EFFECTS OF PHYSICAL DISTURBANCES ON BIOSAND FILTERS USED FOR POINT-OF-USE WATER TREATMENT

EFFECTS OF PHYSICAL DISTURBANCES ON BIOSAND FILTERS USED FOR POINT-OF-USE WATER TREATMENT

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TITLE: Effects of Physical Disturbances on BioSand Filters Used for Point-of-Use Water Treatment

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ABSTRACT

Over 750 million people, 80% of whom live in rural communities, lack access to improved water sources. Even where an improved water source is easily accessible, recontamination and/or inadequate infrastructure may make it unsafe for human consumption. A lack of safe water leads to elevated rates of waterborne diseases and can exacerbate cycles of poverty by forcing individuals to miss school and work and to travel greater distances to secure better-quality water. Households in rural and remote communities may thus choose to use point-of-use treatment as a means of gaining greater control over their water quality and the health of their families. The BioSand Filter (BSF) is one such technology: it is an intermittently-operated household-scale slow sand filter currently used in over 70 nations around the world.

This thesis situates point-of-use water treatment, and specifically the BSF, within the context of the relationship between water and health and the continuum of technologies used for water treatment. From this foundation, it presents the methodology and results of a study carried out to inform best-practices around BSF use by: (a) examining the effects on BSF media and filtration performance of physical disturbances that may commonly occur in the field; and (b) assessing whether the biological community within BSFs promotes nitrification that could produce elevated nitrate/nitrite levels.

Results demonstrated that disturbing the filters through moving and side impacts caused marked sand compaction and decreased flow rates for plastic (Hydraid) BSFs. Although these decreased flow rates may contribute to user frustration and disuse, they were not associated with reduced filtration performance. Nitrate and nitrite concentrations were well below WHO guidelines for all samples, but changes in nitrogen speciation suggested that nitrification was mediated by the biological community within the filters. Recommendations for practitioners and for future research are discussed in light of these findings.

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It is overwhelming to consider the many thoughtful and intelligent people who have walked beside me and helped me to develop, both personally and academically, during my time at McMaster University. Giving adequate thanks to each would fill more pages than anyone cares to read, so I will do my best to be selective and (somewhat) succinct.

I would like to thank my supervisors, Dr. Sarah Dickson and Dr. Corinne Schuster-Wallace, for their encouragement, feedback, humour, and insights throughout my program. Sarah's nurturing care for her students' wellbeing and Corinne's ability to cohesively tie together disparate projects and ideas are traits that I deeply respect and strive to emulate. These women both balance innumerable academic and family commitments with levels of grace and good humour that amaze me.

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Kayla Lucier, our fearless summer student, approached the smelly and unglamorous tasks of setting up filters and mixing sewage with a charming smile and many poppin' playlists. Anna Robertson, Peter Koudys, and my rag-tag team of amazing office mates—in particular Vick, Sean, Graham, Mohammad, and Ahmed—contributed equipment, ideas, manpower, and many positivity points. Our department's administrative staff adeptly helped me navigate the forms and paperwork required from the start to the finish of my degree.

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When I arrived in Cambodia, eager to work with BioSand Filters as a fresh university graduate in 2010, I had no idea how little I would contribute or how much I would learn. I would like to add a note of thanks to the remarkable staff at Samaritan’s Purse and Asian Outreach Cambodia who introduced me to the intricacies of international development, and who continue to strive patiently and persistently towards better water and health alongside the rural communities they serve. I’ll never forget this statement from a crudely translated conversation during my first month overseas: “it’s funny: our God offers us water that makes us thirst no more, but tasting it has somehow made me thirstier for justice.”

May we continue to thirst for a better world together.

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LIST OF ABBREVIATIONS

BSF	BioSand Filter
CAWST	Centre for Affordable Water and Sanitation Technology
cBSF	Concrete BioSand Filter
CFU	Colony-Forming Units
DALY	Disability-Adjusted Life Years
HWT	Household Water Treatment
MI-Flow	Maximum initial flow rate
NTU	Nepholometric Turbidity Units
pBSF	Plastic (Hydraid) BioSand Filter
POU	Point-of-use
RRM	Rural, remote, or otherwise marginalized
SSF	Slow Sand Filter
TDS	Total Dissolved Solids
VS	Volatile Solids
WHO	World Health Organization

Declaration of Academic Achievement

The research presented in this thesis was performed by the author, with special assistance noted in the acknowledgments. The text of Chapter 3 is being submitted for publication in a peer-reviewed journal. Other authors of this paper are:

Sarah Dickson and **Corinne Schuster-Wallace**, who provided guidance throughout the design, interpretation, and editing phases of this work;

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Chapter 1

Introduction

1.1 Overview

In rural, remote, and otherwise marginalized (RRM) communities, a lack of access to water infrastructure often means that household members have to travel greater distances to secure safe water; this exacerbates poverty by detracting from time available for education and income-generating activities (Bartram and Cairncross, 2010; Montgomery and Elimelech, 2007). Insufficient access to infrastructure may also cause people to seek water sources of lower quality but closer proximity, making them more vulnerable to water-related health conditions and further reducing their ability to work and provide for their families. Over 750 million people, 80% of whom live in rural areas, lack access to improved drinking water (WHO/UNICEF JMP, 2013).

Point-of-use (POU) water quality interventions give users a greater degree of control over the quality of their drinking water, and can have significant positive impacts on human health (Fewtrell et al., 2005). One such POU intervention, the biosand filter (BSF) is a household-scale, intermittently operated slow sand filter used by 300,000 or more households in developing countries. The BSF is composed of a concrete or plastic filter body housing a fully-saturated sand media bed that rests on a gravel underdrain. The effectiveness of BSFs in removing microbial contaminants and reducing disease burdens has been well-documented despite the influence of bias in health impact literature (Baumgartner et al., 2007; Elliott et al., 2008; Hunter, 2009; Stauber et al., 2012a, 2006). However, several recent studies identified increases in nitrate and nitrite after water containing ammonia was treated in BSFs (Mangoua-Allali et al., 2012; Murphy et al., 2010a; Wu et al., 2013). This points to nitrification within the filters, which poses a concern because young infants can develop methamoglobinaemia if they consume water that is high in nitrite or nitrate (Fan and Steinberg, 1996). It is assumed that changes in nitrogen speciation observed between when the filter is charged and when its effluent is tested result from biological activity within the filters; however, no study has compared these changes in nitrogen speciation to those that would occur naturally if the influent water was left standing for the same period of time.

BSFs are traditionally contained in concrete filter bodies (cBSFs) made from locally-sourced aggregate, but plastic Hydraid filter bodies (pBSFs) are increasingly popular as they present a lightweight, inexpensive, and mass-producible alternative. Without the weight and strength of concrete walls, the sand inside these filters may behave differently when subjected to disturbances. Current recommendations for BSF installation and use emphasize the importance of keeping the filters stationary and protecting them from jarring impacts. All performance and hydraulic lab studies to date have employed stationary BSFs, making it difficult to confirm or improve upon these recommendations.

1.2 Research Objectives

The objectives of this thesis are:

1) To provide an overview of the relationship between water and health, the role of POU technologies, and the history and mechanisms of the BSF;

2) To affirm, or improve-upon, existing recommendations around filter installation, use, and monitoring by evaluating the impacts of physical disturbances (moving, side impacts, and daily bucket impacts) on the media integrity and filtration performance of BSFs; and

3) To further inform our current understanding of nitrification within BSFs by comparing nitrogen species concentrations in filter effluent to those in influent and “standing bucket” controls.

1.3 Scope

This thesis contains three chapters in addition to this introductory chapter:

Chapter 2 contains a literature review that covers: (1) the relationship between water and health; (2) the need for and types of point-of-use water treatment interventions; and (3) the history and mechanisms of the BioSand Filter.

Chapter 3, which is also being submitted for publication in a peer-reviewed journal, provides the methodology, results, analysis, and discussion used to achieve the above research objectives.

Chapter 4 expands on the conclusions and recommendations of the preceding chapter, with particular attention paid to recommendations for practitioners and for future work.

Extra details regarding analytical methods are provided in Appendix A, with photos of the lab set up provided in Appendix B. Relevant data that were not included or were referred to as “data not shown” in Chapter 3 are provided in Appendix C.

Chapter 2

Literature Review

This chapter will contextualize research presented in Chapter 3 by surveying the following topics:

1. The Water Health Nexus: the known impacts of water availability and water quality on human health, and the need for water quality interventions;
2. Water Quality Interventions for RRM Communities: a broad overview of water quality interventions for improving health in RRM communities, including a discussion of when point-of-use interventions are appropriate; and
3. The BioSand Filter and Slow Sand Filtration: history, theory, and performance of slow sand filtration and of the BSF technology.

2.1 The Water-Health Nexus

Access to potable water at a reasonable distance from the home is crucial for hydration, safe food preparation, and personal hygiene (Bartram and Cairncross, 2010; Howard and Bartram, 2003; Montgomery and Elimelech, 2007). The average person requires 15-100 litres of water per day, at least 7.5 of which should be safe for drinking (Howard and Bartram, 2003). When improved water sources are not available—as is often the case in rural, remote, and otherwise marginalized (RRM) communities--collecting adequate volumes of safe water can consume substantial time and energy, thus detracting from education and income-generating activities (Bartram and Cairncross, 2010; Montgomery and Elimelech, 2007).

The United Nations recognized and affirmed the importance of safe water for ensuring healthy communities and environments through two key actions:

1. setting an ambitious target for drinking water in the Millennium Development Goals (MDG 7 target 3): to "halve, by 2015, the proportion of people without sustainable access to safe drinking water and sanitation" (United Nations, 2012); and
2. declaring access to safe drinking water as a human right in 2010 (WHO/UNICEF JMP, 2011).

The WHO defines "improved drinking water" as a source that "by nature of its construction or through active intervention, is protected from outside contamination, in particular from contamination with faecal matter" (WHO/UNICEF JMP, 2004). Even sources that are considered "improved," such as piped water supplies or rainwater harvesting tanks, may contain contaminants that make them unsafe to drink without additional treatment.

2.1.1 Disease Burdens Linked to Water, Sanitation & Hygiene

Safe drinking water is important because biological contaminants in water present serious health concerns. Water-related diseases are generally classified into four groups according to the mode of transmission of the biological contaminants that cause them (Bradley, 1977; Howard and Bartram, 2003; Zwane and Kremer, 2007):

Water-borne diseases, such as giardiasis, cryptosporidiosis, and cholera, are transmitted through the consumption of contaminated water. Most vectors in this category cause diarrhoea and/or acute gastrointestinal illnesses. Open defecation and a lack of sanitation facilities often contribute to faecal contamination of water and therefore elevated rates of water-borne diseases, which account for up to 90% of the infectious disease burden in developing countries (Esrey et al., 1991; Pimentel et al., 2007).

Water-washed diseases are caused by inadequate volumes of safe water for personal hygiene. Some pathogens in this category, such as trachoma and many skin infections, are transmitted when people wash in unsafe water or fail to attend to personal hygiene. Preventing these diseases requires both (a) knowledge about hygiene practices, and (b) adequate quantities of safe water for personal hygiene (Bradley, 1977; Zwane and Kremer, 2007).

Water-based diseases, such as guinea worm and schistosomiasis, require an intermediate aquatic host. Most prevalent in tropical regions, these are contracted through repeated physical contact with contaminated water (Bradley, 1977; Zwane and Kremer, 2007).

Water-related vector-based diseases are spread through insect vectors that require water as part of their life cycle. In particular, the presence of standing water provides breeding grounds for the mosquito vectors of malaria, dengue fever, chikungunya, and yellow fever (Bradley, 1977).

Water-related diseases may be introduced to the water supply at multiple points from the initial source to final consumption. Pollutants such as faeces in the environment contribute to source water contamination; however, contamination also frequently occurs during collection, storage, and use of household water. Recognizing this, Montgomery and Elimelech (2007) add a fifth disease category for **collection and storage-related diseases**. This includes any disease agents introduced to the water because of poorly designed collection/storage containers and/or improper hygiene or handling.

Chemical contamination of drinking water is also a concern in some regions, particularly where industrial or agricultural activities predominate. In general, the health effects of chemical contaminants tend to be chronic and present themselves after years or even decades of exposure. Examples include aluminum, disinfection by-products, and pesticides that may arise from industrial effluent, municipal sewerage, and agricultural runoff (Prüss-Ustün et al., 2011). When present above critical concentrations in drinking water, these contaminants have been linked to cancers, developmental/reproductive issues, and neurologic conditions (Calderon, 2000; Howard and Bartram, 2003). Naturally-occurring substances may also be harmful at elevated concentrations: in Bangladesh, for example, deep well drilling—which was undertaken to reduce the burden of diseases associated with contaminated surface water—unintentionally exposed over 33 million people to hazardous levels of naturally-occurring arsenic and caused high rates of arsenicosis in the region (Pimentel et al., 2007).

The WHO estimates that 6.3% of all deaths and nearly 10% of the total global burden of disease (measured in disability-adjusted life years, DALYs) could be prevented by improvements in water, sanitation, hygiene, and water resource management (Prüss-Üstün et al., 2008). While some localized regions face high disease burdens due to chemical contaminants in their drinking water, the comparatively acute effects of biological disease vectors present a more pressing and immediate concern for the majority of RRM communities. The greatest burden of disease resulting from insufficient access to water and sanitation is attributed to infectious diarrhoea (Bartram and Cairncross, 2010; Hutton and Haller, 2004). If not prevented or treated, acute diarrhoea may result in severe dehydration, requiring families to draw greater quantities of water to satisfy domestic needs (Zwane and Kremer, 2007). If it becomes chronic, diarrhoea can predispose children to malnutrition, which impairs growth, cognitive development, and congenital immunity (Zwane and Kremer, 2007). Over 2 million preventable child deaths are caused each year by diarrhoea and malnutrition that result from unsafe water, inadequate sanitation, and insufficient hygiene (Prüss-Üstün et al., 2008).

2.1.2 Vulnerabilities and Inequities

Disparities in fresh water resources, financial and social capacities, power, and institutional structures contribute to inequitable access to water and sanitation (Moe and Rheingans, 2006). At a global level, the vast majority of nations with less than 50% coverage in water supply are in sub-Saharan Africa (WHO/UNICEF JMP, 2012). At a regional level, urban-rural disparities persist: 4% and 19%, respectively, of the global urban and rural populations lack access to improved drinking water (WHO/UNICEF JMP, 2012). At the household level, those with incomes under US $1 per day are nine times more likely to lack improved water or sanitation compared to those earning at least US $2 per day (Moe and Rheingans, 2006). For these reasons, interventions are needed to empower poor and rural households to have greater control over their water quality.

Water-related diseases disproportionately affect women and children, two demographic groups whose health, well-being, and education are critical for a healthy economy (Montgomery and Elimelech, 2007). Pregnant women and infants are more likely than other groups to develop symptoms from exposure to chemicals such as disinfection by-products, lead, and sulfates (Calderon, 2000). Pregnant women also have increased susceptibility to malaria, particularly if they are young and expecting their first child, and the resulting anemia can contribute to low birth weights and increased maternal and infant mortality (Desai et al., 2007). High nitrate levels from agricultural activity or sewage can trigger methemoglobinemia in infants under the age of one year; this “blue baby syndrome” is a substantial health concern but is not well-documented and lacks the status of a notifiable disease in most countries (Howard and Bartram, 2003). A report from the WHO recognized that children shoulder a disproportionately high share of the global water-related disease burden: over 20% of deaths and DALYs in children under the age of 14 could be prevented with safe water, sanitation, and hygiene (Prüss-Üstün et al., 2008). Moreover, improved water and sanitation access have both been found to significantly reduce under-five and infant mortality rates (Cheng et al., 2012).

In addition to bearing the brunt of the disease burden, women and children are the primary care-givers and water-collectors in many low-income countries. The time they spend collecting water and caring for ill family members detracts from children’s school attendance and women’s ability to pursue other means of income generation (Montgomery and Elimelech, 2007). A lack of access to safe water can exacerbate cycles of poverty by causing these individuals to devote more time to these time-consuming activities, spend more money on health care and medicines, and consequently miss opportunities for small business endeavors

(Asian Development Bank, 2004; Bartram and Cairncross, 2010; Schuster-Wallace et al., 2008).

2.2 Water quality interventions for RRM communities

When improved water is not available or when the safety of a water source is questionable or inconsistent, treatment is required. How appropriate a particular treatment technology is for a RRM community depends on many factors including cultural acceptability, local knowledge and practices, raw water quality and availability, types of contaminants, cost, maintenance requirements, and local capacity (Nath et al., 2006).

There are two key strategies for water treatment in RRM communities: (1) centralized treatment, and (2) household water treatment (HWT, also referred to as point-of-use treatment or POU). Each of these categories contains a range of technologies and approaches. Centralized systems are generally more cost-effective, but are also more difficult to maintain and more likely to break down without being repaired due to institutional-level problems (Cairncross et al., 2010a). When households in rural communities are encouraged to invest in their own water and sanitation facilities, they are more likely to choose inexpensive and simple technologies and to be involved in their maintenance and operation (Cairncross et al., 2010a). In general, household-level treatment leads to greater health benefits—quantified through reductions in diarrheal morbidity--than community-level systems (Cairncross et al., 2010b; Zwane and Kremer, 2007). In their critical review of interventions to reduce diarrhea in developing countries, Zwane and Kremer (2007) conclude that POU treatment reduces diarrhea more significantly than community-level rural water infrastructure and as such should be further studied and promoted. Cairncross et al. (2010) found that treatment at the source and treatment at the point-of-use (household) caused 27% and 43% reductions, respectively, in incidences of diarrhea. This may, however, reflect a larger placebo effect for those with household treatment, as the technology is located in the home and provides a sense of security that can bias self-reported health indicators. A review by Schmidt and Cairncross (2009) reported 30-40% reductions in diarrheal morbidity due to household water treatment but, given concerns around bias, cautioned that further research and evidence is required before scaling up such approaches.

Despite a lack of rigorous evidence—largely due to the ethical issues involved in conducting fully blinded trials with a technology that affects human health—there are several situations in which POU approaches are generally accepted as the best available option. These are:

1. Post-disaster areas in which centralized systems are inadequate or compromised and/or families have been forced to move to temporary shelters where infrastructure is not available.
2. Cases in which recontamination occurs between water collection and consumption, for example when piped water is not available in the home and thus contamination may readily occur during transportation and storage. In a systematic meta-analysis of 57 studies, Wright et al. (2004) found that bacteriological contamination of water significantly increased between the source and point-of-use. This often resulted from the use of open storage vessels or dirty dipping devices, unsafe practices that can be addressed through education that targets behaviors around water storage and POU treatment (Gundry et al., 2004; Wright et al., 2004). It is important to note that POU approaches will not mitigate the risk of recontamination if safe storage practices are not also adopted.
3. Cases in which the quality or reliability of a water source—even if it is delivered via a piped network—is questionable. In these situations, the implementation of POU approaches should be paired with longer-term solutions to address the reliability of the water source in question (WHO, 2011a).

In evaluating whether POU approaches are appropriate, there is a tension between: (1) the obligation for governments to respect, protect, and fulfil the human right to water; and (2) the recognition that temporary or progressive steps are required—and sometimes must be undertaken by individuals or households—before universal access to safe water can be achieved. On one hand, if individuals or organizations implement POU treatment to address gaps in service provisioning, this may reduce the government's sense of responsibility to ensure that water is accessible and safe to consume for all citizens. On the other hand, it seems unreasonable not to capitalize on POU technologies that could save millions of lives where centralized solutions are not yet available. Mintz et al. (2001) and Mol (2001) argue that relying solely on centralized solutions that are time- and resource-intensive will leave millions without access to water for an unacceptably long period of time; self-sustaining, decentralized approaches can act as viable solutions in cases where capital or human resources are not yet adequate to sustain centralized systems. This approach falls within what de Albuquerque (2012) refers to as "progressive realisation," a principle that acknowledges that incremental steps are required to achieve any economic, social, or cultural right including that to water and sanitation. POU interventions and small-scale community systems can act as intermediate steps towards comprehensive access to improved water. Additionally, improved water sources—that is, those protected from external contamination, especially from faecal matter—are not always safe water sources; Bartram and Cairncross (2010) point out that, for example, one in five water supplies in large Asian cities does not meet national quality standards. There is, therefore, a role for household-level treatment that can act as a secondary barrier against water-related diseases.

2.2.1 Types of POU/HWT interventions

Numerous POU systems are currently available, and the effectiveness of each is highly dependent on situational factors. Following are some of the most promising and accessible approaches for improving water quality at the household level, and some of the key factors that affect their efficacy in RRM communities (Clasen et al., 2007; Sobsey, 2002; WHO, 2011a):

Boiling: having been used for centuries to treat drinking water, boiling is the most common POU practice and, in theory, is the most effective for reducing pathogens (WHO, 2011a). In practice, however, boiling uses large quantities of fuel and is not cost-effective or practical as the primary water treatment system in many regions. It also does not reduce turbidity, and users have to wait for water to cool prior to consumption.

Household chlorination: chlorination has had a dramatic positive effect on public health over the last century, especially at the community/municipality level. Its use at the household level has had mixed success. On one hand, a systematic review of over 20 POU chlorination studies found a pooled risk ratio of 0.71 for diarrheal disease in children from households using chlorination, indicating some success in improving health (Arnold and Colford, 2007). Clasen et al. (2007) found that this intervention was associated with a significant reduction in diarrhoea among all age groups. Conversely, supplies of chlorine are not always available in all regions, and chlorination alone does not remove turbidity and can affect taste and odour. This approach, when used in isolation, is also ineffective against the protozoans *Giardia* and *Cryptosporidium* (Sobsey, 2002).

Solar Disinfection: Sunlight can be used in several ways to disinfect water. Some approaches use the sun’s heat to distill water, or to inactivate microbes through pasteurization, while others such as the SODIS system use clear plastic containers that allow UV light to penetrate and inactivate microorganisms. The latter approach has been associated with reduced diarrhea in all age groups (Clasen et al., 2007; Hunter, 2009), although Hunter (2009) found that its long-term (52 weeks or more) effectiveness was very low when potential biases were considered. Dissolved oxygen and turbidity levels strongly influence the effectiveness of SODIS; if oxygen is scarce or waters are highly turbid, UV radiation cannot disinfect as readily (WHO, 2011a). This approach also has limited applicability during seasons of intense rain or constant cloud cover. Like all POU technologies, it is also limited by the knowledge and attitudes of communities in which it is adopted: one study in Nepal found that only 9% of households that were trained and provided with basic materials for SODIS were still routinely using it several months later, largely because it was perceived as an extra workload and the importance of water treatment was not fully understood (Rainey and Harding, 2005).

Coagulation-Flocculation and/or Sedimentation: these approaches involve using coagulants or precipitants to aid in the settling and removal of suspended particles including microbes. There are many promising low-cost, locally-available coagulants that, when combined with a simple filtration system (usually a fine cloth) can significantly reduce pathogens in source water (WHO, 2011a). While these systems are not always highly effective in isolation, the WHO and several key reviews have identified combination systems—which employ these chemicals alongside filtration and/or chlorination or UV disinfection—as one of the most promising systems for HWT (Clasen et al., 2007; Sobsey, 2002; WHO, 2011a).

Household Filtration: When chemical disinfectants are unavailable and fuel for boiling is expensive, filtration technologies are increasingly becoming a popular alternative for HWT (WHO, 2011a) and have contributed to significant reductions in diarrhea among all age groups (Clasen et al., 2007). Filtration technologies for household use include:

1. Reverse osmosis, nanofilters, and other specialized membrane technologies. These may be used by travelers to developing countries but are rarely an affordable option for local populations; however, some new low-cost applications have been developed and may play a larger role in HWT in the near future (WHO, 2011a).
2. In some regions where other technology is unavailable, cloth filters (i.e. pouring water through a clean sari cloth) are recommended for reducing larger pathogens. Cloth filters cannot remove individual bacteria (which easily pass through the >20 μm pores of fabric), but have contributed to guinea worm eradication programs and have successfully reduced cholera in areas where the bacteria is associated with larger copepods and plankton (Colwell et al., 2003; WHO, 2011a).

3. There are many types of ceramic filters, with the most common being the ceramic "pot" filters and ceramic "candle" filters. These are typically gravity-driven with water flowing down through the ceramic media and into a safe storage container. In a meta-analysis of HWT approaches, Hunter (2009) found that ceramic filters were the most effective at reducing relative risk of diarrhea and were the most likely to still be effective after one year.
4. The BioSand Filter has performed well in many studies, as described below. Its biologically-active layer takes some time to mature, as described below, and must be fed frequently (i.e. cannot be left unattended for more than a few days) to maintain optimal performance. This, combined with its considerable weight and lack of portability, makes the BSF better-suited to stable, long-term communities than to nomadic people or disaster situations.

2.3 The BioSand Filter and Slow Sand Filtration

The BioSand Filter (BSF) was first developed by Dr. David Manz at the University of Calgary, building on principles of slow sand filtration. BSFs were piloted in Nicaragua in 1993 and have since gone through several design modifications and have been implemented in over 300,000 households and 70 countries (CAWST, 2012). It is typically recommended for BSF use to be combined with chlorination or boiling (multi-barrier approach), but the filters are frequently used without subsequent treatment.

This section provides a brief overview of slow sand filter (SSF) technology, how it has been adapted for the BSF, and what is known about its effectiveness in removing contaminants and improving health.

2.3.1 Slow sand filtration: history and overview

Slow sand filters have been used to treat water supplies since 1804, when John Gibb designed and built a sand filter for his bleachery and began selling extra treated water to members of his community for a small profit (Huisman and Wood, 1974). After some modifications, slow sand filtration was adopted by a public water supply company in London in 1829. By the 1850s—decades before scientists understood that diseases were transmitted by microorganisms—the use of SSFs was legislated as a requirement for water withdrawn from the Thames River in London (Huisman and Wood, 1974). Although SSFs have been replaced by faster and more advanced high-throughput methods in many municipalities, they remain an attractive option for rural communities and even some municipalities due to their low cost, ease of operation, minimal maintenance, and high performance (Collins et al., 1992; Lee, 2001).

A slow sand filter consists of four elements: (1) a supernatant water reservoir which maintains a constant head of water; (2) a sand media bed through which filtration occurs; (3) an under-drainage system that supports the media and allows water to emerge freely from the bottom of the filter; and (4) a system of control valves to regulate flow (Huisman and Wood, 1974).

The basic processes and removal mechanisms of SSFs and BSFs are the same. Water first enters the supernatant reservoir and gradually makes its way down through the media. In the supernatant reservoir, heavier particles settle and algal populations take up nutrients from the water. The surface of the sand begins to collect a thin slimy mat of organic matter (including algae, plankton, diatoms, protozoa, rotifers, and bacteria) known as the schmutzdecke. Weber-Shirk and Dick (1997a) found that **surface straining** was the primary physical process operating in the schmutzdecke of SSFs. Surface straining is when particles that are too large to pass through pore spaces become trapped on the surface of the filter. A bed of spherical grains can capture particles about 1/7th of the diameter of the sand used, and a typical sand bed is thus expected to capture anything 30 μm or larger by surface straining (Huisman and Wood, 1974). As more particles become trapped, the pores at the surface of the sand become much smaller and surface straining becomes more effective, entrapping increasingly smaller particles such as cysts (1-20 μm), bacteria (0.1 to 10 μm), and viruses (smaller than 0.1 μm) (Haarhoff and Cleasby, 1991; Lee, 2001).

Surface straining is aided by **inter-particle attraction**, the second primary mechanism of physical removal (Weber-Shirk and Dick, 1997a). Particles attach to the medium (consisting of sand and previously removed particles) due to (1) adhesion to the gelatinous biofilm and (2) electrostatic or Van der Waals forces (Huisman and Wood, 1974). Particle attachment increases when the net attractive force between the medium and the suspended particles is high (Weber-Shirk and Dick, 1997a). Viscous forces are higher at the surface of the filter where pore sizes are smaller and particles are more likely to be sheared from the media; for this reason, attachment of particles to the media may be more efficient slightly below the surface of the filter bed as the water moves down through the sand (Weber-Shirk and Dick, 1997a).

In addition to physical straining and attachment, particles are removed through **biological mechanisms** (Haarhoff and Cleasby, 1991). **Bacterivory** is the process by which larger organisms such as protozoa and rotifers prey upon bacteria. Weber-Shirk and Dick (1997b) found that predatory grazing upon bacteria in the biologically active portion of slow sand filters accounts for a significant portion of the removal of particles smaller than 2 μm. Microorganisms in the schmutzdecke are highly active in entrapping and breaking down microbes, detritus, and nutrients found in the influent water (Huisman and Wood, 1974). Intestinal microorganisms such as *E. coli* are also out-competed by free-living resident bacteria that are better adapted to the temperature and ecosystem found in the filter. Unable to thrive because they are not as well-adapted, non-resident bacteria naturally die off as they fail to obtain sufficient food within the biological layer (Huisman and Wood, 1974). For these reasons, maintaining a healthy biological layer at the surface of the filter is important for enhanced performance. Thus a period of **filter ripening**—during which biological growth occurs at the surface of the sand bed—is required before coliform removal reaches its maximum potential (Bellamy et al., 1985). For BSFs, this ripening period typically occurs over periods ranging from 1-2 weeks (Elliott et al., 2011) to 30 days (Baumgartner et al., 2007; Elliott et al., 2008). The speed of ripening can be affected by the frequency and volume of influent water charges (more frequent charges provide more material for biofilm development, but pause periods between charges are necessary for maturation) and by the levels of turbidity and nutrients in the influent water (more biological material promotes greater biofilm development).

2.3.2 The BioSand Filter: a modified SSF design

Whereas a conventional SSF is 3-5 m tall and 4-15 m wide (Haarhoff and Cleasby, 1991), the BSF is typically less than one metre tall and less than 0.5 m wide (Table 1). With these dimensions, it can easily be installed in a household. The BSF also differs from the SSF because it is designed to operate under intermittent rather than continuous flow: when users pour a bucket of water into the supernatant reservoir, it creates a driving head that pushes water through the filter so that the flow rate gradually declines until the head reaches its static level. The biologically active layer at the top of the filter is protected from desiccation by the presence of this “standing head” of water. An elevated outlet pipe ensures that the resting water level always remains approximately 5 cm above the surface of the sand—enough to keep the surface moist but still allow oxygen to reach the biological layer during pause periods (Buzunis, 1995). A diffuser plate/basin protects the sand surface from being disturbed by the sudden influx of water that occurs when a bucket is poured into the filter (Figure 1 in Chapter 3).

After the filter has been in operation for some time, particles building up in the upper layer of the media may lead to head loss. In SSFs, head is restored by cleaning the filters (i.e. scraping off the top layer of the filter or “harrowing” the sand) (Huisman and Wood, 1974). In BSFs, filter cleaning is only recommended when flow rates become inconveniently low, as the process of cleaning (which involves agitating the very top of the sand and then removing the water right above the surface) can temporarily reduce the BSF’s effectiveness by disturbing the biological layer (Manz, 2009).

Table 1 Comparison of conventional SSF and BSF filters

	Slow Sand Filters[a]	BioSand Filters[b]
Discharge Rate	0.0017 m/min	0.0060 m/minute (first minute of flow)
Filter dimensions	3-4 m tall by 4-15 m wide	0.9 m tall by 0.3 m wide
Sand Depth	0.8 m	0.4 – 0.5 m
Resting water depth above sand	1.5 m	0.05 m
Operation	Continuous	Intermittent

[a]Based on a typical SSF described by (Haarhoff and Cleasby, 1991); other designs also exist.
[b]For a typical square concrete BSF (v10) (CAWST, 2012)

2.3.3 Effectiveness of the BSF

Several studies have demonstrated the effectiveness of BSFs in removing microbial contaminants and reducing disease burdens, and these are briefly discussed in Chapter 3. The filters can remove 99-100% of Giardia cysts and Cryptosporidium oocysts (Palmateer et al., 1997). Field evaluations have found coliform removal rates of 93-98.5% (Duke et al., 2006; Fabiszewski de Aceituno et al., 2012; Stauber et al., 2012b). In the lab, coliform and virus removal rates are variable but generally high (Baumgartner et al., 2007; Elliott et al., 2008; Stauber et al., 2006). Some of the variability can be attributed to differences in pause time, charge volume, and other parameters that will typically be inconsistent between users in the field. For example, Baumgartner et al. (2007) observed that shorter pause times and smaller charge volumes result in significantly greater microbial removal.

2.3.4 Evidence of Nitrification in BSFs

Murphy et al. (2010) found significant increases in both nitrate and nitrite after water was treated in household BSFs in rural Cambodia. The authors concluded that microbially-driven nitrification and denitrification were occurring in the filters.

Nitrification is the oxidation of ammonia to nitrite and then to nitrate by the following reactions, with the first and second steps mediated by the aerobic bacteria *Nitrosomonas* and *Nitrobacter*, respectively:

$$NH_3 + O_2 \leftrightarrow NO_2^- + 3\ H^+ + 2e^- \quad (2.1)$$

$$NO_2^- + H_2O \leftrightarrow NO_3^- + 2\ H^+ + 2e^- \quad (2.2)$$

Denitrification is also microbially-driven, but occurs in anaerobic environments (and thus takes place at lower depths in the BSF). This process occurs by the reduction of nitrate to nitrite and then to nitrogen gas:

$$NO_3^- + 2\ H^+ + 2e^- \leftrightarrow NO_2^- + H_2O \quad (2.3)$$

$$NO_2^- \leftrightarrow NO \leftrightarrow N_2O \leftrightarrow N_2 \quad (2.4)$$

Nakhla and Farooq (2003) found evidence of simultaneous nitrification-denitrification taking place in SSFs in Saudi Arabia, and Murphy et al. (2010) propose that the same can occur in BSFs and can be detected through decreases in both ammonia and nitrate from influent to effluent. Nitrification-denitrification in BSFs has not received a great deal of attention, but several recent studies (Mangoua-Allali et al., 2012; Wu et al., 2013) have confirmed elevated nitrate and/or nitrite in filter effluent. Although these concentrations were generally below WHO guideline values, the fact that nitrification may occur within the filter poses a concern because young infants can develop methemoglobinemia if they consume water from sources high in nitrite and nitrate (Fan and Steinberg, 1996). Wu et al. (2013) explored several practical steps that could be taken to address nitrification in BSFs, such as adjusting operating parameters (ammonia loading, filter cleaning, filter idle time, and effluent storage time) to potentially alter nitrogen speciation in the filter effluent. The only factor that could effectively reduce/mitigate nitrification was using source waters without elevated ammonia; nitrate and nitrite in the effluent only exceeded the WHO guideline values when ammonia loading was high (approximately 18 mg-N/L).

There is generally a time lag between when a filter is charged and when the effluent is used or sampled, and some of the nitrification/denitrification detected in previous studies may have occurred naturally in the water rather than occurring only because of the microbial community in the BSF. For this reason, one of the objectives of the research presented in Chapter 3 is to compare nitrogen speciation in a "standing bucket control" to that in effluent produced over the same period of time.

2.3.5 BSF construction materials

Another important design parameter of the BSF is the materials used to make the filter body. BSFs are traditionally contained in concrete filter bodies made from locally-available aggregate. This results in filters that are made from affordable local materials and are highly durable, easily lasting 10 years or longer. However, there are two drawbacks of the concrete design: (1) in some regions, cement and gravel may be difficult to acquire or technical expertise may be lacking; and (2) the concrete bodies are heavy, weighing 75 – 150 kg before media and water are added (Vanderzwaag, 2008).

Other types of filter bodies are being developed, and plastic Hydraid (pBSF) filter bodies are increasing in popularity due to their portability, durability and anticipated scalability. Some new designs are more portable and utilize light materials such as sheet metal (Smith, 2013) and other plastics (e.g. the plastic bucket design used by Collin (2009)). These have the potential to be highly useful, scalable, and affordable, but without thick concrete walls they may not provide the same protection to the sand media and they may be more frequently relocated or disturbed. A careful assessment is needed of how household disturbances may affect the performance of BSFs that are lighter and more portable than the original concrete design.

Chapter 3

Effects of physical disturbances on media and performance of intermittently operated household-scale slow sand (BioSand) filters

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Abstract

Point-of-use (POU) water treatment provides households in rural and remote communities with a means of obtaining greater control over their water quality and its effects on human health. One of the most prevalent POU interventions, the BioSand Filter (BSF) is a household-scale, intermittently operated slow sand filter used by over 300,000 households in more than 70 countries. The sand and gravel media within BSFs can be housed in concrete (cBSF) or Hydraid plastic (pBSF) bodies, with the latter becoming increasingly popular due to their portability, durability and anticipated scalability. This study evaluated whether pBSFs—which are lighter and thinner than their concrete counterparts—can maintain their integrity and performance after being subjected to disturbances that may occur in a typical household. Eight pBSFs and two cBSFs were run in parallel for 13 weeks, and three disturbances—one-time filter movement, one-time side impacts, and daily bucket impacts—were applied. Moving and side impacts caused marked decreases in sand column height (6 – 29

mm decrease, $p < 0.001$); this was observed to a greater extent for pBSFs than for cBSFs. Sand compaction in the plastic filters contributed to decreases in maximum initial flow rate (18-84% decrease, $p < 0.001$). There was no evidence of preferential flow paths or of decreased filter performance in disturbed filters, aside from brief spikes in pBSF effluent turbidity (0.98 – 15.2 NTU higher than mean effluent levels) immediately after disturbances occurred. Nitrogen speciation in influent, effluent, and standing bucket controls was also assessed to address previously-published concerns around nitrification in BSFs. Decreases in ammonia and increases in nitrate and nitrite, relative to influent levels, were greater and more significant in the effluent than in standing bucket controls. This suggests that the microbial community in the BSF promotes nitrification beyond that which naturally occurs in influent water left standing overnight.

3.1 Introduction

Over 750 million people, 80% of whom live in rural areas, lack access to improved drinking water (WHO/UNICEF JMP, 2013). In rural, remote, and otherwise marginalized communities, a lack of access to water infrastructure often means that household members have to travel greater distances to secure safe water; this detracts from time available for education and income-generating activities (Bartram and Cairncross, 2010; Montgomery and Elimelech, 2007). Insufficient access to potable water may also cause people to seek water sources of lower quality but closer proximity, making them more vulnerable to water-related health conditions and further reducing their ability to work and provide for their families. The World Health Organization (WHO) estimates that 6.3% of all deaths and nearly 10% of the total global burden of disease could be prevented by improvements in water, sanitation, hygiene, and water resource management (Prüss-Üstün et al., 2008).

Point-of-use (POU) water treatment gives users greater control over the quality of their drinking water, and is particularly useful in rural settings where households do not have consistent access to treated piped water. POU approaches can have significant positive impacts on human health; indeed, a meta-analysis by Cairncross et al. (2010) found that household-based water quality interventions contributed to over 40% reductions in incidences of diarrhoeal disease. Hunter (2009) found that household water treatment significantly reduced the risk of waterborne diseases, even after the considerable biases in many published studies were considered.

One such POU intervention, the BioSand Filter (BSF) is a household-scale, intermittently operated slow sand filter used in over 300,000 households. The BSF is composed of a concrete or plastic filter body housing a sand media bed that rests on a gravel underdrain (Figure 1). An elevated outlet pipe ensures that the static water level remains above the sand surface such that the sand column is always saturated and a biological layer forms on its surface.

Several studies have demonstrated the effectiveness of BSFs in removing microbial contaminants and reducing disease burdens. Coliform removal rates are highly variable, but generally close to 1-log reduction or greater on average; virus removal rates are lower, vary substantially by the type of virus, and have been less extensively studied (Baumgartner et al., 2007; Elliott et al., 2008; Stauber et al., 2006).

Although removal rates tend to vary—ranging, for example, from 0% to 99.7% removal of *E. coli* in tests reported by Stauber et al. (2006)—studies to date suggest significant positive impacts of BSF use on household health. In field studies in Cambodia (Stauber et al., 2012b), Kenya (Tiwari et al., 2009), and the Dominican Republic (Stauber et al., 2009), households with BSFs had significantly lower coliform concentrations in drinking water and significantly lower diarrheal disease rates than control communities/households. In Honduras, diarrheal disease rates in children under 5 were approximately 45% lower in households with BSFs, although this finding was not statistically significant due to seasonal fluctuations in morbidity (Fabiszewski de Aceituno et al., 2012). It should be noted, however, that difficulties in conducting fully blinded studies on the health effects of POU technologies have raised questions about the influence of bias on the positive health impacts reported in most literature (Hunter, 2009; Schmidt and Cairncross, 2009).

BSFs have consistently performed well in lab and field studies, but concerns over nitrification during filtration have recently emerged. Murphy et al. (2010) found significant increases in both nitrate and nitrite after water containing ammonia was treated in household BSFs in rural Cambodia. The authors concluded that microbially-driven nitrification and denitrification in the filters caused the majority of effluent samples to exceed WHO guideline values for nitrate and nitrite. Several other studies (Mangoua-Allali et al., 2012; Wu et al., 2013) have confirmed elevated nitrate and/or nitrite in filter effluent, which poses a concern because infants can develop methemoglobinemia if they consume water from sources with high levels of these nitrogen species (Fan and Steinberg, 1996).

As academic and practical reports emerge around nitrification, pathogen removal, and other metrics, BSF implementation practices continue to evolve. BSFs are traditionally contained in concrete filter bodies (cBSFs) made with locally-sourced aggregate; however, plastic Hydraid filter bodies (pBSFs) are increasing in popularity due to their portability, durability, and anticipated scalability. Without the weight and strength afforded by concrete walls, the sand inside these filters may behave differently when subjected to disturbances that occur during regular use. This is particularly important as pBSFs are lighter and—as postulated by Stauber et al. (2012)—may be more frequently relocated or jostled. Preliminary evidence from a Samaritan’s Purse/ Agua Viva pilot project in El Salvador suggests that sand in pBSFs becomes more compact over time than it does in cBSFs (personal correspondence with Ken Morrills, 26 Jan 2013). A lab study at Lehigh University was disrupted after several concrete, plastic, and bucket-style BSFs were relocated and their flow rates dramatically decreased, presumably due to media compaction caused by the move (personal correspondence with Derek Baker, 20 May 2013).

Current recommendations for BSF installation and use emphasize the importance of keeping the filters stationary and protecting them from jarring impacts. The Centre for Affordable Water and Sanitation Technology (CAWST, 2012) recommends installing filters in a safe location where they cannot be disturbed. The Manz Guidance Manual (Manz, 2009) states: "a BSF should not be moved unless absolutely essential…It is practical to remove the media in the filter, as best possible, and then move the filter container itself. The media can then be reinstalled." Triple Quest (2011) also cautions against relocating pBSFs, but provides the following guidelines: "if it must be moved short distances, it may be lifted by its upper rim. Keep the filter upright and level and avoid jarring or dropping."

All performance and hydraulic lab studies to date have employed stationary BSFs, making it difficult to confirm or improve upon the above recommendations with respect to relocating or otherwise disturbing filter bodies. The objectives of this paper are:

1. to assess the extent to which three household disturbances (moving the filter, kicking the filter, and impacting the filter walls with a bucket while filling) affect sand column compaction and flow rate in pBSFs;
2. to determine whether any of these disturbances introduce sand gaps and preferential flow paths that could reduce a BSF's filtration performance; and
3. to explore concerns around nitrification within BSFs by comparing nitrogen species concentrations in filter effluent to those in "standing bucket" controls left to sit for equal time periods.

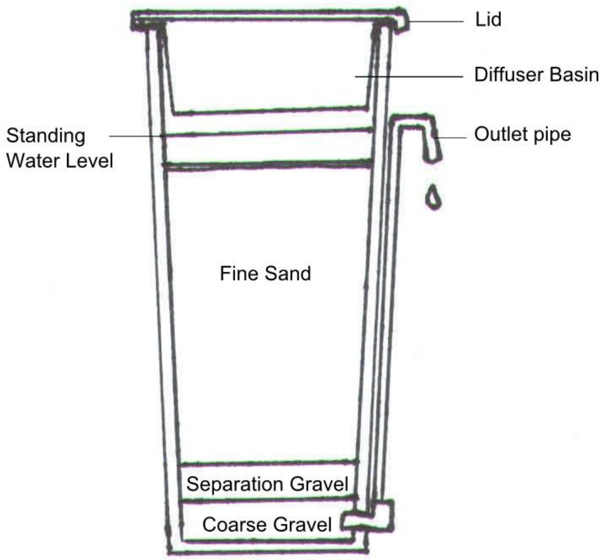

Figure 1 Cross-section showing the main components of a generic BSF.

3.2 Materials and Methods

3.2.1 Set-up

Eight Hydraid pBSFs and two concrete v10 cBSFs were tested for leaks and installed according to the most updated operating manuals (CAWST, 2012; Triple Quest, 2011). Filters contained a layer of underdrain gravel (6.25 – 12.5 mm diameter), topped with a layer of medium-sized separation gravel (3.125 – 6.25 mm diameter), and a bed of sand (ES = 0.19 mm and UC = 1.63). The depths of these three layers were: 7 cm, 5.7 cm, and 42 cm, respectively for the pBSFs; and 5 cm, 5 cm, and 46 cm, respectively, for the cBSFs as specified in the manuals. All 10 filters were run in parallel for 13 weeks and move/kick disturbances were not applied until at least 50 days had passed, to ensure filter ripening and acclimation. Filters were charged daily each Monday through Friday with one “sand pore volume” of influent water: 9.0 L for pBSFs and 7.2 L for cBSFs. The charge volumes and pause times were chosen with the following considerations:

1. charge volumes were approximately equal to the volume of water required to fill all pores within the sand portion of the filters, given the filter body dimensions and a mean porosity of 32% (determined through compaction tests);
2. this enabled the two filter types to be operated similarly from a hydraulic perspective; and
3. using only one charge per day kept the total influent volume manageable for daily preparation (under 150 L).

Influent water was prepared by filling a mixing tub with 148 L of municipal tap water from the City of Hamilton, which was left to stand overnight (18 – 24 hours or ~72 hours if over the weekend). Each morning, 5.3 L of water was removed and replaced with 5.3 L of raw sewage from the Dundas Wastewater Treatment Plant (Hamilton, Ontario) to ensure the presence of nutrients and microorganisms typical of raw water sources in lower-income countries. This dilution was chosen to approximate the coliform count of untreated water, which the WHO describes as containing 0.01% wastewater, where wastewater has approximately 10^6 – 10^{10} fecal coliform CFU/L (WHO, 2011a). Influent water was mixed for 1 - 2 minutes with a 0.25 HP batch mixer immediately after adding the sewage and before each filter was charged.

3.2.2 Disturbances Applied to Filters

A summary of the disturbances applied to each filter is given in Table 2. Filters with bucket disturbances (half of all filters) were charged each day by dropping the rim of the bucket on the side of the filter and then tipping the bucket to fill the reservoir. This represents how filters are typically charged when the user is not strong or tall enough to easily hoist the full bucket above the filter. ‘Kick’ disturbances were applied to filters to simulate side impacts that may occur from, for example, a soccer ball or person bumping into the filter. The kick was standardized through the use of a 10 lb sledgehammer attached to a fixed post. The sledgehammer was pulled back 50 cm and then released, contacting the filter at a height of 29 cm on the opposite side of the filter from the riser pipe. The move disturbance was applied to filters by walking them gently across the floor for a distance of 2.4m (8 ft).

Both cBSFs were moved rather than kicked because side impacts were not expected to have any effect on the concrete walls. Filter C1 was initially planned as a control, but it was deemed important to move it in order to assess whether pH changes observed in its effluent (attributed to flushing of cementitious material from the concrete) were due to filter movement or some other phenomenon occurring within the filter.

Table 2 Summary of disturbances applied to each plastic and concrete BSF.

Filter Number	Type	Disturbance applied (- = none; B = bucket; K = kick; M = move)	Timing of disturbance (number of days after installation)
P1	Plastic	-	**
P2	Plastic	K	51 days
P3	Plastic	-	**
P4	Plastic	M	57 days
P5	Plastic	B, K	63 days*
P6	Plastic	B	**
P7	Plastic	B, K	51 days
P8	Plastic	B, M	57 days
C1	Concrete	B, M	64 days*
C2	Concrete	M	55 days

*Delayed treatment until a long baseline had been collected for comparison as a control
Arbitrary date of 52 days after start was chosen for the control filters in the analysis presented in **Figure 2.

3.2.3 Water Quality and Characterization: Influent and Effluent

After the filters had acclimated for approximately three weeks, influent and effluent water analyses were performed with the methods outlined in Table 3. Temperature, pH, and total dissolved solids (TDS) were regularly monitored to characterize influent and effluent water, with the same parameters being measured in each effluent sample the day after they were measured in the influent. This was to account for the fact that the effluent was composed of a mixture of the influent water from the two previous days. Chemical Oxygen Demand (COD), nitrogen speciation, and alkalinity were measured in a similar manner but only towards the end of the experiment, after disturbances had been applied. Turbidity was measured regularly in influent and effluent as a proxy for filter performance and biological removal. In the last week of the study, *E. coli* removal was also measured through membrane filtration.

Influent water samples were taken by mixing the influent tub with the batch mixer and withdrawing a sample in a sterilized beaker. Effluent samples were taken as soon as the effluent had stopped (or nearly stopped) dripping from the filter; the buckets that captured the effluent were then mixed with a sterilized stir-stick and sampled in a sterilized beaker.

Standing bucket controls were used to confirm that changes in water quality from influent to effluent were the result of filtration through the BSFs and not due to physical settling or biological activity naturally occurring within the water. At least once per week during the second half of the study, a bucket containing 9.0 L of influent was left overnight. The following morning, it was stirred, sampled, and analyzed in the same manner as the effluent samples.

Table 3 Frequency and methods of characterizing influent and effluent water parameters throughout the study.

Parameter (units)	**Detection Limits/Error**	**Frequency**	**Method/Instrument**
Temperature (°C)	± 0.2°C	Every 1-4 days	Thermometer
pH	± 0.002 pH	Every 1-4 days	VWR SympHony pH meter
Total Dissolved Solids, TDS (mg/L)	1 – 1000 mg/L	Every 1-4 days	Hach conductivity/TDS meter 44600
Turbidity (NTU)	0 – 100 NTU, ± 2%	Every 1-4 days	Hach 2100Q Portable Turbidimeter
Chemical Oxygen Demand (mg /L O_2)	3 – 150, ± 3 mg/L	N = 5	Hach USEPA Reactor Digestion Method 8000 (low range)
Nitrate (mg/L N-NO3)	0.5 – 30.0, ± 0.3 mg/L	N = 5	Hach Nitrate Powder Pillows (cadmium reduction) (high range)
Nitrite (mg/L N-NO2)	0 – 0.350, ± 0.001 mg/L	N = 5	Hach Nitrite Powder Pillows (low range)
Ammonia (mg/L N-NH3)	0.015 - 2.00 mg/L	N = 5	Hach Salicylate Method, TNTplus 830 (ultra low-range)
Alkalinity (mg/L as $CaCO_3$)	10 – 250 mg/L	N=3	Titration with 0.0200 N H_2SO_4 (following USEPA SOP for total alkalinity)
Escherichia coli (CFU/mL)		N=3	Hach USEPA Membrane Filtration Method 8074 with m-Endo media, performed in triplicate

3.2.4 Sand Column Height, Flow Rate, Tracer Tests, and Depth Profiling

Implementers recommend that the standing head, which is the vertical distance from the static water level to the top of the sand column, should be 5 cm (see Figure 1). The depth of the standing head was measured every 2-4 days by averaging the distance from the static water level to the media surface at the front, middle, and back of the filter. During the first week of filter operation, the standing head was measured each day and the sand level was adjusted by adding more sand if necessary to maintain the 5 cm head as the media settled. After the first week, the filters were allowed to naturally settle without the sand levels being adjusted. Changes in sand column height were determined by:

$$\text{Change in height of sand column (mm) on day } x = H_b - H_x \qquad (1)$$

where: H_x is the measured standing head (in mm), and H_b is the baseline (mean) standing head over the first two weeks of regular operation (after the initial week of adjustments).

Maximum initial flow (MI-flow) measurements were taken by charging the filter with its regular daily charge (9.0 or 7.2 L) and measuring the volume of effluent produced during the first minute of flow. MI-flow measurements were taken every 2-4 days. Changes in flow rate for each filter were determined by:

$$\text{\% change in MI-flow on day } x = \frac{Q_x - Q_b}{Q_b} * 100 \qquad (2)$$

where: Q_x is the MI-flow rate (mL/min) on day x, and Q_b is the filter's baseline (mean) MI-flow rate over the first 2.5 weeks of filter operation after the flow rates reached equilibrium. It is important to note that MI-flow was not expected to be the same for the cBSFs and pBSFs given that each had a different surface area and driving head.

In the final three days of the study, filters were cleaned by gently agitating the top of the sand surface and then decanting the cloudy water, as is typically done in the field to reduce the biological layer if flow rates become low. This was to confirm that the observed changes in flow rate after disturbances were caused by sand compaction and not by an accumulation of biological material on the sand's surface.

Tracer tests were performed before and after disturbances to evaluate whether the disturbances introduced preferential flow paths caused by the media shifting on impact. On day one of these hydraulic tracer tests, the regular daily charge volume was prepared with the addition of 40 mg/L Acid Yellow 17 and the filter was charged as usual. During the following 4-5 days, regular (no dye added) charges were applied to the filter and the effluent was sampled in 200-500 mL fractions, each of which was weighed to determine its exact volume. Absorbance at 400 nm was used to determine the concentration of Acid Yellow 17 in each sample, and the effluent concentration profile was plotted. Four-day batch tests were conducted with sand from the top of the BSFs to confirm that biodegradation and adsorption did not significantly affect dye concentrations over the length of the tracer tests.

At the end of the 13-week experiment, tall narrow windows were cut into the back of four filters (P1, P3, P5, P8) and the heights of the sand-gravel interface and of the sand column were measured. Two cores (diameter 8.9 mm) were taken from each of the following depths: 0 cm, 5 cm, 20 cm, and 38 cm from the sand's surface. Volatile solids were measured for each core by ignition at 550°C following EPA standard method 1684.

3.3 Results and Discussion

3.3.1 Physical effects of disturbances

There were no visible effects of disturbances on the physical appearance of the filter bodies; no cracks or distortions appeared in the bodies after moving or kicking. The circumferences of the pBSF bodies at three different heights (top, middle, bottom) did not change after they were kicked (data not shown).

Figure 2 depicts the changes in sand column height and flow rate, relative to baseline values, for each filter before and after disturbances occured. The significance of the differences between each filter's pre- and post-disturbance values, as given by Wilcoxon rank-sum tests, are summarized in Table 4 (H_O: no change in sand column height or flow rate after disturbances; $\alpha = 0.05$). Wilcoxon rank-sum tests were chosen due to the small sample size and because the data collected were not normally distributed.

All filters showed significant ($p < 0.001$) decreases in sand column height due to natural settling over the duration of the experiment; however, changes were much greater in the filters that had been kicked and moved than in the controls (Table 4; Figure 2). There was no significant difference between "bucket" and "no bucket" controls ($p = 0.392$; not reported in table). This suggests that when filters are not disturbed in any other way, users can expect some minor settling of sand media that is not affected by whether the bucket contacts the filter wall during charging.

The kicked filters lost 9 to 14 mm of sand column height between pre- and post-disturbance, with the sand level in "bucket" (P5 and P7) dropping significantly more than in "no bucket" (P2). The plastic filters that were moved, P4 ("no bucket") and P8 ("bucket"), showed even greater drops of approximately 29 and 25 mm of sand, respectively. The cBSFs, C1 and C2, only lost 9 and 6 mm in sand column depth, respectively, after being moved the same distance across the floor. In these filters, the concrete walls and/or square design afforded the sand some protection, preventing it from compacting as dramatically.

Two of the control filters (P3 and P6) showed no significant change in MI-flow; control filter P1, on the other hand, showed a slight and significant ($p = 0.033$) increase of 7 mL/min (1.5%). The fact that the control filters did not exhibit decreases in flow during regular use was unexpected; typically, the gradual development of a biological layer is accompanied by a decline in filter flow rate (Stauber et al., 2006). Previous studies reported acclimation periods—the period during which biological removal increases and flow rate decreases—ranging from several weeks (Elliott et al., 2011) to 30 days (Baumgartner et al., 2007; Elliott et al., 2008). In this study, there was an initial drop and leveling of MI-flow during the first week of operation, but this was not followed by a further decline over the following 12 weeks except when disturbances were applied. Filter acclimation may not have occurred as quickly as in other studies due to the relatively low microbial and nutrient loadings of the influent water used here. Additionally, residual monochloramine in the influent water may have reduced biological activity and slowed the rate of biological layer development.

For non-control filters, greater changes in sand column height generally contributed to proportionally greater decreases in flow rate (Figure 2). Decreases in MI-flow (mL/min) from before to after disturbances were significant ($p < 0.001$) for all the three kicked pBSFs (87.5, 161, and 374 mL/min decreases, respectively, for P2, P5, and P7). MI-flow did not significantly change for moved cBSFs C1 or C2, both of which seemed to recover somewhat from an initial dip in MI-flow that occurred shortly after being moved. Moved pBSFs P4 and P8, on the other hand, experienced decreases in MI-flow of of 218 mL/min and 171 mL/min, respectively, from pre- to post-disturbance levels.

Of all the kicked and moved filters, P7 (bucket + kicked) showed by far the most substantial drop in MI-flow rate (decrease of 374 mL/min or 84%) despite being subjected to the same standardized kick as P2 and P5 (Figure 2d, Table 4). Hydraulic testing (results not shown) revealed that it took approximately 2.5 hours to filter 75% of each daily charge for P7, whereas the other filters could process the same volume of water in only 30 – 50 minutes. A user repeatedly charging P7 as soon as the head declined would require over 15 hours to filter 40 L, the minimum WHO recommended volume for two household members. For filter P5 (also kicked), this same volume could be filtered in less than 5 hours despite the filter having decreased flow compared to the controls. With one in three kicked filters developing a constrainingly low flow rate, it is reasonable to conclude that side impacts to pBSFs could affect the convenience of using a BSF and could contribute to frustration and ultimately disuse. If households stop using BSFs because of the increased time cost associated with these disturbances, they may be at higher risk of contracting water-borne diseases. For this reason, users should be strongly urged to install filters in a secure location where they will not be in danger of impacts from passers-by, domestic animals, or other disturbances.

It also became clear during this study that with a maximum charge volume of approximately 9 L for the latest version of the Hydraid pBSFs used here, multiple charges would be required to filter the volumes of water required for daily household use. The pBSFs used in this study have 5 cm more sand than previous versions of the filter; this reduces the charge volume to pore volume ratio as recommended by Elliott et al. (2011), but may also lead to shorter pause times due to the need for more frequent charges. Jenkins et al. (2011) found that shorter pause times between charges contributed to lower removal of bacteria and viruses. Clearly a balance needs to be found between the charge volume and the number of charges required. One option may be to increase the size of the pBSF filter bodies. This would decrease their portability (thereby reducing risk of filters being moved), and would allow for a larger charge volume without compromising the charge volume to pore volume ratio.

It is unclear whether the differences between effects on cBSFs and pBSFs from filter moving are more strongly linked to the shape of the filter bodies (round for pBSFs and square for cBSFs) or to the weight and strength of the filter material. New BSF designs are emerging that are more portable and utilize light materials such as sheet metal (Smith, 2013) and other smaller plastics (e.g. the plastic bucket design used by Collin (2009)). Further work is required to assess which factors most strongly influence the susceptibility of BSFs to sand compaction and associated flow declines so that new designs can take these into consideration.

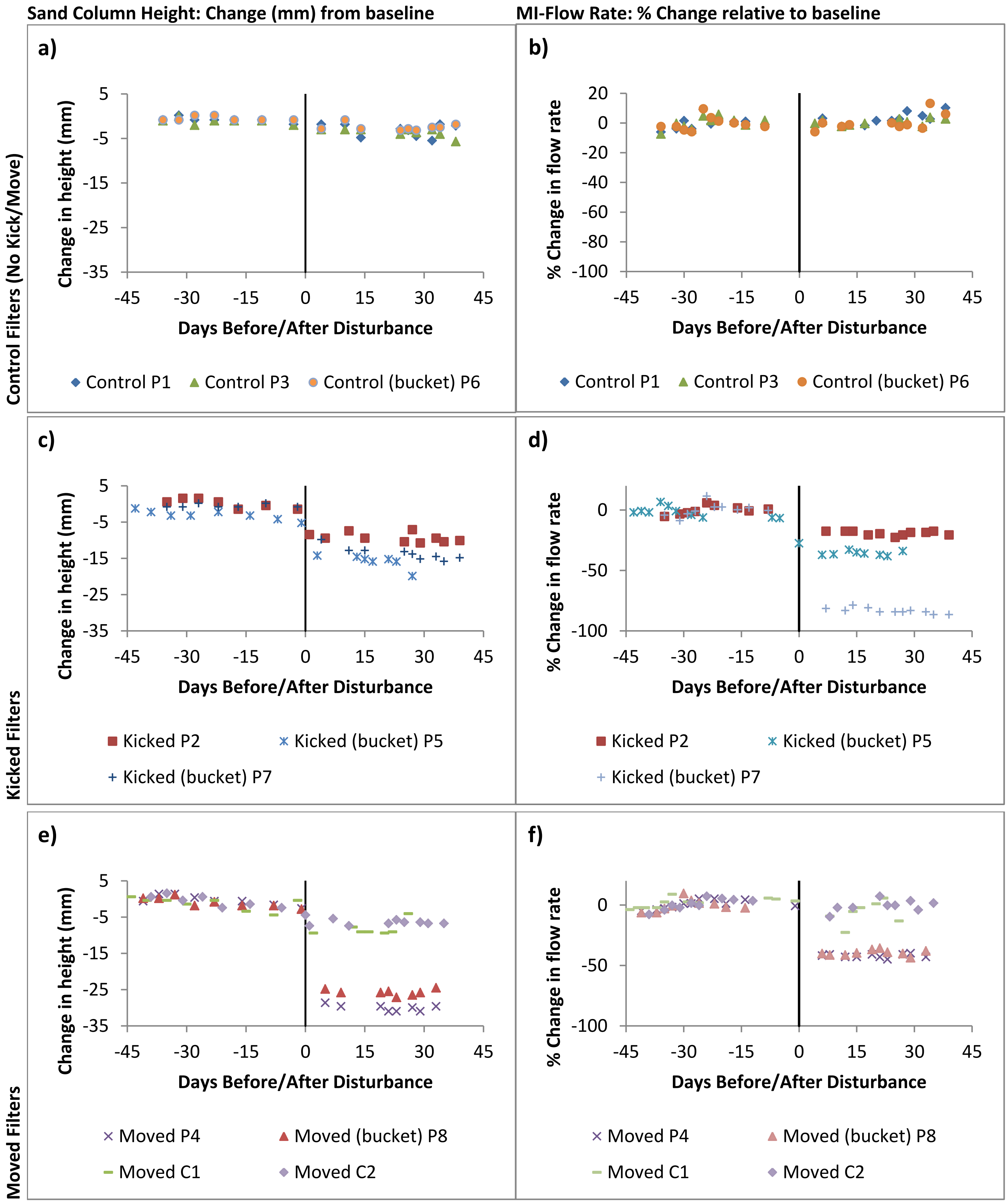

Figure 2 Change in sand column height (left) and percent change in flow rate (right) for control (top, a and b), kicked (middle, c and d) and moved (bottom, e and f) BSFs. An arbitrary date in the middle of the test period was chosen as day 0 for the control filters in panels a and b.

Table 4 Median measurements and p-values from Wilcoxon rank-sum tests for differences in standing head and IM-flow rate before and after disturbances. Bold values indicate significant differences at the $\alpha = 0.05$ level.

		Control Filters			Kicked Filters			Moved Filters			
		P1	**P3**	**P6**	**P2**	**P5**	**P7**	**P4**	**P8**	**C1**	**C2**
Standing Head (mm)	Before disturbance (n)	51 (11)	51 (11)	50 (11)	50 (11)	53 (13)	50 (11)	51 (12)	51 (12)	51 (13)	53 (11)
	After disturbance (n)	53 (9)	53 (9)	53 (9)	59 (10)	66 (7)	64 (9)	80 (8)	76 (8)	60 (7)	59 (10)
	p-value	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**
IM-Flow Rate (mL/min)	Before disturbance (n)	468 (9)	495 (10)	413 (10)	489 (9)	460 (11)	444 (10)	505 (11)	438 (10)	324 (12)	268 (10)
	After disturbance (n)	475 (9)	493 (10)	415 (10)	400 (11)	299 (8)	70 (11)	288 (10)	266 (10)	300 (7)	265 (9)
	p-value	**0.033**	0.849	0.879	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	0.056	0.484

Depth Profiling: The four side-profiled filters had similar sand-gravel interfaces (examples shown in Figure 3) and there was no evidence that sand had fallen into the gravel layers of the disturbed filters even in cases where substantial decreases in sand column height (as much as 29 mm in P4) had been observed. This suggests that the decreases in sand column height resulted from sand compaction rather than sand falling into the supporting media below. In the field, implementers should pay careful attention to their quality control for gravel sizing as larger gravel may not be as effective in preventing sand from falling into the supporting layer. The separation gravel used in this study was carefully sorted and pre-packaged specifically for BSFs (diameter < 6.25 mm) and appeared to perform well in this capacity.

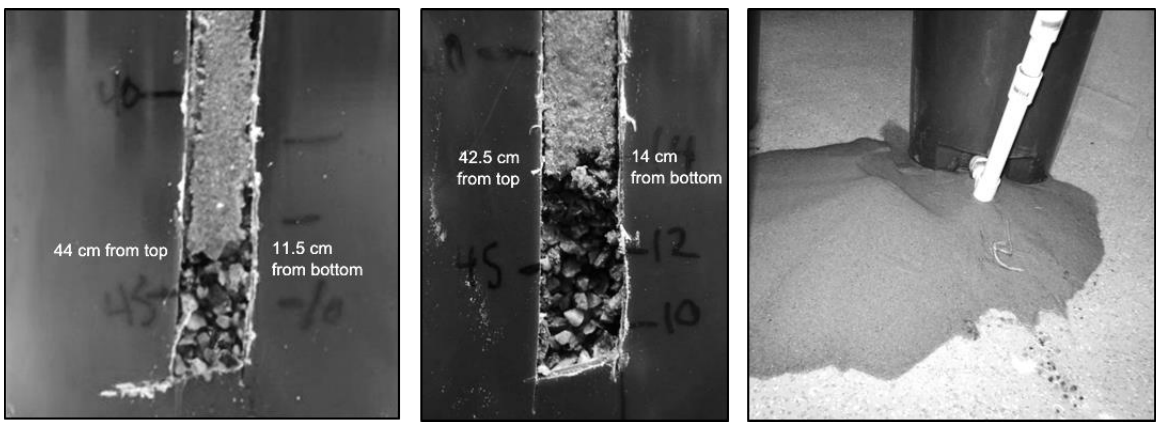

Figure 3 Side profiles of filter P5 (kicked, left image) and filter P3 (control, centre image) show a similar sand-gravel interface at different depths. Right: removing the riser pipe from filter P7 (kicked) caused a surge of sand to rush out of the filter.

The sand-gravel interface was approximately 2.5 cm lower at the back of filter P5 (kicked) than it was in the dismantled control filters (Figure 3 left and centre). This could be a consequence of inconsistent installation of the sand or gravel in the filter or it may have resulted from gravel being forced up and displaced when the kick occurred. The latter is a reasonable conclusion given that P7 (also kicked, but not side-profiled) displayed signs of sand and gravel displacement. When P7 was dismantled, sand poured out of the bottom opening as the riser pipe was removed (Figure 3 right). This did not happen to any of the other filters, and likely means that in P7 a wedge of sand dropped down to the bottom layer of the filter, displacing the gravel when the kick occurred and resulting in the much lower flow rates described above. In cBSFs, the riser pipe connects to the floor of the filter, but in pBSFs it enters on the side. One possible design alteration to prevent sand wedges from entering the outlet pipe if an impact occurs is to add an elbow joint in the pBSF riser pipe so that it connects right at the floor of the filter beneath both gravel layers. It should be noted, however, that this study was unable to evaluate whether side impacts would have the same effect if they were applied at different radial or vertical locations on the filter bodies; the same impact, delivered in a slightly different location, might not have had the same effect.

Volatile solids analysis revealed that only cores taken at the sand surface (depth of 0 cm) had, on average, higher volatile solids than sand prior to installation in the filter (Table 5). These results indicate that very little organic matter was accumulating below the sand surface, even at depths of only 5 cm. Similar findings were reported by Haarhoff and Cleasby (1991), who concluded that microbial populations rapidly decline several centimeters below the sand surface of traditional slow sand filters. Volatile solids values at lower depths were similar for control and disturbed filters; there was no evidence of biomass accumulation lower in the filter bed that could explain the decrease in MI-flow described earlier in this paper for the disturbed filters.

Table 5 Mean % Volatile Solids at given depths in side-profiled pBSFs.

	Mean % Volatile Solids at each depth			
	0 cm	5 cm	20 cm	38 cm
Control Filters (P1 and P3)	**0.425**	0.113	0.125	0.102
Disturbed Filters (P5 and P8)	**0.247**	0.101	0.070	0.073
Average for all filters	**0.336**	0.107	0.097	0.088

Bold values indicate mean %VS greater than 0.126%, the mean VS value for sand prior to installation.

When filters were cleaned during the last week of operation there was no subsequent flow recovery as would be expected if a build-up in the biological layer was responsible for the low flow rates. This finding has important implications for filter monitoring, as technicians are typically instructed to clean filters if flow rates become limitingly low. If decreased flow rates are caused by sand compaction rather than organic matter accumulation, cleaning will not improve the flow rate and could in fact result in decreased filter performance until the biological layer recovers. For this reason, it would be worthwhile to develop a troubleshooting guide so that technicians can be trained to recognize that low flow, when combined with decreased sand column height, may point to a physical disturbance.

Tracer Tests: The filters in this study had similar concentration profiles to those observed by Elliott et al. (2008) in their tracer tests, but with longer tails following the negative input (data not shown). Elliott et al. (2008) used a clean, unripened filter and a smaller tracer molecule (NaCl) while the tests reported here used a comparatively larger tracer molecule and biologically active filters in which some retardation was expected.

While minor variations in curve shape were observed, there was no obvious change in tracer transport caused by the moving or kicking disturbances aside from the decreased flow rates discussed above (data not shown). This indicates that preferential flow paths did not develop in the filters as a result of sand shifting on impact. This is an important finding; although the disturbances tested here may lead to reduced flow and ease-of-use, they should not affect a filter's ability to remove contaminants, as would be the case if preferential flow paths developed. This conclusion is further supported by the filter performance (turbidity and *E. coli* removal) results presented below.

3.3.2 Filter performance and influent/effluent water quality

Water quality parameters for influent and effluent waters and standing bucket controls are shown in Table 6. Effluent typically had higher pH and lower turbidity, COD, and *E. coli* than influent water; this was in agreement with findings published by previous authors such as Chiew et al. (2009) and Stauber et al. (2006). In general, TDS, temperature, and pH of effluent were not affected by the disturbances applied to the filters and did not substantially differ between filters. One exception was concrete filter C2, which had an effluent pH as high as 9.25—just outside WHO guidelines—at the beginning of the study. This pH gradually declined, reaching a similar pH to the other filters after approximately 360 L (7.2 L each weekday for 10 weeks) of water had passed through. A similar pH shift was also observed by Murphy et al. (2010b) who attributed this change to the leaching of calcium carbonate from the concrete filter body.

Mean influent and effluent turbidity levels were 3.38 and 0.28 NTU, respectively (Table 6), which represents an average turbidity removal of 92%. Effluent turbidity was consistently below the WHO guideline value of 5 NTU and the recommended maximum level of 1 NTU. When effluent turbidity was compared to the previous day’s influent levels, removal rates ranged from 71-99% during normal operation. Previous lab studies found similar mean turbidity removal rates of 89% for experimental PVC BSFs (Jenkins et al., 2011) and 88-97% for pBSFs (Kennedy et al., 2012). Turbidity removal in the field, however, is less consistent: for example, a recent field study in Honduras by Fabiszewski de Aceituno et al. (2012) found turbidity removal of less than 5% for a variety of tested water sources, whereas a similar study in Cambodia found 82% removal on average (Liang et al., 2010).

Effluent turbidity was significantly ($p < 0.05$), but not substantially, lower (i.e. decreases of 0.01 to 0.15 NTU) after disturbances for six of the ten filters including control P3 (data not shown). This trend was not correlated to any particular disturbances, and none of the filters had significantly increased effluent turbidity post- disturbances. However, in the first charge immediately following a disturbance, each pBSF demonstrated a sudden and brief spike in effluent turbidity up to between 1.26 NTU (P4) and 15.4 NTU (P5) (Figure 4; note that data from disturbance days when turbidity spiked were excluded from the pre/post disturbance analysis described above). This is likely a result of colloidal material and inorganic salts being mobilized in the plastic filters upon impact. Without knowing whether the colloids released could include pathogens, it is recommended that users do not consume water immediately after disturbances because these elevated turbidity levels may affect potability. The concrete filters had less dramatic spikes of up to 0.42 NTU (C1) and 0.51 NTU (C2) immediately after being moved – well within WHO guidelines—and so their thicker walls may have protected the media from being disturbed to the same extent.

Due to its greater consistency and ease of measurement, turbidity was chosen as a proxy for biological removal during the majority of this study, and filter performance was confirmed through enumeration of *E. coli* in the influent and effluent during the last week of filter operation. Mean influent and effluent *E. coli* concentrations were 1737 and 262 CFU/100mL, respectively (Table 6). Removal rates ranged from 43% to 94%, although there were insufficient data to test for the significance of differences between filter groups.

Table 6 Filter influent and effluent water characteristics.

		Temp (^oC)	pH	Turbidity (NTU)*	TDS (mg/L)	COD (mg/L)	Alkalinity (mg/L CaCO3)	*E. coli* (CFU/100mL)
Influent	Mean	20.7	7.23	3.38	181.7	17.5	100.9	1740
	(Min, Max)	(18.5, 23.0)	(6.94, 7.67)	(1.84, 8.05)	(175.6, 189.4)	(6.3, 26.5)	(99.3, 103.3)	(960, 2460)
	N	32	33	30	28	11	3	3
Effluent	Mean	21.8	7.61	0.28	189.6	3.1	91.6	262
	(Min, Max)	(19.5, 24.3)	(7.17, 9.25)	(0.11, 0.63)	(144.0, 240.2)	(0.6, 6.1)	(69.0, 122.0)	(113, 465)
	N (per filter)	25-29	25-28	21-25	24-28	4-5	1-2	3
Standing Bucket Control	Mean	22.1	7.41	2.41	182.3	-	-	11800
	(Min, Max)	(21.5, 23.0)	(7.33, 7.5)	(1.22, 3.77)	(176.4, 189.2)	-	-	(3180, 27100)
	N	7	6	6	6	-	-	3

*Excluding the 24-hour period immediately after each kick/move, when turbidity spikes of up to 15.4 NTU were measured.

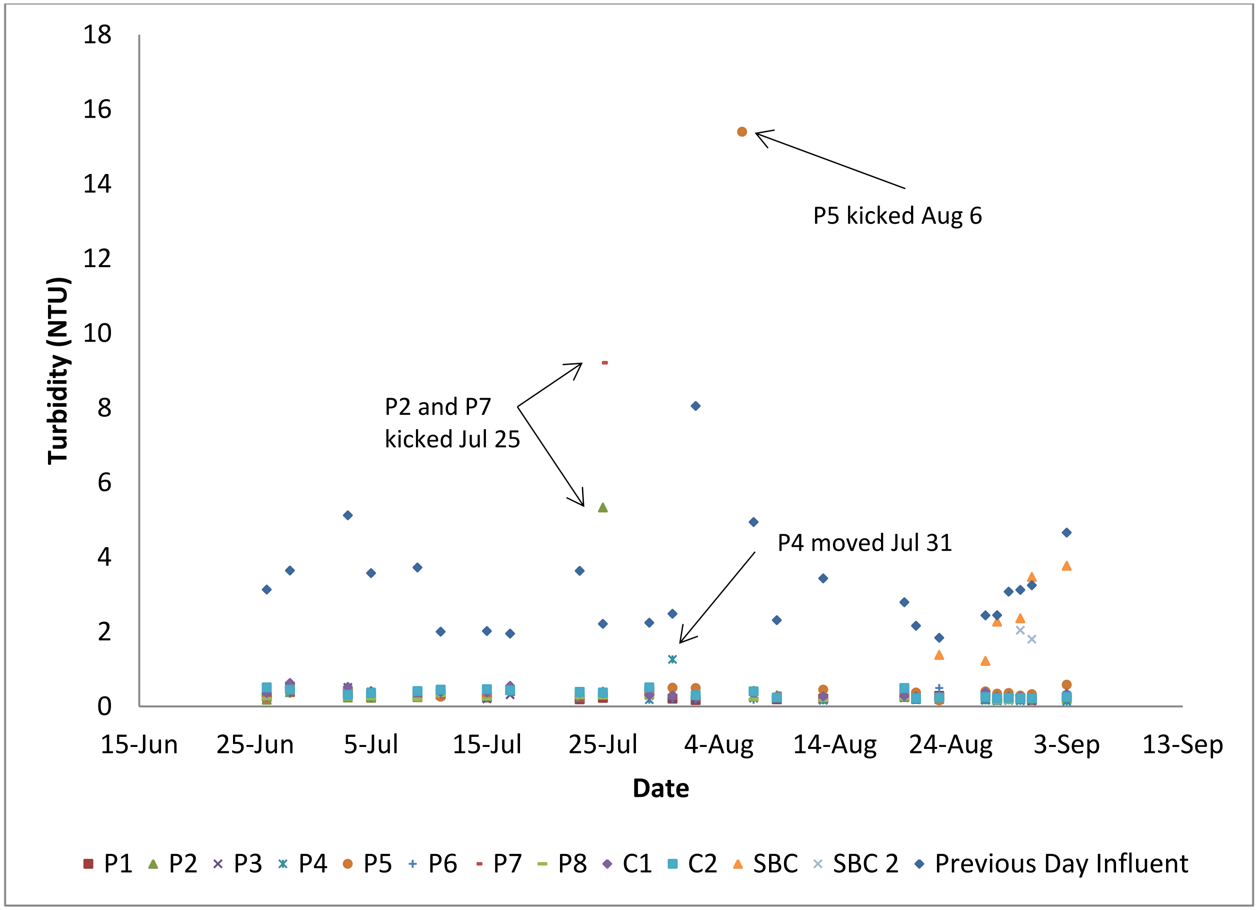

Figure 4 Turbidity measurements over the course of the study for each filter and for influent water. Standing bucket controls left for one or two nights (SBC and SBC2, respectively) are shown for the last 3 weeks.

3.3.3 Nitrogen Speciation

Nitrogen speciation showed substantial variation; there was no pattern to which filter had the highest nitrate or nitrite levels on any given day. Ammonia, nitrate, and nitrite concentrations are shown in Figure 5 for influent, effluent (average of all filters), and standing bucket controls.

The WHO (2011b) specifies that in drinking water: nitrate should be below 11 mg-N/L; nitrite should be below 0.9 mg-N/L; and the combined ratio of each substance to its guideline value should not exceed 1. The concentrations of nitrite and nitrate in all samples were well below these WHO guidelines (Figure 5). While the concentrations were thus not high enough to pose a health concern, results are consistent with other reports of nitrification occurring inside the BSFs. Specifically, there was a dramatic reduction of ammonia in all filters (1.37 mg NH_3-N/L lower in effluent than influent, $p < 0.001$), accompanied by increases in nitrate (0.400 mg $N0_3$-N/L higher in effluent; $p = 0.020$) and nitrite (0.036 mg NO_2—N higher in effluent; $p = 0.002$) (Figure 5). This reduction in ammonia and increase in nitrate and nitrite suggests that nitrification (the bacterially-mediated oxidation of ammonia to nitrite and further oxidation of nitrite to nitrate) was occurring in the filters, as per the findings of Murphy et al. (2010).

The standing bucket controls showed similar trends but much smaller changes in speciation: there was a slight decrease in ammonia (0.22 mg NH_3-N/L, $p = 0.095$) accompanied by a slight but not significant ($p = 0.62$) increase in nitrate and a very slight increase in nitrite (0.009 mg NO_2-N/L, $p = 0.01$). Higher ammonia levels and lower nitrate/nitrite in the standing buckets compared to effluent indicate that bacterially-mediated processes were occurring much more efficiently in the BSF, and that the loss of ammonia in the filter cannot be attributed solely to physical processes such as volatilization.

Wu et al. (2013) also observed significant nitrate and nitrite increases from influent to effluent; however, these changes were only substantial when influent water was high in ammonia, and despite observing both processes in the field, the authors observed only nitrification (not denitrification) in their two control filters in the lab. The results from this study agree with their findings and further demonstrate that the established ecology of the BSF contributes to changes in nitrogen speciation beyond those that would naturally occur within influent water.

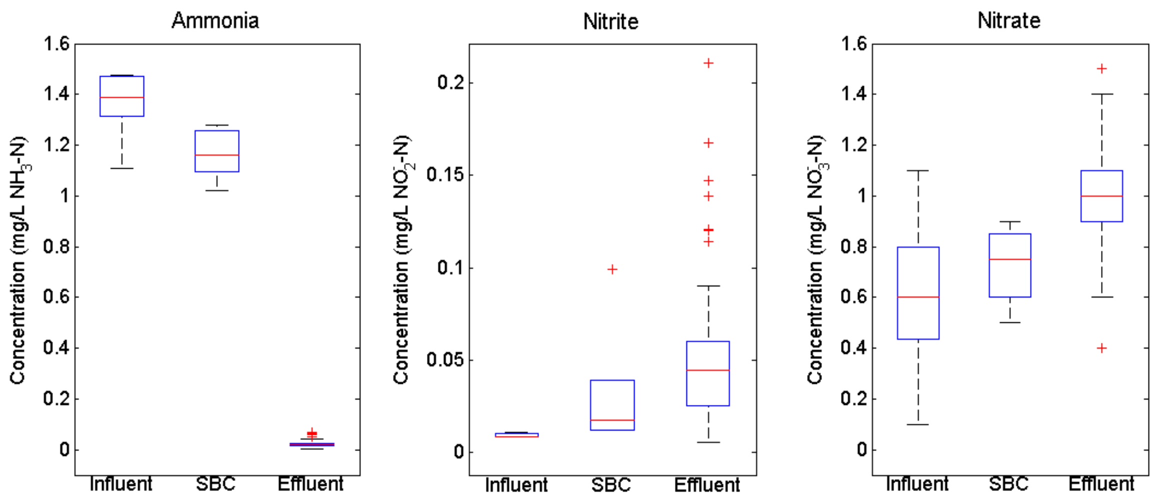

Figure 5 Nitrogen speciation data from five dates in the last three weeks of the study. Boxes represent the first, second, and third quartile values. “Effluent” shows the mean of all 10 filters and SBC refers to standing bucket controls left overnight.

3.4 Conclusions

Key findings:

- Small daily (bucket) impacts did not affect sand compaction, flow, or filter performance.
- Larger one-time disturbances (moving and side impacts) caused significant decreases in sand column height of 6 – 9 mm for cBSFs and 9 – 29 mm for pBSFs. Control filters decreased by only 2 – 3 mm during the same period.

- Sand compaction caused by these one-time disturbances led to significant decreases in flow rate. MI-flow rate decreased by 18 - 84% for disturbed pBSFs but did not significantly decrease for control pBSFs or moved cBSFs. Sand compaction appeared to be the primary reason for these decreases in flow; there was no evidence that it was caused by organic matter accumulation within or on the surface of the media.
- There was no evidence of preferential flow paths introduced by the physical disturbances.
- Effluent quality was generally unaffected in the long term by moving and side impacts. However, substantial turbidity spikes occurred immediately after disturbances.
- Changes in nitrogen speciation were consistent with nitrification occurring within the BSFs, and were shown to be mediated by the ecology of the BSFs.

Key implications for guidelines around installation and use:

- The results presented here confirm the importance of recommending that filters are installed in safe/secure locations where they are unlikely to be moved or bumped. Users should be urged not to relocate their BSFs.
- Technicians could be trained—perhaps through the implementation of a new troubleshooting guide—to recognize that an abrupt decrease in sand level, when accompanied by a decrease in flow rate, may point to a physical disturbance. Technicians otherwise typically attribute low flow rates and decreased sand column height to an accumulation of organic matter and to improper installation or cleaning, respectively.
- To avoid putting users' health at risk, water should not be consumed without additional treatment for the first 1-2 charges immediately after any disturbance or suspected disturbance.

3.5 Acknowledgements

This research was carried out with financial support from the Arthur J.E. Child Foundation and the Natural Sciences and Engineering Research Council of Canada (NSERC). The primary author's graduate study is supported by the NSERC Julie Payette Fellowship and by the Ontario Graduate Scholarship program. Special thanks to Anna Robertson and Peter Koudys for providing equipment and assistance in the lab.

3.6 References

To avoid unnecessary duplication, references from this chapter have been combined with those from the rest of this thesis and can be found following Chapter 4.

Chapter 4

Conclusions and Recommendations

4.1 Findings and relevant recommendations

Conclusions from this study are listed at the end of Chapter 3. These were shared with filter implementers from Triple Quest and Samaritan's Purse Canada during a conference call on 25 November 2013, and again through a presentation to members of the CAWST learning community on 13 January 2014. A set of recommendations was discussed and are described below within the context of the project's key findings. These recommendations were agreed-upon by the practitioners and researchers at both meetings, and thus there has already been some mobilization around the knowledge and ideas described below.

Finding 1: Moving and kicking caused marked decreases in sand column height, especially for pBSFs. This led to substantial decreases in flow rate.

Implications/Recommendations for Practitioners:

It is suggested to keep existing recommendations for installing BSFs in safe/secure locations. Care should be taken to ensure that the filters are protected from impacts; it may be helpful to encourage families to install them in a corner away from any busy activity. Manuals can also emphasize the importance of preventing forceful impacts of any kind and of not relocating the filters once installed. The results of this study clearly show that although the pBSFs are more portable than their concrete counterparts, their flow rates may be more easily jeopardized by unnecessary movement. Gently moving a cBSF short distances (i.e. across a room due to household renovations) may not be detrimental, but the filters should always be reinstalled if moved to a new home.

Practitioners have found that users are highly motivated to keep their water as clean as possible; when users are informed that unnecessary filter cleaning will compromise water quality, they are careful to only clean when necessary. It is recommended that users are informed in a similar manner of the adverse effects of moving on the BSF's performance, to motivate them to keep the filters stationary and protected. As discussed in Chapter 3, a troubleshooting guide could also be added so that technicians can better recognize the symptoms of physical disturbances and take appropriate actions rather than performing unnecessary—and potentially harmful—extra cleanings.

Recommendations for Future Research:

Triple Quest recommends moving pBSFs by gently lifting them from the rim. With limited filters available for testing and with a desire to keep moving treatments consistent between cBSFs and pBSFs, this study was unable to assess the impacts of Triple Quest's recommended moving approach. Future evaluations of how the filters perform after moving in this manner will be useful for the development of more informed recommendations.

Further work is required to assess which factors most strongly influence the susceptibility of BSFs to sand compaction and decreases in flow rate. The differences between effects on cBSFs and pBSFs from filter moving may have something to do with the shape of the filter bodies (round for pBSFs and square for cBSFs) rather than only the weight and strength of the filter material. With the emergence of new BSF designs that are more portable and utilize light materials such as sheet metal (Smith, 2013) and other plastics (e.g. the bucket design used by (Collin, 2009)), this question may become increasingly important.

Finding 2: There was no evidence of preferential flow paths and effluent quality was generally unaffected by disturbances. However, there were brief spikes in turbidity immediately after each disturbance occurred.

Implications/Recommendations for Practitioners:

Following the precautionary principle, it is recommended that water should not be consumed without additional treatment for the first 1-2 charges immediately after any disturbance or suspected disturbance.

Recommendations for Future Research:

Further research is recommended to identify the source of the extra turbidity caused by moving and kicking. This turbidity may originate from one or more of (a) inorganic salts that were dislodged from the underdrain of the filter; (b) colloids that could contain pathogens; or (c) sand from the filter media.

Finding 3: Sand was compacted but did not appear to fall into the gravel layer.

Implications/Recommendations for Practitioners:

This is an important finding and these results validate the existing underdrain design. It is important for practitioners to make sure that gravel size is small enough to prevent sand from collapsing into the supporting layers.

Finding 4: There is evidence of nitrification in BSFs that appears to be mediated by the ecology of the filter.

Implications/Recommendations for Practitioners:

There should be some level of responsibility on the part of filter implementers to ensure that: (1) high levels of nitrate/nitrite are not occurring in filter effluent; (2) users in highly agricultural areas where nitrogenous compounds are prevalent are warned of the possible risks; and (3) education around safe water includes the importance of breastfeeding young infants (rather than using formula) particularly in areas where nitrate/nitrite levels may be high.

4.2 Suggestions/ considerations for improving pBSF designs

Based on the findings above, research and development for improved pBSF designs could benefit from the following considerations. Note that unlike the recommendations above, these ideas represent preliminary suggestions from the author that have not been verified or checked for feasibility during the above-mentioned presentations to other practitioners and researchers in the field.

The pBSFs were more likely than cBSFs to experience decreases in flow after disturbances, which may be frustrating for users. The maximum charge volume on the latest pBSF design is small (9 L) and thus users may wish to charge more often but will be forced to wait for the previous charge to filter through if the flow rate has been compromised. While the pBSF design used in this study (with an extra 5 cm of sand and a decreased supernatent reservoir) reduces the charge volume to pore volume ratio as recommended by Elliott et al. (2011), this may lead to shorter pause times and reduce the filter's performance (Jenkins et al., 2011). Further research may be needed to find the appropriate balance between charge volume and the number of charges required. One option is to increase the size of the pBSF filter bodies, as discussed in Chapter 3.

Although the sand did not fall into the gravel layer, there was evidence that the gravel shifted on impact. In one filter (P7), this caused a wedge of sand to clog the outlet pipe. This is an issue that could be mitigated through improvements to the outlet pipe and underdrain design. There is a tension between (a) providing enough gravel to adequately support the filter media and prevent it from clogging the outlet pipe and (b) ensuring that the media bed is deep enough to ensure effective filtration. One approach would be to add an elbow joint in the pBSF riser pipe after it enters the filter body, so that the opening will be at the bottom of the filter where there is more gravel to protect the opening from getting clogged with sand. Alternatively, a perforated pipe along the very bottom of the filter (below the underdrain gravel) could be used to collect water and connect into the outlet pipe.

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Appendix A: Details Regarding Analytical Methods

A comprehensive list of analytical methods and the instruments/procedures used for analysis is given in Table 2 in Chapter 3. Nitrogen speciation and COD tests used standard HACH reagents and methods, as listed and cited in the table. Further information about the methods used for turbidity and *E. coli* enumeration follows:

Turbidity

The Hach 2100Q Portable Turbidimeter measures turbidity in nephelometric turbidity units (NTU). Test samples were mixed and poured into a 10 mL sample cell, which was inserted into the turbidimeter. The meter detects the amount of scattered light that is deflected 90° from the incident beam by suspended particles in the water sample. Signal Average mode was used to compensate for reading fluctuations caused by drifting of sample particles. This mode rapidly measures 12 times and the final recorded result is the average of all 12 readings.

E. coli

E. coli were enumerated using HACH USEPA Membrane Filtration Method 8074 with m-Endo media. Samples of 1 – 50 mL (depending on concentration) were diluted to a total volume of 50 mL and then filtered through the membrane filtration unit onto a sterile filter paper with grids for enumeration. This was repeated in triplicate and negative controls (distilled water only) were used each time to confirm no cross-contamination. Once the sample was completely filtered, the funnel was removed and the filter was transferred with sterilized tweezers to a petri dish containing an absorbent pad that had been saturated with 2.0 mL of m-Endo media, prepared according to package directions. The membrane filtration unit (Figure A-1) was sterilized between each sample with disinfectant spray and then rinsed five times with deionized water. Petri dishes were incubated upside-down for 24-hours at 36°C and colonies with a dark/shiny sheen were counted to determine the number of *E. coli* colony forming units.

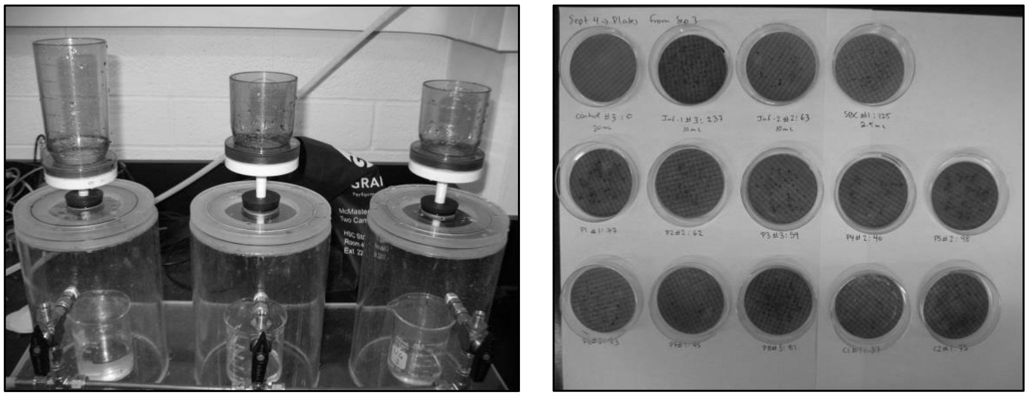

Figure A- 1 Membrane filtration apparatus (left) and examples of plates after incubation (right).

Appendix B: Photos of Set-up and Procedures

Figure B- 1 Bin for mixing influent water (left) and overview of lab set-up with all ten filters (right).

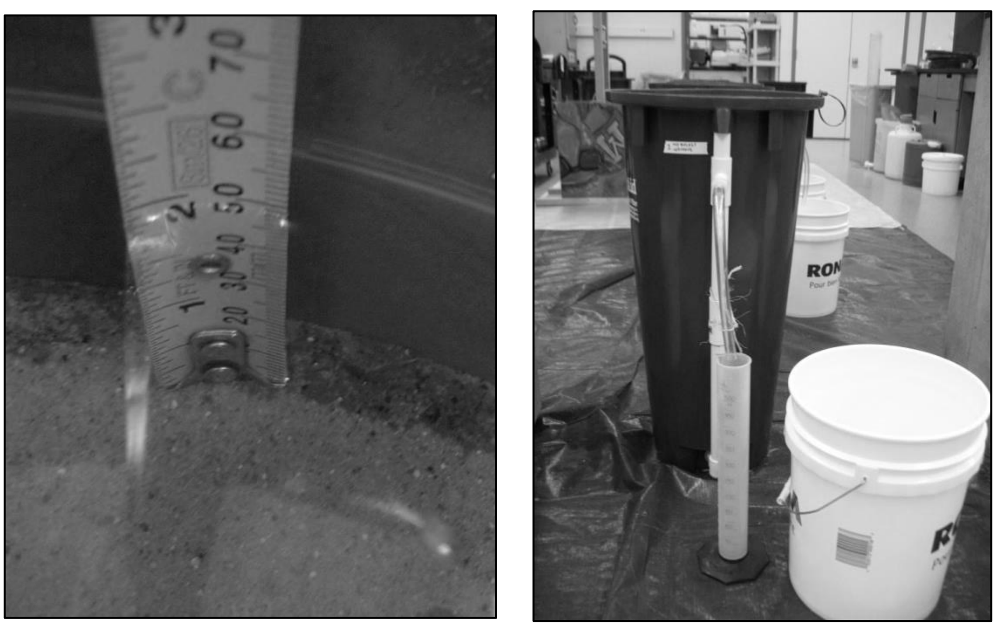

Figure B- 2 Left: standing water level was measured from the static water level to the top of the sand (average of front, middle, and back of filter). Right: Maximum-Initial Flow was measured as the volume eluted in the first minute of flow.

Figure B- 3 Techniques used for bucket (top), kick (middle), and move (bottom) disturbances.

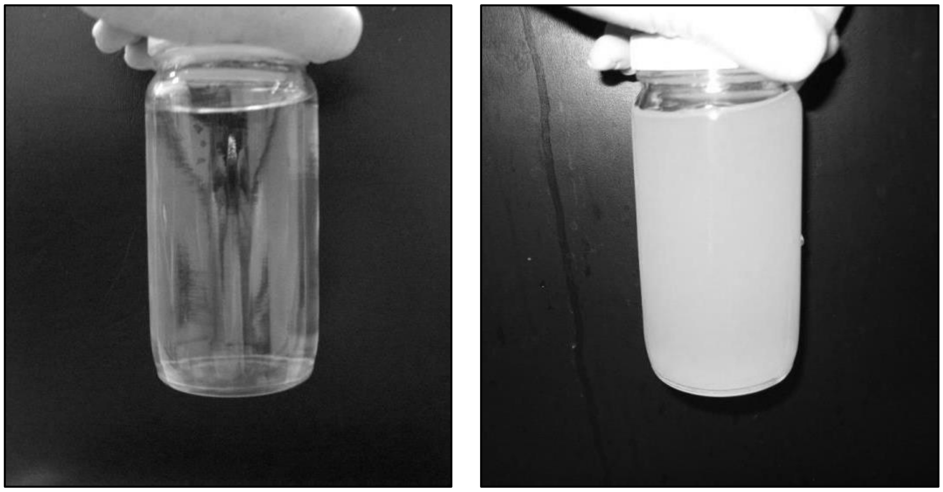

Figure B- 4 Comparison of turbidity before kick (left, 0.49 NTU) and after kick (right, 15.4 NTU) for P5. This was the most dramatic turbidity spike.

Appendix C: Additional Data

Supplemental Influent and Effluent Data

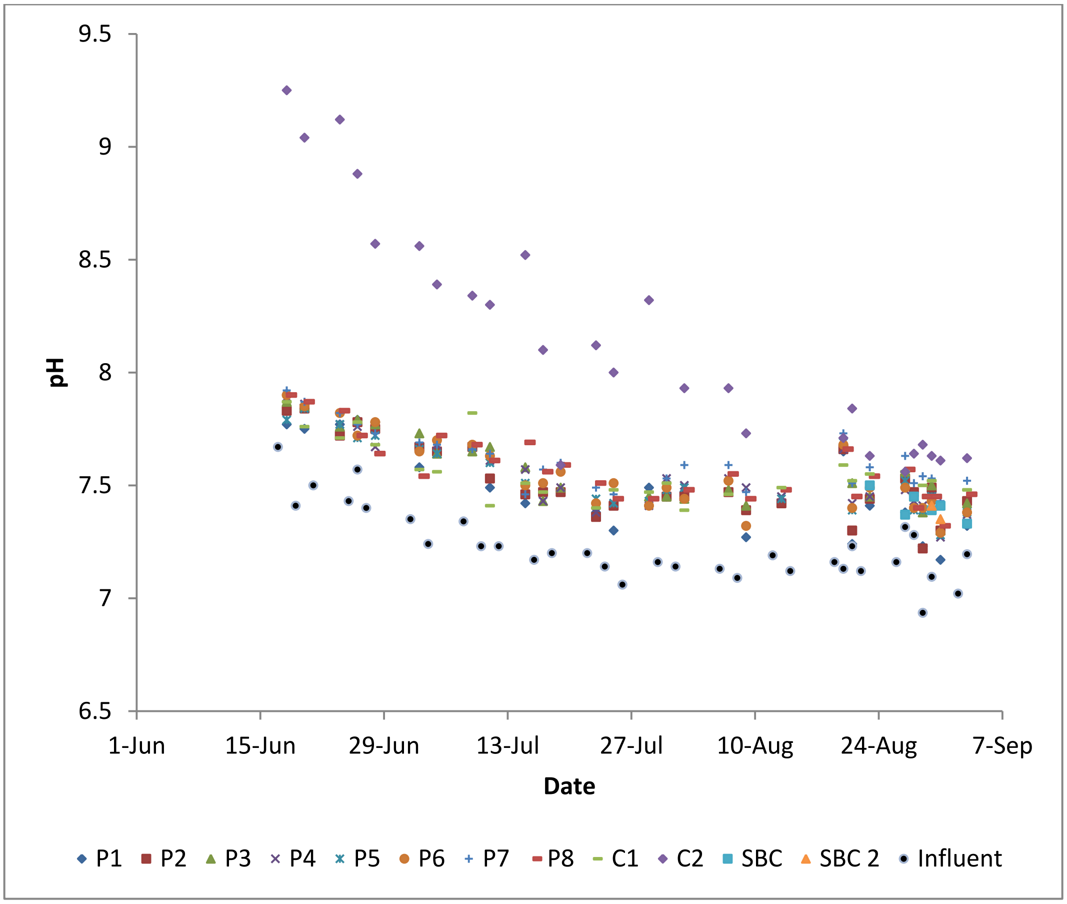

Figure C- 1 pH profiles of the effluent show a gradual decrease and equilibration over time. The pH of C2 started much higher than the other filters, as discussed in Chapter 3. SBC and SBC2 refer to standing buckets left for one and two nights, respectively.

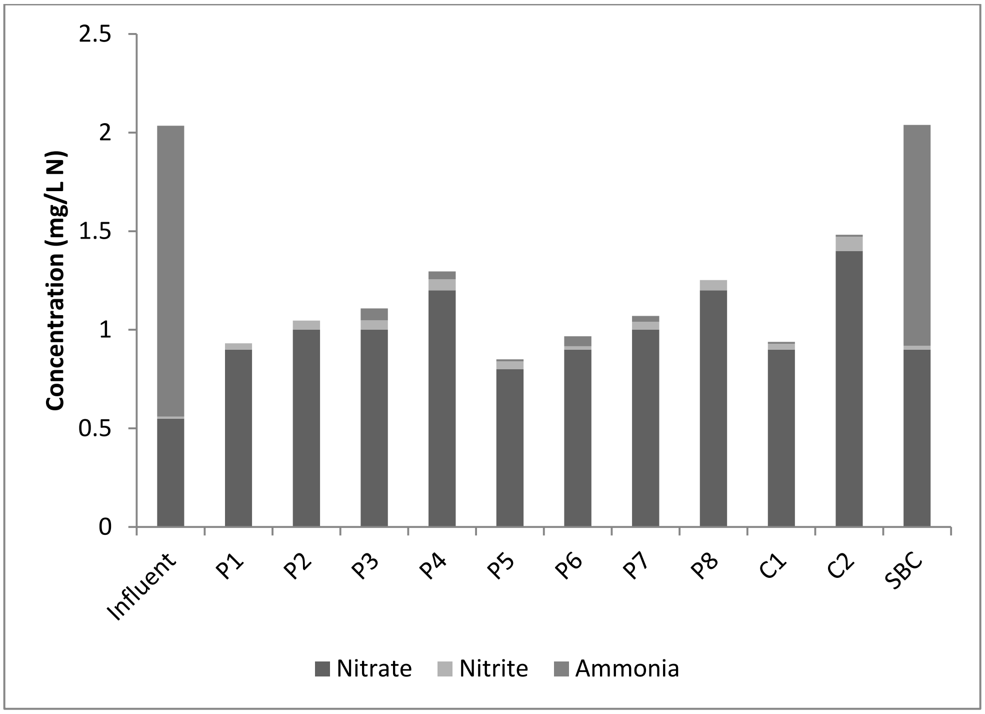

Figure C- 2 Example of nitrogen speciation data disaggregated by filter. SBC refers to standing bucket control.

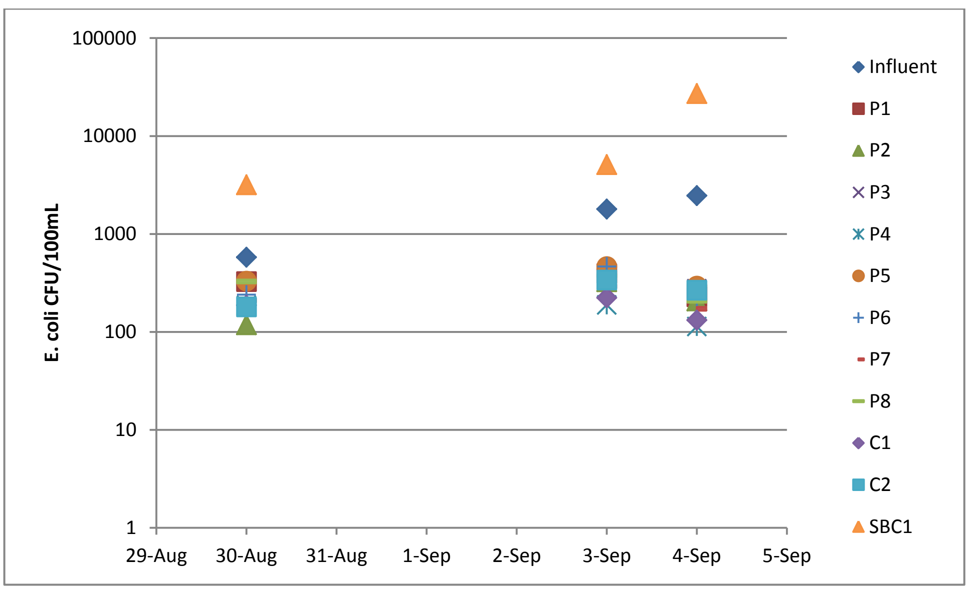

Figure C- 3 *E. coli* coliform counts during the last week of the study. Note the logarithmic scale in the y-axis.

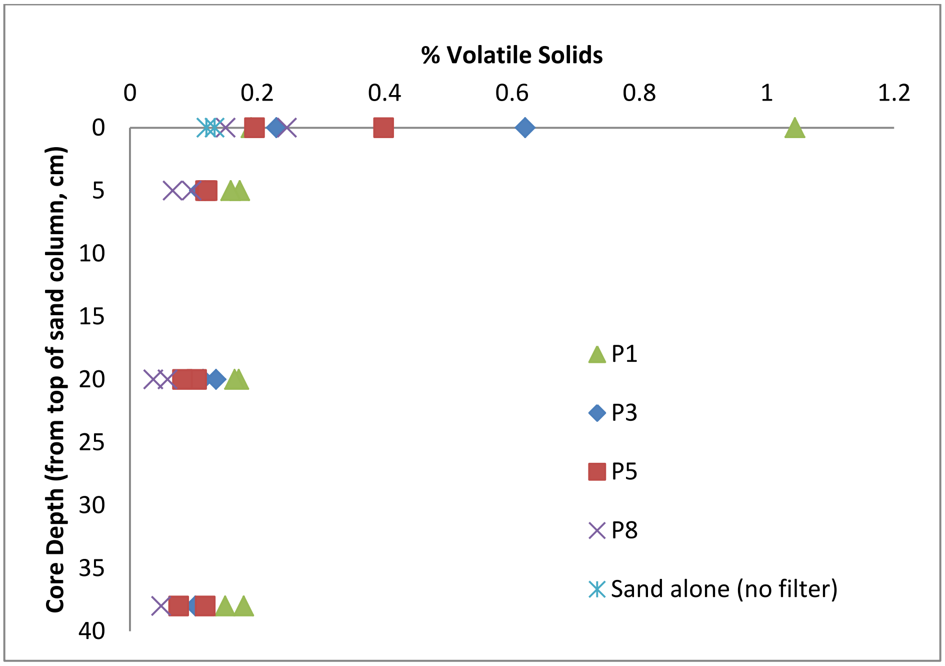

Figure C- 4 Volatile Suspended Solids from cores taken at different depths for two control and two disturbed filters.

Table C- 1 Tap water characterization performed near the end of the study (August 13, 2013)

Water Source	Temp (°C)	pH	TDS (mg/L)	Turbidity (NTU)	COD (mg/L)	Free Chlorine[a] (mg/L Cl_2)	Total Chlorine[b] (mg/L Cl_2)
In bin before sewage addition	20	7.05	165.6	0.25	8.4	0.19	1.87
Straight from tap	21	7.00	166.8	0.2	3	0.25	1.96

[a]Free chlorine refers to both hypochlorous acid (HOCl) and the hypochlorite ion. When ammonia or organic nitrogen is present, chloramines (combined chlorine) will quickly form; this likely occurred in the system used in this lab.
[b]Total chlorine is the sum of free chlorine and combined chlorine.

Tracer Test Data

As described in Chapter 3, tracer tests were performed to evaluate whether there was evidence of preferential flow paths developing within the filters after disturbances. There was very little difference between the curves before and after disturbances (Figure C-5). Curves were also plotted for the cumulative volume eluted over time in each filter (examples shown in Figure C-6); this illustrated the effect of the substantially decreased flow in P7, which had only eluted 2/3 of the charge volume after a full two hours.

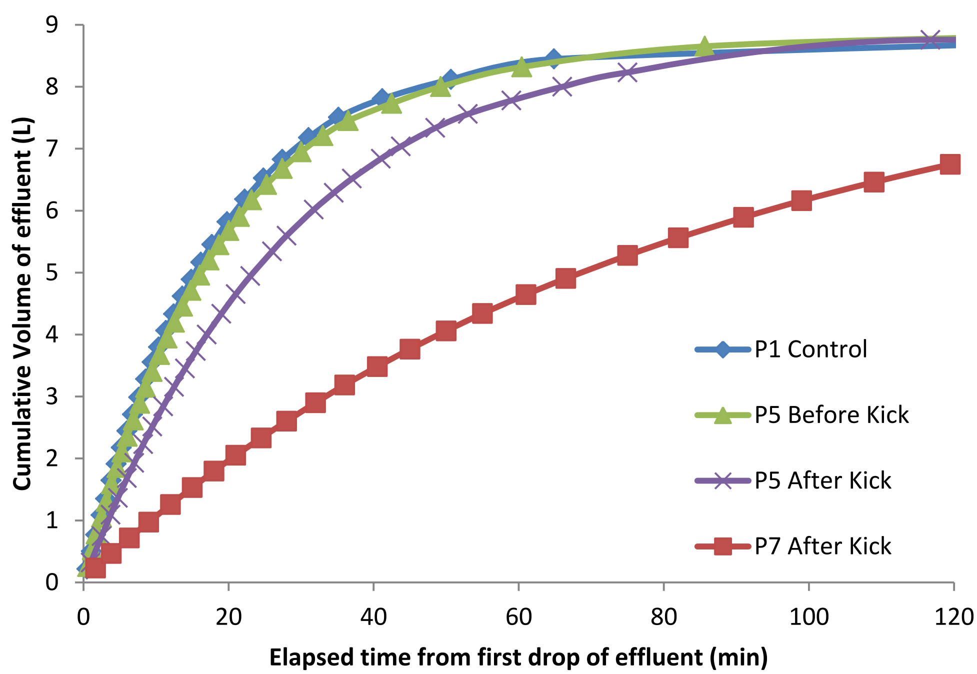

Figure C- 5 Cumulative volume filtered over time for the first two hours after a 9.0 L charge for P1 (control), P5 (kicked + bucket) and P7 (kicked + bucket).

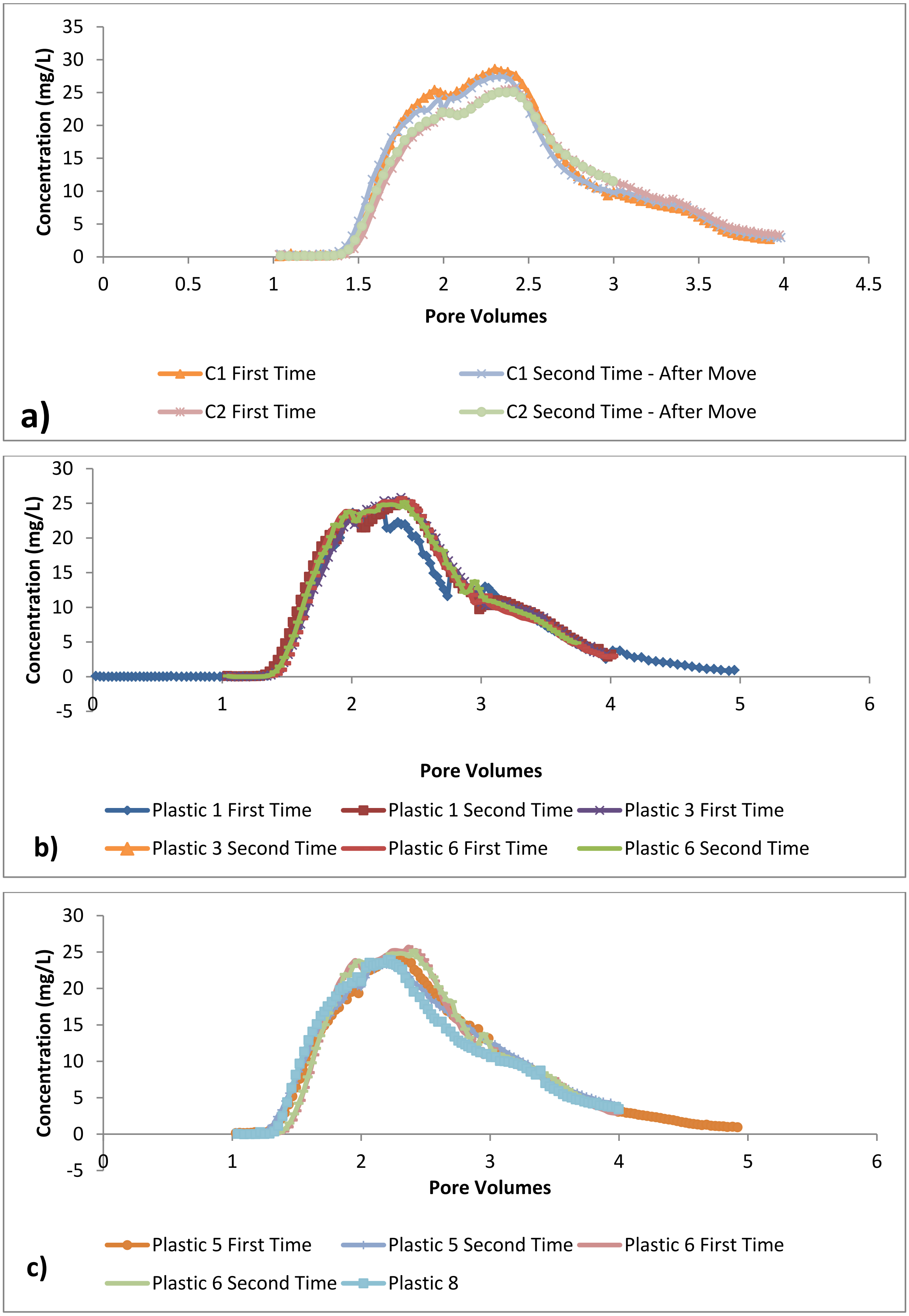

Figure C- 6 Hydraulic tracer test results for before (“First Time”) and after (“Second Time”) disturbances on (a) cBSFs, (b) control pBSFs, and (c) disturbed pBSFs.