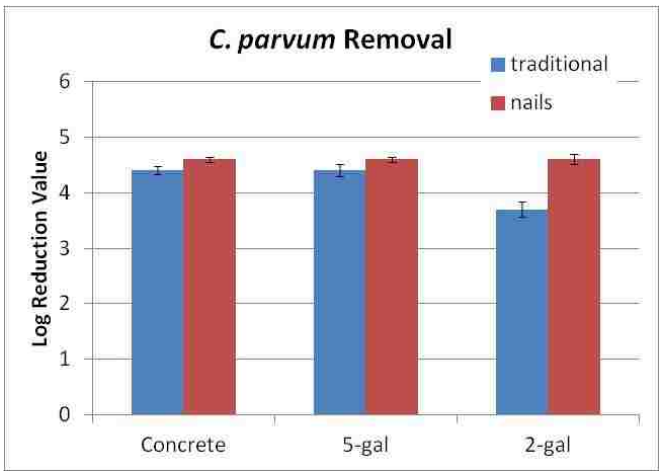

Figure 38. Comparison of median removal values for traditional and modified filters configurations for *C. parvum*. Each bar represents the median value of 24 values (i.e., the results from two trials for duplicate filters over six pause periods) and error bars represent the standard error (stdev/√n).

5.3.5 Bacteria Removal

The *E. coli* and Total Coliforms (TC) removals were high regardless of pause period for the same filter type over the range of pause periods (**Figure 39**). There was no significant difference in the median *E. coli* and total coliform removal rates across the pause periods for each filter type (**Table 16; Figure 40**). The geometric mean for all pause periods (n=36) is presented in **Table 16** for comparison of each filter type. For filters of the same size, the addition of nails to the diffuser basin significantly improved bacteria removal (p-values <0.05 comparing "with" and "without nails" configurations for all filter sizes, as shown in **Table 16**).

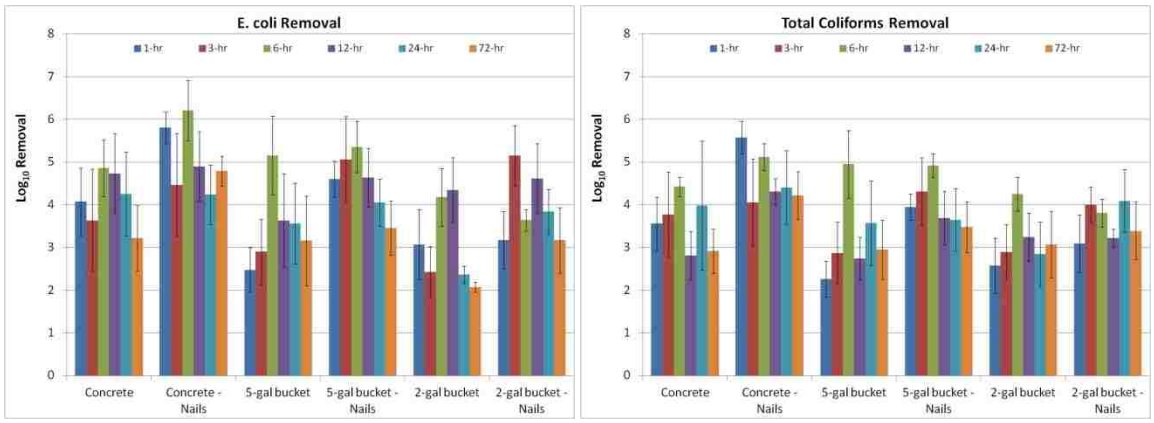

Figure 39. Median removal values for all filter configurations over six pause periods for *E. coli* and Total Coliforms (TC). Each bar represents the median value of three trials performed on duplicate filters and error bars represent the standard error (stdev/√n).

Table 16. Median bacteria removal (log_{10}) for each filter configuration per pause period (n=6). Geometric mean and p-value comparison for all pause periods for each filter configuration (without and with nails).

E. coli Removal (log10)[1]

	1-hr	3-hr	6-hr	12-hr	24-hr	72-hr	p-value[2]	Geo Mean[3]	p-value[4]
Concrete	3.88	3.19	4.32	4.85	4.51	2.75	0.6729	3.8	0.0295
Concrete - Nails	6.20	4.67	6.63	4.85	3.97	4.79	0.2830	5.1	
5-gal bucket	2.26	2.52	5.54	2.95	3.85	3.17	0.2020	3.2	0.0063
5-gal bucket - Nails	4.82	5.54	4.79	4.85	4.01	3.34	0.4080	4.5	
2-gal bucket	2.48	1.44	4.00	2.94	3.76	3.15	0.1022	2.8	0.0491
2-gal bucket - Nails	3.20	5.31	3.60	4.30	3.80	2.76	0.1803	3.7	

TC Removal (log10)[1]

	1-hr	3-hr	6-hr	12-hr	24-hr	72-hr	p-value[2]	Geo Mean[3]	p-value[4]
Concrete	4.06	4.27	2.88	3.89	3.89	2.69	0.5406	3.6	0.0068
Concrete - Nails	4.38	5.44	4.46	3.68	3.68	4.01	0.3543	4.2	
5-gal bucket	2.95	5.06	2.17	3.40	3.40	3.04	0.1312	3.2	0.0112
5-gal bucket - Nails	4.62	4.94	3.53	3.68	3.68	3.40	0.1700	3.9	
2-gal bucket	2.26	4.40	2.34	3.60	3.60	3.38	0.1509	3.2	0.02216
2-gal bucket - Nails	4.22	3.73	3.13	4.01	2.76	3.06	0.4564	3.4	

[1] median values where n=6

[2] overall Kruskal-Wallis p-value for multiple comparison across pause periods for each filter type

[3] geometric mean of removal across all pause periods, n=36

[4] Kruskal-Wallis p-value comparing configurations (i.e., with and without nails) for each filter size

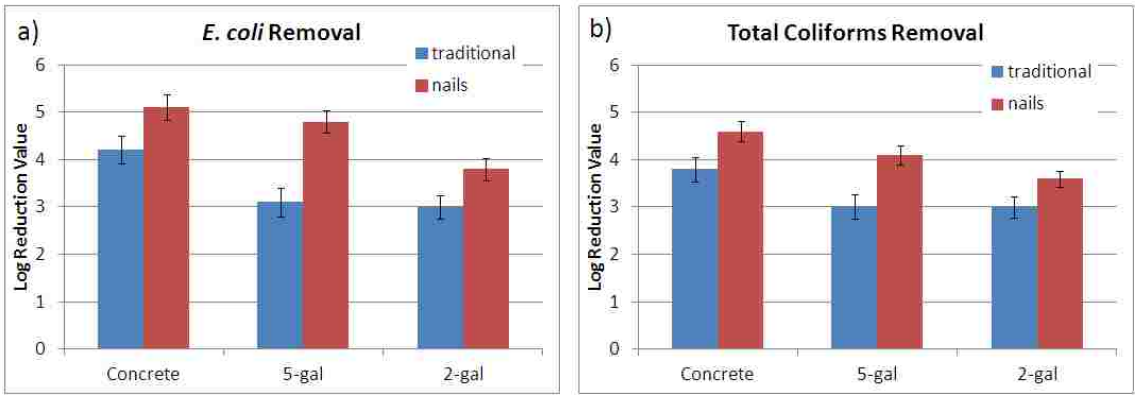

Figure 40. Median removal values for traditional and modified filters configurations for a) *E. coli* and b) Total Coliforms. Each bar represents the median value of 36 values (i.e., the results from three trials for duplicate filters over six pause periods) and error bars represent the standard error (stdev/√n).

5.4 Discussion

In general, virus removal in slow sand filters is primarily attributed to absorption and inferred to increase with depth of bed as removal rates that closely follow the Freundlich isotherm prediction (ASCE 1991). Increasing the pause period was shown to increase MS2 coliphage removal for all filter sizes, and this relationship was generally stronger for filters without nails. Contaminant particle size, sand grain size, and pore water velocity are the primary parameters that influence contaminant removal in filters. For sub-micron particles, such as viruses, molecular diffusion due to Brownian motion is the primary transport mechanism. As pause periods increased, removal also increased, likely due in part to the additional time for diffusion of the virus particles to the sand surface.

Most BSF studies evaluate performance from a single charge per day, or a 24 hr pause period. In this study, all three filter sizes investigated (without the addition of nails)

were able to achieve >2 $\log_{10}$ MS2 coliphage removals with a 24 hr pause period (**Table 12**). Maintaining the same pause period and adding nails to the filter increased MS2 removal in the concrete and 5-gal filters by >1 $\log_{10}$; there was no significant ($p>0.05$) change in MS2 coliphage removal in the 2-gal filters from the addition of nails. MS2 removals observed for the traditional filter configurations (no nails) are in line with previous work that showed MS2 coliphage removal ranging from 2 $\log_{10}$ to >4 $\log_{10}$ in a BSF column study (Elliot et al., 2011). In another previous study, a 60 L plastic-cased BSF with a 40 cm sand bed exhibited fluctuating MS2 removal levels during the first 150 days of operation, between 1 $\log_{10}$ and 3 $\log_{10}$ (Bradley et al., 2011), which is comparable with the 2.3 $\log_{10}$ removal observed in this study for the concrete BSF at the 24 hr pause period.

The low isoelectric point of MS2 coliphage (3.5-3.9) results in repulsive electrostatic forces between the virus particles and negatively-charged sand particles. It is important to note that MS2 coliphage has a lower isoelectric point relative to other virus types (e.g., echoviruses have isoelectric points in the range of 5.0-6.4), which increases the difficulty with which it is removed in filtration units. For this reason, MS2 coliphage is often chosen as a challenge organism for testing, as the results reported for its removal are likely lower than would be observed for other virus types with higher isoelectric points. For the filters with nails, the introduction of positively-charged iron oxide particles led to increased MS2 removal via sorption onto the iron oxide particles by attractive electrostatic forces and capture within the filter bed. Increased microbial removal, for bacteria and viruses, has been demonstrated in other ferric-sand

environments (Mills et al., 1994; Lukasik et al., 1999; Bradley et al., 2011), where the electrostatic repulsion was reduced by modification and/or introduction of positively charged surfaces.

For all filters tested, improved virus removal correlated significantly with increased pause period (**Table 13; Figure 35**). Over the majority of pause periods, a significant positive correlation between sand bed depth and virus removal was observed for the filters with nails, suggesting that iron-enhanced virus removal occurred throughout the entire bed depth. However, only at the 6 and 12 hr pause periods was a significant correlation between MS2 coliphage removal and pause period observed for the filters without nails. Pause periods of 1 and 3 hrs were presumably too short for even deeper bed depths to enhance virus removal. Pause periods of 24 and 72 hrs were presumably long enough that maximum virus removal could be achieved with the shortest bed depths tested. The results confirm that the presence of iron oxide in the system improved the MS2 removal performance of the filters,

Bacteria removal for each filter type was not impacted by the pause periods tested in this study (**Table 16).** The filters with nails yielded statistically ($p<0.05$) higher bacterial removal rates than those without nails for each filter size (**Table 16**). For micron-sized particles, like bacteria, removal in the filter bed is dominated by interception (i.e., particle collision with the sand grains) and sedimentation. The fact that bacterial deposition has been shown to be a reversible process (Lukasik et al., 1999) could explain the lack of observed correlation between increasing removal rates and increasing pause periods.

The increased bacterial removal in filters modified with nails could be attributed to the electrostatic attraction of the bacteria to the iron oxides, neutralizing the negative charge of the bacteria and enhancing their adsorption onto negatively-charged sand grains (Lukasik et al. 1999, Mills et al. 1994). The increased removal could also be attributed to the bactericidal effect of the iron oxide. While iron is an essential micronutrient, high iron concentrations have been shown to have toxic biological effects, reportedly inducing oxidative stress and irreversible damage to protein and DNA during the growth phase (Abdul-Tehrani et al., 1999; Liochev 1999). However, susceptibility to iron toxicity is not universal; the presence of iron oxide in nutrient-limited environments (as found in rusting distribution pipes) was shown to increase the survivability of starved and aging *E. coli* cells (Grandjean et al., 2005).

All BSF types exhibited >3 $\log_{10}$ *C. parvum* removal rates for each of the pause periods tested. The concrete and 5-gal bucket filters yielded >4 $\log_{10}$ *C. parvum* removals over the range of pause periods (geometric mean of all pause periods tested; **Table 15**). These results are comparable to the 3 $\log_{10}$ removal of *Cryptosporidium* oocysts in traditional BSFs reported by Palmateer et al. (1999). No appreciable correlation between *C. parvum* removal and pause period was identified, and the addition of nails did not significantly increase *C. parvum* $\log_{10}$ reduction values for the concrete and 5-gal bucket filters. However, a significant increase in *C. parvum* $\log_{10}$ removal (from 3.7 to 4.4) was observed when nails were added to the 2-gal bucket filters (Table 5).

Past research has shown that collision efficiency will vary for different microbial communities (Abramson and Brown, 2007; Tong et al., 2005; Tufenkji and Elimelch,

2004) and with changes in the solution chemistry, such as ionic strength (Abramson and Brown, 2007). Relative to ionic strength, hardness is an easy parameter to measure in the field and can affect adhesion properties due to the amount of divalent cations; large concentrations of divalent cations correspond greater water hardness. The influent water used for this study had consistently high (overall average 342 mg/L as $CaCO_3$, **Table 11**) hardness concentrations. High levels of hardness have been shown to increase microbial adhesion for waters (Huysman and Verstraete, 1993). The increased adhesion in the presence of large concentrations of divalent cations is due to the reduction of the electric double layer is explained by the DLVO theory (Hermansson, 1999; Stevik, et al., 2004). Thus, the high microbial removal levels are likely, at least, in part due to the hardness of the influent waters.

It is recommended that BSFs are filled a minimum of once per day (CAWST 2012). The 12 L of water produced from one fill of the concrete BSF would be the minimum required to sustain a family of six (2 L per person; WHO 2006) with an adequate volume of drinking water for the day. However, multiple fills per day would be required to obtain the same volume of drinking water from the smaller BSFs. If users filled the smaller units three times per day, a 6-hr pause period is a reasonable approximation of expected operating conditions. The traditional concrete BSF (no nails) operated at a 6-hr pause period achieved a median virus removal of 2.42 $\log_{10}$ (**Table 12**). The unmodified (no nails) 5-gal and 2-gal bucket BSFs required a pause period of 12 hrs to obtain a similar level (>2 $\log_{10}$) of MS2 removal (**Table 12**), which would yield only 7.2 and 3.0 L of water in a 24 hr period. However, with the addition of nails to the filter,

the 5-gal bucket BSF achieved >2 $\log_{10}$ removal at a 3 hr pause period, and the 2-gal bucket filter achieved 1.8 $\log_{10}$ removal at a 6 hr pause. **Table 17** outlines the pause period recommendations for MS2 coliphage removal for the various filter types based on the performance of the traditional concrete filter operated at a 6 hr pause period. Based on the Kruskal-Wallis test on the median MS2 coliphage removal, the given pause periods were not significantly different ($p>0.05$) to the 6 hr pause period removal for the concrete filters.

Table 17. Minimum pause period recommendations for various filter types (size/configuration). Virus recommendation based on an assumed baseline value of 2-3 $\log_{10}$ removal for a traditional (no nails) concrete filter at a 6-hr pause period.

	Virus	Bacteria	Cyst
	2-3 log10	3-4 log10	3-4 log10
Concrete	6	1	1
Concrete - Nails	1	1	1
5-gal bucket	12	1	1
5-gal bucket - Nails	1	1	1
2-gal bucket	12	1	1
2-gal bucket - Nails	6	1	1

These are the first full-scale laboratory tests which confirm that biosand filtration can be effective over a range of pause periods and sand bed depths for removal of protozoan cysts, bacteria, and viruses. The addition of nails to the filters improved virus and bacteria removals in all three filter sizes tested and improved protozoan cyst removal in the 2-gal bucket filters. However, the addition of nails to the filters increased the maintenance required; filters required more frequent cleanings and the volume of water and amount of time required for cleaning was significantly increased. The additional time and increased water requirement would not be practical for most households where BSFs would be deployed. Future work should focus on evaluating the potential to reduce the

amount of nails to the diffuser basin and identifying other means of introducing the iron into the filter system and in particular focus on options that utilize materials that would be readily available in the area where the filters are to be deployed. One option would be evaluating the potential of regional sand types with differing surface properties (e.g., surface potential) and the resulting effects on the removal of contaminants, in particular the sub-micron particles, such as viruses.

Successful and sustainable household water treatment interventions depend on a number of variables, including source water contaminants and end-user volume requirements. Biosand filtration offers the potential for tailoring the solution to the specific needs of a community. In particular, scaled down BSFs could present a viable option for some of the millions of people that still lack access to an improved water source and aid in the attainment of the Millennium Development Goals.

CAWST version 10 Concrete BSF

This research is one of the first comprehensive laboratory performance studies of the standard CAWST version 10 concrete BSF introduced in 2009. The results of this configuration are particularly important because approximately 100,000 of this version of the BSF are in use in households today (out of a total 550,000 BSF) (CAWST 2013) and the following summary provided to assist in future implementation efforts. The iron-amendment to this same filter design is important as well because the cost of the nails is relatively little compared to the full cost of BSFs, and since the nails are simply placed inside the diffuser basin, amending the BSF in this fashion would not be difficult to do if deemed feasible in terms of end-user maintenance requirements.

The traditional (i.e., sand-only) concrete v10 BSF averaged 99.98% (3.8 $\log_{10}$) *E. coli* bacteria removal. The BSF was effective in removing over 99.9% (3 $\log_{10}$) of the bacteria for all pause periods up to 24 hours (1, 3, 6, 12 and 24 hours), with the 72 hour pause period at 99.7%. Removal of *C. parvum* was over 99.99% (4 $\log_{10}$) for all 6 pause periods tested. Virus reduction (MS2 coliphage) was lowest for the 1 and 3 hour pause periods (85% and 51% respectively) but over 99% (2 $\log_{10}$) for the remainder of the pause periods (6, 12, 24, and 72 hours).

The modified concrete v10 BSF (i.e., with nails) exhibited significant bacteria removal efficacy at greater than 99.99% for all pause periods (median of 99.999% or 5.1 $\log_{10}$). Virus reduction for the modified concrete BSF was greater than 10 times more effective over the short pause periods (1, 3 and 6 hours) compared to the traditional sand-only BSF. The virus removal for the modified concrete v10 BSF, with the nails added to the diffuser basin, varied from 99.9% to 99.9999% (3 $\log_{10}$ to 6 $\log_{10}$) over all pause periods. Protozoan removal for the modified configuration was similar to the sand-only filter and was over 99.99% (4 $\log_{10}$) in all experiments.

5.5 Conclusions

The results from this study further substantiate that BSFs are effective at removing microbial contaminants over a range of sand bed depths and identified that the effect of pause period on removal rate was dependent on the type of microbial contaminant. Of the challenge organisms used during this testing, only the MS2 coliphage removal was dependent on sand bed depth and pause period. Furthermore, the

addition of iron nails to the filters significantly increased MS2 removal for all sand bed depths.

All BSF types exhibited >3 $\log_{10}$ *C. parvum* removal rates for each of the pause periods tested. No correlation between pause period and *C. parvum* removal was observed. Only the shortest bed depth exhibited a significant improvement in *C. parvum* removal with the addition of nails.

No correlation between pause period and bacteria removal was observed. However, BSFs with nails exhibited significantly higher bacteria removal rates for all sand bed depths. Bacteria removal rates ranged from >2 $\log_{10}$ to >5 $\log_{10}$ depending on sand bed depth, pause period and configuration (i.e., with nails).

6.0 Field Evaluation of Plastic-cased Filters in Nicaragua

A field study of large and small plastic-cased biosand (BSF) filters was conducted in four rural communities near San Juan del Sur, Nicaragua. Two types of household BSFs were built, installed, and monitored over a six-month period: a large BSF made from (10in diameter) PVC pipe and a smaller one made from a 5-gallon plastic bucket. The objective was to assess 1) the effectiveness of plastic casing biosand filters (BSFs) for improving water quality, 2) user acceptability and use, and 3) operational performance of the units. Source water and treated water, from the filter exit and from the safe storage bucket (SSB), were tested for *E. coli* concentrations. From the results of this study, the 5-gal bucket and PVC BSFs performed similarly ($p>0.05$) with respect to *E. coli* removal. After approximately six months of use, the median log reduction values (LRVs) for treated water from the filter and the SSB were 1.73 and 1.18 for the bucket BSFs, respectively, and 0.95 and 0.70 for the PVC BSFs, respectively.

6.1 Background

In an effort to improve overall health and reduce incidence of waterborne disease, household water treatments are employed in developing countries where economic and/or logistical impediments make community-based treatment systems unfeasible (Baker et al. 2006; Fewster et al. 2004; Samaritan's Purse Canada 2002; Clasen et al. 2007; Sobsey 2002).One of the major benefits of the BSF is the demonstrated long-term adoption of the technology by the end user (Sobsey et al. 2008). Laboratory studies on full-scale filters have shown that depending on design specifications (e.g., sand size distribution, depth of

sand layer) and operating parameters (e.g., HLR, pause period, influent bacteria concentration), log reduction values (LRVs) can range from 2-5 for bacteria (Sections 4.0 and 6.0; Elliott et al. 2008; Hijnen et al. 2004; Baumgartner et al. 2007; Stauber et al. 2006; Baumgartner et al. 2007, Jenkins et al. 2011).

While shown to be effective at improving water quality, the concrete BSF can be extremely difficult to transport in rural settings and can inhibit implementation into the most remote and poorest communities, which are often those most in need of an intervention. This study focused on evaluating two alternative BSFs designs: a large scale filter cased in a 10-inch PVC water pipe and the previously described 5-gal bucket-sized filter. The major difference between the pipe and bucket filter designs was the depth of the sand layer, which is approximately 54 cm and 15 cm, respectively.

6.2 Introduction

A collaborative effort between Lehigh and Tufts Universities, this project was developed and executed in conjunction with support from the Newton/San Juan del Sur Sister City Project (SCP) and the non-governmental organization (NGO) Fundacion Tierra. The SCP has been working on public improvement projects in the San Juan del Sur area for over 20 years, building schools, houses, and smoke-free cook stoves in addition to BSFs. Since 2007, the SCP, with support from Fundacion Tierra, has installed over 600 concrete BSFs in the San Juan area. Filter recipients report markedly lower levels of illness and other communities requested their assistance to help provide them with filters. Some of the communities requesting filters are located in very remote regions

with rough terrain making transportation of the large concrete casings and volumes of filter media very difficult.

The SCP executed a pilot project to manufacture and install a new model of BSF made entirely of lightweight PVC in January 2012 with 12 families receiving filters. This study was the follow-up to that initial project, a programmatic evaluation of the implementation of 90 BSFs, made from either PVC pipe or locally-available plastic buckets. Participating households were chosen by Fundacion Terra in conjunction with the local health center and input from the Ministry of Health based on the communities' need and willingness to participate. Households were surveyed over a six-month period to evaluate 1) changes in water quality with respect to bacteria concentrations and 2) user acceptability of units as determined by a series of surveys to quantify ease of use, consistency of use, filter durability, and maintenance issues.

This field study was conducted to assess 1) the effectiveness of plastic casing biosand filters (BSFs) for improving water quality, 2) user acceptability and use, and 3) operational performance of the units. Two types of household BSFs were built, installed, and monitored over a three month period in four rural communities near San Juan del Sur, specifically a large filter made from (10in diameter) PVC pipe and a small filter made from a 5-gallon plastic bucket. The plastic BSFs were designed based on the proportions of the CAWST v10 concrete BSF (CAWST 2012), that is there were proportionally designed with respect to filter media layers (i.e., sand, rock, and gravel) with the major differences between the types being the sand bed depths and reservoir

volumes, which were 54cm and 15cm, and 12L and 3.6L for the PVC (large) and bucket (small) BSFs, respectively.

The initial field visit was conducted in January 2013 and activities included 1) conducting baseline surveys and analyzing source water samples for all households, 2) procuring materials and building the 5-gal bucket BSFs, 3) installation of the 5-gal bucket BSFs, and 4) conducting the first follow up surveys and analyzing water samples for the bucket BSF households. Another visit was conducted in February 2013 to build and install the 60 large PVC BSFs. During the February 2013 visit, follow-up visits were also conducted several days after installation; however no water samples were obtained or analyzed. The second and third follow-up visits were conducted in March and July of 2013, respectively, during which all households (both bucket and PVC BSF recipients) were visited, surveys conducted, and water samples collected and analyzed.

6.3 Materials & Methods

6.3.1 Test Location

Four rural communities within the municipality of San Juan del Sur, which is located in the Rivas department in the south of Nicaragua, were selected to participate in the study based on their need (as identified by the Ministry of Health) and their willingness to participate (**Figure 41**). Of the four communities that received filters, communities A and B were selected to receive the small, 5-gal bucket BSFs; they are located south-east of San Juan del Sur bordering Lake Nicaragua. Communities C and D

received the large, PVC BSFs and are located south of San Juan del Sur near the Pacific coast. The designation of filter type for each community was determined by the SCP.

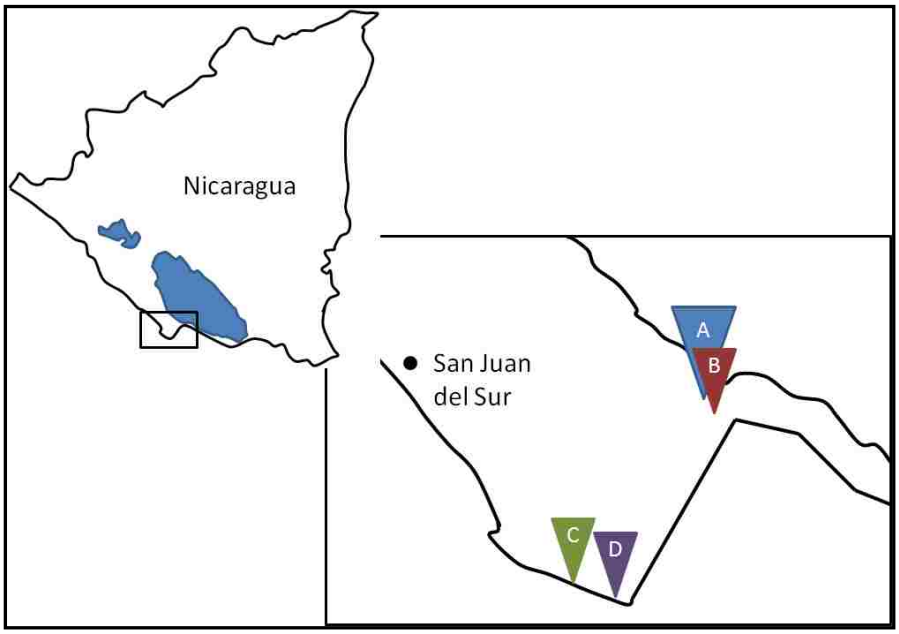

Figure 41. Map of BSF recipient communities. Communities A and B received the 5-gal bucket BSFs and communities C and D received the PVC BSFs.

6.3.2 Surveys & Sample Collection

Initially each household was surveyed (baseline) to obtain information on socio-economic status, water source(s), treatment system(s), and/or water storage containers, and also to collect drinking water samples. Following installation each household was visited to conduct a follow-up survey to obtain information on ease of use, functionality, and user acceptability. In addition, three water samples were collected at each household: 1) an untreated water sample from inside the home, 2) a sample from the outlet spout of the filter, and 3) a sample from the tap of the safe storage bucket.Testing the water at both the filter outlet and from the safe storage bucket was conducted to identify an defenicies in proper maintenance (i.e., proper and regular cleaning) of the entire

treatment system that would render the technology ineffective. All water samples were analyzed for bacteria concentrations at a field laboratory site setup at the Fundacion Tierra offices in the city of San Juan del Sur.

Baseline. During two separate trips, one in January and the other in February 2013, the project team constructed the BSF casings, prepared the filter media and then transported the casings and media to the communities and oversaw the installation of the BSFs at each household. Prior to installation, the project team's enumerators with assistance from the community leader conducted an educational training session with all BSF recipients. During the training, the connection between proper hygiene and sanitation to water quality and the subsequent influence on health/disease was reviewed. The team reviewed how the BSFs work, including proper maintenance and operation, and demonstrated how to install the various media layers of the filters.

The team installed 82 of the targeted 90 BSFs, resulting in a 91% installation rate. **Table 18** delineates the numbers and percentages of the number of BSFs that were targeted (per the test plan) and successfully installed. Of the total 82 installed filters, 35% (29/82) were 5-gal bucket BSFs and 65% (53/82) were PVC BSFs.

Table 18. Numbers of BSFs targeted (per the test plan) and installed by community.

	Community				
	A	B	C	D	Total
Targeted	10	20	30	30	90
	11%	22%	33%	33%	
	5-gal Filters		PVC filters		
	30		60		
	33%		67%		
Installed	9	20	30	23	82
	11%	24%	37%	28%	
	5-gal Filters		PVC filters		
	29		53		
	35%		65%		

Follow Up Visits. The first follow-up visit was conducted 1-2 days after installation for the households that received the 5-gal bucket BSFs. For the PVC BSFs, the first follow-up visit was conducted 1-2 weeks following installation. Due to personnel and time constraints, water samples were only collected from the 5-gal BSF households. For the initial survey, samples were taken from the source water storage vessel inside the house and from the safe storage bucket that is used to collect water coming out of the filter. The second and third follow up visits were conducted in March and July of 2013, respectively. Water samples were collected from the source (in the bucket/container that was used to fill the BSF), directly from the outlet spout of the filter, and from the tap of the SSB; **Figure 42** depicts the water sample locations. A summary of the water samples collected for each community during each site visit is outlined in **Table 19**.

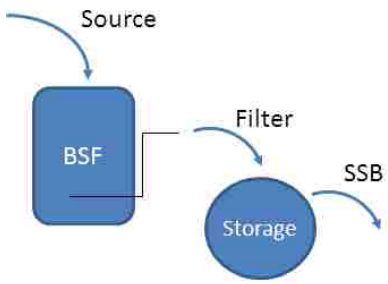

Figure 42. Water sample type/location.

Table 19. Number and type of water samples collected and analyzed for *E. coli*.

		Community				Community			
		A	B	C	D	5-gal Filters		PVC Filters	
Installed filters		**9**	**20**	**30**	**23**	**29**		**53**	
Baseline (January 2013)	Source	10	20	15	20	30	103%	35	66%
First Follow Up (January 2013)	Source	7	19	—	—	26	90%	0	0%
	Filter	—	—	—	—	0	0%	0	0%
	SSB	6	19	—	—	25	86%	0	0%
Second Follow Up (March 2013)	Source	9	20	27	18	29	100%	45	85%
	Filter	9	20	27	17	29	100%	44	83%
	SSB	9	20	27	18	29	100%	45	85%
Third Follow Up (July 2013)	Source	8	18	26	23	26	90%	49	92%
	Filter	9	16	24	21	25	86%	45	85%
	SSB	9	16	24	21	25	86%	45	85%

6.3.3 Microbial Analysis

Approximately 100ml water samples were collected in Whirl-Pak® bags (Nasco, Fort Atkinson, Wisconsin), stored on ice, and analyzed within 6-8hrs from time of collection. All samples were analyzed via membrane filtration (MF) for *E. coli* and total coliforms (Rice et al. 2012). Field membrane filtration units (Microfil Stand Alone Support, Millipore Corp., Billerica, MA) with syringe vacuum source were used to filter samples through a 47mm, 0.45um pore size cellulose ester membrane filter (Millipore Corp., Billerica, MA). Two volumes of each sample were filtered (e.g., volumes of 1, 5, 10, or 50 mL) and analyzed for bacterial growth after 24-hr incubation period at 35 ±

0.5°C using m-ColiBlue 24® broth. Bacteria concentrations were calculated based on the number of colonies observed per volume of sample filtered to yield results in CFUs/100ml. If more than one plate could be enumerated (including duplicates), concentrations were averaged to yield average *E. coli* concentration. For all test days, 10% duplicates and 20% blanks were performed.

Water samples were tested and categorized by risk posed to human health associated with *E. coli* concentrations. There are five risk levels are based on WHO guidelines (WHO 1997) and are defined as follows: 1) conforms: 0-<1 CFU/100ml, 2) low risk: 1-<10 CFUs/100ml, 3) moderate risk: 10-<100 CFUs/100ml, 4) high risk: 100-<1000 CFUs/100ml, and 5) very high risk: ≥1000 CFUs/100ml.

To compare performance of the two BSF types, log reduction rates, or log reduction values (LRVs) were calculated based on *E. coli* concentrations recorded for source water, filter effluent, and the safe storage bucket according to **EQN 3 (as previously described in Section 3.3.5)**:

$$log_{10}Removal = -log_{10}(\frac{C_{eff}}{C_{inf}}) \qquad \textbf{(EQN 3)}$$

where, C_{inf} = concentration or turbidity of the influent (CFU/100mL or NTU)

C_{eff} = concentration or turbidity of the effluent (CFU/100mL or NTU)

6.4 Results & Discussion

6.4.1 Baseline Results

Source water samples from 65 households (**Table 19**) were collected during the baseline survey and tested, specifically 10, 20, 15, and 20 source samples were collected from communities A, B, C and D, respectively. Six source water types were identified: closed well, open well, surface water, bottled water, spring, and water system. Closed and open wells were defined as hand-dug, open pit water wells with concrete well heads, differing in the presence and type of ground surface enclosure to keep out foreign materials (e.g., animals, tree litter, etc.) Open wells either had no enclosure at all or one that was not permanent, such as a piece of board, whereas for closed wells the enclosure was permanent and most often was constructed from concrete. Springs were defined as shallow water sources, typically located near a surface water source; these were shallow wells with no constructed well head. The water system was sourced from a drilled (deep) well located in the highlands outside the community with a piped distribution system to the community and individual households. **Figure 43** shows the breakdown of source water type for each community.

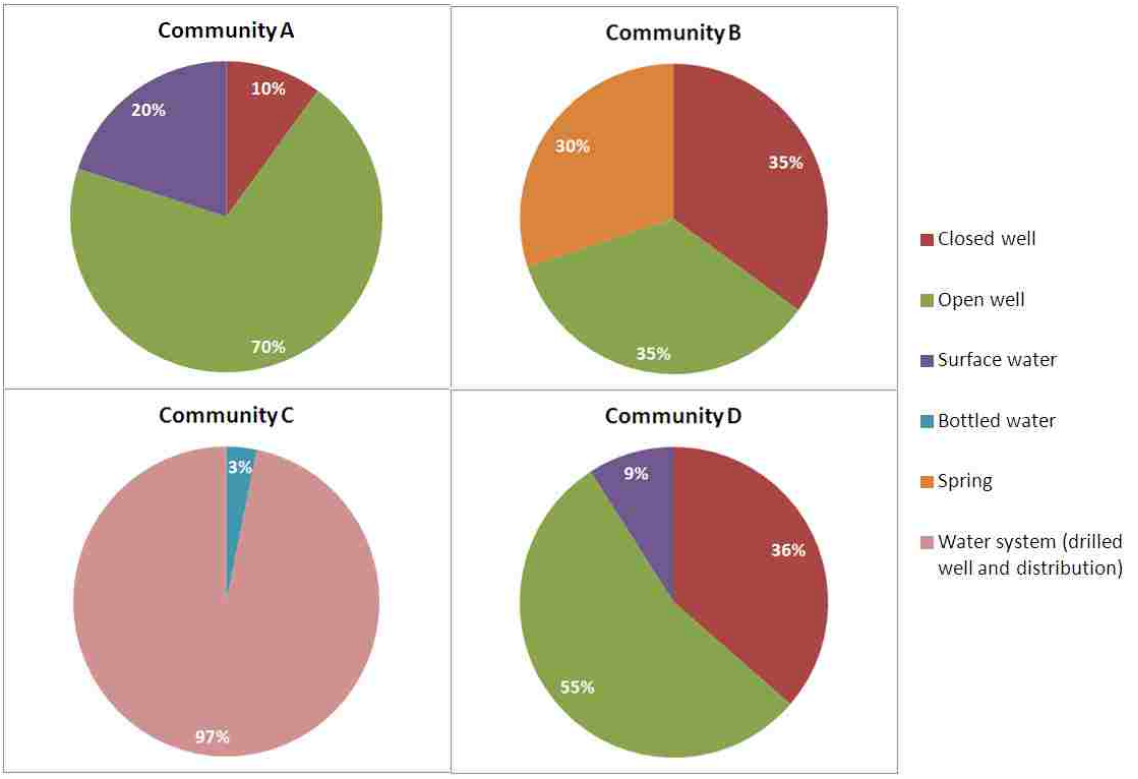

Figure 43. Source type of baseline samples for all four test communities (A, B, C, D).

The source water from the majority of households in communities A, B, and C came from open and closed wells, representing 80%, 70%, and 91% of the samples, respectively. The remainder of the samples from these communities came from surface water, either directly or from springs. The water system was located in Community C; only one household in that community did not use the tapped water as a drinking water source and reportedly used bottled water. **Figure 44** combines the source type data to show the distribution based on the type of BSF each community received, thus combining the data for communities A and B (the 5-gal BSF communities) and communities C and D (the PVC communities). For the 5-gal BSF communities, 73% of source water was from wells (27% closed and 46% open) and the remaining 27% were from surface water

sources (with 20% of those from springs). For the PVC communities, the majority of source samples were from a water system, consisting of 56% of the samples, where the majority of the remaining source samples were from wells (15% from closed and 23% from open). Surface water and bottled water sources represented small percentages of households at 4% and 2%, respectively.

The reported water source for the four communities also displays a similar trend, (**Figure 43**). For communities A, B, and D, the majority of households reported wells were their primary water source, at 80%, 70%, and 91% (for closed and open wells), respectively. In contrast, community C had a community drilled well with a piped distribution system; the majority (97%) of households surveyed reported using tap water for drinking and only one household (3%) reported using purchased bottled water.

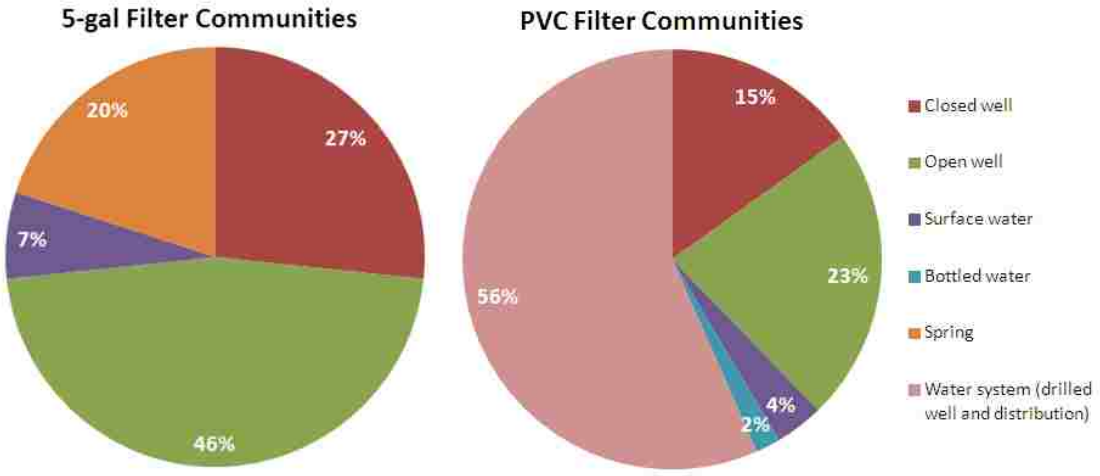

Figure 44. Source type of baseline source samples by BSF type, 5-gal bucket communities (A and B) and PVC communities (C and D).

The majority of the source samples from both the 5-gal and PVC communities presented a moderate to high risk based on *E. coli* concentration ≥10 CFUs/100ml, specifically 75% of all samples (49/65), see **Figure 45**. Communities A, B and D all exhibited positively skewed distributions (i.e., more samples in the higher risk categories. In Community A, all 10 source water samples tested contained *E. coli* concentrations ≥10 CFUs/100ml, with 40% (4/10), 50% (5/10), and 10% (1/10) of samples classified as moderate, high, and very high with respect to human health risk, respectively. The majority of source samples from Community B were also within the moderate and high risk levels at 55% (11/20) and 35% (7/20), respectively; two households, representing only 10% of the total from community B had *E. coli* concentrations that presented an acceptable level of risk to human health (<1CFUs/100ml). Source water samples from community D exhibited a similar trend to those from community B specifically the majority of samples were in the moderate and high risk levels with 50% (10/20) and 35% (7/20), respectively, with the remaining 15% (3/20) within the acceptable risk level.

The source water from community C displayed a negatively skewed distribution; all samples had concentrations <100 CFUs/100ml yielding no samples in either the high or very high risk classifications. The majority of samples from community C, 60% (9/15), had *E. coli* concentrations <1 CFU/100ml, 13% (2/15) were between 1-<10 CFUs/100ml and 27% (4/15) were between 10-<100 CFUs/100ml.

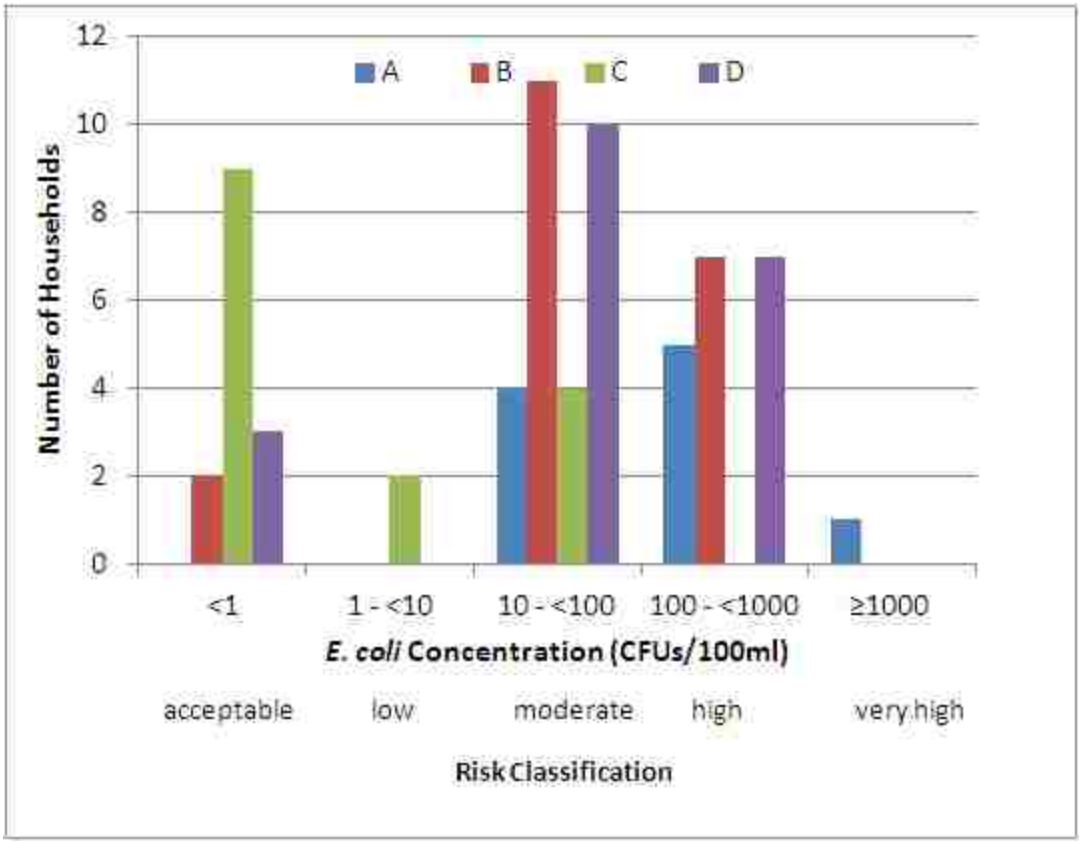

Figure 45. *E. coli* concentrations (CFUs/100ml) in untreated source water, where n = 10, 20, 17, and 18 for communities A, B, C, and D, respectively.

The baseline source water *E. coli* results were combined for communities that received the same BSF type (i.e., bucket BSFs: communities A and B and PVC BSFs: communities C and D) and the results presented in **Figure 46**. The moderate risk level contained the majority of baseline source samples for both BSF communities, with 50% and 40% for the bucket and PVC BSF communities, respectively. The remaining samples for the 5-gal bucket BSF communities were largely in the high risk level (40%), whereas for the PVC BSF communities 34% were considered acceptable with respect to *E. coli* concentration.

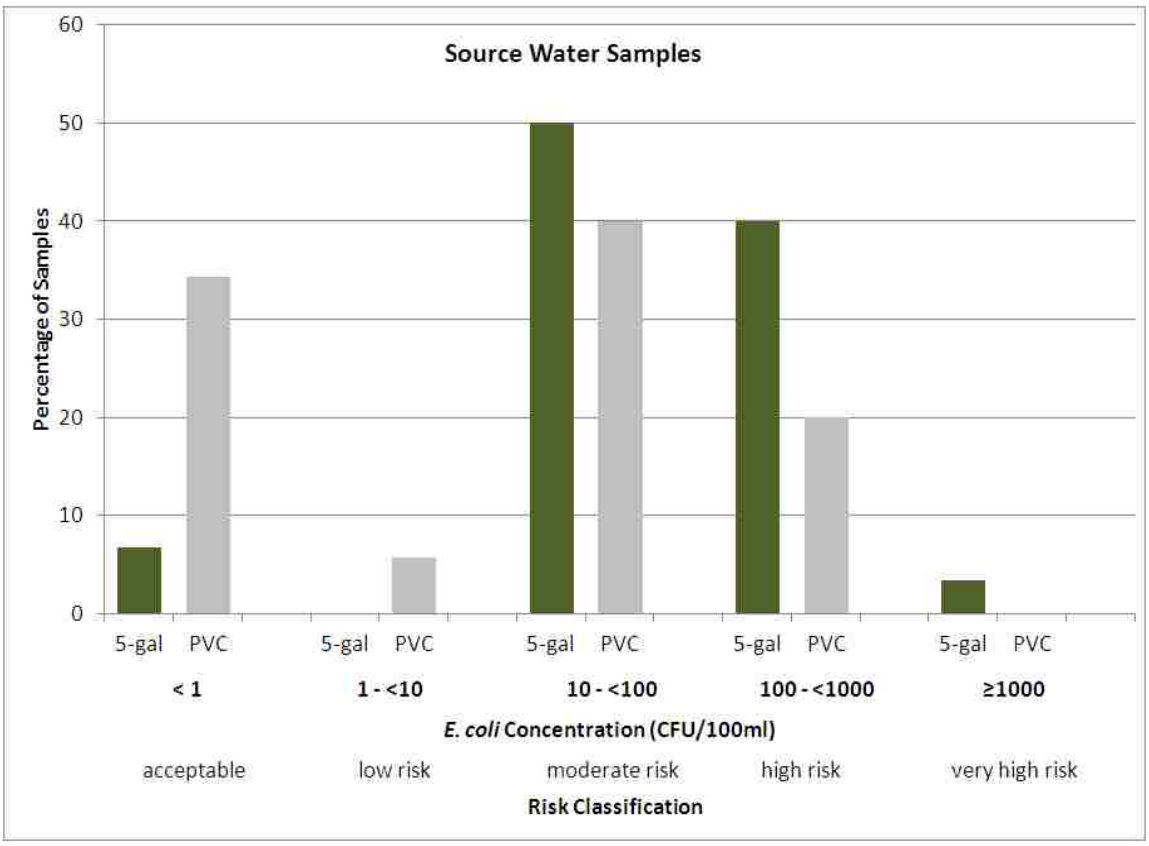

Figure 46. *E. coli* concentration and risk level of source water samples by BSF type.

6.4.2 First Follow-up (January 2013) Results

The water quality data from the first follow up visits, conducted 1-2 days after installation, were limited. Water samples were only collected from the 5-gal BSF communities (A and B) and only source water and SSB water were collected (**Table 19**). The summary of the data is shown in **Figure 47**. The majority of the source samples were represented moderate to very high risk with respect to human health. Specifically, 4% of samples had *E. coli* concentrations ≥1000 CFUs/100ml, 35% were between 100 to <1000 CFUs/100ml, and 46% were between 10 to <100 CFUs/ml, corresponding to very high, high, and moderate risk levels for human health, respectively. Comparing the source

water samples to those from the SSB showed that treatment and storage resulted in the elimination of samples with *E. coli* concentrations ≥1000 CFUs/100ml, but a slight increase in the percentage of samples that were between 100 to <1000 CFUs/100ml occurred, from 35% to 36%. In general, the treatment and storage of the water resulted in a decrease in the percentage of samples in the moderate to high risk levels and an increase in the percentages for the lower risk categories.

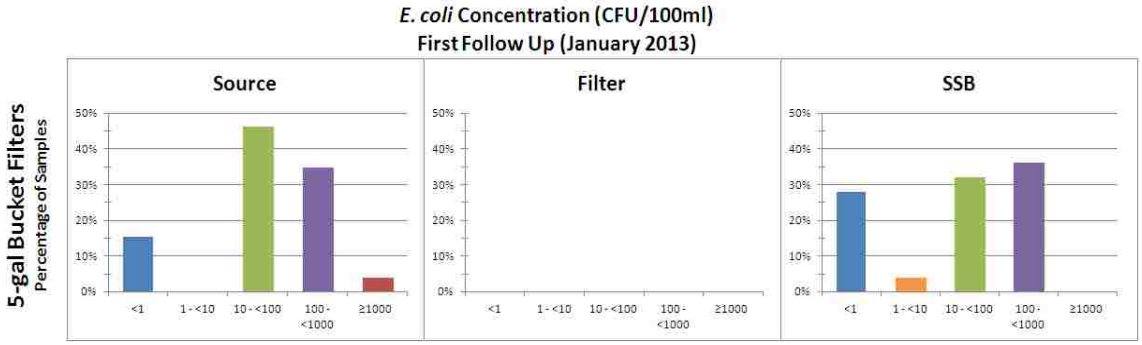

Figure 47. Comparison of *E. coli* concentrations for water sample types from the first follow-up visit in January 2013, conducted 1-2 days after installation. Percentages based on total number of samples analyzed. Data presented for the 5-gal bucket BSF communities (A and B) only; no data available for the PVC communities (C and D).

6.4.3 Second Follow-up (March 2013) Results

In March 2013, a second round of follow-up visits was conducted at both the bucket and PVC BSF communities. Of the 82 households visited, 90% (74/82) of BSFs were in use, 3% (2/82) were not in use, and the status of the remaining 7% (6/82) could not be ascertained (no one was home). As outlined in **Table 19**, 74 source water and safe storage bucket samples and 73 samples from the filter outlet were collected. The missing filter sample was at a residence where source water was unavailable at the time of the visit as the well was being cleaned (i.e., users empty out all the water and any debris and then allow the well to recharge).

A comparison of the *E. coli* concentrations for each sample type is presented in **Figure 48**. The 5-gal BSF communities had a larger percentage of source samples that were either high (100->1000 CFUs/100ml) or very high risk (≥1000 CFUs/100ml) as compared to the PVC communities, at 37% and 24%, respectively. Of the remaining source samples, the 5-gal BSF communities had 40% that were at moderate risk level (10-<100 CFUs/100ml) and 23% that were at an acceptable risk level (<10 CFUs/100ml), in comparison the PVC communities had 22% and 54% at moderate and acceptable risk levels, respectively.

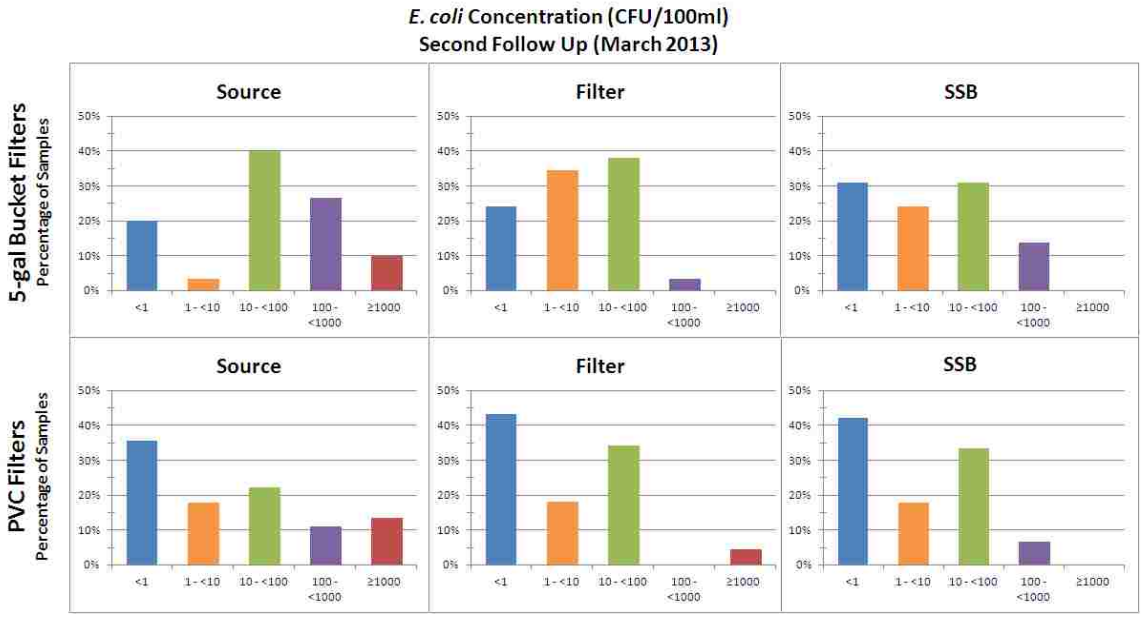

Figure 48. Comparison of *E. coli* concentrations for water sample types from the second follow-up visit in March 2013 (after 1-2 months of use). Percentages based on total number of samples analyzed (e.g., sum of bars for each plot equals 100%).

A greater percentage of both sample types from the PVC BSFs were within the recommended level of <1CFU/100ml than was observed for the 5-gal BSFs; the PVC

samples had 43% and 42%, from the filter and SSB, respectively. In comparison, the percentages of samples from the bucket BSFs with a concentration <1CFU/100ml were 24% and 31% from the filter and SSB, respectively. Conversely, the bucket BSFs had greater percentages of samples from both the filter and the SSB in the low risk level (1–<10) as compared to the PVC BSFs. Subsequently, the two BSF types exhibited similar percentages of samples that were <10 CFUs/100ml, for the 5-gal BSFs: 59% and 55% from the filter and SSB, respectively, and for the PVC BSFs: 61% and 60%, respectively.

Both BSF types exhibited similar percentages of samples in the moderate (10->100) and higher (≥100) risk levels. For the 5-gal BSFs, the percentages of filter and SSB samples in the moderate risk level were 38% and 31%, respectively, and correspondingly the sample percentages were 34% and 35% for the PVC BSFs. The higher risk levels contained 3% and 14% of the filter and SSB samples, respectively, from the 5-gal BSFs; whereas, the PVC BSFs had 3 and 7% of filter and SSB samples, respectively.

Comparing the source water to the water collected from the filter outlet, in general resulted in a decrease in the percentage of samples within the higher risk levels (>100 CFUs/100ml) and an increase in the lower risk levels (<10 CFUs/100ml). Specifically for the 5-gal BSFs, comparing the source to the filter, the percentage of samples decreased from 37% to 3% for the higher risk categories and increased from 23% to 59% in the lower risk categories. Correspondingly for the PVC BSFs, comparison of the source to filter, the overall percentage of samples in the higher risk levels decreased from 24% to 3% and in the lower risk levels increased from 54% to 61%. For the percentage of samples within the moderate risk level (10-<100 CFUs/100ml) comparing source to

filter, a slight decrease was observed for the 5-gal BSFs, from 40% to 38%; whereas for the PVC BSFs, the overall percentages increased in this risk category from 22% to 34%.

The general trends identified above when source to filter samples were compared were the same when source to SSB samples were compared; the percentages of samples in the lower risk levels increased, in the higher risk categories decreased, and were variable between BSF types for the moderate risk level. Specifically for the 5-gal BSFs, comparing the source to the SSB, the percentage of samples decreased from 37% to 14% for the higher risk categories and increased from 23% to 55% in the lower risk categories. Comparison of the source to filter samples for the PVC BSFs, the overall percentage of samples in the higher risk levels decreased from 24% to 7% and in the lower risk levels increased from 54% to 60%. For the moderate risk level (10-<100 CFUs/100ml) comparing source to filter, a decrease was observed for the 5-gal BSFs, from 40% to 31%; whereas for the PVC BSFs, the overall percentages increased in this risk category from 22% to 33%.

6.4.4 Third Follow Up (July 2013) Results

In July 2013, third and final follow-up visits were conducted with the communities. Of the 82 households visited, 85% (70/82) of BSFs were in use, 12% (10/82) were not in use, and the status of the remaining 3% (2/82) could not be ascertained (no one was home). As outlined in **Table 19**, 75 source water samples and 70 filter outlet and safe storage bucket samples were collected. For the 10 BSFs that were not in use, 4 were bucket BSFs and 6 were PVC BSFs. For the four bucket BSFs not in

use, one was broken (crack in the bottom of the bucket), two had blocked flow, and one was abandoned (owner moved away). For the six PVC BSFs not in use, one was abandoned (owner moved away), four were given away and moved to other households, and one was not being because it produced bad tasting water.

A comparison of the *E. coli* concentrations for each sample type is presented in **Figure 49**. The bucket BSF communities had a larger percentage of source samples that were either high (>100 CFUs/100ml) or very high risk (≥1000 CFUs/100ml) as compared to the PVC communities, at 77% and 24%, respectively. Of the remaining source samples, the bucket BSF communities had 15% that were at moderate risk level (10-<100 CFUs/100ml) and 8% that were at an acceptable risk level, in comparison the PVC communities had 41% and 35% at moderate and acceptable risk levels, respectively.

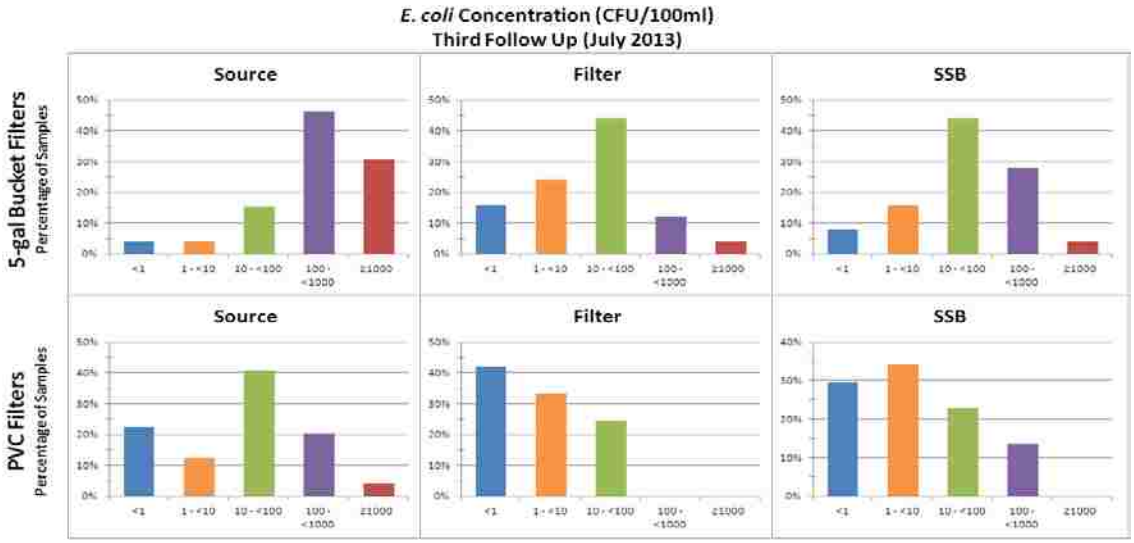

Figure 49. Comparison of *E. coli* concentrations for water sample types from the third (and final) follow-up visit conducted in July 2013 (after 5-6 months of use). Percentages based on total number of samples analyzed.

In general, for both filter and SSB samples, the PVC BSFs resulted in a greater percentage of samples at the lower risk levels (<10 CFUs/100ml) and a smaller percentage of samples in the higher risk levels (≥100 CFU/100ml) than observed for the bucket BSFs. For the PVC BSFs, 75% of the filter samples (33% at 1-<10 and 42% at <1) and 63% (34% at 1-<10 and 29% at <1) of the SSB samples were in the low risk level. In comparison, the percentages of the bucket BSFs samples in the lower risk levels were 40% (24% at 1-<10 and 16% at <1) and 24% (16% at 1-<10 and 8% at <1) from the filter and SSB, respectively. For the two higher risk levels, the PVC BSFs only had 14% of the SSB samples with *E. coli* concentrations from 100-1000 CFUs/100ml; there were no SSB samples in the very high level and there were no filter samples in either of the high risk levels. The bucket BSFs presented samples in both high risk levels for both filter and SSB samples. For the filter samples from the bucket BSFs, 12% and 4% were in the high and very high risk categories, respectively; for the SSB samples, 28% and 4% were in the high and very high risk categories, respectively. However, for the bucket BSFs, the largest percentages of samples were in the moderate risk level with results of 44% for samples from both the filter and the SSB.

The general trends identified during evaluation of the data from the second follow up (March 2013) were also observed for the data collected during the third follow up visit (July 2013); in particular, the percentages of samples in the lower risk levels increased, in the higher risk categories decreased, and in the moderate risk level increased for the bucket BSFs and decreased for the PVC BSFs.

Comparing the source to the filter samples for the bucket BSFs, the percentage of samples in the lower risk levels increased from 8% to 16% and in the moderate risk level increased from 15% to 44%, while the higher risk levels decreased from 77% to 16%. For the PVC BSFs, comparison of the source to filter samples also showed an increase in lower risk levels from 35% to 75% and decreases in both the higher risk levels, from 25% to 0, and the moderate risk level from 41% to 25%.

Percentages of source samples to SSB samples also followed the aforementioned trends. The distribution of percentages of the SSB samples from the bucket BSFs from the low to high risk categories were as follows: 8%, 16%, 44%, 28%, and 4%. And percentages for the SSB samples from the PVC BSFs were 23%, 34%, 23%, 14%, and 0% for the low to high risk levels, respectively.

From the preceding summation of the sample percentages represented in the various risk categories, a cursory review could yield the conclusion that the PVC BSFs were more effective at removing *E. coli* than the bucket BSFs. However, in both cases (i.e., for the second and third follow up datasets) the source water from the PVC BSF communities had large percentages of samples in the higher risk categories as compared to the bucket BSF communities. Thus, perhaps it was not that the PVC BSFs were more effective as compared to the bucket BSFs; rather that both types offered the same removal capability and the difference in the finished water quality distributions was a result of the difference in the concentrations of the source waters.

6.4.5 Log Reduction Values

To test the hypothesis that both BSF types offer a similar removal rate for *E. coli*, the LRVs of the individual filters was calculated and the resulting populations were compared across the test communities to identify if there was a statistical significance between the populations.

The calculated LRVs for each BSF type plotted by collection time (follow up visit), community (A, B, C, D), and treated sample type (filter, SSB) are presented in **Figure 50**. Overall, the LRVs ranged from -3.30 to 3.52, with the resultant median LRVs for each data set (sample type/community/date) ranging from 0.00 to 1.62. A Kruskal-Wallis test was performed on the data sets for each sample type and date (e.g., A, B, C, and D data sets from the plots in **Figure 50 b-f**) to identify significant differences in the median LRVs; **Table 20** outlines the summary parameters of the data sets and the resultant p-values.

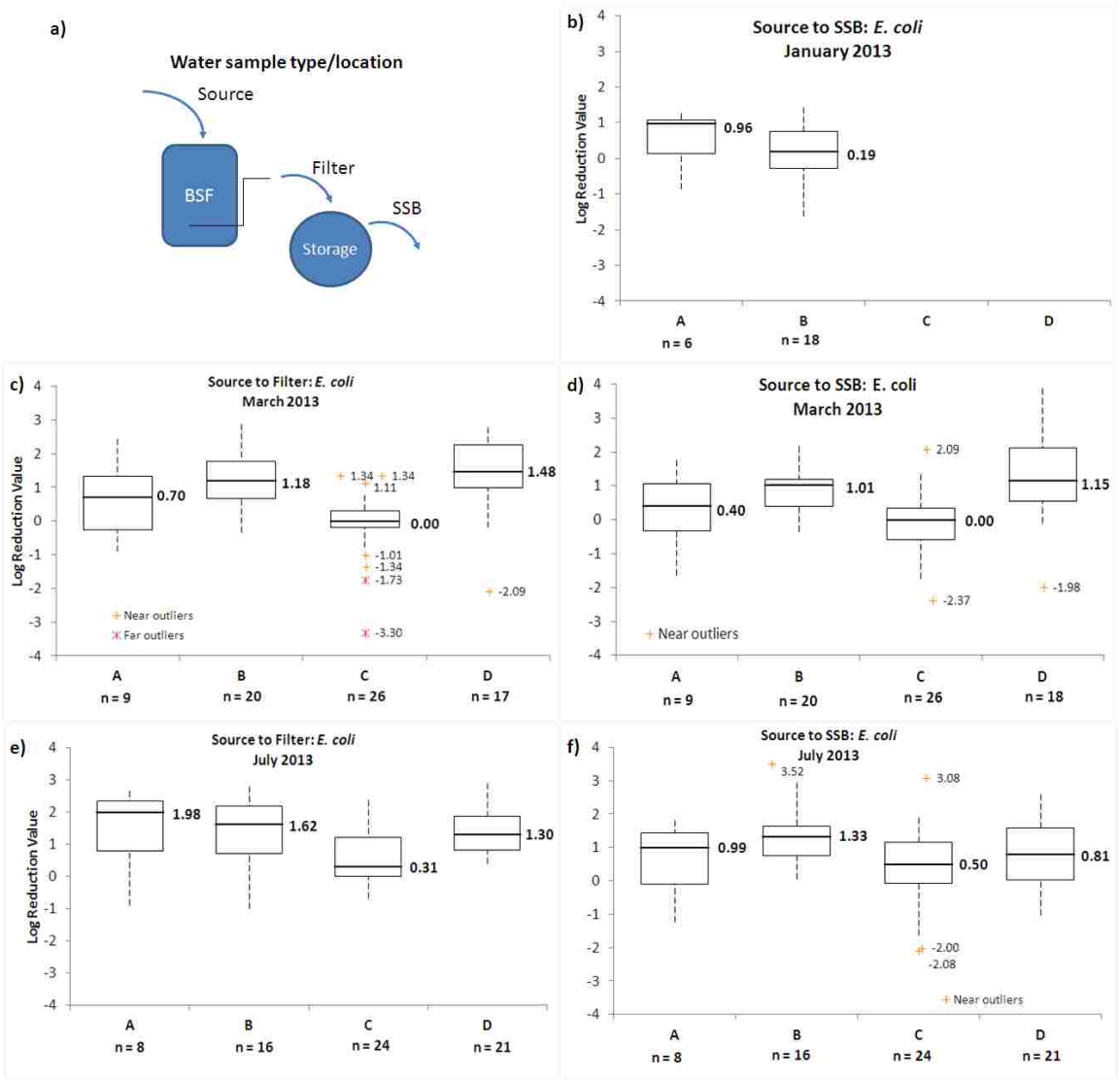

Figure 50. Comparison of *E. coli* log reduction values (LRVs) for source waters to filter exit samples (c and e) and source waters to safe storage bucket (SSB) samples (b, d, and f). Individual plots group data sets by sample type and collection date. Bold data values are the median LRVs and correspond to the bold center line within the box plots. Sample size, n, values are presented for each data set. Water sample type/location diagram (a) included for reference.

Table 20. Median *E. coli* log reduction values (LRVs), standard deviation, and sample size for filter and SSB samples by collection date and community/BSF type. Calculated Kruskal-Wallis p-values from comparison of median LRVs. Bold values indicate a significant difference in the median LRVs between two sample groups.

Community (BSF type)	Log Reduction Value	First Follow Up January 2013		Second Follow Up March 2013		Third follow Up July 2013	
		Filter	**SSB**	**Filter**	**SSB**	**Filter**	**SSB**
A (bucket)	Median	—	0.96	0.70	0.40	0.98	0.99
	stdev	—	*0.80*	*1.08*	*1.08*	*1.22*	*1.08*
	n	—	*6*	*9*	*9*	*8*	*8*
B (bucket)	Median	—	0.19	1.18	1.01	1.62	1.33
	stdev	—	*0.77*	*0.70*	*0.59*	*1.02*	*0.99*
	n	—	*18*	*20*	*20*	*16*	*16*
C (PVC)	Median	—	—	0.00	0.00	0.31	0.50
	stdev	—	—	*0.97*	*0.95*	*0.76*	*1.28*
	n	—	—	*26*	*26*	*24*	*24*
D (PVC)	Median	—	—	1.48	1.15	1.30	0.81
	stdev	—	—	*1.22*	*1.36*	*0.72*	*1.02*
	n	—	—	*17*	*18*	*21*	*21*
p-values							
overall		—	0.1096	**<0.0001**	**0.0004**	**0.0016**	0.0928
	A v B	—	0.1096	0.2889	0.6576	1.0000	1.0000
	A v C	—	—	0.6076	1.0000	**0.0197**	1.0000
	A v D	—	—	0.0775	0.3106	1.0000	1.0000
	B v C	—	—	**<0.0001**	**0.0019**	**0.0059**	0.0688
	B v D	—	—	1.0000	1.0000	1.0000	0.8065
	C v D	—	—	**<0.0001**	**0.0005**	**0.0083**	1.0000

From the first follow up (January 2013), there were only two data sets available for comparison: LRVs from source to SSB samples for the two bucket BSFs communities 1) A and 2) B; there was no significant difference in the median LRVs between these two data sets. For the second follow up (March 2013), comparison of the four data sets (by community/BSF type) for each LRV (i.e., filter and SSB) yielded similar results: There was no difference between communities B and D; A was similar to B, C, and D; however, C was significantly different from B and D. For the third follow up (July 2013),

the same analysis was performed and yielded the following results: the filter LRVs for community C were significantly different from the other communities (i.e., A, B, and D) whereas the SSB LRVs were all statistically similar.

A comparison test of the median values for the LRVs (by visit and sample type) for each community was also conducted, for the five data sets from communities A and B: 1-SSB, 2-filter, 2-SSB, 3-filter, and 3-SSB and the four data sets from communities C and D: 2-filter, 2-SSB, 3-filter, and 3-SSB. For communities A, C and D there was no significant difference in the LRVs from either the filter or the SSB (data not shown, all p-values >0.05). For community B, the SSB LRVs from the first follow up (1-SSB) data set was significantly different from three other data sets, specifically the filter LRVs from the second follow up (2-F, p=0.0003), the filter LRVs from the third follow up (3-F, p=<0.0001), the SSB LRVs from the third follow up (3-SSB, p=0.0003).

6.5 Conclusions

From the results of this study, the 5-gal bucket and PVC BSFs performed similarly with respect to *E. coli* removal. After approximately 6 months of use (third follow up visit), the median LRVs for treated water from the filter and the SSB were 1.73 and 1.18 for the bucket BSFs, respectively, and 0.95 and 0.70 for the PVC BSFs, respectively. These results are comparable to field results obtained by others for the performance of concrete BSFs (Sobsey et al. 2008, Fiore et al. 2010).

While the *E. coli* concentrations of the treated (either filter or SSB) samples from the bucket and PVC BSFs, at times, produced differing distributions (**Figures 48 and**

49), there was no statistically significant difference in the LRVs between BSF types. Community C exhibited slightly lower median LRVs (not always significant) as compared to the other communities. This was attributed to the fact that the source waters from community C had consistently lower *E. coli* concentrations.

Slight variations were observed in the *E. coli* concentrations between filter and SSB samples for the same BSF type (bucket or PVC). While there was no significant difference in the LRVs from filter to SSB for any of the communities, almost all median LRVs were less for SSB samples as compared to the filter samples. In addition, of the total 307 LRVs calculated, 16% (49/307) were negative (i.e., treated water concentration was greater than source water concentration). As shown in **Figure 50**, both types of BSFs yielded negative LRVs either from the filter and/or the SSB; the only exceptions were from the third follow up visit for SSB LRVs from community B and for filter LRVs from community D. Community C had the greatest number of negative LRV values, 45% of the total (22/49). Negative LRVs from communities A, B and D constituted 16% (8/49), 18% (9/49), and 20% (10/49) of the total, respectively. One of the disadvantages with the BSF technology (regardless of size or casing material) has always been the potential for recontamination of the filtered water either from the filter hose and/or from the SSB. The results of this study further support other recommendations that emphasis the need for household water programs to incorporate additional follow-up training with technology recipients (Lantagne and Clasen 2012).

While the removal rates for the BSFs in the field are substantially less than those reported from the laboratory studies, field conditions and parameters cannot be as

controlled as in laboratory environments. Thus, there were several limitations to the preceding data analysis. First, it was only an approximate measure of the LRVs of the BSFs. Because the source water and treated water samples were taken at the same time, a direct comparison of influent conditions to effluent conditions of the same water sample, as is done in a laboratory setting, could not be performed to evaluate the true filter performance. The actual source water that corresponded to the treated samples would have been previously charged to the filters and could potentially have different concentrations than the source sample collected. In addition, for the first follow up conducted in January, only source and SSB water samples were collected at the bucket BSF communities (A and B) therefore full data sets (with source, filter and SSB samples for both BSF types) were only available from the second (March 2013) and third (July 2013) follow up visits.

Other investigators have recognized the difficulty to replicate microbial reduction rates obtained in laboratory settings to in field performance (Elliot et al. 2008). For example, a field study of 55 concrete BSFs in the Dominican Republic demonstrated *E. coli* reductions of 84-88% (Stauber et al. 2009). However, the results obtained during this field study are similar to those from a previous SCP requested study to evaluate the performance of concrete BSFs, where the median filter efficiency was reportedly 80% (Fiore et al. 2010).

6.6 Observations & Recommendations

The primary source for community C is piped water from a drilled well and source water was consistently of higher quality with respect to *E. coli* concentration than for the other communities. However, biological contaminants are not always the only constituents of concern. For this particular community (C), one of the driving factors for the residents to request BSF from the SCP was a concern about chemical constituents in the drinking water. According to one of the filter recipients, the source water from the tap produced scaling in pots and pans (e.g., when boiling water for coffee). While chemical water quality analysis was outside the scope of this investigation, the filter recipient that acknowledged this concern produced her water pot, with no evidence of scaling, during the second follow up visit as proof that her BSF was working.

The ability of the BSFs to remove other chemicals of concern was also identified by residents of the 5-gal bucket BSF communities. A calcium mine is situated in close proximity to both of these communities (A and B), and was a motivating force for the residents to seek out a household treatment option. During the second follow up visit, several recipients reported they had observed contaminant accumulation (**Figure 51**) along the side of the bucket in the reservoir area; it was not typical turbidity settling that is observed on top of the sand bed but rather a hard, flaky deposit along the sides of the bucket. When asked if she would purchase another filter if something happened to hers, one of the recipients responded by saying: *Of course, look at all that stuff that we used to*

drink, that would be in our stomachs; I can't go back to drinking water that is not filtered.

Figure 51. Photo of characteristic contaminant deposition observed in the 5-gal bucket BSF communities. Photo credit: J. Napotnik

In March, another BSF recipient said that as the summer was progressing and the dry season was starting, her well water normally begins to smell foul but when put through the filter the water no longer smelled bad. This recipient offered glasses of source and treated water as visual evidence (**Figure 52**) of how well her filter was working. Thus, even though the focus of this study was on the *E. coli* levels of the water, the filters were clearly providing additional water quality benefits.

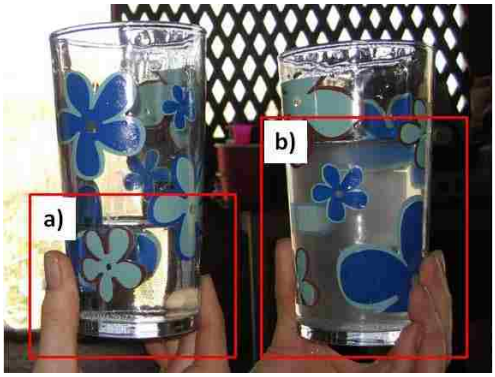

Figure 52. Visual comparison of a) treated (from the filter exit) and b) source waters from a 5-gal bucket BSF household. Photo Credit: J. Napotnik

7.0 Conclusions

Overall smaller BSFs can provide similar removal capabilities as compared to the traditional concrete configuration. While virus removal capabilities were consistently higher for the larger concrete filters, the 5-gal bucket filters provide similar performance with respect to turbidity, bacteria, and protozoan cyst removal. The 2-gal bucket filters did offer similar removal with respect to these contaminants; however, the smaller filtering capacity (reduced reservoir volume) may not be practical for real world use for most households, at least not as the sole treatment device. One of the inherent problems of HHWT devices is the potential for recontamination of the treated water from time of treatment to time of use. The 2-gal bucket filters could offer a possible solution as a clarifying, or secondary treatment, as a counter top filter. However, the potential for contamination from a dirty filter outlet will always be present with any sized BSF.

Even though the smaller 5-gal bucket filters can offer similar performance, it is important to point out that it is not intended that smaller filters should arbitrarily replace the offering of the large BSFs. For some households, the recipients may prefer the smaller filter, requiring less space and offering a smaller filtering volume. However, other households, in particular those with a larger number of family members, may desire the larger BSF. One of the most important aspects of successful implementation of a household treatment is the initial technology selection phase that must be done with each community. During this phase (or even prior to it), it is important to characterize the

target communities, or households, current water supply. This will effectively help determine which household treatment options are potentially viable.

The addition of nails to the diffuser basin was shown to enhance virus removal for all filter sizes. The rusting of the nails resulted in a substantial increase in the amount of maintenance for those filters. Filters with nails not only required more frequent cleanings but the cleanings took longer and required more water to clean the top of the schmutzdecke layer. The additional time and increased water requirement may not be practical for most households where BSFs would be deployed. However, additional research should focus on evaluating the performance of the filters using a reduced amount of nails. Another option would be to identify other means of introducing the iron into the filter system. It is recommended that potential options should focus on utilizing materials that would be readily available in the area where the filters are to be deployed. One potential option is to investigate different filter media options, or different sand types. The sand used for the laboratory studies reported herein, used locally-sourced playground sand, which will be different from the sand available in areas where BSFs will be deployed. Additional research should focus on evaluating the potential of regional sand types with differing surface properties (e.g., surface potential) and the resulting effects on the removal of contaminants, in particular the sub-micron particles, such as viruses. Experimenting with differing sand types coupled with a smaller amount of nails may present the best solution: high contaminant removal levels and low user maintenance.

While not a universal option for every situation, the smaller 5-gal bucket filter was shown to be an option for household water treatment, especially for remote locations where implementing large concrete filters cannot be deployed. Overall, the testing results from these studies have shown that biosand filtration can be successfully executed in smaller, cheaper housing units, if necessary and desired.

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Lehigh University Bethlehem, PA
Ph.D. Candidate, Environmental Engineering
M.S., Environmental Engineering, May 2008

The Pennsylvania State University State College, PA
B.S., GeoEnvironmental Engineering, December 1999

Certifications: OSHA 1910.120 HAZWOPER, GIS Post-baccalaureate Certification

Laboratory assays: detection and enumeration of microbes in water samples: EPA Method 1620, membrane filtration, immunomagentic separation (IMS), immunomagnetic flourescent assay (IFA), double/single agar layer method; DNA analysis for source-tracking: DNA extraction, nested polymerase chain reaction (PCR), gel electrophoresis, gene production via host cloning, infectivity assay of HCT-8 cells with *Cryptosporidium*, atomic adsorption spectroscopy, UV-VIS spectroscopy, ion chromatography, titration.

RESEARCH

Long Term Study on the Performance of Full-scale Biosand Filters

- Investigated the long-term performance of full-scale units, depth of sand bed on microbial removal, the effect of filter transport on hydraulic loading rates and bacterial removal, operator influence (cleaning and pause period) and effects on particulate removal rates, effects of iron oxide on virus removal, and the role of biological mechanisms on removal.
- Programmatic evaluation of the field implementation of units in the San Juan del Sur area of Nicaragua. Collaborative effort with Tufts University, the Newton/San Juan del Sur Sister City Project and Fundacion Tierra, installation of 90 BSFs in households in rural Nicaragua in January 2013, including baseline and initial use surveys. Follow-up surveys are scheduled for March 2013 and Summer 2013.

Source-Tracking for *Cryptosporidium* in the Wissihickon Watershed

- Analyzed water and fecal samples from the Wissihickon watershed to identify sources (i.e., human, agriculture, and/or wildlife) of *Cryptosporidium* spp. oocysts and to determine if and where watershed control strategies should be implemented.

Silver Disinfection Studies

- Researched inhibitory effects silver nitrate on *E. coli* growth and *Cryptosporidium* viability and infectivity.

E. coli monitoring in the Little Lehigh Watershed.

- Identified *E. coli* concentrations for (i) base flow for several streams in the Lehigh Valley, (ii) storm flow for several sub-basins of the Little Lehigh watershed, and (iii) throughout storm discharge hydrographs on the Little Lehigh Creek at the Pool Wildlife Sanctuary. Developed sampling and testing plan. Oversaw field sampling and data collection activities.

TEACHING EXPERIENCE

- **CEE 275: Environmental, Geotechnics and Hydraulics Laboratory**
  Teaching Assistant for Dr. Kristen Jellison — every Spring 2009 – 2012
- **Drinking Water in Developing Countries Laboratory: Teaching Seminar for Professors from South East Unviersity, China**
  Teaching Assistant for Dr. Kristen Jelllison — Summers 2011 & 2012
- **CEE 379: Environmental Case Studies**
  Administrative Teaching Assistant for Dr. Kristen Jellison — Spring 2010
- **CEE 170: Introduction to Environmental Engineering**
  Teaching Assistant for Dr. Wei-xian Zhang — Spring 2007

WORK EXPERIENCE

Lehigh University
Graduate Research Assistant
August 2006 – Current — Bethlehem, PA

Concurrent Technologies Corporation (*CTC*)
Assistant Process Engineer
October 2000 – June 2006 — Johnstown, PA / Largo, FL

GEO-CON, Inc.
Project Engineer
March 2000 – October 2000 — Monroeville, PA

PUBLICATIONS

Napotnik J, Jellison K, and Baker D. (Draft). "Effect of sand bed depth and media age on microbial removal in biosand filters." Research Paper.

Napotnik J and Jellison K. (In Press). "Transport effects on hydraulic loading rate and removal performance." *Journal of Water and Health*.

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Napotnik J, Lantagne D, Jellison K. (2009). "Efficacy of silver-treated ceramic filters for household water treatment." Extended Abstract/Report, Water Environment Federation's Disinfection 2009 - International Ceramic Pot Filter Workshop, Atlanta, GA, February 28, 2009

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Napotnik J. 1999. "Thermo-mechanical Effects on Fractured Tuff at Yucca Mountain, Nevada." Senior Research Papers. Pennsylvania State University.

PRESENTATIONS (*Presenter)

Napotnik J*, Doup K, Smith N, and Jellison K. "Good Things Come in Small Packages." Poster Presentation and Exhibition. EPA People, Prosperity, Plant (P3) National Sustainable Design Expo and Awards, Washington DC, April 15-17, 2011.

Napotnik J*, Jellison K. "Optimizing the Biosand Filter for Household Drinking Water Treatment in Developing Countries." Oral presentation, Water Environment Federation's Disinfection 2011 – International Ceramic Pot Filters and Biosand Filters Workshop, Cincinnati, OH, April 10, 2011.

Jellison K, **Napotnik J***, Smith N, Doup K, Rayner J, Schubert J, Oyanedel-Craver, V, Lantagne D. "Evaluating the Impact of Production Variables on the Effluent Water Quality of Ceramic Pot Filters." Oral Presentation, Water Environment Federation's Disinfection 2011 – International Ceramic Pot Filters and Biosand Filters Workshop, Cincinnati, OH, April 10, 2011.

Napotnik J*, Doup K, Smith N, Zientarski, Wilson M, Jellison K. "Optimizing the Biosand Filter for Household Water Treatment." Poster Presentation. Earth & Environmental Sciences Research Symposium April 2011.

Napotnik J*, Mayer A, Lantagne D, Jellison K. "Efficacy of silver-treated ceramic filters for household water treatment." Oral presentation, Water Environment Federation's Disinfection 2009 - International Ceramic Pot Filter Workshop, Atlanta, GA, February 28, 2009

Napotnik J. "Sustainable Approaches to Landfill Diversion Opportunities". 11th Annual Joint Services Environmental Management Conference and Exhibition March 23, 2006.

Napotnik J. "Relocatable Buildings – Opportunity to Turn Waste into Product." 11th Annual Joint Services Environmental Management Conference and Exhibition March 23, 2006.

MEETINGS/CONFERENCES/TRAINING

- 5th Lehigh Valley Watershed Conference "Rising Waters: What a Wetter PA Means for Local Communities." Lehigh University, Bethlehem, PA October 9, 2012
- Biosystems Dynamics Summer Institute Mentoring Workshops, May 2012

- Panel Discussion. “Marcellus Shale Development: Communities, People, Health, Economics” April 18, 2012
- Teacher Development Series, Lehigh University, Fall 2011 – Spring 2012
- Edward Tufte Seminar: Presenting Data and Information, New York, NY March 2011
- WEF Disinfection 2011
- Access Database Seminar – Lehigh University (Spring 2011)
- 4th Annual Lehigh Valley Watershed Conference “ Watershed Science” Lehigh University, Bethlehem, PA, March 11, 2011
- PA/Chesapeake AWWA Joint Conference April 24-27, 2007 Hershey, PA
- International *Giardia* and *Cryptosporidium* Conference May 13-18, 2007 Morelia, Mexico
- The Joint Service Environmental Management Conferences, The Guide to Creating a Sustainable Installation, Sustainability Training (2005-2006) – Fort Benning, GA; Fort Lewis, WA; Fort Jackson, SC; Fort Bliss, TX; and Fort Buchanan, Puerto Rico
- UXO Basic Training Monterey, California Interstate Tecnologies and Regulatory Cooperation (ITRC) April 2004
- US EPA Environmental Technology Verificaiton (ETV) Coatings and Coatings Equipment Program (CCEP) Conference Wilmington, DE September 2001
- Air & Waste Mangement Association’s 94th Annual Conference and Exhibition, June 24-28, 2001 Orlando, FL

PROFESSIONAL MEMBERSHIPS

- PA Water Environment Association (PWEA) – Laboratory Practices Committee
- American Water Works Association (AWWA)
- Water Environment Federation (WEF)
- American Society of Civil Engineers (ASCE)
- Engineers Without Borders (EWB)
- American Society for Microbiology (ASM)
- New York Academy of Sciences (NYAS)
- Academy of Certified Hazardous Materials Managers (ACHMM)