Impact of Design and Operational Parameters on Rapid, Deep Bed Biological Filtration of Drinking Water

by

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AUTHOR'S DECLARATION

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Abstract

A series of pilot and full-scale experiments were carried out at the Mannheim Water Treatment Plant in Kitchener, Ontario to examine the impact of backwash technique, filter media characteristics, and combinations thereof on single stage drinking water biological filter performance. The media characteristics investigated were effective size, uniformity coefficient, and media type (GAC and anthracite). Backwash techniques investigated were the collapsed pulse backwash, the extended terminal subfluidization wash (ETSW), and the presence of chlorine in the wash water. Single stage biological filters must serve the dual purpose of biologically mediated removal of biodegradable organic matter (BOM), as well as meeting traditional filter performance criteria such as turbidity removal with minimal head loss accumulation. Accordingly, dissolved organic carbon removal, biodegradable dissolved organic carbon removal, biological respiration potential, turbidity removal, filter ripening time, and head loss accumulation were all quantified as measures of biological filtration performance. The results of this study have several implications for optimized design and operation of biological filters during drinking water treatment.

An increase in effective size of media grains from 1.0 mm to 1.3 mm was shown to significantly extend filter run time by minimizing head loss accumulation without compromising turbidity or BOM removal. Uniformity coefficient however, showed no significant effect on biological filter performance; indicating that the performance benefits associated with highly uniform media may not be commensurate with cost. GAC was found to be significantly more resilient to backwashing in collapsed pulse and chlorinated modes, which impaired BOM removal in anthracite filters. This resilience imparts a high degree of operational flexibility to backwashing GAC filters. The significant decrease in BOM removal by anthracite filters can be minimized; however, by using an optimized backwashing technique.

Collapsed pulse backwashing was found to have a significant effect on biological filter performance. When chlorinated collapsed pulse was used, filter cycles were significantly shortened by approximately 30 – 50% due to a sudden surge in effluent turbidity. This effect is thought to be the result of biofilm, damaged during the course of backwashing sloughing from the media. Extended terminal subfluidization wash was found to significantly reduce, and often eliminate filter ripening entirely. Additionally, the extended contact time with chlorine associated with chlorinated ETSW did not appear to have a significant effect on filter BOM removal. By eliminating filter ripening without compromising biological performance, ETSW shows promise for significant water and production cost savings by minimizing the filter-to-waste period during filter ripening. The presence of chlorine however, was associated with decreased DOC, 24 hours in to the filter cycle. This factor, combined with the negative interaction between chlorine and collapsed pulse suggests chlorinated wash water should be avoided in biological filtration systems like the ones investigated.

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Dedication

To Lindsay

Table of Contents

List of Figures

List of Tables

List of Abbreviations

AOC	Assimilable Organic Carbon
AOC-NOX	AOC measured by *Spirillum sp.* strain NOX
AOC-P17	AOC measured by *Psuedomonas fluorescens* strain P17
ATP	Adenosine Tri-Phosphate
BAS	Biologically Active Sand
BDOC	Biodegradable Dissolved Organic Carbon
BOM	Biodegradable Organic Matter
BRP	Biological Respiration Potential
BW	Backwash
CFBR	Continuous Flow Bioreactor
CP	Collapsed Pulse
DBP	Disinfection By Products
DO	Dissolved Oxygen
DOC	Dissolved Organic Carbon
DOM	Dissolved Organic Matter
EPS	Extracellular Polymeric Substance
ES	Effective Size
ETSW	Extended Terminal Subfluidization Wash
FRS	Filter Ripening Sequence
GAC	Granular Activated Carbon
HES	High Energy Sonication

HPC	Heterotrophic Plate Count
LES	Low Energy Sonication
MLSS	Mixed Liquor Suspended Solids
MWTP	Mannheim Water Treatment Plant
NOM	Natural Organic Matter
OCR	Oxygen Consumption Rate
PLB	Phospholipid Biomass
THM-FP	Tri-Halomethane Formation Potential
TOX-FP	Total organic halide formation potential
UC	Uniformity Coefficient
X*	Dimensionless Contact Time

Chapter 1
Introduction

1.1 Introduction

Biological rapid filtration offers economical and flexible solutions to a variety challenges faced by North American drinking water suppliers. By exploiting the natural ability of aquatic biofilms to efficiently oxidize dissolved organic matter in a controlled environment, biological filters are able to: remove many disinfection by-product precursor molecules, reduce chlorine demand, stabilize residual distribution system disinfectant concentrations, oxidize taste and odor compounds, and prevent distribution system fouling by depriving downstream micro-organisms of essential nutrients. The ability of biological filters to remove dissolved biodegradable organic matter (BOM) facilitates use of ozone as a viable alternative to chlorinated disinfectants by efficiently removing ozonation by-products.

Ozone has increasingly been applied as a primary disinfectant in North America as a result of an increased awareness of chlorinated disinfection by-products, as well as the ineffectiveness of chlorine in *Cryptosporidium* oocyst inactivation (Finch et al., 1994). Ozone has also been demonstrated to oxidize taste and odour compounds (Camel and Bermond, 1998), and certain micro-pollutants (Von Gunten, 2003). In addition to taste and odour molecules and micro-pollutants, ozone oxidatively fractionates large, uncharged humic molecules, thereby significantly increasing the amount of lower molecular weight, charged molecules (ozonation by-products) (Siddiqui et al., 1997) that are microbiologically labile (Van der Kooij et al., 1982; Werner and Hambsch, 1986).

If this increased concentration of biodegradable dissolved organic matter reaches the distribution system, it can result in several deleterious health, aesthetic and/or regulatory consequences. These include: pathogen shielding (Percival and Walker, 1999), pathogen harboring (Camper et al., 1991; Rice et al., 1991), taste and odour complaints (Bruchet et al.,

1992), and development and maintenance of a trophic food web that can lead to the proliferation of invertebrate organisms (Levy, 1990).

As a result of their highly biodegradable nature, many ozonation by-products are readily removed by biological filtration; accordingly, these processes are often coupled. Biological filters consist of a community of indigenous microbes in the form of a biofilm affixed to a support medium and have demonstrated BOM removal capacities ranging from 50-80% in drinking water (eg. Krasner et al., 1993; Miltner et al., 1995; Emelko et al., 2006) while not compromising traditional filtration performance such as particle counts, turbidity removal, head-loss accumulation and filter run time (e.g. Emelko et al., 2006; Goldgrabe et al., 1993; Ahmad and Amirtharajah, 1998).

The selection of an appropriate support medium is a key design parameter in biological filtration. The performance implications of common media characteristics such as size and uniformity coefficient have not been extensively studied in biological filtration systems. In contrast, the impact of media type on biological filtration performance has received more attention; however, the conclusions of such studies are often confounded because important factors such as media size and size distribution, as well as operational parameters such as backwashing technique are frequently excluded from the analysis.

Like traditional filters, biological filters must be periodically backwashed to maintain turbidity removal and prevent terminal head loss accumulation. The backwashing of a biological filter can have a significant impact on both the quantity and quality of water produced. Proper backwashing of biological filters requires sufficient particle removal to prevent high head loss and/or turbidity breakthrough in the subsequent filter run concurrent with adequate biomass retention to ensure sufficient BOM removal. A combination of sub-fluidization air and water wash called “collapsed pulse” backwashing has been demonstrated as an optimal backwashing regime for traditional filters (Amirtharajah, 1978). The impact of collapsed pulsing on biological filtration performance has not been extensively investigated; as a result, collapsed pulse has been applied to biological filtration with varying degrees of success (Liu et al., 2001; Emelko et al., 2006; Ahmad et al., 1998).

In additional to particle/turbidity and BOM removal, appropriately optimized filter backwash protocols should also aim to minimize the filter ripening sequence (FRS). The FRS is a period of increased particle passage immediately following backwash; it can represent up to 90% of particle passage in a single filter cycle (Amirtharajah, 1988) and significant periods of filter downtime or loss of production water during filter-to-waste. The extended terminal subfluidization wash (ETSW) backwash strategy was developed to minimize the impact of the FRS (Amburgey, 2003). ETSW removes the majority of backwash remnant particles from the filter before it is placed back in service, often minimizing and even eliminating the FRS altogether. The biological filtration performance impact of an increased period of contact time with chlorinated wash water during this backwash has not been investigated to date.

1.2 Research Objectives

The overall goals of this research were to determine the impact of media characteristics, backwash technique, and the interaction between the two on biological filter performance, which is defined herein as the ability of a filter to remove BOM while concurrently meeting traditional filtration goals such as turbidity and particle removal without excessive head loss accumulation. The specific research objectives related to media characteristics and their potential impacts on biological filtration performance are listed below.

- To evaluate the impact of decreased contact time associated with larger media effective size on filter BOM and turbidity removal and head loss accumulation;
- To assess the impact of the interaction between biomass and the higher concentration of small media grains at the top of filters (resulting from the use of higher uniformity coefficient media);
- To evaluate the impact of media type (GAC or anthracite); and

- To evaluate the impact of the interaction between each of the above media characteristics with backwash strategy.

The specific research objectives related to backwash regime and their potential impacts on biological filtration performance are listed below.

- To investigate the impact of backwash strategies optimized for traditional filtration such as collapsed pulse and ETSW;
- To evaluate the impact of chlorinated wash water; and
- To assess the impact of the interaction between chlorinated wash water and advanced backwash strategies collapsed pulse and ETSW on biological filter performance.

1.3 Research Approach

To address the above listed research needs, three pilot-scale factorial experiments were conducted. Three full-scale experiments were also conducted during a three-year period of traditional full-scale performance monitoring at the Mannheim Water Treatment Plant (MWTP) in Kitchener, Ontario. The pilot-scale factorial experiments were designed to investigate the impact of media characteristics, backwash strategy, and the interaction between the two on biological filter performance. BOM, turbidity, and particle removal, as well as head loss accumulation and filter ripening were evaluated.

Although backwash investigation was not possible at full-scale, the impact of media configuration on both traditional and biological filter performance was studied. Traditional performance was assessed by analysis of minute-by-minute operational data during all of the three year study period. Full-scale biological performance was assessed during three sampling periods. The first experimental phase involved assessment of cold water biodegradable dissolved organic carbon (BDOC) removal by four media configurations throughout the winter of 2008; the second involved the evaluation of phospholipid biomass,

biological respiration potential, and BDOC removal to generate a comprehensive picture of the quantity, state, and performance of the biomass supported on each media type. The third experimental phase involved an evaluation of chlorine demand and tri-halomethane precursor molecule removal by the filters to elucidate the operational significance of the first two experimental phases. To assess the impact of backwash regime on filter performance, the analyses during the second and third experimental phases were conducted immediately following backwash and after 24 hours of filter operation.

1.4 Thesis Organization

A literature review containing relevant background information and justification of research and method selection is presented in chapter two. This is followed by chapter three, which presents a description of the experimental procedures, equipment and analytical methods used in this research. Pilot- and full-scale results are presented, discussed, and compared to similar findings in chapters four and five respectively. The conclusions drawn from this investigation are presented in chapter 6, and chapter 7 contains both recommendations for optimized filter design and operation, as well as recommendations for future investigations of biological filtration.

Chapter 2 - Literature Review

2.1 Natural Organic Matter

The term “natural organic matter” (NOM) describes the organic compounds found in natural waters. NOM is present in all natural waters and is largely composed of decaying organic matter such as plant, animal, and microbial biomass (Pelekani et al., 1999). The composition of NOM is highly variable, and specific to a given matrix. The range of organic molecules present in NOM is diverse and includes: proteins, amino acids, polysaccharides, amino sugars, refractory hydrocarbons (aliphatic and aromatic) and lignins (Biber et al., 1996; Hedges et al., 1994).

Water soluble NOM is termed “dissolved organic matter” (DOM). It is this fraction of NOM that is of greatest concern in drinking water treatment. The majority of disinfection by-product precursors exist as DOM (Reckhow et al., 1990), and it is this fraction that can result in reduced organoleptic water quality and compromised treatment efficiency (Bruchet et al. 1992). In the study of drinking water, organic carbon is frequently used as a surrogate parameter for NOM (Kaplan et al., 2005). Therefore dissolved organic carbon (DOC) is considered to be a key parameter in determining the ability of a given treatment process to remove NOM.

The fraction of NOM that is subject to microbial attack is known as “biodegradable organic matter” (BOM). BOM consists of a complex mixture of aliphatic and aromatic hydrocarbons that is highly matrix specific and varies temporally, seasonally, and geographically (Kaplan et al., 1994). In the majority of drinking water distribution systems, carbon is the limiting nutrient for bacterial growth (Chandy and Angles, 2001; van der Kooij, 1992; Owen et al., 1995). Therefore, the biodegradable fraction of DOC is a critical for assessing the biological stability of water entering the distribution system.

Biological growth in drinking water distribution systems, typically referred to in the literature as “re-growth”, can lead to a number of regulatory, aesthetic, organoleptic, infrastructure, and health concerns. The growth of heterotrophic bacteria in the distribution

system can lead to violations of heterotrophic plate count (HPC) regulations and guidelines; but these same bacteria can also cause coliform concentrations to be underestimated by as much as 80% by inhibiting coliform growth during standard coliform tests (Percival and Walker, 1999). These organisms can also become the basis for a trophic food web that can include invertebrate organisms, and while these do not pose a significant risk to human health, they are understandably considered to be highly objectionable by consumers (Levy, 1990). Biofilms in the distribution system are associated with pathogen harboring, allowing coliforms such as *Escherichia coli* to persist and multiply within the biofilm structure (Camper et al., 1991; Rice et al., 1991). DOC is also associated with the formation of disinfection by-products (DBPs) (Reckhow et al., 1990) and taste and odor complaints (Bruchet et al., 1992).

With current technology it is possible to accurately detect DOC concentrations as low as 10 µg C/L (APHA, 1998); but the value of these measurements is limited due to the variability of DOC composition. DOC represents a large and significantly variable pool of organic compounds that likely have different significance depending on the purpose of the measurement. Therefore, to estimate BOM from DOC, bioassays are frequently employed. The two most common bioassays used for this purpose are assimilable organic carbon (AOC), which is a biomass based method, and biodegradable dissolved organic carbon (BDOC), which is a DOC based method.

The objective of quantifying AOC and BDOC is to assess the removal of dissolved organic carbon compounds by a treatment process or to determine the biological stability of finished water. AOC quantifies the fraction of biodegradable dissolved organic carbon that is assimilated into cell mass, converted to a carbon concentration through a conversion factor or calibration (Huck, 1990). BDOC quantifies the fraction of biodegradable organic carbon that is assimilated and metabolized by heterotrophic micro-organisms (Servais et al., 1987). BDOC can therefore be conceptualized as including AOC, with the addition of the organic substrate necessary for cellular energy and upkeep.

The selection of AOC or BDOC as a system evaluation parameter is based on the purpose of the measurement, although they are best considered complementary measures. BDOC is typically used for assessing biologically mediated reductions in chlorine demand or disinfection by-product formation potential (Huck, 1990), although BDOC also has been correlated to biological stability of drinking water (Servais et al., 1993; Volk et al., 1994). This is because BDOC describes the entire quantity of organic material removed, whereas AOC describes only that fraction which is assimilated into cell mass. For this reason, AOC is typically used for assessing the potential of water to support bacterial growth in the distribution system. Both AOC and BDOC are discussed at length in section 2.4.

The presence of NOM is ubiquitous in natural waters, and aquatic bacteria have evolved to efficiently utilize NOM as an energy source. Where a solid substratum is present, micro-organisms form stable, highly diverse aggregates of cells embedded in a stabilizing matrix called extracellular polymeric substance (EPS). The dense cell concentrations and high degree of diversity supported by this mode of growth facilitate sequential and synergistic degradation of NOM molecules that would otherwise be highly resistant to microbial attack (Flemming et al., 1999).

2.2 Biofilms in Biological Filtration

Biological filtration exploits biofilm growth in a controlled environment to deprive downstream distribution system biofilms of BOM. By doing so, chlorine demand is lowered (Reckhow et al., 1990), the necessary residual disinfectant concentration in the distribution system is diminished, and biological growth is impaired by removing the growth limiting nutrient, carbon (LeChevallier et al., 1992). Biofilms are composed of bacterial cells growing within an organic matrix of EPS, which is synthesized by attached bacteria, and is similar to a porous gel containing 90-95% water (Characklis and Marshal, 1990). As a biofilm grows on the support medium, it develops into a complex porous structure containing dense cell clusters separated by interstitial voids and open channels (Yang and Lewandowski, 1995). These clusters of cells consist mainly of five distinct regions, each with varying chemical properties that affect mass transfer; these regions are summarized in table 2.1. This structure

can significantly complicate the bulk fluid biofilm interface, and the convective transport occurring therein, as depicted in table 2.1, and visualized figure 2.1.

Table 2.1: Summary of Biofilm Sorption Sites

Biofilm Region	Abbreviation in Figure 2.1	Description
Extracellular Polymeric Substance	EPS	Contains cationic groups in amino sugars and proteins (e.g. $-NH^{4+}$), anionic groups in uronic acids and proteins (eg $-COO^{-}$, $-HPO^{4-}$), apolar groups from proteins (such as aromatic amino acids), and groups with a high hydrogen bonding potential such as polysaccharides
Outer Membrane (Gram Negative)	OM	Lipopolysaccharides (LPS)
Lipotechoic Acids	LT	Gram-Positive outer membrane
Cell Wall	M	Consists of N-acetylglucosamine and N-acetylmuramic acid, offering cationic and anionic active groups.
Cytoplasmic Membrane	CM	A lipophilic region,
Cytoplasm	CY	Aqueous phase separated from surrounding aqueous phases

Bacteria in drinking water biofilms derive energy and nutrients from oxidizing dissolved organic compounds. This process occurs in two stages, the first being sorption (both adsorption and absorption) of the contaminant into and onto the biofilm and the second being biodegradation (Carlson and Silverstein, 1998). Conceptually, these processes occur concurrently and the dominant removal mechanism can often depend on system configuration and substrate properties (Simpson, 2008). Biodegradation must occur at a sufficient rate to drive a concentration gradient between the biofilm and bulk liquid phase. Mass transfer rates depend on the nutrient properties and how the nutrients interact with the biofilm. Upon “entering” the biofilm, biodegradable substrate is oxidized by embedded

microorganisms. More recalcitrant and partially degraded substrates are often able to diffuse through the biofilm to the substratum surface; however, and in the case of granular activated carbon (GAC), into micropores where they may be adsorbed (Sontheimer et al., 1988). When substrate concentration in the bulk fluid, or at the biofilm-support medium (e.g., biofilm-GAC) interface becomes sufficiently low, substrate can desorb from the surface of the support medium and diffuse back in to the biofilm for further biodegradation, due to concentration gradient (Simpson, 2008).

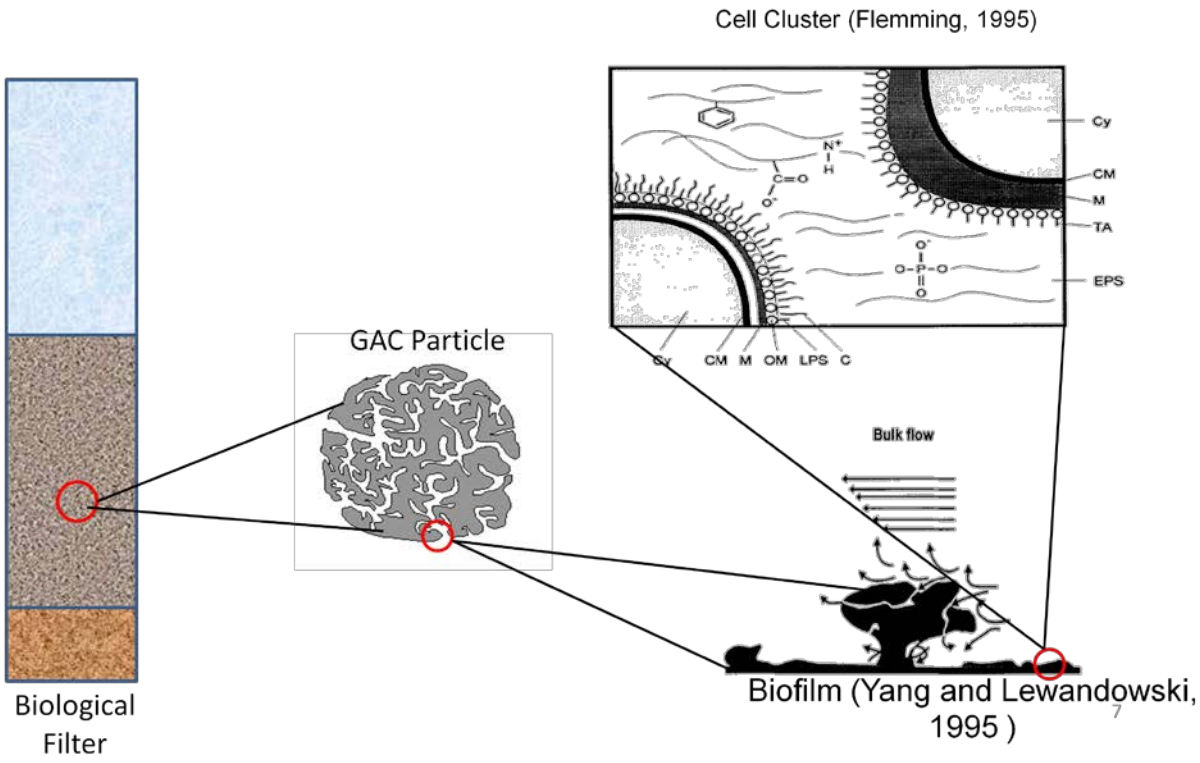

Figure 2.1: Schematic of biofilms in drinking water biological filters

Substrate mass transfer into biofilms is dominated by molecular diffusion (De Beer et al., 1993). The most abundant functional groups on the surface of the biofilm are carboxyl and hydroxyl groups that are typically ionized at filtration conditions, and therefore contribute to an overall negative charge on the biofilm surface (Morgan et al., 1990). Carlson and Silverstein (1998) demonstrated that biofilm sorption is governed by molecular size and charge. For smaller molecules (below 500 amu), they reported that charge was the dominant factor governing sorption, with sorption decreasing with increasing anionic charge. This study was carried out using model compounds that were highly recalcitrant, but of biological origin rather than using an inactivated biofilm, which can often have significantly altered properties relative to indigenous/untreated biofilms. In this work, Carlson and Silverstein (1998) elucidated earlier results that had reported a non-significant difference in biofilm

sorption before and after ozonation (Carlson and Silverstein, 1997): it was discovered that the reduced molecular size and increased anionic charge resulting from ozonation had counteracted one another with respect to their impacts on substrate sorption by the biofilm.

2.3 Assessing Biological Filter Performance

2.3.1 Adenosine Tri-Phosphate (ATP) Method

The ATP method for assessing microbial activity has been applied to drinking water biological filtration systems to assess the state and activity of the biomass adhering to filter media (Magic-Knevez and Van der Kooij, 2004; Velten et al., 2007). In the presence of oxygen, ATP extracted from biomass will combine with luciferin to produce luciferase, a compound that emits light that can be measured by a luminometer. This light can be calibrated to an ATP concentration, which can then be converted to a biomass concentration using a conversion factor (Karl, 1980). Speed and accuracy are two of the commonly cited advantages of this method, which has a detection limit of 1 ng ATP L^{-1} (Magic-Knevez and Van der Kooij, 2004). Measurement sensitivity in biological filtration applications is often hindered; however, by the highly heterogeneous nature of the biofilm that forms on biological filters. Relatively small amounts of filter media are typically sampled, making it difficult to obtain representative samples because of the uneven or "patchy" distribution of the biofilm in biological filters (Velten et al., 2007). Converting ATP concentration to active biomass concentration presents an even greater analytical challenge. It is this conversion that results in much of the loss of analytical precision in ATP measurements (Karl, 1980).

The validity of the ATP method hinges on four key assumptions: 1) all living organisms contain ATP, 2) ATP is easily and equally extracted from all microbial communities, 3) ATP does not associate with inactive cells, and 4) there is a relatively constant ratio of ATP to total cell carbon for all microbial taxa, independent of microbial activity or environmental conditions (Karl, 1980). Of these assumptions, the first two are generally valid under strictly controlled experimental conditions. The third assumption, that

ATP is only associated with active cells if often inaccurate. It has been found that free ATP concentrations in natural waters, or ATP not bound by microbial cells, can comprise 75-90% of the total ATP measured in natural aquatic ecosystems (Karl, 1980). It has also been demonstrated that ozonation, a treatment step that commonly precedes biological filtration, can lead to significant increases in the concentration of free ATP in filter influents (Hammes et al., 2008). The interference of this ATP in the analysis filter biomass could potentially be mitigated by carefully washing media samples before extracting attached cellular ATP; however, this does not take the potential of variable intracellular concentrations resulting from cells scavenging free ATP from the surrounding water. The fourth assumption also can be inaccurate because ATP content can vary significantly because of growth conditions, even within taxa (Karl, 1980).

For these reasons, the practice of citing published values for the conversion of ATP concentration to biomass carbon concentrations or cell numbers negates much of the quantitative value of the ATP measurements. Magic-Nevez and van der Kooij (2004) cited a conversion value of C = 250 x ATP to convert a reported ATP concentration (ATP) from the literature to biomass carbon concentration (C). This conversion was derived from free living marine bacteria and unicellular marine algae (Hamilton and Holm-Hansen, 1967; Holm-Hansen, 1970), which would not necessarily have similar cellular ATP concentrations to those of cells attached to a biofilm in the freshwater environment of a drinking water biological filter. Rather than relying on literature-based conversions that may not be relevant to the aquatic ecosystem being studied, a more appropriate approach is to determine case specific ATP/cell conversion factor (e.g., Velten et al., 2007).

To determine case specific ATP concentrations, Velten et al. (2007) detached bacteria from GAC surfaces using gentle manual shaking and quantified detached cells using a flow cytometer. The ATP concentration of these cells was then determined and it was assumed that GAC-associated cells exhibited similar ATP concentrations. This treatment only quantifies a certain subgroup of the biofilm community that is readily detachable, and it is possible that this subgroup would have a different ATP concentration within cells than would

the greater biofilm community. It was found that six to eight two minute high energy sonication (HES) treatments detached approximately 90% of attached active biomass (Magic-Knevez and van der Kooij, 2004); they also damaged a significant portion of the bacterial population. Flow cytometry does not distinguish between viable and non-viable cells; however, so determination of a cell number using this method provides an estimation of total biomass based on ATP concentration, rather than total active biomass. Accordingly, this method provides an estimate of total biomass, similar to the phospholipid biomass method discussed below in section 2.3.2, but with the added uncertainty of an ATP to biomass conversion factor. Given that flow cytometers are capable of quantifying different wavelengths of fluorescence on different channels, it might be possible to distinguish and quantify viable cells using a live/dead staining technique, provided a method was developed to remove sufficient biomass from the filter media without damaging a significant portion of the bacteria. Conversion factors generated in this manner would be more accurate; but would also have to be re-determined seasonally and as filter conditions changed (i.e. flocculation, backwash strategies, primary disinfection, etc.), which would compromise the advantage of speed and ease of measurement associated with ATP analysis.

2.3.2 Phospholipid Biomass

Each microbial cell present in a biological filter is bound by a cell membrane. This membrane is composed of phospholipids. Phospholipids are not stored within cells and are hydrolyzed within minutes to hours following cell death, and so quantifying the phospholoipids should provide a reasonable estimate of viable biomass (White, 1993). The phospholipid method, as described by Findlay et al. (1989), extracts phospholipids by separating them into a hydrophobic phase (chloroform) while a methanol phase containing non-lipid bound phosphate is decanted from the sample. The chloroform is then evaporated, leaving the lipids behind. The lipids are then digested in potassium persulfate and complexed into a phosphomolybdate compound which absorbs light at 610nm. Absorbance at 610nm is then converted to nmol p/cm^3 dried media via calibration curve.

The advantage of the phospholipid method is that it is a simple method and does not require any specialized or costly laboratory equipment. The disadvantages to this method however, are that it is labor intensive, time consuming, and most importantly: phospholipid biomass has no direct relationship to actual microbial activity and BOM removal (Wang et al., 1995; Miltner et al., 1995; Perrson et al., 2006). For example, in a study of the relationship between phospholipid biomass and biological filter performance, Wang et al. (1995) found that, while GAC filters were able to support three to eight times more top of filter biomass than anthracite filters, no detectable difference in AOC removal was found, and only very minor differences in THM-FP reduction were measured. In another study evaluating biological filter performance, phospholipid biomass was found to have no discernable correlation with either BDOC or AOC, and while phospholipid biomass was found to be highly dependent on temperature, both BDOC and AOC removal were not (Persson et al., 2006).

In addition to the lack of a discernable relationship between phospholipid biomass and BOM removal, the phospholipid method also presents representative sampling issues. To obtain phospholipid concentrations in the commonly used calibration range of 0 to 40 nmol P, as little as 0.1 g of media are used in each sample. Given the heterogenous nature of biofilms both within the biofilm (Yang and Lewandowski, 1995) and throughout the filter media (Velten et al., 2007), it would be difficult to draw conclusions regarding full filter performance without a large number of samples. Given that purpose of a biological filter is to reduce the concentration of BOM present in water, and the lack of correlation between phospholipid biomass and BOM, the value of phospholipid biomass to drinking water biological filtration research is limited. This method is not suitable as a stand alone method for evaluating filter performance.

2.3.3 Biological Respiration Potential

The biological respiration potential (BRP) method was developed to provide a more representative estimate of actual microbial activity within biological filters, while maintaining the simplicity of the phospholipid method (Urfer and Huck, 2001). The

biological activity of given sample of biomass attached to filter media is determined by measuring the dissolved oxygen (DO) consumed as biofilm bacteria oxidize known concentrations of BOM molecules (Urfer and Huck, 2001). This method incubates a sample of biological filter media in a cocktail of common ozonation by-products for five hours. Oxygen is consumed as the bacteria attached to the filter media degrade the BOM cocktail. The subsequent decrease in oxygen concentration is an indirect measure of microbial BOM removal.

The BRP method may hold promise as a rapid, sensitive method to assess biological filter performance without the need of costly specialized laboratory equipment. DO measurements are highly sensitive, and allow quantification of respiration of low substrate concentrations (Ellis et al., 1996). DO measurements have also been successfully applied to quantifying microbial activity in wastewater processes (Metcalf and Eddy, 1991). This specific method has not been directly compared to BOM removal (i.e. AOC or BDOC removal) in biological filters.

The oxygen consumption rate (OCR) method is a modified BRP method that has been compared to filter BOM removal. The OCR method uses water sampled from the same location as the media in place of a BOM cocktail, incubates the sample for 2 hours rather than 5, and incubates at prevailing surface water temperatures rather than room temperature (Persson et al., 2006). When this method was compared to BOM removal, no correlation to AOC removal ($R = 0.1$) was found and a very weak correlation to BDOC removal ($R = 0.4$) was found (Persson et al., 2006). Given the incubation temperature and shorter incubation time, the OCR method may be expected to be more correlated to BOM removal, but the lack of a standardized incubation solution may complicate the measurements, as the BOM concentration in sample water could vary significantly from test to test.

Because the BRP and OCR methods quantify respiration potential, it is not surprising that Persson et al. (2006) found BRP to have a stronger correlation to biomass rather than actual BOM removal. The relatively stable batch test conditions used during BRP measurement favor full utilization of the biomass present on the media, and allow for

prolonged contact time for enhanced mass transfer, which would not be possible during filtration. Like phospholipid biomass, however, BRP/OCR measurements concurrent to BOM removal measurements can potentially help to confirm or disprove that observed increases or decreases in BOM removal are associated with changes in biomass activity.

2.3.4 AOC

AOC is defined as the concentration of dissolved organic carbon present in water that can be converted into microbial biomass (Van der Kooij, 1982; Huck, 1990). This fraction represents only a small percentage of the TOC, but is a critical parameter governing heterotrophic growth in the distribution system (Van der Kooij, 1982; Escobar and Randall, 2001). Conventional AOC determinations are based on the method developed by Van der Kooij et al. (1982). The AOC method is a batch method that seeks to estimate the fraction of carbon available for microbial growth by determining maximum cell density achieved during the course of a batch incubation of known organisms in sterile filtered sample water. These organisms are typically *Psuedomonas fluorescens* strain P17 (AOC-P17) and *Spirillum sp*. strain NOX (AOC-NOX). The cell density is then converted to acetate carbon equivalents using an empirically derived conversion factor. The specifics of the AOC method can be found in detail in standard methods (APHA, 1998) and the various modifications to the procedure have been reviewed elsewhere (Kaplan et al., 2005; Huck, 1990). This method is based on the assumption that the metabolic characteristics of the test organism(s) are representative of the much more diverse microbiota present in the drinking water distribution system. This assumption is generally true and a strong correlation between AOC and bacterial growth has been observed and documented (Escobar and Randall, 2001; Van der Kooij, 1992).

The AOC method has the advantage of being able to detect extremely low concentrations of organic carbon; with a theoretical detection limit below 1µg C/L. In practice however, this same sensitivity renders the method vulnerable to even very low levels of organic carbon contamination, leading to a practicable detection limit of 5 to 10 µg C/L (APHA, 1998). As detailed in standard methods AOC determinations are tedious, time

consuming, and labor intensive, although attempts have been made at automating the procedure (Hammes and Egli, 2005).

2.3.5 BDOC

BDOC is measured by determining the difference between initial DOC concentration of a sample and the minimum DOC concentration observed during the course of an incubation period (Frias et al., 1991; Joret and Levi, 1988; Lucena et al., 1990; Ribas et al., 1991; Servais et al., 1987, 1989). Methods for determining BDOC can be divided into two broad subcategories: those which employ a bioreactor (Lucena et al., 1991; Ribas et al., 1991; Frias et al., 1995) and batch methods, which involve an incubation step (Servais et al., 1987, 1989; Joret et al., 1988). Bioreactor based methods involve the water travelling through a column or series of columns containing an inert support medium that has been colonized by indigenous micro-organisms. Batch methods involve measuring the difference between an initial sample DOC concentration and the minimum concentration observed after inoculating the sample with micro-organisms and incubating it for a set period of time. The batch methods can be distinguished by the type of inoculum that they use, which varies widely and can be planktonic (Servais et al., 1987, 1989) or a biofilm affixed to a support medium such as sand (Joret et al., 1988).

The key parameters that must be considered when choosing a BDOC method include the duration of the analysis, the nature of the inoculum, and the characteristics of the water to be measured. Test duration is an important factor and BDOC analysis methods vary widely in the time required for analysis, ranging from two hours (Frias et al., 1991) to six weeks (Trulleyova and Rulik, 2004). Table 2.2 provides a comparison of various batch methods for BDOC analysis and underscores that the nature of the inoculum can have a significant impact on the accuracy and reproducibility of the test; therefore, it must be carefully selected. The characteristics of the water also can have a significant impact on analytical method selection because different methods are better suited to different water characteristics (Frias et al., 1995, Lucena et al., 1991; Block et al., 1992). For example, the column method was found to produce lower estimations when oxidants are present, or when testing raw water with a

higher concentration of refractory BDOC, while the longer batch incubations with biologically active sand (BAS) tend to produce higher and more variable estimates of BDOC (Frias et al., 1995; Lucena et al., 1991; Volk et al., 1994).

2.3.5.1 **Batch BDOC Methods**

Batch methods quantify BDOC by incubating a discrete sample with an inoculum of micro-organisms for a defined period of time. BDOC is considered to be the difference between initial sample DOC and the DOC concentration measured at the end point of the analysis. Samples are sterilized by filtration through a 0.45 µm nominal porosity membrane and then inoculated with a microbial consortium. Batch methods for BDOC quantification can be divided into general two groups based on inoculum type(i.e., suspended [$BDOC_{susp}$] bacteria (Servais et al., 1987, 1989) or bacteria attached to a solid substrate [$BDOC_{sand}$] (Joret et al., 1988)).

The $BDOC_{susp}$ methods involve the sterile filtration of a water sample, which is then inoculated with unfiltered water from the same source as the sample and incubated for three to six weeks, depending on the type of water and the inocula used (Servais et al., 1987, 1989). BDOC is then determined by subtracting the final DOC concentration from the initial DOC concentration of the sample. The benefits of the $BDOC_{susp}$ method are that it is a simple method with a readily available inoculum that contains an assemblage of bacteria native to the water sample. It is also a sensitive method, with a reported detection limit in the range of +/-11 to 162 µg C/L in a study of 109 American waters (Kaplan et al., 1994). The origin of the inoculum has not been found to be a significant source of variability (Block et al., 1992, Servais et al., 1987), most likely due to the long incubation time allowing for enzyme expression patterns to adapt to the sample DOC, as well as selection and growth of organisms capable of degrading sample DOC. The main source of variability in the $BDOC_{susp}$ method is sample handling because sample filtration can be a significant source of contamination unless appropriate precautions are taken (Khan and Subramania-Pillai, 1997). Glassware must also be prepared appropriately (baked and acid washed) to avoid carbon contamination (APHA, 1998).

The use of a suspended inoculum, though simple and straightforward, has two significant limitations: a long incubation period, and possible underestimation of actual BDOC concentrations. Due to the limited diversity of suspended organisms, the long incubation period required by this method renders it of little use to utilities seeking to optimize or adapt operationally in real time, in response to changing water BDOC concentrations. The second limitation of the $BDOC_{susp}$ method is underestimation; although there is a long incubation period, planktonic bacteria are not necessarily capable of biodegrading the full spectrum of BDOC that can be degraded by biofilm micro-organisms such as those present in the distribution system, which utilize diversity and complex co-metabolism present in biofilm associated micro-organisms that allow them to metabolize a larger pool of BDOC molecules (Volk et al., 1994).

To decrease incubation time and increase inoculum diversity, biofilm microbes affixed to a solid support medium such as sand are commonly employed in BDOC bioassays (Joret et al., 1988; Park et al., 2005; Volk et al., 1994). The use of fixed bacteria takes advantage of the highly diverse nature of biofilm bacteria and their inherent co-metabolism capabilities to significantly decrease the time of analysis from 21-28 days to 3-5 days (Joret et al., 1988). The $BDOC_{sand}$ method (Joret and Levi, 1988) involves an initial DOC measurement of a 300mL water sample, followed by the addition of 100 grams of biologically active sand, collected from a biologically active sand filter. The samples are then aerated and a DOC sample collected daily until a minimum is reached. The immediately apparent disadvantage of this method is that it requires many more DOC analyses than the $BDOC_{susp}$ method. This frequent sampling is due to a currently unexplained phenomenon of DOC release from biologically active sand that commonly occurs during the course of the $BDOC_{sand}$ method and can represent a DOC increase above up to 38% above the minimum measured during the incubation period (Volk et al., 1994). Though well documented (Volk et al., 1994; Park et al., 2005; Joret and Levi, 1988), the mechanisms of DOC release are currently unknown.

Unlike the $BDOC_{susp}$ method, inoculum selection can have a significant impact on the accuracy, duration, and reproducibility of $BDOC_{sand}$ measurements. The most common inocula involve sand that has been colonized with active biofilm, often from a biological sand filter (Joret et al. 1988). The original $BDOC_{sand}$ method (Joret and Levi, 1986; Joret et. al. 1988) utilized biologically active sand harvested from a slow sand biological filter.

A number of studies have directly compared BDOC batch methods using replicate samples. In most cases (table 2.2) the $BDOC_{sand}$ method provides equal or higher estimates of sample BDOC, although in some cases a suspended inoculum may generate similar results to the $BDOC_{sand}$ method. In the studies where the $BDOC_{susp}$ appears to underestimate the BDOC value of a sample it is likely that some combination of the limited initial cell number and diversity of the inoculum (Volk et al., 1994), and the more refractory nature of the sample DOC are limiting the ability of the method to achieve complete biodegradation. The possibility that DOC contamination from fixed bacterial inocula is leading to artificially high BDOC measurements must first be ruled out, however.

DOC leaching from the support medium in the $BDOC_{sand}$ method has been discussed in the literature (Park et al., 2005; Joret et al., 1988; Trulleyova and Rulik, 2000). There are several techniques that address this problem, including: washing the sand until wash water yields no detectable DOC in comparison to distilled water (Joret et al., 1988); ensuring that the DOC of the sample after BAS addition does not differ from DOC prior to BAS addition (Park et al., 2005), washing and storage of inocula in distilled water for 24 hours (Trulleyova and Rulik, 2004), and ten repeated rinses with 500 mL organic free distilled water (Block et al., 1992). Repeated washing, as well as an extended storage period, combined with check samples to ensure that DOC after BAS addition does not differ from DOC prior to BAS addition is likely an effective method to ensure DOC contamination is negligible.

Presently, BAS appears to be the optimal inoculum for the $BDOC_{sand}$ method (provided adequate precautions during pretreatment such as washing and the use of check samples) because it provides accurate measurements over a range of water types (table 2.2) without contributing significant variability to the method through DOC leaching (Khan and

Subramania-Pillai, 1997). Inocula such as stream sediment (Trulleyova and Rulik, 2004) and mixed liquor suspended solids (MLSS) (Khan et al., 2005) contribute excessive DOC to the sample, leading to increased variability and detection limits. Other more standardized inocula such as commercial BOD seeds (Khan et al., 2005) lack the necessary microbial diversity to achieve a representative estimate of sample BDOC. When BAS is not readily available, the best alternative batch method is the $BDOC_{susp}$ method, which in most cases provides an adequate estimate of BDOC (Frias et al., 1991; Park et al., 2005; Lucena et al., 1994).

Table 2.2: BDOC Batch Method Comparisons

Sample Water	Inoculum	Time of Analysis (days)	Reported BDOC (BDOC/DOC)%	Reference
Range given for raw, ozonated, and finished waters	Surface water	21	24-29	Block et al. (1992)
	BAS	3	34-39	
Standard acetate solution (100mg/L)	Surface water	28	94 (± 8)	Park et al. (2005)
	BAS	5	89 (± 1)	
	recirculating column	5	77 (± 4)	
Dechlorinated tap water	Surface water	21	25.86 (± 3.44)	
	BAS	5	22.41 (± 3.44)	
	recirculating column	5	5.17 (± 1.72)	
Surface water	Surface water	42	34.46 (± 4.15)	Trulleyova and Rulik (2004)
	BAS	14	40.90 (± 5.12)	

Sample Water	Inoculum	Time of Analysis (days)	Reported BDOC (BDOC/DOC)%	Reference
River Water	Surface Water	21	36.74 (± 15.56)	Frias et al. (1995)
	BAS	10	39.17 (± 11.09)	
	BAS	21	46.41 (± 12.10)	
	Frias et al., 1992	5	36.07 (± 14.35)	
	Ribas et al., 1991	2 hours	25.8 (± 9.2)	
Prechlorinated Water	Surface Water	21	18.88 (± 9.54)	
	BAS	10	24.75 (± 12.26)	
	BAS	21	30.37 (± 14.11)	
	Frias et al., 1992	5	22.04 (± 11.83)	
	Ribas et al., 1991	2 hours	14.70 (± 9.8)	
Settled Water	Surface Water	21	23.76 (± 13.41)	
	BAS	10	18.36 (± 9.92)	
	BAS	21	27.87 (± 10.86)	
	Frias et al et al. 1992	5	22.04 (± 11.83)	
Filtered Water	Surface Water	21	27.17 (± 13.26)	
	BAS	10	25.05 (± 10.34)	
	BAS	21	35.86 (± 16.22)	
	Frias et al., 1992	5	29.24 (± 12.42)	
	Frias et al., 1991	2 hours	32.4 (± 9.1)	

Sample Water	Inoculum	Time of Analysis (days)	Reported BDOC (BDOC/DOC)%	Reference
Finished Water	Surface Water	21	23.33 (± 10.94)	
	BAS	10	16.18 (± 9.17)	
	BAS	21	29.63 (± 14.82)	
	Frias et al. 1992	5	27.01 (± 11.54)	
	Frias et al., 1991	2 hours	31.9 (± 17.0)	
River Water	Surface Water	28	34.11 (± 8.30)	Lucena et al. (1991)
	BAS	10	37.34 (± 14.67)	
	BAS	21	41.19 (± 14.90)	
	CFBR (SAND)	1 hour	20.78 (± 8.95)	
GAC Filtered Water	Surface Water	28	24.35 (± 11.97)	
	BAS	10	24.60 (± 11.73)	
	BAS	21	33.06 (± 17.76)	
	CFBR (SAND)	1 hour	32.47 (± 7.95)	
Distribution Water	Surface Water	28	27.10 (± 17.71)	
	BAS	10	12.30 (± 15.92)	
	BAS	21	23.71 (± 19.59)	
	CFBR (SAND)	1 hour	20.63 (± 7.41)	

Sample Water	Inoculum	Time of Analysis (days)	Reported BDOC (BDOC/DOC)%	Reference
Ozonated Water	Surface Water		15.16	Volk et al. (1994)
	Surface Water (aerated)		13.87	
	BAS (aerated)		37.74	
	BAS (non-aerated)		19.35	
Settled Water			18.00	
	Surface Water		16.50	
	Surface Water (aerated)		22.50	
	BAS		31.50	
	BAS(aerated)			

2.3.5.2 **Quantifying BDOC using bioreactors**

Regardless of the choice of inoculum, the critical limitation of batch BDOC methods is incubation time. Ranging from three days (Joret et al., 1988) to six weeks or more (Trulleyova and Rulik, 2006), the time of analysis is often too long for use by drinking water utilities interested in using the method to respond to operational problems. Lucena et al. (1991) addressed the issue of analytical time by introducing the use of continuous flow bioreactor technology ($BDOC_{cfbr}$), which allows for the dynamic determination of BDOC on time scales of hours, rather than days. The bioreactor based methods consist of glass columns, alone or in series, filled with a support medium on which a biofilm is grown from indigenous bacteria. The water to be evaluated passes through the bioreactor and BDOC is reported as the difference between measured DOC at the column influent and measured DOC at the column effluent. It is the large microbial community supported by the reactor bed that allows for a high inoculum to sample water ratio, which facilitates the rapidity of measurement (Maclean et al., 1996a). Because of this large population, and resultant excess

metabolic capacity, the reactors are buffered against operational changes such as fluctuating DOC concentration or temperature. (Kaplan and Newbold, 1995).

The $BDOC_{cfbr}$ method originally utilized BAS as a support medium (Lucena et al., 1991); but sintered borosilicate glass beads are now the preferred support because they are able to achieve comparable results while allowing for a higher degree of standardization of support media (Ribas et al., 1991). Columns employing sintered glass beads must first be colonized and achieve a relatively steady state before reliable measurements can be obtained. The colonization of columns presents somewhat of a limitation, as colonization may require one week (Frias et al., 1991) to nine months (Volk and LeChevallier, 2000).

Colonization of BDOC columns is typically achieved by circulating water through the reactor and allowing indigenous micro-organisms to develop a biofilm on the glass bead support (Frias et al., 1991; Lucena et al., 1991; MacLean et al., 1996; Sondergaard and Worm, 2001). In some cases the intended sample water is used for colonization (Frias et al., 1991). In others, sample water amended with untreated surface water (Frias et al., 1991) or with additional nutrients (Cormier, 1994) is used to speed the colonization process. Obtaining an accurate estimate of BDOC requires that the columns be colonized with water from the intended sampling site, as transfer to different water matrices results in a second stabilization period that can last for several months (Volk and LeChevallier, 2000; Camper et al., 2000). There are several measures of the "completeness of colonization." The most common is to consider the columns colonized when the measured DOC begins to approach the DOC measured using a BDOC batch method on parallel samples (Camper et al., 2000; Volk and LeChevallier, 2000). Others consider colonization complete when the %BDOC/DOC ratio stabilizes or reaches 20% BDOC/DOC (Frias et al., 1995; Lucena et al., 1991, Ribas et al., 1991), and still others simply consider colonization complete after a set period of time (MacLean et al., 1996a). Given the natural spatial and temporal variability of BDOC (Volk and LeChevallier, 2000), the approximate rule of 20% BDOC/DOC proposed by Frias et al., (1995) is likely not a sufficient measure of completeness of colonization. Verifying the columns performance using a reference method such as Servais et al., (1989) or Joret and

Levi, (1988) would ensure that the column is operating properly and providing an accurate estimate of BDOC. The BDOC columns studied by Camper et al. (2000) required nine months to achieve similar BDOC concentrations to the $BDOC_{sand}$ method.

The time of colonization is dependent on several characteristics of the sample water, including nutrient concentration, water temperature, and the presence of oxidants. Waters with higher concentrations of nutrients such as raw river water or ozonated water will typically become stabile faster than low DOC waters such as filter effluent (Frias et al., 1991), due to the higher growth rates associated with higher nutrient concentrations. Temperature also impacts the length of colonization. For example, at one location in Pennsylvania, columns that were put into operation in January took approximately seven months to achieve a steady BDOC/DOC ratio, in July at the same facility only four months was required to achieve stabilization (Kaplan and Newbold, 1995).

When operated properly, the $BDOC_{cfbr}$ method can achieve levels of accuracy and precision that are comparable to the batch BDOC methods. Replicate columns have produced precision values of 6.3% (Kaplan and Newbold, 1995). Recovery studies using known concentrations of model compounds have yielded accuracy rates of 73-100% for fulvic acid and 111-124% for acetate (Kaplan and Newbold, 1995). This recovery is higher than that reported for both the $BDOC_{susp}$ and $BDOC_{sand}$ method using acetate, at 94% and 89% respectively (Park et al., 2005). Of note, however, is that Park et al. (2005) failed to properly execute the BDOC sand method by not aerating the sample and by choosing an end point based on time, rather than minimum DOC concentration.

Depending on operating conditions, the $BDOC_{cfbr}$ method has been shown to consistently measure an equivalent BDOC concentration to batch methods (Frias et al., 1995; Lucena et al., 1991; Ribas et al., 1991; Sondergaard and Worm, 2001; MacLean et al., 1996a). The reason BDOC CFBRs are able to achieve this degree of performance on such short time scales is the high inoculum to sample ratio that exists within the column, allowing for a large and diverse population of biofilm associated micro-organisms to degrade the bioavailable pool of dissolved organic carbon molecules (MacLean et al., 1996a).

The performance of BDOC bioreactors is impacted by several water quality parameters; in particular, nutrient concentrations and the presence of oxidants. The type of sample water can significantly impact the performance of BDOC bioreactors; for exmaple, raw water BDOC is often underestimated by the $BDOC_{cfbr}$ method (Lucena et al., 1991; Frias et al., 1995). The presence of higher concentrations of more refractory BDOC compounds in raw water could result in this underestimation, suggesting experimentation for optimized contact time, either through the addition of multiple columns or a reduction of flow rate may improve performance. However, the addition of a second column by Ribas et al., 1991 yielded only a minor increase in BDOC removal (6.9%) in raw water, which is in the range of column abiotic DOC adsorption reported by Kaplan and Newbold (1995). Another possibility is that the BDOC batch methods overestimate the quantity of BDOC present in raw water, and that the bioreactors are more reflective of operational reality. Given that the batch methods allow a stable period of up to 42 days for the inoculum bacteria to multiply and develop enzyme expression patterns for optimized biodegradation, along with strong selective pressure in the later weeks of the incubation period, it could be that the $BDOC_{cfbr}$ method provides an estimate that is more reflective of operational reality, with water flowing across a biofilm. It is also possible that differences in adsorption isotherms between the methods lead to this discrepancy, for example the batch methods are in a closed system that can reach equilibrium, whereas a CFBR is an open system that is more representative of treatment processes and distribution systems. Further experimentation in this area could elucidate whether these discrepancies result from a lack of metabolic capacity in the BDOC CFBR or due to differences in adsorption and biodegradation caused by a comparison of a closed system given up to six weeks to up-regulate enzyme expression patterns for breakdown of more refractory BDOC that would not be degraded in an open system such as a CFBR or distribution pipe.

The presence of chlorine in sample water during BDOC analysis, such as in finished or pre-chlorinated waters, and consequent requirement for thiosulfate dechlorination can also significantly impair the performance of a BDOC CFBR either through increased variability

(Frias et al., 1995) or underestimation of BDOC (Lucena et al., 1991). Left untreated, the presence of a disinfectant like chlorine would impair the ability of test organisms in a CFBR to degrade DOC, leading to an underestimation of BDOC. Therefore, when a chlorinated sample is tested for BDOC, chlorine quenching is necessary. This is often achieved through thiosulfate addition, calcium or sodium thiosulfate is often employed. Unless the thiosulfate concentration is carefully controlled, however, it is metabolized by naturally occurring thiobacilli which produce sulfuric acid as a by-product, resulting in a pH drop and consequent underestimation of BDOC (MacLean et al., 1996b). This can be avoided by dosing thiosulfate at near-stoichiometric concentrations with chlorine; a difficult task in a continuous flow system. In cases where finished water is measured for BDOC it may be necessary to either dechlorinate in a carefully pH controlled reservoir or to choose a batch method.

In contrast to oxidants and sample water characteristics, no substantial impacts of temperature on the concentration of BDOC measured by CFBRs have been reported. Using parallel $BDOC_{cfbr}$ measurements on the same influent water, with one stored at a temperature range of 8-11^{o}C and the other at 19-23^{o}C, it was found that temperature did not impact CFBR performance when measuring both ozonated and BAC filtered water (MacLean et al., 1996a). Although a short term drop in temperature has been shown to negatively impact performance (Kaplan et al., 1994), a similar short term temperature drop from 14^{o}C to 1.7^{o}C, but with well acclimated CFBRs, had a negligible impact on BDOC measurement (Kaplan and Newbold, 1995). The resilience of acclimated CFBRs to temperature change is likely due to their excess metabolic capacity (MacLean et al., 1996a).

Optimized CFBR performance involves balancing sufficient contact time for adequate DOC removal with column flow rate, which will dictate sampling speed. The most commonly reported EBCTs of CFBRs range from one to three hours (Ribas et al., 1991; Frias et al., 1995; Kaplan and Newbold, 1995). Tracer studies have determined that retention time has been reported to be approximately 75% of EBCT (Kaplan et al., 1994). Two columns in series are typically employed to ensure a reasonable balance between flow rate

and sufficient contact time (Ribas et al., 1991, Kaplan and Newbold, 1995). The benefit of using a second column has yet to be clearly demonstrated, however. Ribas et al. (1991) reported an increased %BDOC/DOC measurement of 6.9 +/- 5.9 % for raw water, 9.2 +/- 5.4% for GAC filtered water, and 10.5 +/- 9.8% for finished water, all of which are comparable to the rate of abiotic DOC uptake of 3 +/- 3% reported by Kaplan and Newbold (1995). Other studies have shown that more than 85% of biodegradation occurs within the first ten minutes of contact time within the reactor (Cormier, 1994) and that an 85% increase in contact time led to only a 10% increase in measured BDOC concentration (Kaplan et al., 1994). Optimum contact time is likely site specific, and will depend on the composition of the DOC present in water. Rather than implementing published values it is likely beneficial to determine the appropriate contact time through experimentation, taking into account the impact of increased BDOC loading associated with seasonal variability or storm surges. This approach would allow for efficient BDOC estimation at maximum sampling speed and in some cases could result in cost savings if a second column is not necessary.

To summarize, properly operated CFBRs offer a robust, dynamic, and rapid measurement of BDOC in drinking water treatment scenarios, although careful dechlorination is necessary for finished waters, and perhaps substantially extended contact time is necessary for raw waters. Due to high cell densities within CFBRs and the short contact times at which the majority of BDOC is biodegraded (Cormier et al., 1994), CFBRs are resistant to temperature (MacLean et al., 1996a; Kaplan and Newbold, 1995) and influent DOC concentration fluctuations such as may be expected during storm surges (Kaplan and Newbold, 1995a). Consistent performance, in combination with speed of measurement, and precision make CFBRs an ideal currently available tool for assessing BDOC at drinking water treatment plants.

2.3.5.3 **BDOC Method Summary**

Relative advantages and disadvantages of the different BDOC methods are summarized in table 2.3. The $BDOC_{susp}$, $BDOC_{fixed}$, and $BDOC_{cfbr}$ methods, when executed properly, are all capable of providing precise and accurate measurement of BDOC concentration in a variety

of water types. The $BDOC_{susp}$ method has the advantage of being a simple method with a readily available inoculum. Although using environmental inocula results in a different inoculum being used for each sample, this has not been found to contribute to variability amongst replicate samples (Block et al., 1992; Prévost et al. 1992). Additionally, this method is capable of a high degree of accuracy in recovery studies using standard solutions of BOM molecules (Park et al., 2005) and, depending on the nature of the tested water, often provides comparable results to the $BDOC_{fixed}$ and $BDOC_{cfbr}$ methods (Frias et al., 1995; Lucena et al., 1991; Trulleyova and Rulik, 2004; Park et al., 2005). Due to the varying degree of biodegradability of DOC molecules and the limited diversity and quantity of biomass present in suspended inocula, the $BDOC_{susp}$ method is not necessarily well-suited to all water matrices. For some samples the $BDOC_{susp}$ method underestimates sample DOC (Frias et al., 1995, Lucena et al., 1991, Volk et al., 1994, Block et al., 1992). The method also requires an impractically long incubation period of approximately 21-28 days (Servais et al., 1987, 1989). Because of its simplicity the $BDOC_{susp}$ method is ideally suited for measuring samples where there is no ready supply of a fixed inoculum such as BAS, where there is no requirement for operational changes in response to BDOC fluctuation, and where the method has been verified to be appropriate for the specific water matrix being tested.

Table 2.3: Advantages and Disadvantages of Three Methods for BDOC Quantification

	$BDOC_{susp}$	$BDOC_{fixed}$	$BDOC_{cfbr}$
Advantages	• Simple, readily available inoculum • Only initial and final DOC measurements necessary	• Analysis complete in days rather than weeks • More representative estimation of BDOC (biofilm)	• Fastest method (hours rather than days) • Most representative (water continuously flows across a biofilm)

Disadvantages	• Much longer incubation times due to lower initial concentration of micro-organisms • Underestimation of BDOC in some water matrices due to limited biodiversity	• Higher variability due to DOC leaching from media • Extensive pretreatment • Multiple daily samples necessary until a minimum can be determined	• Continuous dechlorination required for chlorinated waters • Possible poor performance with some raw waters • Long colonization period

2.4 Impact of Filtration Media on Biological Filter Performance

As discussed in section 2.2, biofilms require a solid support on which to attach and proliferate. In biological filtration, this support is typically a synthetic membrane or granular media. A range of materials are employed as granular media, including: engineered ceramics, expanded clay, charcoal, diatomaceous earth, sand, etc. Media selection is a critical design parameter with significant short and long term cost and performance implications. For the present study, the effect on filter performance of anthracite, a non-adsorptive medium, and granular activated carbon (GAC), an adsorptive medium are compared, both in terms of biological and traditional filter performance parameters. Because different media types may differ in optimal backwashing procedures, the interaction between backwash parameters and media characteristics was also tested. In addition to media type, the size of the media grains, and the uniformity of their distribution can have a significant impact on traditional filter performance, but their impact on biological performance has not previously been studied.

2.4.1 Effect of Media Size and Uniformity Co-efficient

In the drinking water treatment industry, granular filtration media are characterized by effective size and uniformity coefficient. The size distribution of granular media is determined by sieve analysis, where a given weight of sample is passed through a series of sieves and the fraction of sample retained by each sieve size is determined (ASTM, 2001). Effective size (E.S.) is defined as the media grain diameter at which 10% of the media are smaller (d_{10}). The uniformity coefficient of a given medium is the ratio of the 60^{th} percentile

grain diameter (d_{60}) to the 10th percentile grain diameter (d_{10}) according to equation 2.1 (Crittenden et al., 2005).

$UC = d_{60}/d_{10}$ (equation 2.1).

There are several equations used to predict clean bed head loss through granular media, and the correct equation is chosen based on the flow regime of the system, which is identified based on the Reynold's number for flow around a sphere. The filtration rate employed in this study is 11.2 m/h, and the effective size of the media ranged from 1.06mm to 1.3mm, resulting in associated Reynold's numbers of 2.03 to 2.80. When the Reynold's number is above 1.0, flow in filtration systems is described by the Forchheimer flow regime, which was related to media properties by Ergun (1952), and is presented in equation 2.2. From equation 2.2, it is clear that head loss is inversely proportional to media size, with increasing effective size resulting in decreased head loss through the filter bed.

$$h_l = \kappa_v \frac{(1-\varepsilon)^2}{\varepsilon^3} \frac{\mu L v}{\mu_w g d^2} + \kappa_I \frac{1-\varepsilon}{\varepsilon^3} \frac{Lv^2}{gd} \quad \text{(Equation 2.2)}$$

Where: κ_v and κ_I are head loss coefficients, ε = porosity of filter bed, μ_w = viscosity of water, L = filter bed depth, v = superficial velocity, g = acceleration due to gravity, and d = diameter of a media grain. Therefore, to increase the duration of the filter run by decreasing the rate of head loss accumulation, a larger effective size is desirable. As effective size increases, however, particle removal by filters is generally decreased (figure 2.2). Consequently, optimized design will require the selection of a media size that will minimize head loss accumulation while maximizing particle removal. The design of biological filters requires a third factor to be considered: BOM removal. At present, however, no published data exist on the impact of media effective size on BOM removal performance of biological filters.

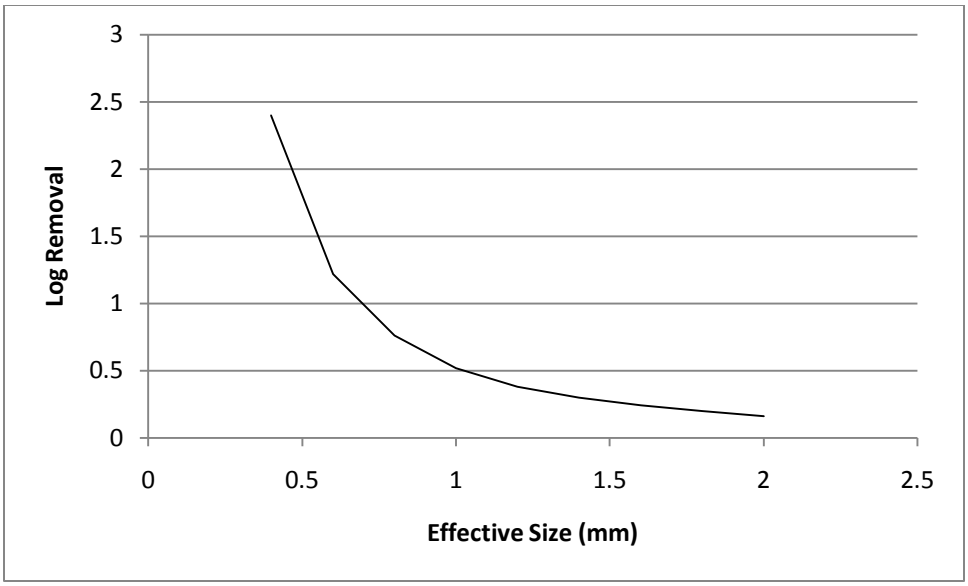

Figure 2.2: Relationship between log removal of 0.1µm particles and effective size of filtration media (adapted from data presented in Crittenden, 1985)

A key mechanism for any potential impact of media size on biological BOM removal will be contact time. BOM removal has been correlated to dimensionless contact time, X*, which is described by equation 2.3 (Zhang and Huck, 1996).

$$x^* = \theta \frac{\alpha D_f}{\tau}$$ Equation (2.3)

Where:

x^* = Dimensionless contact time

θ = EBCT

α = Specific surface area (biofilm surface area per unit volume of filtration media)

D_f = Diffusivity of substrate into biofilm

τ = Biodegradation rate

The term α from equation 2.3 is directly affected by the surface area of the media. Although biofilm distribution on filter media is highly heterogeneous, the surface area of the media will dictate the surface area available for biofilm growth. The surface area of a given filtration medium is approximately related to effective size according to equation 2.4.

$S = \frac{\xi(1-\varepsilon)}{d}$ (Equation 2.4)

Where:

S = specific surface area (m^{-1})

ξ = dimensionless shape factor (dimensionless)

ε = porosity of the filter bed (dimensionless)

d = effective size (m)

It is important to note that equation 2.4 does not take into account the intra-particle surface area of porous media such as GAC. The micropores of GAC, with an average diameter of 1-100 nm, are too small to allow colonization by most bacteria, which are typically larger than 200nm (Madigan et al., 1996). Therefore, given the relationship between effective size and surface area available for microbial growth, although there are no published data demonstrating this, it is not unreasonable to speculate that biological removal may be impacted by the effective size of a given medium. This represents a significant knowledge gap, as media size is a key filtration design parameter not only for BOM removal, but also for traditional filter performance such as head loss accumulation and particle removal.

Like ES, there are no published data available regarding the impact of media uniformity on particle and BOM removal by biological filters. Uniformity coefficient affects filter performance by influencing media stratification: over time smaller grains will accumulate at the top of the filter, with particle size increasing with increasing bed depth (Crittenden et al. 2005). This will lead to increased top of filter head loss; it is possible that biological growth could act as an antagonistic factor with the increased concentration of smaller media grains at the top of filter to increase the rate of terminal head loss accumulation. This too represents a knowledge gap, as more uniformly distributed media require a higher degree of mechanical sorting, and are significantly more expensive as a result. It is therefore desirable to have data regarding of the impact of specific uniformity coefficients on biological filter performance to facilitate informed decisions on selecting media.

2.4.2 Effect of Media Type

When selecting filtration media, the first priority of a given drinking water utility will be to ensure that the filtration media will allow the utility to meet all regulatory standards and guidelines. Subsequently, the economic benefits of selecting optimal filter media will be balanced against initial installation, and long term operations and maintenance costs. For example, a given unit of GAC can cost four times the amount of an equivalent unit of anthracite. Assuming both media were able to meet all regulatory standards and guidelines, the economic benefit of potentially enhanced BOM removal in the GAC media (ie: reduced disinfectant demand and distribution system fouling) would have to offset the significant initial investment, as well as the long term operations and maintenance cost.

The two types of media tested in the present study were non-adsorptive anthracite, and GAC, an adsorptive medium. Raw material for GAC can be a range of carbonaceous materials, typically coal or wood based. GAC production is a two stage process of carbonizing the raw material, and then activating it through oxidation. During the carbonization process, raw material such as lignite or coal is pyrolized in the absence of oxygen, a process that yields a carbonaceous residue (Weber et al., 1980). Preferential oxidation of approximately fifty percent of the pyrolized residue in the subsequent oxidation process results in the formation of a series of fissures, interstices, and channels throughout the remaining material and conveys adsorptive surface properties (Wolff, 1959). The end product is a highly porous material (figure 2.3), with channels and fissures forming macro- and mesopores, and interstices between channels forming micropores (Weber, 1972). This highly porous structure results in high surface to volume ratio in the range of 800-1200 m^2/g (Allan, 1974).

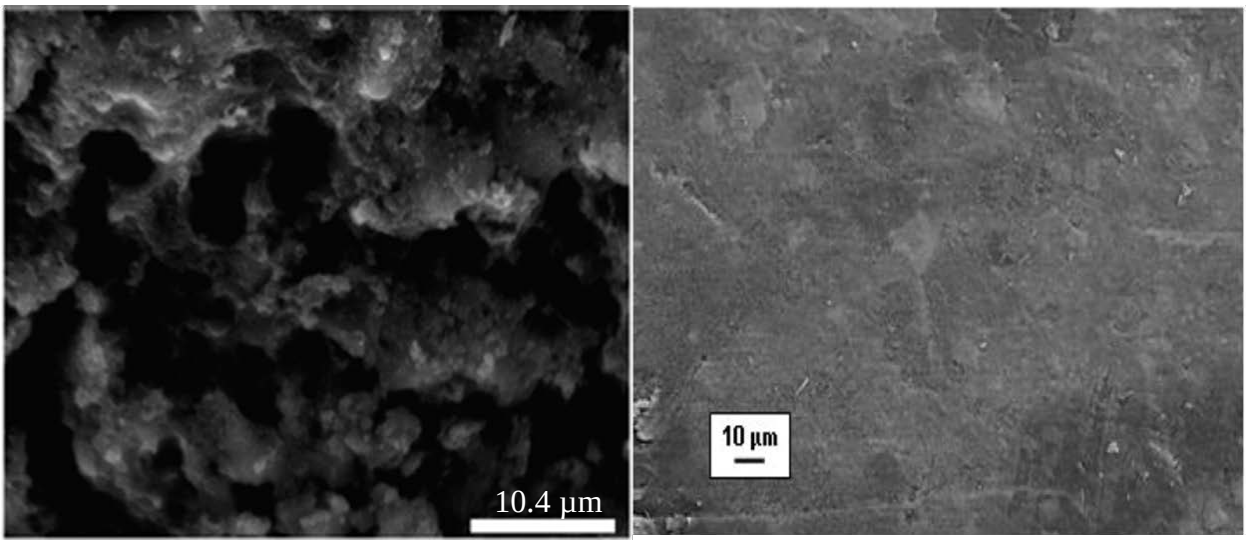

Figure 2.3: Comparison of Left: GAC surface (Velten et al., 2007) and Right: Anthracite surface (Scott, 2008).

The porosity of GAC results in both a high surface to volume ratio, and a high degree of surface roughness. Both of these properties likely contribute to the ability of GAC to support a higher concentration of biomass per unit volume of media. The increased surface area of GAC allows for a much higher biofilm-bulk fluid interface per unit volume of filter media. Substratum micro-roughness, a characteristic of GAC, also favours biological attachment by three mechanisms: 1) enhanced convective transport rates near the GAC surface, 2) shelter from hydrodynamic shear forces, and 3) more efficient cell and EPS attachment due to increased cell-substratum contact area (Characklis, 1990).

As discussed above in section 2.4.2, biomass concentration does not necessarily directly correlate to BOM removal by biological filters (Miltner et al., 1995, Wang et al., 1995; Perrson et al., 2006). GAC filters are often cited as achieving higher BOM removal rates than anthracite (LeChevallier et al., 1992, Krasner et al., 1993, Wang et al., 1995, Liu et al., 2001). However, as will be discussed below, these performance differences are often due to design and operational conditions, rather than inherent properties of GAC.

Table 2.4: Comparison of relevant media properties of GAC and Anthracite

	Anthracite	GAC
Surface to volume ratio	Low	High
Roughness	Smooth	Rough
Adsorption	Non-Adsorptive	Adsorptive
Friability	Low	High
Cost	$	$$$$

Despite a number of studies comparing AOC removal in GAC and anthracite filters, a significant improvement in AOC removal has yet to be incontrovertibly linked to GAC media. In the cases where differences have been observed, media type is confounded with other design and operational factors such as media size (Chien et al., 2008; Wang et al., 1995), backwash technique (Krasner et al., 1993), actively adsorbing GAC (LeChevallier et al., 1992), and EBCT (Chien et al., 2008).

Table 2.5: Studies comparing AOC removal in GAC and anthracite filters

Study	Media (E.S.) (mm)	% Removal (SD)	EBCT (min)	Design Influences
Wang et al. (1995)	Anth (1.02) GAC (0.64) GAC (0.64) GAC (1.52)	39 (26) 51 (23) 47 (22) 42 (16)	9.2	Un-matched media sizes
Krasner et al. (1993)	Anth (N.R.) GAC (1.0-1.1) Anth (N.R.) GAC (1.0-1.1)	68 74 57 72	4.2 4.2 1.4 1.4	Chloraminated backwash (0.2-0.4 mg/L)
LeChevallier et al. (1992)	Anth (1.0-1.2) GAC (0.8-0.9)	30.8 57.1	7.5	GAC not exhausted
Chien et al. (2008)	Anth (1.0-1.2) GAC (0.5-0.6)	17 58	4.8 6.6	Depth = 80 cm Depth = 120 cm

Study	Media (E.S.) (mm)	% Removal (SD)	EBCT (min)	Design Influences
Huck et al., 2000	Anth (n.r.)	64 (6*)	3.6	Non-chlorinated BW
	GAC (n.r.)	63 (22*)		Non-chlorinated BW
	Anth (n.r.)	57 (14*)		Chlorinated BW
	GAC (n.r.)	54 (16*)		Chlorinated BW
	Anth (n.r.)	21 (21*)		Chloraminated BW
	GAC (n.r.)	12 (19*)		Chloraminated BW

** - Inter-Quartile Range*

n.r. – Not Reported

In a study to determine the impact of biomass on biological filter performance, Wang et al. (1995) evaluated four media types: bituminous coal (GAC1), lignite based coal (GAC2), wood based GAC (GAC3), and anthracite. Although the wood based GACs had higher mean AOC-NOX removal than the anthracite and coal based GAC filters, the difference was not statistically significant, and it is unclear whether the increased removal was attributable to increased biological activity, or simply due to increased contact time associated with the much smaller effective size of the wood based GAC. Another pilot study comparing AOC removal in anthracite and GAC filters demonstrated significantly higher total AOC removal (P17 and NOX) in GAC filters relative to a parallel anthracite filter (Chien et al., 2008). Like Wang et al. (1995), this finding is confounded because the effective sizes of the media were not matched; here, a 0.6 mm E.S. GAC was compared to a 1.1 mm E.S. anthracite. Moreover, a tri-media anthracite filter with a total depth of 80 cm of anthracite was compared to that of a mono-medium GAC filter with a total depth of 110 cm. Accordingly, the higher AOC removal by GAC could be attributable to enhanced biological activity or differences in media contact time. Often these types of comparisons are necessary in practice because they utilize media configurations necessary for achieving optimal traditional filtration performance (e.g. maximum filter run time), although sometimes filter design can be based on what has worked in a particular environment historically. While these types of studies are practically useful, they are limited in their capacity to provide mechanistic insight; caution should be exercised to prevent over-interpretation. These studies

highlight the importance of matching media characteristics such as effective size, media size distribution (and therefore uniformity coefficient), bed depth, and possibly other surface characteristics such as surface roughness to achieve an accurate assessment of the impact of media type when the interest is mechanistic elucidation of filter design and operation optimization.

In addition to media characteristics and configurations, operational factors such as backwash technique can also impact the accuracy of media performance comparisons. Krasner et al. (1993) compared AOC removal in an anthracite filter to a parallel GAC filter at EBCTs of 4.2 and 1.4 min respectively. At the longer EBCT, no difference in AOC removal was detected between GAC and anthracite. At the shorter EBCT however, GAC appeared to have a higher AOC removal (72% vs 57%). This result is confounded, however, because at the high loading rate the filters were backwashed every eight hours with water containing a chloramine residual. As discussed in detail below (section 2.5), the presence of chlorine in backwash water can impair the biological performance of anthracite filters, while GAC filters are much more resilient (Liu et al., 2001). This period of impaired BOM removal can last up to twelve hours (Miltner et al., 1995), and it is possible the differences in AOC removal between the anthracite and GAC filters that were reported by Krasner et al. (1993) were the result of inadequate biomass regeneration on the anthracite filter between backwashes. LeChevallier et al. (1993) compared AOC removal in parallel anthracite and GAC filters, using roughly matched E.S. and non-chlorinated backwash water, and reported that GAC was capable of higher AOC removals than anthracite. In this case, however, relatively fresh GAC was compared to anthracite, leaving in doubt whether the increased AOC removal observed in the GAC filter was attributable to microbial activity or residual adsorptive capacity.

To summarize: a *conclusive* demonstration of the relative AOC removal advantages of GAC relative to anthracite does not exist. This represents a significant research gap, as one of the key objectives of biological filtration is to prevent biofilm growth in the distribution system and AOC has been shown to be a key parameter in determining whether this will occur (LeChevallier et al., 1993). Urfer et al. (1997) presented a set of guidelines for carrying

out studies of biological filtration that, if adhered to, should allow for more conclusive analyses of the impact of media type on BOM removal; these are presented in table 2.6.

Table 2.6: Recommendations for conducting studies of biological filtration (reproduced from Urfer et al., 1997)

Parameter	Recommendations
Filter Media	• Sufficient time should be allotted to ensure the adsorptive capacity of GAC is exhausted prior to any assessment of BOM removal. • When comparing filtration media, grain size distributions (E.S. and U.C.) should be matched.
Backwashing	• Backwash procedures at bench or pilot-scale should be matched to full-scale backwash. • Document detailed backwashing procedures, and consider how manual backwashing (potentially variable) could impact results. • When possible, at least one filter backwashed with non-chlorinated water should be included.
BOM Removal	• Filters should be allowed to achieve pseudo-steady state removal prior to study. • If possible, experiments should begin under warm-water conditions • The point in the filter cycle at which samples were taken must be documented.
Oxidants	• Ozone residual or other influent oxidants should be minimized, unless the impact of such residuals on filter performance is being studied.
Temperature	• Filter influent temperatures must be documented
Other Issues	• Filter out of service time such as on weekends or between experiments should be avoided.

BDOC removal in parallel anthracite and GAC filters has been compared in only one full-scale study (Huck et al., 2000; table 2.7). BDOC is an aggregate measure which quantifies the extent of biologically mediated BOM removal through biological filters. BDOC therefore represents an aggregate measure of biologically mediated: AOC, DBP precursor, and chlorine demand simultaneously. In that study, BDOC removal was found to be comparable in both full-scale anthracite and GAC filters, regardless of whether chlorinated wash water was used. The dearth of studies on comparing BDOC removal by

GAC and anthracite filters is a significant research gap, particularly given that it has not been compared extensively at cold water conditions, at which it is commonly believe that GAC offers better BOM removal than anthracite.

Table 2.7: Studies comparing BDOC removal in GAC and Anthracite Filters

Study	Media (E.S.) (mm)	% Removal (S.D.)	EBCT	Design/Operational Influences
Huck et al., 2000	Anth (n.r.)	31 (10*)	3.6	Non-chlorinated BW
	GAC (n.r.)	34 (15*)		Non-chlorinated BW
	Anth (n.r.)	31 (3*)		Chlorinated BW
	GAC (n.r.)	21 (18*)		Chlorinated BW
	Anth (n.r.)	15 (20*)		Chloraminated BW
	GAC (n.r.)	12 (19*)		Chloraminated BW
Huck et al., 2000	Anth (n.r.)	65 (10)	15	Temperature ranged from 5 – 10^{o}C
	GAC (n.r.)	67 (20)		
Huck et al., 2000	Anth (n.r.)	50 (10)	15	Temperature > 10^{o}C
	GAC (n.r.)	48 (15)		

* - Inter-Quartile Range

n.r. – not reported

In addition to increased biological stability, another of the primary goals of biological filtration is increasing the efficiency of disinfection. This includes: reducing chlorine demand, increasing the stability of residual disinfectant concentration through the distribution system, and removal of disinfection by-product precursors. Information regarding the comparative removal of chlorine demand by GAC and anthracite is lacking. Two studies have reported the relative removal of disinfection by-product precursors by these media (Wang et al., 1995; Huck et al., 2000). Wang et al. (1995) used two lignite based GAC media, a bituminous coal based GAC medium, and anthracite. Both of the wood based GAC media removed more THM-FP and TOX-FP than the anthracite and coal based GAC, but as discussed above, the use of unmatched media sizes preclude mechanistic interpretation of the outcomes because they may be attributable to differences in microbial activity and/or media size. Accordingly, there is a knowledge gap regarding the impacts of media type, particularly

selection between GAC and anthracite for improving disinfection efficiency by biological filtration.

Table 2.8: Studies comparing DBP/DBP-precursor Removal in GAC and Anthracite Filters

Study	Media (E.S. mm)	DBP/Precursor Quantified	% Removal	EBCT	Design Influences
Wang et al. (1995)	Anth (1.02) GAC (0.64) GAC (0.64) GAC (1.52)	THM-FP	23 (6) 40 (5) 34 (5) 27 (3)	9.2	Unmatched media sizes
Wang et al. (1995)	Anth (1.02) GAC (0.64) GAC (0.64) GAC (1.52)	TOX-FP	28 (4) 52 (5) 44 (6) 31 (7)	9.2	Unmatched media sizes
Huck et al. (1998)	Anth (n.r.) GAC (n.r.)	THM-FP	17.5 (9*) 19.2 (7*)	3.6	
Huck et al. (1998)	Anth (n.r.) GAC (n.r.)	HAA-6	18.3 (10*) 21.7 (5*)	3.6	

Aldehyde removal is the most commonly reported parameter for describing and quantifying BOM removal by filtration (Table 2.9). Unlike AOC removal, GAC has been demonstrated to have a clear advantage over anthracite in the removal of certain aldehydes; particularly more recalcitrant aldehydes such as glyoxal and methyl-glyoxal (Krasner et al., 1993; Liu et al., 2001). Like AOC removal, however, in most reported cases it is unclear whether higher aldehyde removal observed in GAC filters is attributable to an inherent advantage of GAC or to operational factors such as chlorinated backwash (Krasner et al., 1993; Emelko et al., 2006) or ozone residuals in filter influent water (Niquette at al., 1998; Wobma et al., 2000).

GAC and anthracite filters appear to removal comparable levels of the more biologically labile aldehydes (e.g., formaldehyde) and carboxylic acids so long as a non-chlorinated backwash is employed (Liu et al., 2001). When chlorinated backwash is employed; however, both aldehyde and carboxylic acid removal is significantly impaired in

anthracite filters, particularly at cold water temperatures (Emelko et al., 2006, Krasner et al., 1993, Liu et al. 2001). The antagonistic effect of chlorinated backwash at low temperatures is evident in the work reported by Liu et al. (2001) who compared parallel anthracite filters treating water at 5^{o}C and 20^{o}C with both chlorinated and non-chlorinated backwash. In that study, biologically labile compounds such as formaldehyde, acetate, and formate were removed comparably by anthracite and GAC filters at 20^{o}C, regardless of whether a chlorinated backwash was used; however, at 5^{o}C chlorinated backwash severely impaired the removal of each compound in anthracite filters, while GAC filter removal was quite robust. This effect is consistent with the results of Emelko et al. (2006), who reported oxalate removal in backwash chlorinated anthracite filters to be negligible while GAC filter removal was approximately 90%. This result is also consistent with the findings of Krasner et al. (1993), who reported comparable formaldehyde removal by anthracite and GAC filters at warm water operating conditions, despite the presence of a chlorinated backwash. Chlorinated backwash appears to only impair the removal of carboxylic acids, and the more labile aldehydes in anthracite filters at low operating temperatures.

Unlike formaldehyde, glyoxal is an aldehyde that is recalcitrant to biological degradation. GAC filters have consistently demonstrated greater glyoxal removal than anthracite filters (table 2.9), regardless of operating temperature (Krasner et al., 1993; Liu et al., 2001). It is important to note that the relative significance of the removal of this one specific aldehyde with respect to the goals of biological filtration is unclear. This point more broadly highlights one of the inherent difficulties in interpreting aldehyde or carboxylic acid concentration data because those data not provide conclusive information regarding the ability of a given filter to: reduce chlorine demand, remove DBP precursors, or increase biological stability. In contrast, is specific ozonation by-products have are of health significance, it is quite useful to have a quantitative measurement of a given filter configuration's ability to remove them. For example, glyoxal has demonstrated cytotoxic effects (Shangari and Obrien, 2004) and may at some point become a regulated ozone

disinfection by-product, in which case the use of GAC filter media would potentially represent a potential advantage for achieving regulatory compliance.

The apparent enhanced removal of glyoxal could indicate that GAC offers an advantage in removing the more recalcitrant organic contaminants, possibly due to adsorptive capacity retaining the more slowly biodegradable nutrients. Such an advantage would be highly dependent on the chemical properties of the adsorbed substrate. The process of biodegradation of adsorbed substrates is dependent on compound specific desorption kinetics (Aktas and Cecen, 2007). This process could also account for the faster recovery of GAC after out of service time or system perturbations (Krasner et al., 1993). Given that AOC removal remains comparable even when glyoxal removal is impaired (Krasner et al., 1993, Liu et al., 2001), a utility would need to consider whether the goal of biological filtration was BOM reduction, the removal of specific OBPs, or both.

Table 2.9: Studies comparing Aldehyde/Carboxylic Acid removal in GAC and anthracite filters

Aldehyde	Media (E.S.) (mm)	% Removal	EBCT (min)	Design Influences	Study
Glyoxal	Anth GAC Anth GAC	57 >90 37 >90	4.2 4.2 1.4 1.4	Chloraminated backwash	Krasner et al. (1993)
Glyoxal	Anth (1.1) GAC (0.9) Anth (1.1) GAC (0.9) Anth (1.1) GAC (0.9) Anth (1.1) GAC (0.9)	55 (30) 82 (24) 71 (16) 77 (6) 13 (30) 65 (3) 58 (30) 82 (6)	5.6	20°C (Chlorinated Backwash) 20°C (Chlorinated Backwash) 20°C (No Chlorine) 20°C (No Chlorine) 5°C (Chlorinated Backwash) 5°C (Chlorinated Backwash) 5°C (No Chlorine) 5°C (No Chlorine)	Liu et al. (2001)
Glyoxal	Anth (1.4) GAC (1.5)	- 5.6 83.3	13.2	Ozone residual of 0.4 mg/L loaded on to filters.	Niquette et al. (1998)
Glyoxal	Anth (1.4) GAC (1.5) Anth (1.4) GAC (1.5)	-18.5 66.7 -35.1 78.4	5	Winter Winter Summer Summer Flocculent loaded directly onto filter	Weinberg et al. (1993)
Glyoxal	Anth (n.r.) GAC (n.r.) Anth (n.r.) GAC (n.r.) Anth (n.r.) GAC (n.r.)	82 (6*) 82 (4*) 81 (7*) 79 (6*) 75 (9*) 73 (9*)	3.6	Non-chlorinated BW Non-chlorinated BW Chlorinated BW Chlorinated BW Chloraminated BW Chloraminated BW	Huck et al., 2000
Methyl-Glyoxal	Anth (1.4) GAC (1.5) Anth (1.4) GAC (1.5)	-8.6 70 -43.9 71.9	5	Winter Winter Summer Summer Flocculent loaded directly onto filter	Weinberg et al. (1993)
Methyl-Glyoxal	Anth (1.4) GAC (1.5)	0 72.7	13.2	Ozone residual of 0.4 mg/L loaded on to filters.	Niquette et al. (1998)

Aldehyde	Media (E.S.) (mm)	% Removal	EBCT (min)	Design Influences	Study
Methyl-Glyoxal	Anth GAC Anth GAC	62 88 75 88	4.2 4.2 1.4 1.4	Chloraminated backwash	Krasner et al. (1993)
Formaldehyde	Anth GAC Anth GAC	>92 >92 >92 >92	4.2 4.2 1.4 1.4	Chloraminated backwash	Krasner et al. (1993)
Formaldehyde	Anth (1.1) GAC (0.9) Anth (1.1) GAC (0.9) Anth (1.1) GAC (0.9) Anth (1.1) GAC (0.9)	89 (9) 92 (6) 94 (3) 97 (6) 13 (3) 84 (3) 88 (15) 94 (19)	5.6	20^oC (Chlorinated Backwash) 20^oC (Chlorinated Backwash) 20^oC (No Chlorine) 20^oC (No Chlorine) 5^oC (Chlorinated Backwash) 5^oC (Chlorinated Backwash) 5^oC (No Chlorine) 5^oC (No Chlorine)	Liu et al. (2001)
Formaldehyde	Anth (1.4) GAC (1.5)	59.4 87.5	13.2	Ozone residual of 0.4 mg/L loaded on to filters.	Niquette et al. (1998)
Formaldehyde	Anth (1.4) GAC (1.5) Anth (1.4) GAC (1.5)	-23.5 100 -86.7 100	5	Winter Winter Summer Summer Flocculent loaded directly onto filter	Weinberg et al. (1993)
Formaldehyde	Anth (n.r.) GAC (n.r.) Anth (n.r.) GAC (n.r.) Anth (n.r.) GAC (n.r.)	93 (3*) 93 (3*) 95 (7*) >95 (8*) 94 (0*) 94 (1*)	3.6	Non-chlorinated BW Non-chlorinated BW Chlorinated BW Chlorinated BW Chloraminated BW Chloraminated BW	Huck et al., 2000
Acetaldehyde	Anth (1.4) GAC (1.5)	44.7 87.5	13.2	Ozone residual of 0.4 mg/L in filter influent.	Niquette et al. (1998)
Total Aldehydes	Anth (1.1) GAC (1.1)	25.4 (9.5) 68.8 (16.3)	3.6	Ozone residual of 0.6 mg/L in filter influent.	Wobma et al. (2000)

Aldehyde	Media (E.S.) (mm)	% Removal	EBCT (min)	Design Influences	Study
Acetate	Anth (1.1)	85 (9)	5.6	20°C (Chlorinated Backwash)	Liu et al. (2001)
	GAC (0.9)	89 (18)		20°C (Chlorinated Backwash)	
	Anth (1.1)	87 (19)		20°C (No Chlorine)	
	GAC (0.9)	89 (19)		20°C (No Chlorine)	
	Anth (1.1)	50 (19)		5°C (Chlorinated Backwash)	
	GAC (0.9)	87 (13)		5°C (Chlorinated Backwash)	
	Anth (1.1)	85 (18)		5°C (No Chlorine)	
	GAC (0.9)	92 (19)		5°C (No Chlorine)	
Formate	Anth (1.1)	88 (12)	5.6	20°C (Chlorinated Backwash)	Liu et al. (2001)
	GAC (0.9)	91 (9)		20°C (Chlorinated Backwash)	
	Anth (1.1)	90 (13)		20°C (No Chlorine)	
	GAC (0.9)	90 (13)		20°C (No Chlorine)	
	Anth (1.1)	32 (13)		5°C (Chlorinated Backwash)	
	GAC (0.9)	87 (13)		5°C (Chlorinated Backwash)	
	Anth (1.1)	85 (18)		5°C (No Chlorine)	
	GAC (0.9)	91 (19)		5°C (No Chlorine)	
Oxalate	Anth	54.4 (14.7)	n.r.	21-23°C (BWC)	Emelko et al. (2006)
	GAC	91.2 (17.6)		21-25°C (BWC)	
	Anth	- 5.9 (11.8)		1-3°C (BWC)	
	GAC	91.2 (11.8)		1-3°C (BWC)	
Oxalate	Anth (n.r.)	82 (10*)	3.6	Non-chlorinated BW	Huck et al., 2000
	GAC (n.r.)	79 (8*)		Non-chlorinated BW	
	Anth (n.r.)	76 (5*)		Chlorinated BW	
	GAC (n.r.)	80 (8*)		Chlorinated BW	
	Anth (n.r.)	48 (10*)		Chloraminated BW	
	GAC (n.r.)	53 (6*)		Chloraminated BW	

n.r. – not reported

* - Inter Quartile Range

Despite the inconclusiveness (due to operational or experimental design factors) of most comparative studies of BOM removal by GAC and anthracite, these studies consistently demonstrate that relative to anthracite, BOM removal by GAC is more robust during non-

ideal operational conditions . GAC has been conclusively demonstrated to be more resistant to chlorinated backwashing (Krasner et al., 1993; Liu et al., 2001; Emelko et al., 2006), cold operating temperatures (Emelko et al., 2006; Liu et al., 2001), and the presence of oxidants in the influent (Niquette et al., 1998, Wobma et al., 2000). The resilience of GAC is a clear advantage over anthracite; however, the mechanistic explanation for this advantage is uncertain. Accordingly, it is possible that this relative advantage can be counteracted by the design process by making accommodations for a non-chlorinated backwash. For utilities retrofitting for biological filtration, the cost of conversion to non-chlorinated backwash would have to be balanced against the installation and maintenance costs associated with using GAC media rather than anthracite. The effect of chlorinated wash-water on biological filter performance is further discussed in section 2.6.3.

To summarize, the observed differences in BOM removal by GAC and anthracite are likely the result of GAC resilience to operational factors such as chlorinated backwash water or are confounded with experimental design factors such as un-matched media characteristics and residual adsorptive capacity of GAC media. Further work is required to address the associated knowledge gaps, which include:

- BDOC removal by various media types must be demonstrated and better understood at a variety of operating temperatures. BDOC removal in parallel anthracite and GAC filters was compared in only one study at warm water conditions (Huck et al., 2000); however, there is a lack of such data at cold operating conditions.
- AOC removal by various media types must be demonstrated and better understood at a variety of operating temperatures. There is a lack of conclusive comparative AOC removal data from studies comparing GAC and anthracite media performance. Differences in AOC removal by these media that have been reported in the literature are confounded with operational factors such as the use of chlorinated backwash, or experimental design factors such as un-matched media characteristics rather than the relationship between biological activity and media type.

- DBP/chlorine demand removal by various media types must be demonstrated and better understood at a variety of operating temperatures. There is a lack of data demonstrating the comparative DBP/chlorine demand removal performance of GAC and anthracite, particularly by parallel filters operated at cold water conditions.

2.5 Impact of Backwash

During the course of a biological filter cycle, both biological and non-biological particles accumulate within the filter media, eventually leading to terminal head loss or turbidity/particle breakthrough. To remove these accumulated solids it is necessary to periodically backwash the filters. Backwashing of biological filters entails reversing the flow of water through the filters to remove accumulated particles and biomass. Particle detachment occurs when either physico-chemical forces, hydrodynamic forces, or some combination of the two are sufficient to overcome the inter-particle and particle-media grain forces of attachment.

The micromechanics force model, developed and experimentally verified by Ahmad and Amirtharajah (1998) explores the differences between non-biological (clay, flocculent, etc) and biological particle forces of adhesion. It was found that the overall hydrophobic nature of the bacterial cell surface plays a significant role in bacterial attachment and demonstrated that bacteria are much more difficult to detach from GAC than more hydrophilic particles such as clay and floc particles. These findings suggest that there is may be an optimal backwash procedure for removing non-biological particles without excessive loss of biomass.

An optimized backwash procedure balances biomass support and particle removal, so as not to compromise traditional filter performance, while leaving enough biofilm on the media to not compromise biological performance at the start of the subsequent filter cycle. Insufficient backwashing can lead to excessive head loss build-up, mud ball formation, and media clumping over the long term course of filter operation (Amirtharajah, 1993). Important

backwash considerations include minimization of filter ripening, sufficient particle removal without compromising BOM removal, and the impact of chlorine in the backwash water.

2.5.1 Collapsed Pulse Backwashing

It has been established that backwashing with water alone is an ineffective process for both non-biological (Amirtharjah, 1978) and biological filters (Hozalski and Bouwer, 1998). A combination of air and subfluidization wash water to generate collapsed pulse conditions has been found to generate the best particle removal in non-biological filters, by maximizing inter-particle scouring for solids removal (Amirtharjah, 1993). The effects of collapsed pulse on biological filter performance however, are not well understood. Several studies have indicated that collapsed pulse backwashing does not impact biological filter performance (Ahmad and Amirtharajah, 1998; Emelko et al., 2006; Servais et al., 1991). Other studies however, have indicated a possible negative effect of collapsed pulse (Liu and Huck, 2001). Servais et al., (1991) found that C^{14} glucose removal in GAC filters was not impaired by a non-chlorinated air scour backwash, anthracite filters were not studied. C^{14} is not an optimal performance measure for determining operational impacts on a treatment process, however, because even highly stressed microbial communities are able to oxidize glucose. Carboxylic acid and aldehyde removal has been shown to be impaired in anthracite filters, but only at low temperatures (Liu and Huck, 2001). In contrast, Ahmad and Amirtharajah (1998) found that both NPOC and AOC removal in anthracite filters were not significantly impacted by collapsed pulse. These findings are also in agreement with a full-scale study, which found TOC removal in GAC and anthracite filters to be unaffected by collapsed pulse backwashing (Emelko et al., 2006). TOC removal was not impaired in the full-scale study of Emelko et al. (2006), but the implementation of collapsed pulse led to a significant decrease in full run time due to turbidity breakthrough, an effect which warrants further investigation.

The lack of information regarding the impact of collapsed pulse backwashing on biological filters represents a significant knowledge gap. Collapsed pulsing has been shown to be the most effect method for prevention of long term performance degradation resulting from preferential flow pathways, media clumping, and mudball formation in conventional

(predominantly non-biological) filters (Amirtharajah, 1978). The possible interaction between collapsed pulse and chlorinated wash water also has yet to be studied. It is possible that the enhanced scouring associated with collapsed pulse backwashing could enhance chlorine penetration by stripping away the negatively charged outer layers of the biofilm.

2.5.2 Extended Terminal Subfluidization Wash (ETSW)

The filter ripening sequence (FRS) describes the period of increased particle passage through a filter immediately following backwash. It has been estimated that up to 90% of particle passage in a well operated filter occurs during the FRS (Amirtharajah, 1988). This presents a significant operating challenge for drinking water utilities, regardless of the ability to filter-to-waste. When filter-to-waste is available this can represent significant losses in production, due not only to the loss of large quantities of coagulated, settled, and disinfected water, but also to filter out of service time. When filter-to-waste is not available, the FRS represents a period of potential pathogen passage. The filter ripening sequence can be divided into five stages, which are summarized in figure 2.4 and table 2.10 (Amburgey et al., 2003).

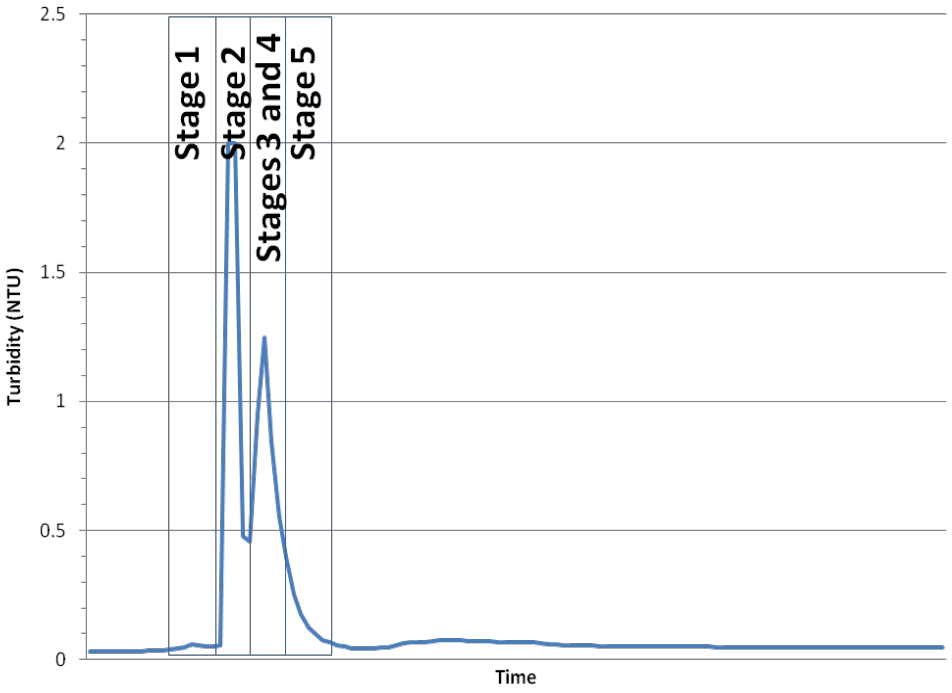

Figure 2.4: A filter ripening spike in a pilot-scale GAC filter at the Mannheim Water Treatment Plant

Table 2.10: Stages of the filter ripening sequence (adapted from Amburgey et al., 2003)

1. Lag phase	• Clean water remaining in the underdrain of a filter results in an initially low turbidity.
2. Media disturbance and intra-media remnant Stage	• Particles dislodged from the media and remaining in pore water pass through the filter, increasing turbidity. • It is possible that the settling bed also dislodges further particles.
3. Upper filter remnant stage	• Particles removed from the media during backwash, but remaining in the water above the filter media pass through the filter.
4. Influent mixing and particle stabilization stage	• Influent water enters the filter and mixes with the backwash remnant water in the upper region of the filter. • Although stages three and four are two separate processes, they occur simultaneously and there is significant intermixing.
5. Dispersed remnant and filter media conditioning stage	• Newly attached particles become collectors of other particles within the filter and improve filtration performance. • In actual treatment scenarios this stage may be almost unnoticeable; media is typically already coated with a significant number of particles despite backwash.

The increased ability of backwash remnant particles to pass through the filter media is due to particles reverting towards their raw water zeta potential as the backwash progresses (Amburgey and Amirtharajah, 2005). The mechanism of this process is currently not well understood, but it has been speculated to be due to partitioning of soluble NOM passing through the filter bed onto retained particles, aging of metal hydroxide floc particles, and/or new biological growth (Amburgey and Amirtharajah, 2005).

The FRS is caused by the chemistry of the backwash remnants and can be highly variable in resultant maximum turbidity levels and duration. As a result, the FRS is influenced by influent water characteristics, backwash water chemistry, backwash procedural adjustments, coagulation efficiency, coagulant type, raw water conditions, and other factorss (Amburgey and Amirtharajah, 2005). A backwash step, called the extended terminal subfluidization wash (ETSW), that targets the removal of backwash remnant particles has shown promise in significantly shortening, and in some cases eliminating the FRS under certain conditions (Amburgey et al., 2003).

ETSW is a period of backwash flow at sub-fluidization velocity of sufficient duration to displace one volume of water from within and above the filter bed and not detach a significant number of additional particles from the filter media (Amburgey et al., 2003). The goal of ETSW is to remove the majority of the filter remnant particles detached during backwash so as to significantly reduce the duration and severity of the FRS. This seemingly minor additional step has been shown to significantly reduce or even eliminate the FRS in both biological and conventional filters, even to a point where effluent turbidity following backwash is ≤ 0.05 NTU and increases to (pseudo) steady-state effluent turbidity as the filter run progresses (Amburgey, 2005). Although it seems counterintuitive to remove particles that could act as additional collectors and therefore decrease filter ripening, in most drinking water filtration situations there are a significant number of collectors remaining on the media after backwash. Accordingly, the majority of the FRS is due to the passage of backwash remnant particles (Amburgey, 2005). Although in some cases the ETSW would increase the quantity of water necessary for backwash, this is balanced by the potential savings of lost water due to prolonged filter-to-waste periods, or the increase in effluent quality at locations that are unable to filter-to-waste.

The success of ETSW depends in part on flow rate. A flow rate that is too high will generate shear forces sufficient to remove more particles from the media and a flow rate that is too low will not remove a sufficient number of remnant particles (Amburgey, 2005). The optimal flow rate is dependent on temperature, coagulation conditions, and size and type of media, and would therefore have to be optimized for each specific application (Amburgey, 2005). This optimal backwash water flow rate for ETSW implementation is a variable in water treatment scenarios. Changing temperatures results in changes in water viscosity and therefore the minimum fluidization velocity (V_{mf}). As a result, ETSW flow rates have to be adjusted seasonally (Amburgey, 2005). To provide the shortest period of degraded backwash water quality, it has been shown that the V_{mf} calculated for the d_{10} sized grains and the coldest annual water temperature should be used and then flow should be incrementally increased post backwash turbidity begins to increase (Amburgey, 2005).

The ETSW backwash has been demonstrated to be capable of significantly reducing, and often eliminating filter ripening, thereby presenting an opportunity for substantially reduced particle passage or wasted production quality water at locations that filter-to-waste. It is therefore important and potentially quite useful to understand the impact of ETSW on biological filter performance. Intuitively, this gentle wash period at the end of a backwash procedure should not have any effect on biofilm. When chlorinated wash water is used, however, this represents an increase in chlorine contact time with the biofilm, and a potential for reduced biological performance. The effect of this extended contact time immediately following the vigorous scouring of collapsed pulsing must also be studied.

2.5.3 Chlorinated Wash Water

In many drinking water treatment plants, particularly those that have been "retrofitted" with biological filtration, chlorine is often present in backwash water. In a pilot study of two anthracite/sand biological filters operated in parallel with one backwashed with a free chlorine residual of 1.0 mg/L and the other with non-chlorinated water, chlorinated backwash water was found to significantly impair mean OBP removal; particularly glyoxal, which was 97% removed in the non-chlorinated filter and 21% removed in the backwash chlorinated filter (Miltner et al., 1995). Mean AOC removal was also impaired, dropping from 53% to 33%. Interestingly, mean BDOC removal was slightly higher (25 %) in filter with chlorinated backwash, compared to 18% in the non-chlorinated filter. Filter run times in this study ranged from 80 to 100 hours. In a similar pilot study comparing the performance of a filter with chlorinated backwash (1.0 mg/L free chlorine residual) to a parallel non-chlorinated anthracite/sand biofilter, effluent AOC concentrations were approximately 88 and 50 µg/L respectively (Ahmad et al., 1998). Although AOC removal was impaired in this study, it is important to note that the filter with chlorinated backwash was only chlorinated during the shorter collapsed pulse step and then washed with non-chlorinated water. In most cases the water wash step of backwash would be performed with the same water used in the air scour step, and therefore the filters would be exposed to chlorine for even longer periods of time, with potentially more deleterious impacts on AOC removal.

As was noted above in section 2.4, chlorinated backwash water appears to have a more significant impact on BOM removal in anthracite filters than on GAC filters. Liu et al. (2001) further examined this phenomenon in a bench scale factorial investigation of the impact of chlorinated backwash water, filter media, and temperature on aldehyde and carboxylic acid removal removal by biological filtration. The studied compounds were formaldehyde, glyoxal, acetate, and formate; in all cases, the percentage of target compounds removed in the anthracite filter was significantly lowered by the presence of 0.5 mg/L chlorine residual in the backwash water and a water temperature of 5^{o}C. Except for glyoxal, the GAC filter did not show significant impairment in target compound removal at either warm or cold temperatures. This outcome is consistent other investigations of parallel comparison of anthracite and GAC that demonstrated that chlorinated backwash only significantly impaired BOM removal in anthracite filters only, particularly at cold water conditions (Emelko et al., 2006; Krasner et al., 1993; Wang et al., 1995). With the exception of Emelko et al. (2006); however, many of these studies were carried out at bench or pilot-scale. There is some evidence that both GAC and anthracite performance is more robust to chlorinated backwash at full-scale, anthracite is still deleteriously impacted (Huck et al., 1998). A full-scale investigation at a demonstration plant in California revealed no significant differences in BOM removal measured by BDOC, AOC, aldehyde, and DBP removal in anthracite and GAC filters, regardless of the presence of free chlorine in the backwash water. It should be underscored that this study was carried out at warm water conditions only and the negative impact of chlorinated backwash on BOM removal is expected to be most significant at cold water conditions (Liu et al., 2001).

The mechanism that gives GAC biofilters an increased resistance to chlorinated backwash is not well understood. GAC can remove chlorine and chloramines by catalytic reduction to non-oxidative chloride. While chlorine removal by new GAC is very fast, chloramines removal is much slower. Moreover, the catalytic (and adsorptive) capacity of GAC is exhausted over time and varies with operational conditions such as raw water quality and temperature, contact time, etc. It was postulated by Liu et al. (2001) that GAC reacting

with chlorine could potentially protect the biofilm from exposure to free chlorine. Snoeyink et al. (1981) found that free chlorine could react with virgin GAC when the chlorine was at a concentration of 2.5 g chlorine/g GAC; but concluded that in a drinking water treatment plant scenario there would be no significant reactions. It is possible though, for free chlorine to react with adsorbed organics such as humic acids (McReary and Snoeyink, 1981) and phenolic acids (McReary, 1982). Both of these studies where however, carried out at free chlorine concentrations (10 mg/L and 15 mg/L respectively) that are unlikely in drinking water treatment situations. Regardless, it is possible that these types of reactions occur at lower concentrations of chlorine and could be responsible for the increased resistance of GAC biofilters to backwash chlorination.

Overall, the available data indicate that backwash chlorination tends to have a negative impact on BOM removal, particularly in anthracite filters and at low water temperatures (Emelko et al., 2006, Liu et al., 2001). Based on this information, it is advisable to avoid the presence of chlorine in the backwash water wherever possible. Despite relatively extensive study in this area, several knowledge gaps continue to exist, particularly in regard to the interaction between chlorine and backwash techniques optimized for traditional filtration such as collapsed pulse and ETSW.

2.6 Research Needs

An examination of the literature related to the optimization of biological filtration revealed several knowledge gaps that need to be addressed to optimize future efforts in the design and operation of biological filters, these include:

Media Selection

- No published data exist regarding the impact of media size on biological filtration. Media size is a critical parameter in optimized performance of traditional filters, but the impact of decreasing contact time associated with increased media size is currently unknown. If the key mechanism is increased contact time, it can be achieved by a variety of approaches.

- The impact of uniformity coefficient on biological filter performance is also unknown. Higher uniformity coefficients result in stratification of the filter media such that smaller grains accumulate at the top of filter, resulting in increased head loss. A possible antagonistic effect of this accumulation and biological growth on filter head loss accumulation has not been investigated.
- BDOC represents the full extent of biologically mediated BOM removal in biological filters; yet, BDOC removal in parallel GAC and anthracite filters has been rarely at warm water conditions (Huck et al., 2000) and even less so at cold water conditions. Accordingly, further examination of BOM removal by biological filtration as measured by BDOC removal is warranted.
- Differences in the BOM removal performance of GAC and anthracite filters have yet to be conclusively demonstrated. In each case where a difference in AOC removal between GAC and anthracite filters has been reported, the conclusion has been confounded with operational factors such as chlorinated backwash, or experimental design factors such as un-matched media size and size distribution characteristics (table 2.5). Although GAC has shown an ability to remove certain aldehydes more efficiently, the operational significance of this ability is unknown.

Backwash

- The impact of collapsed pulse backwashing on filter BDOC removal has not been studied. Collapsed pulse has been demonstrated to be the most effective method for prevention of long term performance degradation due to mud-ball formation and media clumping in conventional filters. Further study of the impact of collapsed pulse on BOM removal is necessary before it can be recommended for biological filtration systems.
- ETSW is a backwash technique optimized for traditional filtration. Its impact on biological filtration; however, has yet to be studied. Non-chlorinated ETSW is not expected to impact biological filter performance; however, the impact of extended

contact time with chlorinated wash water during ETSW must be studied prior to implementing ETSW in biological filtration systems.

- The deleterious impact of chlorine on the BOM removal performance of biologically active anthracite filters has been repeatedly demonstrated. The interaction effect of chlorine with collapsed pulse or ETSW on biological filter performance has yet to be investigated, however.

Chapter 3- Materials and Methods

3.1 Research Approach

To address the research needs identified above, biological filtration was studied at full- and pilot-scales at the Mannheim Water Treatment Plant (MWTP) in Kitchener, Ontario, Canada. The overarching goal of the pilot-scale and full-scale work was to determine the impact of media characteristics, backwash techniques, and the interaction between the two on the concurrent BOM removal and traditional performance of the filters. The full-scale study focused on quantifying and comparing the performance of the four full-scale biological filter media configurations. The pilot-scale study enabled operational factors to be varied significantly, without impacting potable water production. Detailed descriptions of both the full- and pilot-scale experimental designs can be found in section 3.4 and 3.5 respectively. The knowledge gaps identified in section 2.6 and the scale at which they are addressed by this research approach are summarized in table 3.1.

Table 3.1: Knowledge gaps addressed by pilot and full-scale experiments

Identified Knowledge Gaps	Full-scale	Pilot-scale
Impact of media type on BOM removal	✔	✔
Impact of media type on traditional performance	✔	✔
Impact of media size on BOM removal	✔	✔
Impact of media size on traditional performance	✔	✔
Impact of media uniformity on BOM removal		✔
Impact of media uniformity		✔

Identified Knowledge Gaps	Full-scale	Pilot-scale
on traditional performance		
Impact of backwash on BOM removal	✔	✔
Impact of collapsed pulse on BOM removal		✔
Impact of collapsed pulse on traditional performance		✔
Impact of ETSW on BOM removal and traditional performance		✔
Impact of interaction between chlorine, CP, and ETSW on BOM removal		✔
Impact of interaction between chlorine, CP, and ETSW on traditional performance		✔
Interaction effects between media type and backwash strategy on BOM removal		✔
Interaction effects between media type and backwash strategy on traditional performance		✔

3.2 Assessing biological performance of filters

Careful consideration was given to the selection of the biological performance criteria to be measured during this study. Both biomass- and molecular-based methods were

considered for assessing the activity of the biofilm on each of the media types. Regardless of the goals of a biological filtration system, the success of the system will always depend on the ability of filters to remove BOM. Given the uncertainty and difficulty in determining seasonal conversion factors and the added complexity of achieving representative sampling, it was decided that the ATP method was not appropriate for the goals of this project. Additionally, the water being tested in this study was ozonated and would therefore contain artificially high ATP concentrations (Hammes and Egli, 2008).

The phospholipid biomass method was selected for use at full-scale. Due to analytical time constraints, it was not used during the pilot-scale experiments. Given that the purpose of a biological filter is to reduce the concentration of BOM present in the water and the well-documented lack of correlation between phospholipid biomass and BOM removal, the value of phospholipid biomass measurements for informing biological filtration research is limited. Although this method is not suitable as a stand alone method for evaluating filter performance, it was used at full-scale in conjunction with BRP and BDOC analyses during sampling phase 2 to generate a comprehensive picture of the impact of media type, media size, and backwash on the quantity, state, and performance of the biofilm micro-organisms in each filter.

Ideally, both AOC and BDOC would have been quantified to achieve the goals of this study. Time, equipment, and labor limitations allowed for only one parameter to be assessed. For this reason, BDOC was selected because it is a more comprehensive measure of BOM removal. BDOC represents the full extent of biologically mediated DOC removal, and therefore encompasses biological stability, chlorine demand, and DBP pre-cursor removals; it is therefore a more practical parameter for meeting the goals of this study.

The $BDOC_{fixed}$ method overcomes several of the limitations of the $BDOC_{susp}$ method by providing a reliable estimate of BDOC regardless of water type and at the time scale of days rather than weeks. The requirements of extensive pretreatment and ready access to BAS, however, render this method sub-optimal for the purposes of this project. The $BDOC_{cfbr}$ method was therefore selected to quantify BDOC at both pilot- and full-scale. The

water types tested included biological filter effluent and ozonated filter influent, neither of which contained disinfectant residuals. Additionally, ample time was available for adequate column colonization. The columns were exposed to water below 5^{o}C over the winter months, but with a year of colonization it was unlikely to have a significant impact on CFBR performance (MacLean et al., 1996a, Kaplan and Newbold, 1995). Given that the goal of the experiment was to assess biological performance, rather than to predict distribution system regrowth potential, BDOC was chosen over AOC as a measure of BOM.

3.3 The Mannheim Water Treatment Plant

All pilot and full-scale sampling and experimenting occurred on site at the Mannheim Water Treatment Plant. The Mannheim Water Treatment Plant in Kitchener, Ontario is a 16 MGD water treatment utility that treats surface water from the Grand River and delivers treated drinking water to several communities within the Region of Waterloo. Raw water from the Grand River is stored in a 38 million gallon reservoir before entering the plant, at which point it is diverted into two similar treatment trains. Both treatment trains contain the same processes, as outlined in figure 3.1, but coagulation on side 1 is optimized for a smaller filter media, whereas it is optimized for a larger filter media on side 2. The separate coagulation regimes for side 1 (predominant influent to filters 1 and 2) and side 2 (predominant influent to filters 3 and 4) limits the extent to which comparisons can be made of filters 1 and 2 to filters 3 and 4. In contrast, during this study, all pilot plant filters received influent water from side 2.

The MWTP employs poly-aluminium chloride (PACl) is used. This is a potentially significant factor given that some evidence has indicated that PACl may inhibit biological growth, and could therefore influence metabolic activity that is quantified by the BRP and BDOC methods (Springthorpe et al., 2009; Perez et al., 2010). All pilot-scale experiments were carried out at cold water conditions with PACl addition.

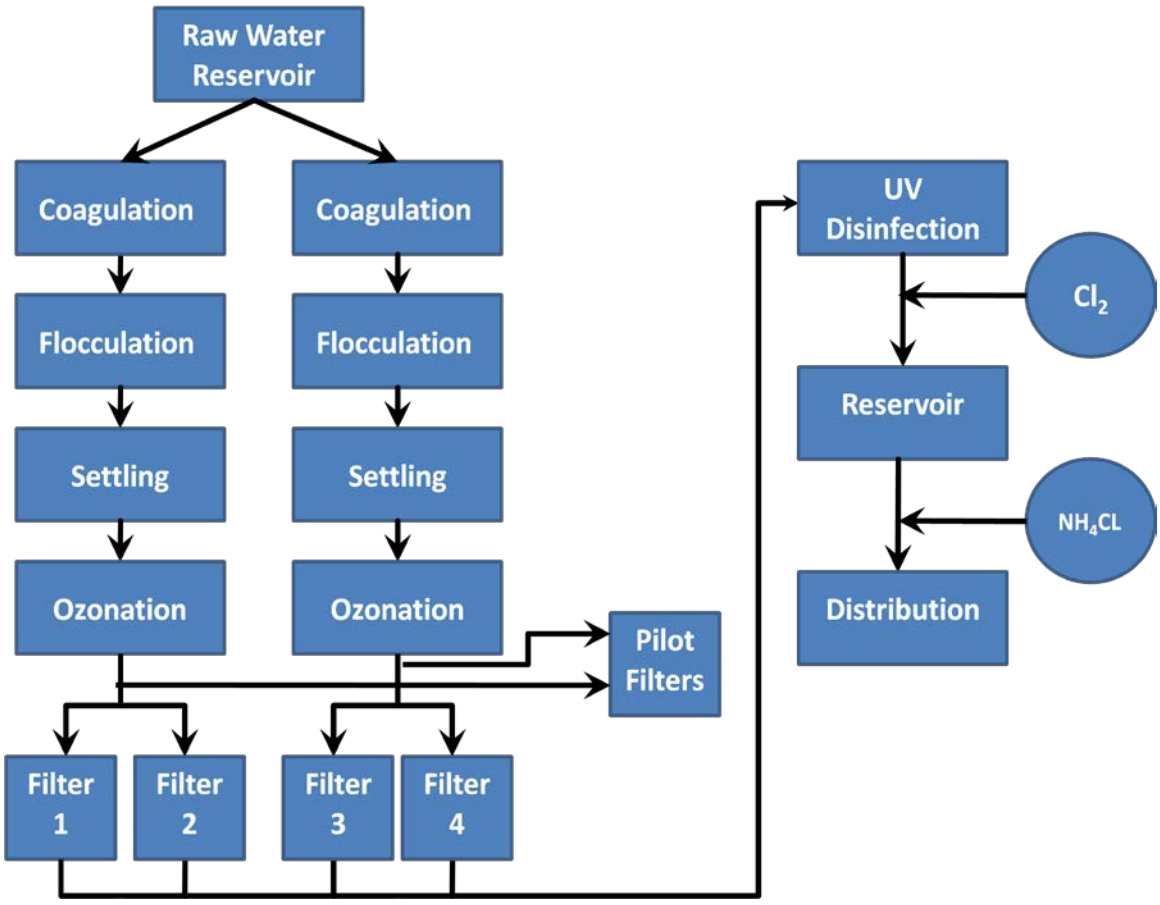

Figure 3.1: Mannheim Water Treatment Plant process schematic

After coagulation, flocculation, sedimentation, and ozonation, water is filtered through four single stage, granular media biological filters. The design loading rate for these filters is 11.2 m/h, and typical loading rates range from 7 to 10 m/h. On March 2nd 2007, the small GAC media present in full-scale filter 3 was replaced with a larger, deeper bed of GAC media, and on April 28th 2007 the small GAC present in full-scale filter 4 was replaced with an anthracite media similar to the large GAC filter media. The purpose of this media change out was to evaluate both media configurations to select a new larger deep bed media to be put into service in all four filters. The media specifications for each filter are presented in table 3.2.

Table 3.2: Full-scale filter media configurations. All media configurations are over a 300 mm layer of sand, for a total bed depth of 1.6m.

	Filter 1	Filter 2	Filter 3	Filter 4
Media Type	GAC	GAC	GAC	ANTH
E.S. (mm) (d_{10})	1.06	1.05	1.3	1.3
U.C. (d_{60}/d_{10})	1.48	1.50	1.28	1.3
Depth (mm)	1300	1300	1300	1300

3.4 Pilot-scale Experiments

The pilot filters at the MWTP permitted the freedom to vary backwash parameters significantly without impacting full-scale production water quality. A series of factorial experiments were conducted at pilot-scale to assess the impact of backwash parameters, media characteristics, and the interaction between the two on both traditional and biological filter performance. In keeping with the previously stated need to take a holistic approach to the design and operation of biological filters, both traditional and biological parameters were quantified simultaneously in all experiments.

3.4.1 Mannheim Water Treatment Plant Pilot Filters

The media configurations used during the pilot experiments are outlined in table 3.3.

Table 3.3: Pilot filter media configurations

	Filter 1	Filter 2	Filter 3	Filter 4
Media Type	GAC	GAC	ANTH	ANTH
E.S. (mm) (d_{10})	1.06	1.46	1.3	1.3
U.C. (d_{60}/d_{10})	1.48	1.27	1.6	1.3
Depth (mm)	1300	1300	1300	1300
Full-scale Analog (Filter #)	1/2	3	n/a	4

The filter influent water is full-scale, post-ozonated water, with valves that allow water from either side 1 or side 2 to be used individually or simultaneously. Figure 3.3 is a simplified schematic of the MWTP pilot filters.

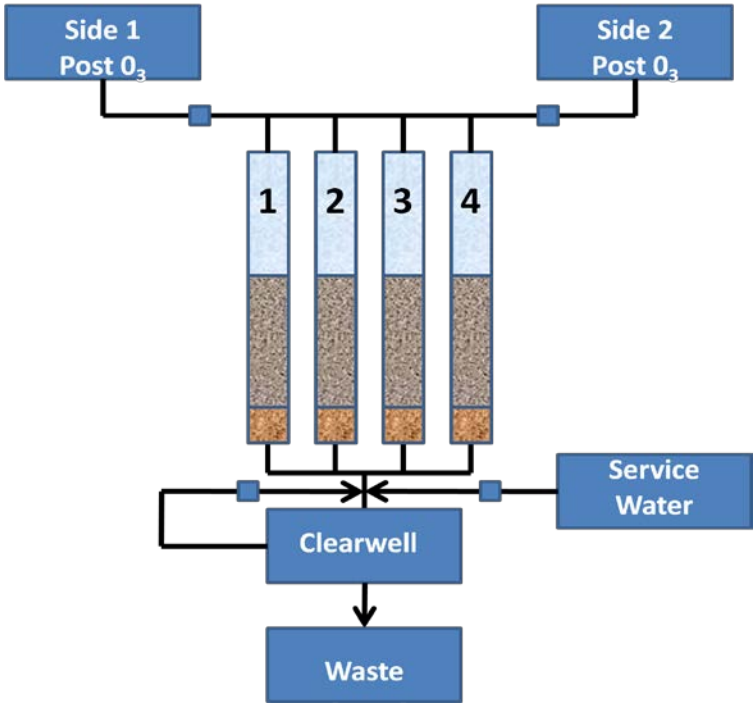

Figure 3.2: Simplified pilot plant schematic

During the pilot-scale experiments reported herein, filter influent water was drawn exclusive from side 2 of the MWTP because side 2 chemical pretreatment was optimized for the larger filter media that were the focus of this study. Figure 3.4 presents a schematic of an individual pilot filter. Filter effluent flowed to a clearwell, and then to the plant waste stream. The filters could be backwashed with chlorinated service water or non-chlorinated water from the clearwell. Effluent flow was controlled by automated flow meters. Individual effluent turbidities were monitored using Hach sc100^{TM} 1720E low range turbidity meters (Hach Company, Loveland, CO). Filter head loss, as well as effluent turbidity and flow rate were monitored automatically by the MWTP's SCADA system. Each of the four pilot filters contained a different media configuration, which allowed for the assessment of the impact of media size, uniformity coefficient, and type on filter performance.

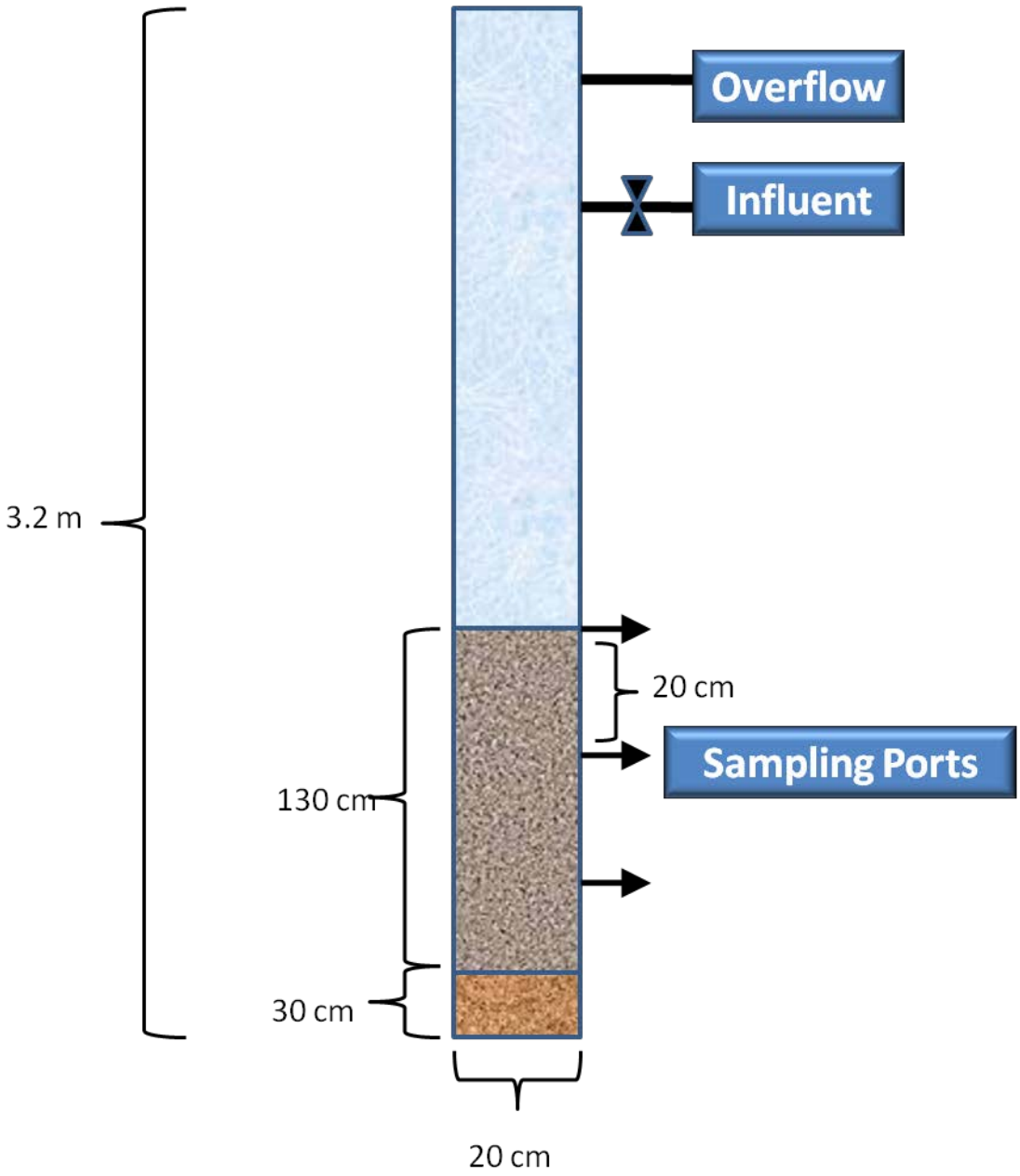

Figure 3.3: Pilot filter schematic

3.4.2 Backwash Design

The baseline backwash used for the factorial experiments was matched to the MWTP baseline backwash. This backwash consisted of air scour, followed by a high rate wash, followed by a low rate wash. The specific flow rates for each filter are presented in table 3.4.

Table 3.4: Baseline MWTP backwash

	Filter 1		Filter 2		Filter 3		Filter 4	
	Duration (min)	**Rate (m/h)**	**Duration (min)**	**Rate (m/h)**	**Duration (min)**	**Rate (m/h)**	**Duration (min)**	**Rate (m/h)**
Air Scour (scfm/ft^2)	5	3	5	3	5	3	5	3
Media Settling	Based on visual inspection							
Low Wash	5	13.0	5	13.0	5	13.0	5	13.0
High Wash	8	28.2	8	28.2	8	39.1	8	39.1

The collapsed pulse backwash was calculated according to (Amirtharajah, 1991). The specific flow rates for each filter are presented in table 3.5. Because of the translucent material the pilot filters were constructed from, it was possible to observe each backwash. As such no set duration was employed, the collapsed pulse backwash was carried out until the water level reached approximately 15cm below the filter influent. The media was then allowed to settle. Adequate settling was determined visually.

Table 3.5: Collapsed Pulse backwash rates

	Filter 1		Filter 2		Filter 3		Filter 4	
	Duration (min)	**Rate**	**Duration (min)**	**Rate**	**Duration (min)**	**Rate**	**Duration (min)**	**Rate**
Q_w (m/h)	n/a	10.6	n/a	11.1	n/a	17.8	n/a	13.6
Q_{air} (scfm/ft^2)	n/a	3	n/a	3	n/a	3	n/a	3

Scfm – standard cubic feet per minute

The ETSW backwash was designed according to (Amburgey et al., 2003). The specific flow rates for each filter are presented in table 3.6. Ideally, when designing an ETSW backwash, the flow is calculated based on the V_{mf} for the d_{10} particle size, and then flow is incrementally increased until performance begins to degrade. In this case, to ensure that any potential performance effects were detectable for each backwash, the most conservative case (ie. V_{mf} for d_{10}) was used. When used in combination with collapsed pulse, collapsed pulse was followed by a settling period, a high rate wash, a second settling period, and then ETSW.

Table 3.6: ETSW backwash rates and duration

	Filter 1		Filter 2		Filter 3		Filter 4	
	Duration (min)	**Rate (m/h)**	**Duration (min)**	**Rate (m/h)**	**Duration (min)**	**Rate (m/h)**	**Duration (min)**	**Rate (m/h)**
ETSW	29.5	5.9	17.0	10.2	21.36	8.2	16.86	10.5

3.4.3 Experimental Design

A series of three factorial experiments were conducted at pilot-scale. Each experiment was based on a 2^4 design, with factors such as run time and sampling and filter bed depth added when biological response variables were used. The primary drawback to choosing to break the experiment into three experiments rather than a fractional 2^6 design was that this method does not allow for interactions between media characteristics. This was not a concern for uniformity coefficient, but it was unfortunate that the interaction between media size and media type was not quantified. The rationale for this design was simply that, without a small effective size anthracite filter, a fractional factorial would require needlessly complicated analysis to attain an uncertain estimate of the anthracite-size interaction based upon the GAC-size interaction. The study as designed allowed for the effect of media type, backwash strategy, and the interaction between the two to be quantified.

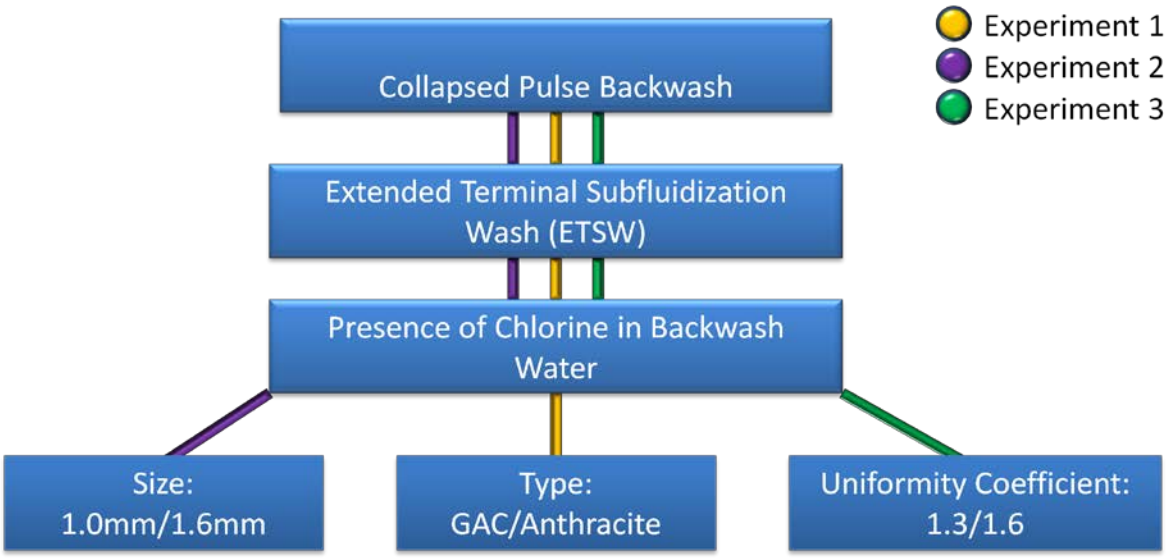

Figure 3.4: Basic design for pilot factorial experiments. Additional factors of time and filter bed depth were added for biological response variables DOC, BDOC, and BRP.

Because of the large number of filter cycles required to complete the factorial experiment series, the large sample volume associated with those experiments, and the challenge of conducting experiments at a full-scale water treatment plant with variable influent raw water quality, only a single replicate of each experiment was possible. Accordingly, the mean square for error was estimated by pooling non-significant high-order interactions, whose mean squares all have expectation σ^2. Prior to this, a normal probability plot of effects was examined to check for potential significance and to prevent an inflated estimate of error through the erroneous inclusion of significant effects (Daniel, 1959). Where possible, a second method for estimating error also was employed for comparison, which involves identifying a non-significant factor in a 2^k, removing this factor, and projecting the data onto a 2^{k-1} with replication in the remaining factors (Montgomery, 1984).

The BRP experiments provided the only opportunity for factorial data to be analyzed in two ways, thus providing two separately calculated error estimates for comparison. In each of the three experiments, no significant five or six factor interactions were detected, and in experiments two and three, the factor for media characteristic (E.S. and U.C. respectively)

was also found to not be significant. This allowed for error to be estimated both from the fifth and sixth order interactions, as well as by projecting the 2^6 experiment onto a 2^5 with replication in the remaining factors. The error estimates are presented in table 3.7 and are quite similar in both cases, providing confidence in the analysis.

3.7: Comparison of Mean Square Error Estimates

Experiment	4^{th} and 5^{th} level interactions	2^6 converted to 2^5 with replication
2	$4.7x10^{-4}$	$4.8x10^{-4}$
3	$2.2x10^{-3}$	$2.0x10^{-3}$

3.4.4 Study Conditions

Study conditions remained relatively constant throughout the course of the pilot experiments, which were conducted between February 9th and April 13th, 2010. The large range in temperature was due to an unexpected, but brief winter warm spell.

Table 3.8: Water conditions for pilot factorial experiments

Parameter	Mean and Range
Loading Rate	11.2 m/hour
Water Temperature	Mean: 2.60, Range: 0.72 – 11.02
Influent DOC	Mean: 4.58, Range: 3.59-5.50
pH	Mean: 7.99, Range: 7.35 – 8.23

3.4.5 DOC

Water for DOC analysis was sampled in 40 mL borosilicate glass EPA vials that were prepared according to standard methods (APHA, 1998) Vials were washed, rinsed with 10 N HCl, triple rinsed with Milli-Q^{TM} water, then baked at 500^{o}C for 1 hour. Vials were then rinsed three times with sample water prior to sample collection. Samples were immediately acidified to pH 2 using concentrated orthophosphoric acid. Samples were filtered through a 0.45µm Supor® and then analyzed along with appropriate bottle and filter blanks to ensure against contamination.

DOC was then analyzed using an OI analytical TOC 1010 analyzer (OI Analytical, Texas, USA), which employs the wet oxidation method. Due to operational issues with this TOC analyzer, pilot-scale samples from March 10th to April 14th 2010 were sent to the Earth Sciences lab at the University of Waterloo for analysis. This laboratory also employs an OI analytical TOC 1010 analyzer that utilizes the wet oxidation method.

DOC removal was one of the parameters chosen to assess biological filter performance. This was done by subtracting the filter effluent DOC concentration from the influent. DOC was chosen because this parameter reflects the pool of molecules that exert chlorine demand, act as nutrients in the distribution system, react with chlorine to form DBPs, and contribute to taste and odor issues. As discussed previously; DOC alone is not an ideal method to quantify biological filter performance, as only a small and highly variable percentage of DOC is biodegradable. The variability of , as well as abiotic adsorption effects

limit the utility of DOC removal as an indicator of biological activity. For this reason, BDOC was quantified as well.

Statistical analysis of the DOC factorial data was conducted based on the base 2^4 design outlined in figure 3.7, with time added as an additional factor for a full 2^5 design. Influent and effluent DOC was sampled immediately following backwash, allowing two hours to pass to account for residence time in the BDOC columns, and then again at twenty four hours to determine if filter DOC had improved or deteriorated during the course of the filter run. The design generators for this experiment are presented in table 3.9.

Table 3.9: 2^5 factorial using DOC as response variable

Factor	Code	+	-
Collapsed Pulse	A	On	Off
ETSW	B	On	Off
Chlorinated Wash Water	C	Absent	Present
Media Characteristic	D	GAC/Large E.S./Low U.C.	Anthracite/Small E.S./High U.C.
Time	E	24 hours	1 hour

3.4.6 BDOC

BDOC was determined using the column method. Columns were constructed and operated according to (Camper et al., 2000). No pre-filtration was used. A 140 µm stainless steel screen was used on the influent column lines to prevent excessive headloss build up due to floc carry over. This screen was thoroughly cleaned with a test tube brush and rinsed with influent water on a daily basis to prevent potential biofilm buildup that could alter influent DOC characteristics prior to water reaching the column. Water was collected from filter effluent sampling lines for filter effluent BDOC, and directly from the post O_3 sampling line for influent BDOC. In both cases the water then passed through inert Pharmed^{TM} tubing and stainless steel Swagelok fittings before arriving at the BDOC column. Columns were

operated at a flow rate of 3mL/min, for a residence time of approximately 2.5 hours, using the 60 x 2.5 cm column size of Kaplan and Newbold, (1995). BDOC was calculated as the difference between BDOC column influent and effluent DOC concentration in mg/L. BDOC removal, which was selected as a metric of BOM removal or biological filter performance, was calculated by subtracting the filter influent BDOC from the filter effluent BDOC.

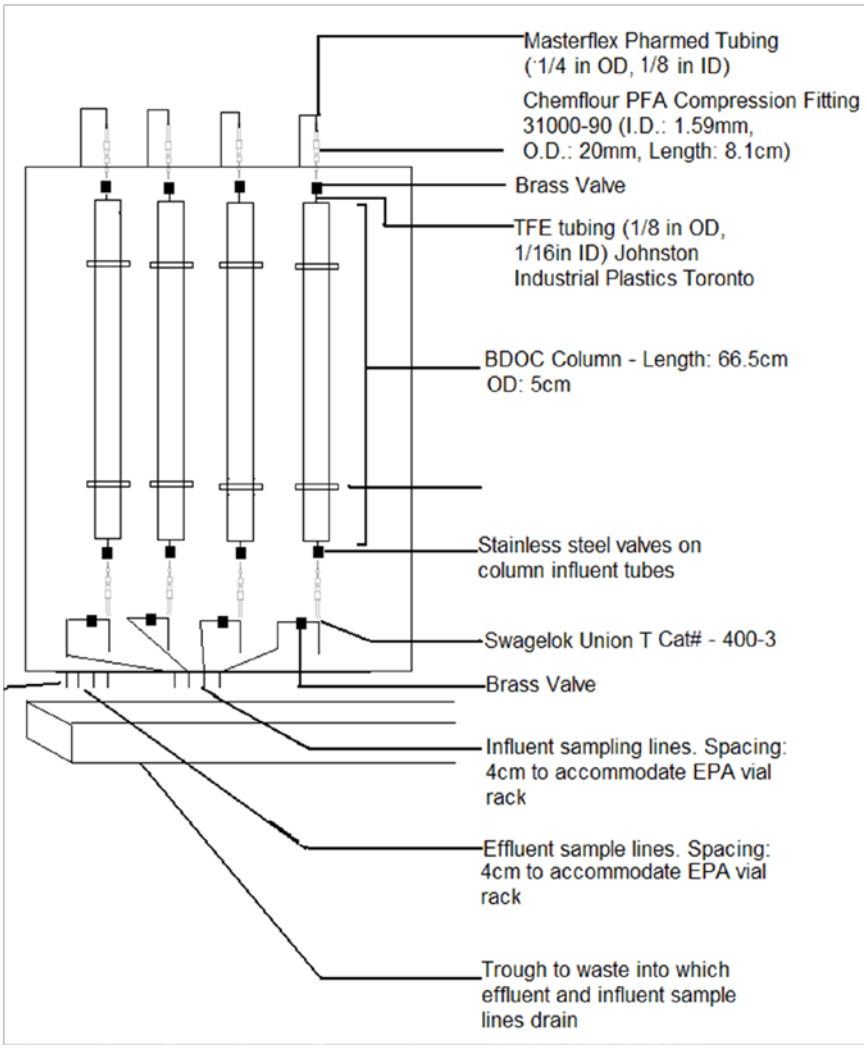

Figure 3.5: BDOC column set up

The following modifications to the (Camper et al., 2000) method for BDOC column set up were made:

- One column was used rather than two. The residence time was by 50% to approximately 2.5 hours. Given the large column size (60 x 2.5 cm), these changes were deemed adequate because previous work demonstrated that two columns were only necessary for chlorinated or raw waters (Ribas et al., 1991) and an 85% increase in contact time yielded only a 10% increase in BDOC removal (Kaplan et al., 1994).
- Column influent water was divided using a union-T fitting allowing continuous influent sample flow, eliminating the need to adjust flow for sampling purposes.
- Pilot-scale BDOC columns were fed by gravity and full-scale columns were fed by inline pressure, rather than a pump.

BDOC factorial results were statistically analyzed using the base 2^4 design outlined in figure 3.4, with time added as an additional factor for a full 2^5 design. Column influent and effluent DOC were sampled immediately following backwash, allowing two hours to pass to account for residence time in the BDOC columns, and then again after twenty four hours of run time to determine if BDOC removal had improved or deteriorated during the course of the filter run. The design generators for this experiment are presented in table 3.10.

Table 3.10: 2^5 factorial using BDOC as response variable

Factor	Code	+	-
Collapsed Pulse	A	On	Off
ETSW	B	On	Off
Chlorinated Wash Water	C	Absent	Present
Media Characteristic	D	GAC/Large E.S./Low U.C.	Anthracite/Small E.S./High U.C.
Time	E	24 Hours	1 hour

3.4.7 BRP

Biological respiration potential was measured according to the procedure outlined in (Urfer and Huck, 2001) and is summarized in figure 3.7. The only modification made to the method was that commercial mineral water was used as a source of micro-nutrients rather than dechlorinated tap water. Mineral water has previously been used in biological assays because it is well buffered and provides a source of all necessary micro-nutrients to support biological growth (Hammes and Egli, 2005). It was also used to provide a compositionally more consistent source of micronutrients than tap water.

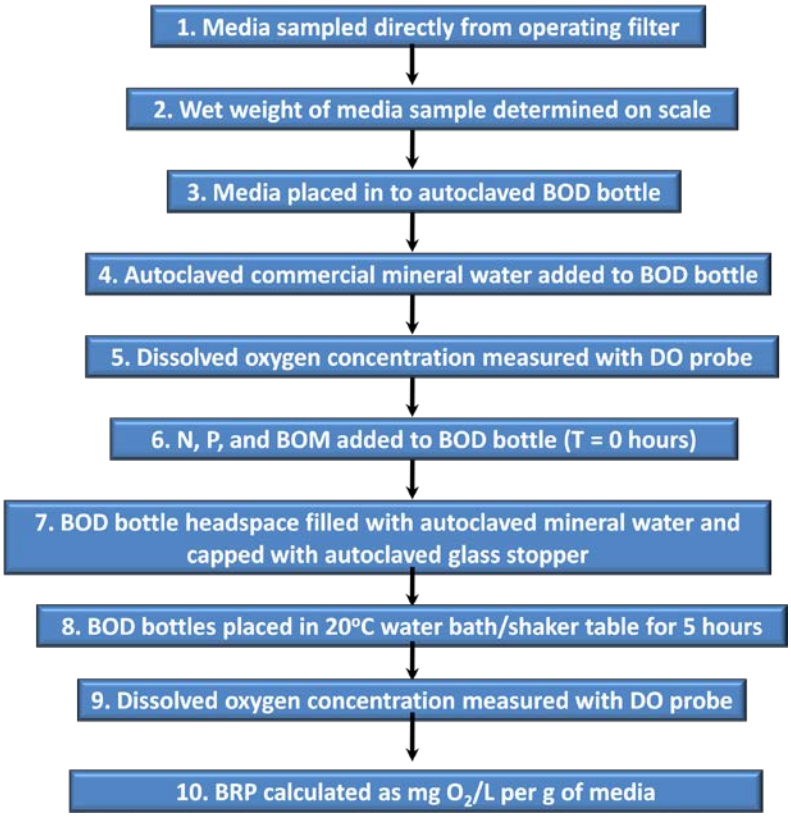

Figure 3.6: Procedure for determining BRP of filter media (modified from Urfer and Huck, 2001)

The method detectin limit (MDL) for this method was determined to be 0.070 mg O_2/cm^3 media for GAC and 0.064 mg O_2/cm^3 media for anthracite. The average measured BRP for autoclaved samples used for controls was 0.097 mg O_2/cm^3 media (s.d 0.019, n=5) and 0.069 mg O_2/cm^3 (s.d. 0.017, n=5) for GAC and anthracite respectively. Dissolved oxygen was measured using a DO probe (VWR symPHony, VWR, Radnor, Pennsylvania). The oxygen consumption of autoclaved GAC using this method has not previously been reported, but the values of 0.069 mg O_2/cm^3 measured for autoclaved anthracite in this study were higher than those reported by (Urfer and Huck, 2001), which were 0.007mg O_2/cm^3 (s.d. 0.004, n=2). This difference could be due to the larger number of control samples used in this study (five vs two); differences in the anthracite media surface area, volume, or supplier; or perhaps a less sensitive DO probe.

BRP factorial results were statistically analyzed using the base 2^4 design outlined in figure 3.4, with time and media depth added as additional factors for a full 26 design. BRP was sampled immediately following backwash from the surface of the filter, and from 20cm into the filter bed, and then again after twenty four hours of filter operation to determine if BRP had improved or deteriorated during the course of the filter run. The design generators for this experiment are presented in table 3.11.

Table 3.11: 2^6 factorial using BRP as response variable

Factor	Code	+	-
Collapsed Pulse	A	On	Off
ETSW	B	On	Off
Chlorinated Wash Water	C	Absent	Present
Media Characteristic (Exp 1/Exp 2/Exp 3)	D	GAC/Large E.S./Low U.C.	Anthracite/Small E.S./High U.C.
Time	E	24 Hours	1 hour
Bed Depth	F	Surface	20 cm

3.4.8 Filter Run Time

The beginning of the filter cycle was established as the moment effluent turbidity dropped below 0.2 ntu for the remainder of the filter run following post-backwash filter ripening. The filter cycle end point was based on one of the following three criteria being met: 1) head loss accumulated to the point that flow was reduced by 25% (this measure was used due to problems with pressure transducers used to estimate head loss), 2) turbidity exceeded 0.1ntu, or 3) filter operation reached 48 hours. The final criterion represents a study limitation, as this may lead to the underestimation of the run time that may be associated with a particularly effective backwash or media type. This restriction on filter run time was required to enable the completion the factorial experiments. The 48 hour filter run length was also generally consistent with the longer filter run times experienced at the full scale plant.

3.4.9 Filter Ripening Time

Pilot-scale filter ripening time was determined using a slightly modified version of the approach used to determine full-scale ripening time at the MWTP. At full-scale and at pilot-scale, the start point of filter ripening was the moment at which the filter effluent valve was opened following backwash. At full-scale, the ripening end point was after ten consecutive data points were below 0.2 ntu. Full scale turbidimeters sample turbidity once per minute. After this point, the filters go back in to service. At pilot-scale, however, the filters operate to waste. As a result, filter ripening was determined retroactively, which allowed for the exact point at which effluent turbidity dropped to consistently below 0.2ntu to be determined. Because turbidity at full scale is measured once per minute, this shortens pilot-scale filter ripening by ten minutes.

3.5 Full-scale Experiments

All four filters full-scale filters at the MWTP were evaluated. Biological performance was assessed using BDOC, DOC, and THM formation potential removal, as well as BRP and phospholipid biomass. Traditional performance assessment was based on filter run time and filter ripening time. This general approach allowed for an assessment of how the different media configurations perform at full-scale. Observed trends could then be compared to pilot-scale results, and pilot-scale results could be used to speculate on the mechanisms driving full-scale performance.

3.5.1 Full-scale Performance Data

The MWTP provided comprehensive filter operational data logged by a SCADA system that allowed for a comparison of the traditional performance of the full-scale filter media configurations. Operational data from the date large anthracite and large GAC filter media were installed (March 2nd, and April 28th 2007 respectively), to May 2010 were analyzed. Filter effluent turbidities at the MWTP are consistently below 0.1 NTU, as they were during all of the experimental periods at the MWTP. Accordingly, no discernible

differences between the filters' particle removal capacities were observed. For this reason, filter run time was selected as a key parameter for assessing full-scale performance. The drawback to this method is that this does not account for whether filters are being backwashed primarily due to headloss, turbidity breakthrough, or time. This is easily accounted for however, by assessing what factors trigger backwash. Backwash triggers for full-scale filters at the MWTP are presented in table 3.12.

Table 3.12: Full-scale backwash triggers at the MWTP

Backwash Trigger	Value
Turbidity	> 0.1ntu
Headloss	75%
Time	60 hours (max)

3.5.2 THM Formation Potential

A critical aspect of biological filtration is that in enables utilities to economically achieve effluent DBP concentrations below regulatory standards and guidelines. Because the MWTP employs chloramines to lower the potential formation of DBPs, effluent DBP concentrations are typically well below detection limits. Therefore, to draw comparisons between the ability of the different media to remove THM precursor molecules from production water, THM formation potential was quantified. Although THM concentrations are not currently an issue at the MWTP, the ability of a filter to lower DBP precursor concentrations is a critical aspect of filter performance, and could become vital if conditions change in the future.

All THM-FP samples were collected from the filter influent and effluent immediately following backwash and at 24 hours in to the filter cycle. The difference between influent and effluent was then used to assess the ability of the filters to remove THM precursors. Samples were collected in 1 L amber glass bottles which were pre-washed in acid and milli-

Q^{TM} water. Bottles were then sent to a commercial lab (SGS Lakefield, Lakefield ON) for analysis. Sample dates are presented in table 3.13.

Table 3.13: Sampling dates for THM-FP and chlorine demand analysis

Date	Filter Sampled
15/03/2010	3
16/03/2010	4,3
17/03/2010	4, 2
18/03/2010	2
22/03/2010	1
23/03/2010	1
06/04/2010	4
07/04/2010	4, 3
08/04/2010	3
13/04/2010	2
14/04/2010	4

3.5.3 DOC and BDOC

DOC and BDOC were sampled according to the same procedures previously outlined in sections 3.2.5 and 3.2.6. Two separate BDOC and DOC sampling events were carried out at full-scale. The first sampling event was carried out between October 2008 and May 2009. During this time period bi-monthly samples were collected from all filters to assess the performance differences between the media during a time period when cold water would likely maximize any BOM removal differences between anthracite and GAC. Sampling dates are listed in table 3.14.

Table 3.14: Sample dates for cold water full-scale BDOC performance analysis

Date	Filters Sampled
16/10/2008	1, 2, 3, 4
04/11/2008	1, 2, 3, 4
24/11/2008	1, 2, 3, 4
18/12/2008	1, 2, 3, 4
27/01/2009	1, 2, 3, 4
11/02/2009	1, 2, 3, 4
28/02/2009	1, 2, 3, 4
16/03/2009	1, 2, 3, 4
01/04/2009	1, 2, 3, 4
07/05/2009	1, 2, 3, 4

The second sampling event occurred during July 2009. The purpose of this sampling period was to generate a comprehensive picture of the quantity, state, and activity of biomass on each filter media configuration both before and after backwash. To achieve these goals, BDOC, phospholipid biomass, and BRP were all assayed simultaneously immediately following, and immediately prior to filter backwashing. Sampling dates are listed in table 3.15.

Table 3.15: Sample dates for BDOC, DOC, phospholipid biomass, and BRP

Date	Filter Sampled
03/06/2009	1
09/06/2009	2
18/06/2009	3,4

3.5.4 Phospholipid Biomass

Phospholipid biomass was assayed according to the procedure decribed by Findlay et al., (1989). Samples were concurrently with BDOC and DOC samples on the dates presented in table 3.13. To sample phospholipid biomass, media were collected from the top of the filters. A calibration curve using inorganic phosphate standards ranging from 5 nmol to 40 nmol was prepared. The procedure followed for the phospholipid assay is outlined below:

Extraction

1. Transferred approximately 0.05-0.1 g of media to a 20mL EPA vial.

↓

2. Add 1.8mL of Mill-Q^{TM} water, 5 mL methanol, and 2.5 mL of chloroform in order.

↓

3. Mix at low speed on shaker for 10 minutes, let stand overnight for extraction.

↓

4. Add 2.5mL chloroform and 2.5 mL Milli-Q^{TM} water in order, let stand for approximately 30 minutes, until phase separation has occurred.

↓

5. Remove and discard upper layer (MeOH-H2O) with Pasteur pipette.

↓

6. Transfer lower layer (chloroform) to COD vial with Pasteur pipette.

↓

7. Remove solvent (chloroform) under a stream of nitrogen.

↓

Digestion

8. Add 1.1mL potassium persulfate solution (5% potassium persulfate in 0.36 N sulfuric acid)

↓

9. Close vial tightly and digest at 95-100^{o} C overnight on a heating plate.

↓

Quantification

10. Let cool, then add 0.2mL ammonium molybdate solution (2.5% $(NH_4)_6Mo_7O_{24}$-4H2O in 5.72 N sulfuric acid), wait 10 minutes

↓

11. Add 0.9 mL malachite green solution (0.011% malachite green in 0.111% polyvinyl alcohol solution), wait 30 minutes.

↓

12. Convert to nmole of lipid phosphate using a standard curve established using inorganic phosphate (K_2HPO_4)

3.5.5 BRP

BRP was sampled using the same sampling, analytical, and quality control procedures outlined in section 3.2.7. BRP samples were collected concurrently with BDOC, DOC, and phospholipid biomass samples. The sampling dates correspond to those presented in table 3.13.

Chapter 4– Pilot-scale Experiments

The MWTP pilot filters were used to conduct three parallel factorial experiments that quantified five response variables: DOC removal, BDOC removal, biological respiration potential, filter run time, and filter ripening time. The goals of these experiments were to identify key design and operational factors impacting both biological and traditional filtration performance and to identify factors that optimize both. To do so, the impacts of backwash technique (chlorinated wash water, collapsed pulse, and ETSW), media characteristics (type, effective size, and uniformity coefficient), and combinations thereof were quantified in a realistic drinking water treatment plant environment. Specific areas of interest included:

- To determine the impact of media size on head loss accumulation, turbidity removal, and BOM removal.
- To investigate whether the accumulation of small grains at the top of filters with less uniform media interacts with biomass to increase the rate of head loss accumulation.
- To compare BDOC removal in parallel anthracite and GAC filters with matched media characteristics at cold water conditions using both chlorinated and non-chlorinated backwash.
- To determine the impact of collapsed pulse backwashing on head loss accumulation, turbidity removal, and BRP in both anthracite and GAC filters, particularly when chlorinated wash water is used.
- To assess the efficacy of ETSW in minimizing the filter ripening sequence, and to determine if the associated extended contact time with chlorine effects BDOC removal and BRP.

To achieve the stated experimental goals the results of the factorial experiments were analyzed statistically using ANOVA. In the ANOVA analysis the test statistic used for assessing the significance of treatments is the *F*-ratio. The probability of obtaining a test statistic at least as extreme as the one that was actually observed is the p value. Smaller p

values are associated with more extreme test statistics and can be generally considered as an indication that the effects are "more significant". The statistical significance level of the statistical test is indicated by the *p* value. As indicated in the footnote of each table in which ANOVA results are presented herein, the data are summarized so that the *F*-ratios that are significant at the 0.1% significance level (or "99.9% confidence level") are noted with the superscript "a". Subscripts "b", "c", and "d" denote 1%, 2.5%, and 5% significance levels (or "99%, 97.5%, and 95% confidence levels") respectively. The calculated effects of each factor (backwash or media characteristic) on filter performance (biological and traditional) from the factorial experiments were either positive or negative. A positive effect indicates that the factor or factor interaction had a positive effect on the response variable, for example: if factor D (media type) were to have a positive effect on DOC removal, it would indicate that DOC removal in GAC filters was higher than parallel anthracite filters. A negative effect indicates that the factor or factor interaction had a negative effect on the response variable, for example: if factor B (ETSW) were found to have a negative effect on filter ripening, it would indicate that filter ripening time was reduced when ETSW backwash was used compared to filter cycles when ETSW was not used.

4.1 Design and Operational Impacts on DOC Removal

The DOC data for all three experiments were analyzed using single replicates of a 2^5 factorial. Normal probability plots were used to check for significance of higher order interactions (Appendix A, figure A.13). In all three cases, fourth and fifth level interactions were not significant, and therefore these effects were used to estimate mean squared error. With this estimate, results were then analyzed using ANOVA. The results of the ANOVA analysis are presented in table 4.1.

Table 4.1: ANOVA table for factorial analysis of DOC removal by pilot-scale biological filtration

	Experiment 1			Experiment 2			Experiment 3		
Source	**D**	**MS**	**F_o**	**D**	**MS**	**F_o**	**D**	**MS**	**F_o**

	Experiment 1			Experiment 2			Experiment 3		
	F			F			F		
A	**1**	**0.488**	**25.948[b]**	1	0.030	0.628	**1**	**0.443**	**94.311[a]**
B	**1**	**1.284**	**68.359[a]**	1	0.216	4.535	**1**	**1.345**	**286.668[a]**
AB	**1**	**0.261**	**13.881[b]**	1	0.012	0.260	**1**	**0.296**	**62.979[a]**
C	**1**	**0.633**	**33.679[b]**	1	0.170	3.579	**1**	**0.424**	**90.418[a]**
AC	**1**	**1.539**	**81.912[a]**	**1**	**0.520**	**10.931[c]**	**1**	**1.726**	**367.755[a]**
BC	**1**	**0.294**	**15.645[b]**	1	0.001	0.029	**1**	**0.348**	**74.158[a]**
ABC	**1**	**0.822**	**43.751[a]**	1	0.181	3.807	**1**	**0.859**	**183.078[a]**
D	**1**	**0.208**	**11.052[c]**	1	0.275	5.774	1	0.003	0.545
AD	1	0.014	0.748	1	0.165	3.477	1	0.012	2.465
BD	1	0.029	1.540	1	0.249	5.228	1	0.010	2.114
ABD	1	0.005	0.292	1	0.106	2.227	1	0.007	1.542
CD	1	0.001	0.050	1	0.171	3.595	1	0.025	5.276
ACD	1	0.029	1.567	1	0.121	2.543	1	0.003	0.634
BCD	1	0.001	0.074	1	0.219	4.601	1	0.003	0.634
E	1	0.008	0.436	1	0.042	0.881	1	0.025	5.422
AE	1	0.081	4.317	1	0.034	0.717	**1**	**0.059**	**12.660[c]**
BE	1	0.001	0.027	1	0.020	0.416	1	0.000	0.033
ABE	1	0.048	2.573	1	0.154	3.230	1	0.005	1.052
CE	1	0.025	1.347	1	0.017	0.347	1	0.003	0.638
ACE	**1**	**0.394**	**20.974[b]**	1	0.129	2.722	**1**	**0.416**	**88.565[a]**
BCE	**1**	**0.407**	**21.641[b]**	**1**	**0.348**	**7.305[d]**	**1**	**0.223**	**47.604[a]**
DE	1	0.002	0.098	1	0.025	0.519	1	0.000	0.068
ADE	1	0.051	2.715	1	0.016	0.332	**1**	**0.050**	**10.593[c]**
BDE	1	0.005	0.269	1	0.002	0.047	1	0.000	0.013
CDE	1	0.003	0.177	1	0.008	0.164	1	0.000	0.002

Experiment 1			Experiment 2		Experiment 3		
error	6	0.019	6	0.048	6	0.005	1.000

a: $F_{obs} > F_{(0.001)}$, b: $F_{obs} > F_{(0.01)}$, c: $F_{obs} > F_{(0.025)}$, d: $F_{obs} > F_{(0.05)}$

Significant effects are indicated in bold font in table 4.1. Each of the treatments identified as significant in table 4.1 is summarized and described in tables 4.2 through 4.4, which are respectively associated with experiments 1, 2, and 3. Brief examination of table 4.1 demonstrates both more and smaller *p* values during experiments 1 and 3, in which the media characteristic factor (D) included anthracite filters (i.e. experiment 1 compared GAC and anthracite whereas experiment 3 compared anthracite media with uniformity coefficients of 1.3 and 1.6). This observation provides an immediate indication of greater resilience of GAC to the backwash techniques investigated herein. The design and operational factors investigated in Experiment 1 demonstrated few significant effects of GAC on DOC removal. The comparison of large to small sized GAC media in Experiment 2 yielded even fewer significant operational impacts on DOC removal.

Table 4.2: Summary of factors found to be significant for experiment 1, comparing GAC to anthracite biological filters

Factor/Interaction Direction (+/-)	P-Values	Description
A (-)	<0.01	CP backwash negatively impacted DOC removal.
AB (-)	<0.01	CP backwash combined with ETSW had a negative impact on DOC removal.
AC (-)	<0.001	CP backwash with non-chlorinated wash water negatively impacted DOC removal.
ABC (-)	<0.001	CP backwash combined with ETSW and non-chlorinated water negatively impacted DOC removal.
B (+)	<0.001	ETSW positively impacted DOC removal.
C (+)	<0.001	Non-chlorinated wash water positively impacted DOC removal.
BC (+)	<0.01	ETSW in combination with non-chlorinated wash water positively impacted DOC removal.
D (+)	<0.025	GAC filters removed more DOC than anthracite filters.
ACE (+)	<0.01	Collapsed pulse with non-chlorinated wash water positively impacted DOC removal 24 hours into the filter cycle.
BCE (+)	<0.01	ETSW with non-chlorinated wash water positively impacted DOC removal 24 hours into the filter cycle.

The high ES, low UC anthracite filter was compared to the high ES, low UC GAC filter in Experiment 1. This comparison demonstrated that backwash strategy and media type can significantly affect DOC removal by biological filtration. Collapsed pulsing in particular

tended to have a negative impact on DOC removal in the anthracite filter. Each backwash strategy that employed collapsed pulsing resulted in reduced DOC removal, but in GAC filters, this effect was reversed 24 hours in to the filter cycle. The key factors that positively influenced DOC removal were the ETSW backwash, non-chlorinated backwash water, and use of GAC filter media.

Collapsed pulsing negatively impacted DOC removal by biological filtration regardless of whether chlorinated or non-chlorinated wash water was used (factor A, and interaction AC respectively). While collapsed pulsing with both chlorinated and non-chlorinated wash water led to decreased filter DOC removal, after 24 hours DOC removal increased in filters backwashed with a non-chlorinated, collapsed pulse (ACE interaction). One possible mechanistic explanation for this positive effect on performance is decreased heterogeneity of the biofilm on the filter media, which will be further discussed in section 4.2. Further work is necessary to determine whether the positive effect on performance at 24 hours offsets the impaired performance at the beginning of the filter cycle, and whether this may ultimately lead to greater overall DOC removal during the course of the filter cycle.

The negative effect collapsed pulsing had on DOC removal by biological filtration is in contrast to several previous studies of the impact of collapsed pulse on biological filter performance (Ahmad and Amirtharajah, 1998; Emelko et al., 2006; Servais et al., 1991). Other studies, however, have indicated a possible negative effect of collapsed pulse backwashing on filter performance (Liu and Huck, 2001; Lu and Huck, 1993). Servais et al., (1991) found that C^{14} glucose removal in GAC filters was not impaired by a non-chlorinated air scour backwash, however, the effect of collapsed pulse on DOC removal in the present study was only detected in anthracite filters. This finding was consistent with the findings of Liu and Huck (2001), who found that backwashing with collapsed pulse resulted in decreased carboxylic acid and aldehyde removal only in anthracite filters at low temperature, when chlorinated wash water was used. In contrast, Ahmad and Amirtharajah (1998) found that both NPOC and AOC removal in anthracite filters were not significantly impacted by collapsed pulse backwashing. The temperature at which that study was carried out was not

documented; however, it may be that the effect of collapsed pulse backwashing is more relevant in cold water conditions such as those in this study. The overall implications of this finding are that DOC removal in GAC filters is more resilient to the deleterious effects of collapsed pulsing than in anthracite filters, and that the presence of chlorine in the wash water may impair DOC removal in anthracite filters for a period of up to 24 hours.

In addition to, or perhaps consequent to the resiliency of GAC biological filters to vigorous backwashing, better DOC removal was achieved by GAC relative to an anthracite filter of similar ES and UC (factor D in Experiment 1). The GAC filter was also resistant to the more vigorous backwashing protocols and chlorine exposure, as evidenced by the lack of interactions between any of the backwash factors with the media factor in Experiment 2 (i.e., no significant interactions between factors A, B, or C and factor D). It may be that this resistance to the highly negative impacts of collapsed pulse backwashing that led GAC filters being detected as having a positive effect on DOC removal. Figure 4.1 demonstrates that much of the DOC removal advantage that GAC has over anthracite is the result of the resilience of GAC filters to the negative impacts of collapsed pulse backwashing and that at 24 hours of filter run time this advantage has dissipated except when chlorinated collapsed pulse is used. A further discussion of the impact of media type on biological filter performance can be found in section 4.2.

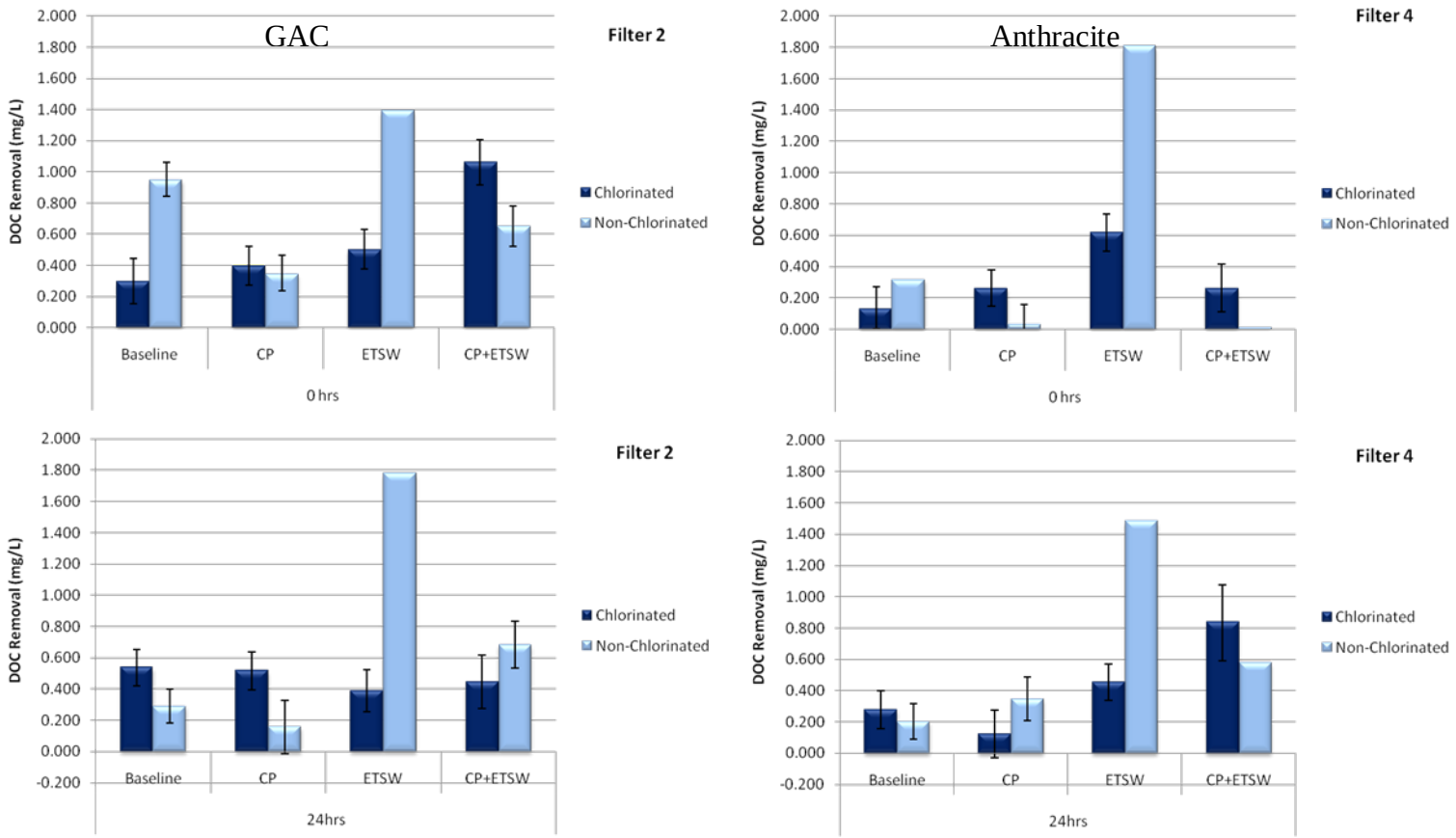

Figure 4.1: DOC removal in GAC (filter 2) and anthracite (filter 4) filters of comparable E.S. and U.C. (mean ± standard deviation). Baseline backwash is the MWTP full scale backwash, with no CP or ETSW. (mean +/- standard deviation)

Finally, the absence of chlorine had a positive impact on DOC removal. This effect was masked by the strong negative effects of the collapsed pulse backwash, but when CP was absent, such as in the baseline and ETSW runs, the non-chlorinated backwash filters tended to be capable of higher DOC removals immediately following backwash. It should also be noted that none of the negative effects on DOC removal interacted with the factor E, the factor for time, suggesting that anthracite filters are able to recover from the impact of collapsed pulse backwashing at some point during the first 24 hours of the filter cycle. Further work is necessary to determine the duration of impaired DOC removal following collapsed pulse.

Table 4.3: Summary of significant effects for experiment 2, comparing large GAC to small GAC

Factor/Interaction Direction (+/-)	P -Value	Description
AC	<0.025	Collapsed pulse with non-chlorinated water is associated with reduced DOC removal immediately following backwash.

Experiment 2, which compared the large GAC filter to the small GAC filter to assess the impact of media size on filter performance, yielded only one significant interaction effect (interaction AC). Collapsed pulse with non-chlorinated water was found to result in lower DOC removal immediately following backwash (interaction AC), but not 24 hours in to the filter cycle. In all three experiments collapsed pulse in the absence of chlorine appeared to have a more significant impact on DOC removal than chlorinated collapsed pulse immediately following backwash (figure 4.1). It is possible that the presence of chlorine leads to a higher rate of EPS production by cells colonizing the filter media, which is a well documented stress response in microbial communities. This thicker biofilm may be more resistant to the vigorous scouring of collapsed pulse backwashing. When non-chlorinated wash water was used, however, collapsed pulsing had a positive impact on DOC removal 24 hours in to the filter cycle. It is possible the collapsed pulse backwash serves to increase biofilm homogeneity by detaching and redistributing organisms throughout the bed.

Assuming successful colonization, at 24 hours in to the filter cycle this would lead to higher biofilm surface area and thus higher DOC removals by biological filtration.

Unlike collapsed pulsing and media type (Experiment 1), GAC media size was not found to significantly impact DOC removal by biological filtration. This is a critical finding, as one of the primary research goals of this study was to determine whether larger media sizes can compromise DOC removal at high flow rates in biological filters. This study demonstrated that larger GAC filtration media could achieve DOC removals at least comparable to those achieved by smaller GAC media (figure 4.2).

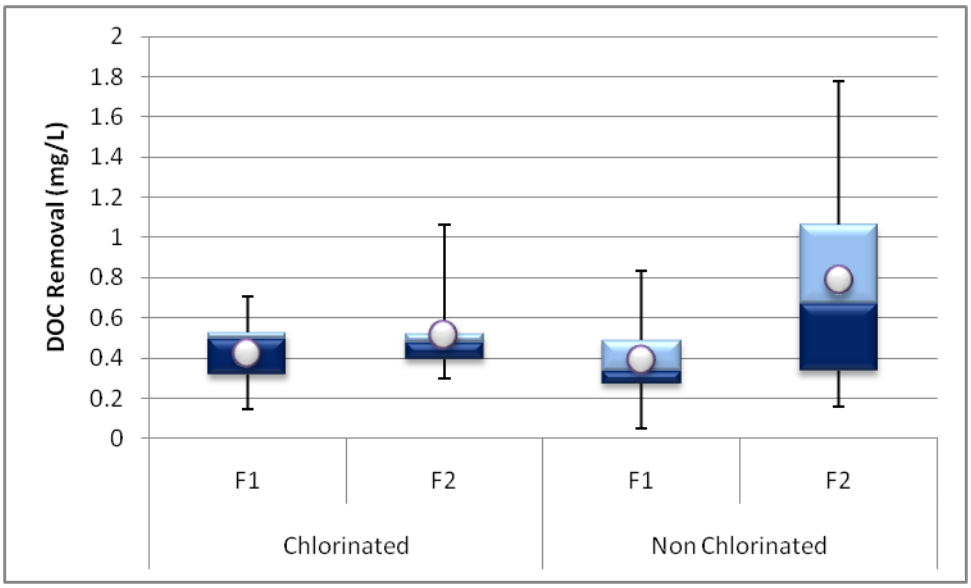

Figure 4.2: Box plots comparing DOC removal in filter 1 (small GAC) and filter 2 (large GAC) for all pilot factorial filter cycles. Light blue box represents DOC removals in from the median to 75th percentile, dark blue box the 25th percentile to the median, and the whiskers the maximum and minimum values.

Table 4.4: Summary of significant effects for experiment 3, comparing high and low uniformity coefficient anthracite filters

Factor/Interaction Direction (+/-)	P-value	Description
A (-)	<0.001	CP backwash with chlorinated wash water negatively impacts DOC removal immediately following backwash.
AB (-)	<0.001	CP backwash combined with ETSW and chlorinated wash water negatively impacts DOC removal immediately following backwash.
AC (-)	<0.001	CP backwash with non-chlorinated wash water negatively impacts DOC removal immediately following backwash.
ABC (-)	<0.001	CP backwash combined with ETSW and non-chlorinated water negatively impacts DOC removal, immediately following backwash.
B (+)	<0.001	ETSW positively impacts DOC removal.
C (+)	<0.001	Non-chlorinated wash water positively impacts DOC removal.
BC (+)	<0.001	ETSW in combination with non-chlorinated wash water positively impacted DOC removal.
ACE (+)	<0.001	Collapsed pulse with non-chlorinated wash water positively impacted DOC

Factor/Interaction Direction (+/-)	P-value	Description
		removal 24 hours into the filter cycle.
BCE (+)	<0.001	ETSW with non-chlorinated wash water positively impacted DOC removal 24 hours into the filter cycle.

High and low uniformity coefficient anthracite media were compared in Experiment 3. This experiment yielded not only more significant effects and interactions and higher degrees of significance compared to the preceding experiments. This means that the DOC removal by the anthracite filters was generally more sensitive to the study factors (e.g., chlorinated wash water, the use of collapsed pulse) than DOC removal by GAC filters (as was demonstrated in experiment 1). Anthracite uniformity coefficient (factor D) did not significantly affect DOC removal by biological filtration, nor did it interact with any other factors at the conditions investigated. Collapsed pulsing on the other hand was found to negatively affect filter DOC removal, both for chlorinated and non-chlorinated wash water (factor A and AC interaction respectively). Similar to experiment 1, collapsed pulsing with non-chlorinated wash water led to an had a positive effect on filter DOC removal later in the filter cycle (ACE interaction).

ETSW backwash was found to be beneficial to DOC removal, unless combined with collapsed pulse, this finding is in agreement with the findings of experiment 1. When non-chlorinated wash water was used, the ETSW wash led to positively impacted DOC removal 24 hours in to the filter cycle (BCE interaction). This effect was also found to be significant in experiment 1. Initially it was thought that this effect may have been an artifact of the unusually high DOC removals that occurred during the non-chlorinated ETSW backwash in all four filters (figure 4.1), which suggested a surge in biodegradable organic matter loading onto the filters. The ability of the ETSW wash to mitigate the negative effects of collapsed pulse backwash, and to positively impact filter performance 24 hours following collapsed pulse (as well as corresponding BRP data in section 4.3), however, suggest that ETSW does

have a positive effect on DOC removal at 24 hours in to the filter cycle. This finding is supported by the positive effects of ETSW on DOC removal being only associated with non-chlorinated wash. The presence of chlorine in the wash water would be expected to decrease the colonization efficiency of bacteria detached during the course of backwash.

Summary of Design and Operation Impacts on DOC Removal by Biological Filtration

Media characteristics, backwash technique, and the interaction between the two impacted biological filter performance. Collapsed pulse backwashing had the most substantive effect on biological filter performance, particularly on anthracite filters and immediately following backwash. The other backwash parameter found to have an impact on DOC removal was ETSW, which had a positive effect DOC removal. This was result was unexpected because intuitively, a non-chlorinated water flow at sub-fluidization velocity would not be expected to have any impact on biological filter performance. GAC was also found to improve DOC removal and was the only media characteristic that exerted a detectable effect.

There are three possible mechanistic explanations for the observation of reduced DOC removal following collapsed pulse backwash: 1) biofilm damage and removal, 2) increased oxidation efficiency of chlorine due to enhanced inter-particle scouring, and 3) the efficient scouring of collapsed pulse leading to very few collectors remaining on the media, thereby resulting in reduced abiotic adsorption of DOC following vigorous backwashing. The fact that collapsed pulse only had a detectable effect immediately following backwash suggests that a portion of this effect may be due to a limited number of collectors leading to reduced sorption of DOC…immediately following collapsed pulse. This effect is only detected in anthracite filters however, which may indicate that biomass removal also played a role.

Non-chlorinated collapsed pulse was found to have a positive effect on DOC removal 24 hours in to the filter cycle. When chlorinated water was used however, no such positive effect was observed, suggesting that increased disinfection efficiency during collapsed pulse

also plays a role in decreasing DOC removal efficiency throughout the filter run. Likely, the lower DOC removal associated with collapsed pulse backwashing is a result of an interaction between all three mechanisms (i.e.: physical damage to the biofilm, increased chlorine disinfection efficiency, and a paucity of collectors following vigorous backwashing. There is limited research available in the literature regarding the effect of collapsed pulse on biological filtration performance. In one similar bench scale study, the same effects were observed; collapsed pulse backwashing was found to be associated with lower DOC removal by anthracite filters at low temperature but not GAC filters (Liu and Huck, 2001). The duration of the negative effects of collapsed pulse on DOC removal following backwash is unclear, and further study is necessary to determine whether this has significant implications for DOC removal throughout the course of the filter cycle.

The positive impact ETSW had on DOC removal was an unexpected result. No previous research on the effect of ETSW on biological filtration performance has been reported. Intuitively, a gentle sub-fluidization wash step of 20 to 30 minutes would not be expected to affect DOC or BDOC removal, unless perhaps extended contact time with chlorine exerted a negative effect. ETSW was found to have a positive effect on DOC removal however, and was able to mitigate the negative effect of the collapsed pulse backwash. This study does not address the mechanism driving this effect, but it is speculated that perhaps ETSW serves to distribute detached bacteria throughout the filter bed, and improve their re-attachment efficiency through extended contact time with the media, leading to a higher biofilm surface area in the subsequent filter run.

The only media factor found to have a significant impact on DOC removal was the use of GAC over anthracite. Both media size and uniformity coefficient were not found to have a detectable effect on DOC removal, indicating that larger filter media may be employed to extend filter run length, without compromising DOC removal. As expected, uniformity coefficient was also found to have a negligible effect on DOC removal, though this factor is of more concern with regards to traditional filter performance. Table 4.5 summarizes the impacts of design and operational parameters on filter DOC removal.

Table 4.5: Impacts of backwash technique and media characteristics on DOC removal in biological filters

Parameter	Effect on DOC Removal	Notes
CP	↑/↓	Negative - Immediately following backwash for anthracite filters, chlorinated and non-chlorinated Positive – 24 hours into filter run for non-chlorinated
ETSW	↑	Immediately following backwash for non-chlorinated and chlorinated 24 hours in to filter cycle for non-chlorinated
Absence of Chlorine	↑	The absence of chlorine positively impacted DOC removal immediately following backwash. Chlorine led to negatively impacted DOC removal 24 hours into filter cycle.
GAC vs Anthracite	↑	Advantage of GAC likely related to resilience to collapsed pulse and chlorinated wash water.
Media Size	None	
Uniformity Coefficient	None	

4.2 Design and Operational Impacts on BDOC Removal

The ripening time factorial data for all three experiments was analyzed using single replicates of a 2^5 factorial. Normal probability plots were used to check for significance of higher order interactions (Appendix A, figures A.4, A.9, and A.14). In experiments 1 and 2, no significant four factor interactions were detected, and these were used as an internal estimate of error. In experiment 3, no three level interactions were detected, and these were used as an internal estimate of error. Using this estimate, results were then analyzed using ANOVA. The results of the ANOVA are presented in table 4.6.

Table 4.6: ANOVA table for factorial analysis of BDOC removal by pilot-scale biological filtration

Experiment 1				Experiment 2				Experiment 3			
Source	**DF**	**MS**	**F_o**	**Source**	**DF**	**MS**	**F_o**	**Source**	**DF**	**MS**	**F_o**
A	1	0.227	3.163	A	1	0.056	1.531	**A**	**1**	**0.950**	**12.348[a]**
B	1	0.218	3.031	B	1	0.042	1.145	B	1	0.351	4.566
AB	1	0.423	5.886	AB	1	0.005	0.136	AB	1	0.089	1.155
C	1	0.028	0.396	C	1	0.009	0.245	C	1	0.027	0.352
AC	1	0.353	4.912	AC	1	0.000	0.001	**AC**	**1**	**0.910**	**11.834[a]**
BC	1	0.010	0.140	BC	1	0.119	3.231	BC	1	0.029	0.374
ABC	1	0.326	4.540	ABC	1	0.188	5.124	D	1	0.155	2.013
D	**1**	**0.545**	**7.578[c]**	D	1	0.204	5.565	AD	1	0.103	1.339
AD	1	0.031	0.435	AD	1	0.052	1.412	BD	1	0.091	1.180
BD	1	0.031	0.428	BD	1	0.033	0.891	CD	1	0.008	0.106
ABD	1	0.001	0.007	ABD	1	0.015	0.413	ABCD	1	0.008	0.105
CD	1	0.059	0.822	CD	1	0.034	0.928	E	1	0.039	0.502
ACD	1	0.042	0.586	ACD	1	0.028	0.763	AE	1	0.094	1.223
BCD	1	0.098	1.360	BCD	1	0.043	1.164	BE	1	0.005	0.062
E	1	0.253	3.523	**E**	**1**	**0.297**	**8.080[c]**	CE	1	0.004	0.049

Experiment 1				Experiment 2				Experiment 3			
AE	1	0.023	0.317	AE	1	0.132	3.584	**ABCE**	**1**	**0.383**	**4.975[c]**
BE	1	0.008	0.113	BE	1	0.101	2.749	DE	1	0.116	1.504
ABE	1	0.001	0.017	ABE	1	0.008	0.208	ABDE	1	0.113	1.466
CE	1	0.002	0.023	CE	1	0.001	0.025	ACDE	2	0.024	0.310
ACE	1	0.055	0.769	ACE	1	0.119	3.241	BCDE	3	0.019	0.242
BCE	1	0.230	3.202	BCE	1	0.010	0.273	ABCDE	4	0.007	0.096
DE	1	0.001	0.015	DE	1	0.087	2.377	error	10	0.077	
ADE	1	0.076	1.052	ADE	1	0.001	0.027	Total			
BDE	1	0.044	0.618	BDE	1	0.206	5.594				
CDE	1	0.008	0.112	CDE	1	0.013	0.349				
ABCDE	1	0.322	4.481	ABCDE	1	0.074	2.008				
Error	5	0.0719		Error	5	0.037					
Total				Total							

a - $F_{obs} > F_{(0.01)}$, b – $F_{obs} > F_{(0.025)}$, c – $F_{obs} > F_{(0.05)}$

Table 4.7: Summary of significant effects for pilot factorial experiment 1, comparing GAC to Anthracite

Factor/Interaction (+/-)	P-Value	Description
D (+)	0.05	GAC provided higher BDOC removal than anthracite.

Of all the factors analyzed in pilot factorial 1, the only parameter found to have a potentially significant effect on filter BDOC removal was media type. GAC was found to have a positive effect on filter performance at the 5% significance level. This result is in agreement with DOC removal results. Like the DOC results, there is no interaction between GAC and time, suggesting the advantage of GAC over anthracite has dissipated by the 24 hour point of the filter cycle. Figure 4.3 demonstrates the resilience of GAC to collapsed pulse backwashing, a factor which likely played a role in the advantage GAC held over anthracite immediately following backwash. Further work is necessary to determine the duration of this advantage and its significance to overall organic removal through the filter cycle.

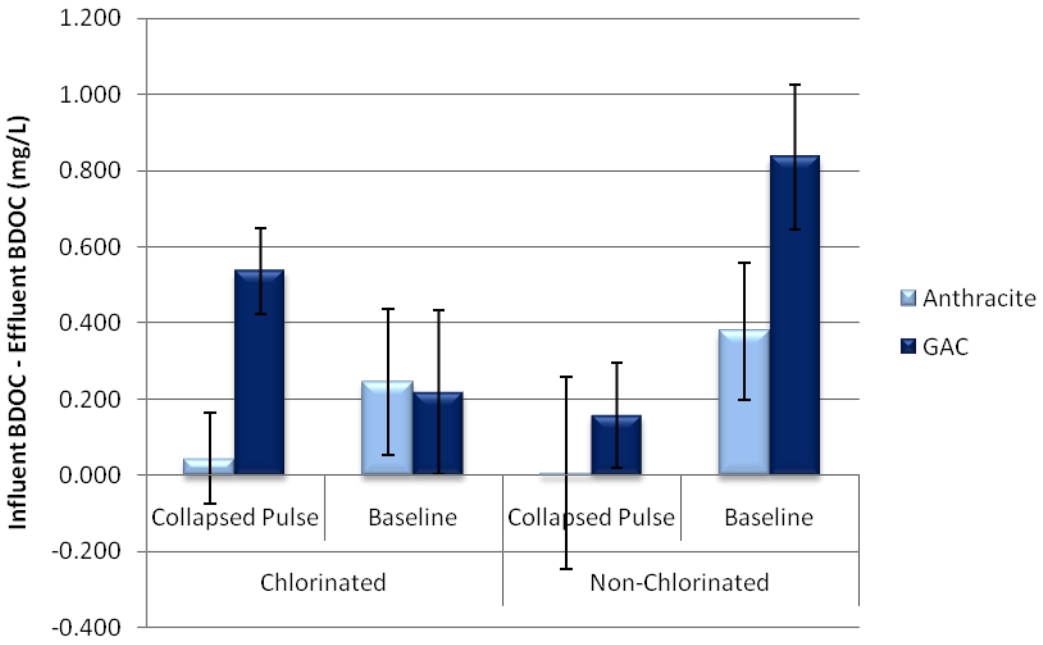

Figure 4.3: Impact of collapsed pulse on BDOC removal following backwash (mean +/- standard deviation)

The increased DOC and BDOC removal observed in GAC for this experiment corresponds to previous research comparing GAC to anthracite filters which have also noted the advantage of GAC (Lechevallier et al., 1992, Krasner et al., 1993., Wang et al., 1995, Liu et al., 2001). In the majority of cases in such studies however, the comparison of GAC to anthracite is confounded by operational or design factors. Krasner et al. (1993) observed a difference between GAC and anthracite at an EBCT of 1.4 min, a loading rate at which filters had to be backwashed every eight hours with chloraminated water. As has been noted in this study and others (Miltner et al., 1995; Liu et al., 2001; Emelko et al., 2006; Krasner et al., 1993; Wang et al., 1995), the presence of disinfectant in the backwash water tends to impair anthracite biological performance to a higher degree than GAC. At lower loading rates, with correspondingly longer filter runs between chlorinated backwashes GAC showed no advantage over anthracite. The results of LeChevallier et al. (1992) were complicated by the fact that inexhausted GAC to anthracite, artificially increasing AOC removal through the

residual adsorptive capacity of the GAC. Finally, the higher removal observed in the bituminous coal GAC filter in (Wang et al., 1995) was more likely due to a much smaller effective size (0.64mm) than the anthracite filter it was being compared to (1.02mm), thereby increasing the surface area and EBCT for the GAC filter. An earlier study of BDOC removal in anthracite and GAC filters at the MWTP revealed no differences between the media types in both warm and cold water conditions; this study however was carried out under a different filtration regime with a much longer EBCT of 15 minutes compared to the present filtration regime of 8 minutes. A bench scale study that was similar to the present study in taking media characteristics and backwash technique in to consideration came to very similar conclusions: that anthracite had lower removals than GAC only at low temperatures with chlorinated backwash (Liu and Huck, 2001).

Table 4.8: Summary of significant effects for pilot factorial experiment 2, comparing large and small GAC

Factor/Interaction (+/-)	P-value	Description
E (-)	<0.05	BDOC removal in GAC filters is impaired 24 hours into the filter cycle.

In experiment 2, which compared the large GAC filter to the small GAC filter, backwashing with chlorinated water appeared to reduce BDOC removal 24 hours in to the filter cycle. From figure 4.4, it is clear that this effect was most pronounced in filter one which is likely the reason why it was only detected in experiment 2, the only experiment to employ filter one. From figure 4.4 however, it is clear that performance in backwash chlorinated filters, particularly anthracite filters, does not increase appreciably after 24 hours of filter operation for any filter. There are two potential mechanistic explanations for this effect: a delayed effect of chlorine on BDOC removal, or an increase in BDOC release from the filter 24 hours following backwash. A BDOC release from the filter would be consistent with the proposed mechanism driving the delayed turbidity surge that follows chlorinated collapsed pulse backwashing (section 4.4).

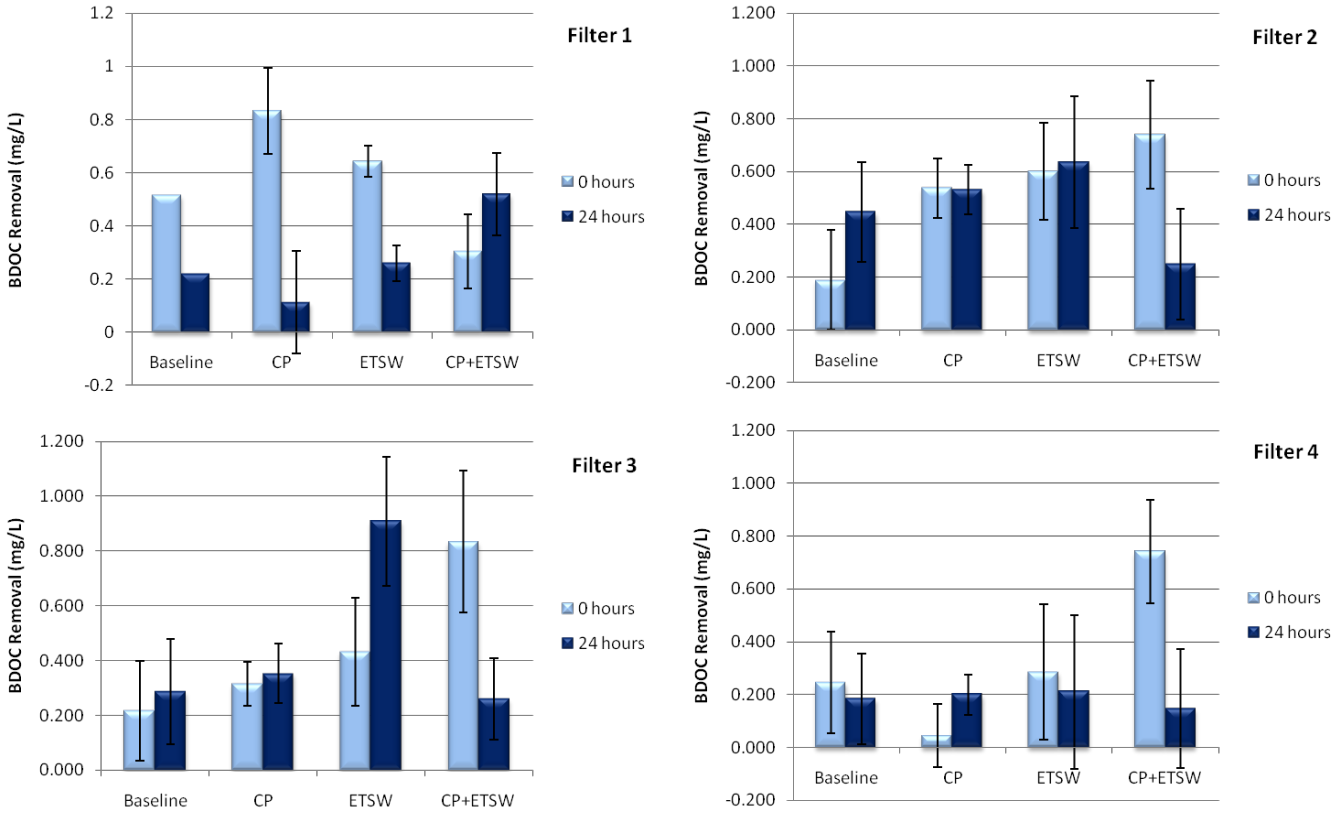

Figure 4.4: Delayed impact of chlorine on BDOC removal (mean +/- standard deviation)

As with DOC removal, media size was found to have no impact on BDOC removal. This finding provides further evidence that the size of filter media can be increased significantly without compromising DOC or BDOC removal, even at low water temperatures. Figure 4.5 demonstrates that the larger media are capable of similar, and in some cases higher DOC removals than the smaller media. This is consistent with previous research on the impact of contact time on BOM removal, which has demonstrated that while BOM removal is improved with increasing contact time, the relationship is less than proportional (Zhang and Huck, 1996). Thus, the incremental decrease in contact time realized from changing media size from 1.06 mm to 1.3 mm did not have a significant impact on the ability of the filter to remove of both BDOC and DOC. As will be discussed in section 4.4 however, this incremental increase did have a significant impact in reducing head loss accumulation and extending filter run time.

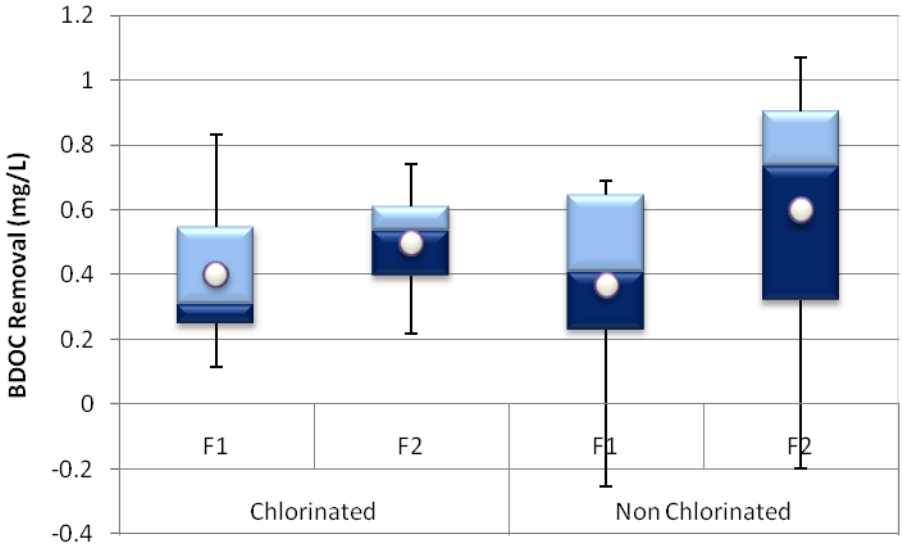

Figure 4.5: Comparison of BDOC removal in filter 1 (small GAC) and filter 2 (large GAC) for all runs of pilot factorial.

Table 4.9:Summary of significant effects for pilot factorial experiment 3, comparing high and low uniformity coefficient anthracite

Factor/Interaction (+/-)	P-Value	Description
A (-)	<0.01	Collapsed pulse backwash, chlorine present in wash water had a negative effect on BDOC removal
AC (-)	<0.01	Collapsed pulse backwash, chlorine absent from wash water had a negative effect on BDOC removal
ABCE (+)	<0.05	Collapsed pulse backwash, ETSW, chlorine absent from wash water, t = 24 hours. had a positive effect on BDOC removal

In experiment 3, the BDOC removal of anthracite filters proved to be vulnerable to collapsed pulse backwashing, regardless of chlorinated wash water, which is consistent with experiments 1 and 2, as well as the DOC removal data. This effect was not detected after 24 hours had passed however, and therefore further study is necessary to assess the duration of this period of impaired BDOC removal when chlorinated collapsed pulse is used to backwash anthracite filters. When the non-chlorinated collapsed pulse was used in combination with ETSW, a positive effect on filter BDOC removal after 24 hours of filter operation is observed, which is also consistent with experiments 1 and 2, as well as the DOC removal data. When chlorinated collapsed pulse was used in combination with ETSW however, performance was significantly impaired in both anthracite filters 24 hours following backwash (figure 4.4).

Summary of Design and Operation Impacts on BDOC Removal

Like DOC, BDOC removal was also impacted by both backwash technique and media characteristics. There were far fewer significant effects however, due to the fact that BDOC removal is largely biological, while DOC removal is confounded with sorption. While it is important to understand the impact of operational and design factors on DOC removal, the BDOC data gives a stronger estimate of biological performance. Collapsed pulse and chlorine were found to impair BDOC removal, while GAC and a backwash consisting of

non-chlorinated collapsed pulse and ETSW were found to positively affect BDOC removal after 24 hours. These findings are in agreement with the results of the DOC experiments.

Experiment 1, which compared anthracite and GAC filters of similar size and uniformity coefficients yielded only one detectable effect: the use of GAC over anthracite was shown to improve BDOC removal. Figure 4.4 however, demonstrates that this stems more from the resiliency of GAC to vigorous backwashing. Additionally, no interaction between collapsed pulse and the time effect was observed, suggesting the advantage of GAC over anthracite is transient. Further study is necessary to determine the duration of reduced BDOC removal following collapsed pulse. If this period extends significantly in to the filter run it would rule out the use of collapsed pulse on anthracite biological filters.

The effect of collapsed pulse on BDOC removal followed a similar pattern to its effect on DOC removal. Immediately following backwash, BDOC removal in anthracite filters was lower than BDOC removal following non-collapsed pulse backwashes, while GAC filters remained resilient to the vigorous scouring. At 24 hours however, this effect had dissipated. Interestingly, when non-chlorinated collapsed pulse backwash was used the effect had not only dissipated, but reversed, improving BDOC removal. It is possible that this is the result of the same mechanism that was proposed for improved DOC removal following this same backwash procedure: increased biofilm homogeneity. The benefit of this increased removal will depend on the duration of impaired removal immediately following the collapsed pulse backwash. The implications of these findings regarding collapsed pulse backwashing are that, firstly the duration of impaired removal following collapsed pulse of anthracite filters must be studied, secondly collapsed pulse can be used on GAC filters without compromising BDOC removal, and finally collapsed pulse may actually have a positive effect on DOC removal when non-chlorinated wash water is used.

The presence of chlorine in the wash water, while not found to directly impair biological performance, exerted a negative effect in a number of indirect ways. Chlorine interacted with collapsed pulse to impair the ability of the filter to recover from this vigorous

scouring procedure. Chlorine also led to a decreased filter BDOC removal 24 hours into the filter cycle.

Table 4.10: Summary of impacts of backwash technique and media characteristics on BDOC removal in biological filters

Parameter	Effect	Notes
CP	↓/↑	Negative - Immediately following backwash for anthracite filters, chlorinated and non-chlorinated. Positive - 24 hours into filter run for non-chlorinated when ETSW is used.
ETSW	↑	24 hours in to filter cycle for non-chlorinated CP wash
Absence of Chlorine	↑	Chlorine interacted with collapsed pulse to impair BDOC removal and may contribute to impaired performance 24 hours following backwash.
GAC vs Anthracite	↑	Advantage of GAC likely related to resilience to collapsed pulse and chlorinated wash water.
Media Size	None	
Uniformity Coefficient	None	

4.3 Biological Respiration Potential

The BRP factorial data for all three experiments was analyzed using single replicates of a 2^6 factorial. Normal probability plots were used to check for potentially significant factors (appendix A, figures A.5, A.10, and A.15). Due to the extreme length of the ANOVA table for the 2^6 factorial it has been added to appendix A (figure A.1, page 189)

Biological respiration potential was selected with the intention of verifying DOC and BDOC data. While BDOC and DOC data give the best measure of how a filter is performing by quantifying the activity of the entire filter in situ, there are challenges in comparing separate filter runs due to varying influent concentrations. Filter removal was chosen as a comparative measure in this study to help normalize the performance by how much organic

material was removed from the water, rather than effluent concentration. If an unusually large quantity of biodegradable matter was loaded onto the filter however, and the biofilm was not previously utilizing its entire respiratory potential, an unusually high removal would occur that was not necessarily the result of backwash. BRP on the other hand, takes the media from its environment, and quantifies the ability of biofilm on that media to degrade a cocktail of ozonation by-products common in the filter influent water, each test is carried out with identical concentrations and compositions of DOC.

Due to the issues with representative sampling and test conditions discussed in chapters 2 and 3, effects where BRP and DOC or BDOC removal data are in agreement are cautiously considered as further evidence of the likely significance of a given factor and an indication that trends observed in DOC or BDOC removal are associated with an impact on biological activity.

Table 4.11: Summary of significant effects on BRP for pilot factorial experiment 1, comparing GAC to anthracite biological filters

Factor/Interaction (+/-)	P-Value	Description
AC (-)	0.001	Non-Chlorinated collapsed pulse negatively impacts BRP immediately following backwash.
BC (-)	0.001	Non-Chlorinated ETSW negatively impacts BRP immediately following backwash.
BDE (-)	0.01	Chlorinated ETSW negatively impacts BRP 24 hours in to the filter cycle in GAC filters.
D (+)	0.01	GAC has a higher BRP than anthracite.

Depth in the filter (surface vs 20cm in to the filter bed) was not found to have a significant impact on BRP. This result does not agree with previously published results regarding the impact of filter bed depth on BRP (Urfer and Huck, 2001). This could be due to a number of factors. One possible reason was that each backwash carried out in the present study included an air scour step, in the case of collapsed pulse this air scour was combined with water wash at a sub-fluidization rate. This would result in a relatively even scouring of the upper portion of the filter, regardless of backwash strategy, and perhaps by 24 hours the BRP of the media 20 cm below the surface was approximately the same as that at the surface. Another possibility was that the lower loading rates (7.5 m/h vs 11.2 m/h) in Urfer and Huck, (2001) led to a higher degree of biomass stratification. Regardless, the purpose of this study was to carry out experiments in a drinking water treatment plant environment, and to measure effects that have significant implications for the optimization of biological filtration as it is applied in the drinking water treatment industry, and as such minor differences in respiration potential between media surface and 20 cm in to the filter bed are not of great concern.

The negative impact of collapsed pulse on both DOC and BDOC removal immediately following backwash observed in the DOC and BDOC data was confirmed by the BRP data. Due to the proximity of sampling to the backwash, it was speculated that

perhaps the reduced removal of DOC and BDOC was due to reduced sorption due to a deficiency of collectors on the media, similar to filter ripening. The BRP data however, indicates that this was due to decreased biomass activity in the filter. The BRP data also indicated that non-chlorinated collapsed pulse tended to have a more significant impact on BRP immediately following backwash than chlorinated collapsed pulse. This effect was also observed in the experiments using BDOC and DOC as response variables, and could potentially be the result of increased EPS production in response to the presence of chlorine.

As expected, GAC was found to have a higher BRP than anthracite. This finding is consistent with previous studies demonstrating that GAC supports a higher quantity of biomass (Wang et al., 1995). This higher concentration of biomass, though not necessarily capable of removing more BOM in the drinking water treatment environment, would be expected to produce higher BRP results in a batch culture with relatively simple substrates such as the BRP test. Figure 4.6 highlights the significant difference in biological respiration potential between GAC and anthracite, particularly when chlorine, collapsed pulse, or both are used. This is further evidence of the resiliency of GAC to vigorous backwashing. From figure 4.6 it is clear that the advantage of GAC over anthracite in terms of BRP is derived from its resilience to chlorinated wash water and collapsed pulse backwashing. When neither of these factors where used during backwash (baseline and ETSW only backwashes) the anthracite filters had very similar BRP to GAC. The impact of chlorine observed in this study is in contrast to observed BDOC and DOC results, which showed anthracite to be capable of similar levels of removal to GAC in chlorinated backwash runs so long as collapsed pulse was not used. This suggests that while anthracite filters are capable of achieving similar removal levels to GAC, the reduction in biodegradation potential following chlorinated backwash could impair their ability to respond to surges in organics loading, such as those following a heavy precipitation event.

Table 4.12: Summary of significant effects on BRP for pilot factorial experiment 2, comparing small GAC to large GAC biological filters

Factor/Interaction (+/-)	P-Value	Description
B (+)	<0.01	ETSW positively impacted BRP immediately following backwash.
AB (+)	<0.01	Collapsed pulse combined with ETSW positively impacted BRP immediately following backwash.
AC (-)	<0.01	Non-Chlorinated collapsed pulse negatively impacted BRP immediately following backwash.
BC (-)	<0.01	Non-Chlorinated ETSW negatively impacted BRP immediately following backwash.
AE (+)	<0.01	Collapsed pulse positively impacted BRP 24 hours in to the filter cycle
BE (-)	<0.01	Chlorinated ETSW negatively impacted BRP 24 hours in to the filter cycle.
ABE (-)	<0.01	Chlorinated collapsed pulse combined with ETSW negatively impacted BRP 24 hours in to the filter cycle.
ABCE (-)	<0.01	Non-Chlorinated collapsed pulse combined with ETSW negatively impacted BRP 24 hours in to the filter cycle for anthracite filters.
AF (+)	<0.01	Chlorinated collapsed pulse positively impacted positively impacted BRP 20 cm into the filter bed.
ACF (-)	<0.01	Non-Chlorinated collapsed pulse negatively impacted BRP 20cm into the filter bed.
ABEF (-)	<0.01	Chlorinated collapsed pulse, combined with ETSW negatively impacted BRP 20 cm into the filter bed.

In experiment 2, many of the same trends seen in BDOC and DOC removal were observed: collapsed pulse inhibiting biological activity immediately following backwash, particularly when non-chlorinated (AC and ACF interactions); but improving performance 24

hours in to the filter cycle (AE interaction), ETSW improving biological activity and ameliorating the negative effects of collapsed pulse (factor B and AB interaction), chlorine impairing biological performance 24 hours in to the filter run (BE, ABE, ABCE, and ABEF interactions). These trends are also evident in figure 4.6.

In addition to the expected results, experiment 2 also yielded contradictory results. As previously discussed, non-chlorinated ETSW was found to have an inexplicably negative impact on BRP (BC interaction), and chlorinated collapsed pulse was found to actually improve BRP immediately following backwash (AF interaction).

Table 4.13: Summary of significant effects on BRP for pilot factorial experiment 3, comparing large and small anthracite biological filters

Factor/Interaction (+/-)	P-Value	Description
BCEF (+)	<0.01	Non-Chlorinated ETSW positively impacted BRP 20 cm in to the filter bed after 24 hours of filter operation.
ACEF (+)	<0.01	Non-Chlorinated collapsed pulse positively impacted BRP 20 cm in to the filter bed after 24 hours of filter operation.
BCE (+)	<0.001	Non-Chlorinated ETSW positively impacted BRP 20 cm in to the filter bed after 24 hours of filter operation.
CE (-)	<0.01	Absence of chlorine negatively impacted BRP 24 hours in to the filter run.
BE (+)	<0.01	Chlorinated ETSW positively impacted BRP 24 hours in to the filter run.
E (-)	<0.001	Chlorinated ETSW negatively impacted BRP immediately following backwash.
AB (+)	<0.01	Chlorinated Collapsed pulse and ETSW positively impacted BRP immediately following backwash
C (+)	<0.01	Absence of chlorine positively impacted BRP immediately following backwash.
AC (-)	<0.001	Non-Chlorinated collapsed pulse negatively impacted negatively impacted BRP immediately following backwash.
BC (-)	<0.001	Non-Chlorinated ETSW negatively impacted BRP immediately following backwash.
ABC (-)	<0.01	Non-Chlorinated collapsed combined with ETSW pulse negatively impacted BRP immediately following backwash.

In experiment 3, many of the same trends seen in BDOC and DOC removal, as well as experiment 2 were again observed: collapsed pulse inhibiting biological activity immediately following backwash, particularly when non-chlorinated (AC, ABC); but

improving performance 24 hours in to the filter cycle (ACEF). ETSW improving biological activity and ameliorating the negative effects of collapsed pulse (BE, BCE, BCEF, AB).

Experiment 3 yielded two interesting results: in both non-chlorinated and chlorinated backwash runs BRP was found to decrease over time in the anthracite filters (E and CE). Decreased biological performance late in the filter cycle has been previously documented in the literature and is thought to result from inhibitory effects of floc loading onto the filter (Prevost et al., 1995; Carlson et al., 1996). This effect was not detected in GAC filters, which is consistent with the resilience of GAC to influent floc and oxidant loading that was noted in chapter 2 (Niquette et al., 1998; Weinberg et al., 1993).

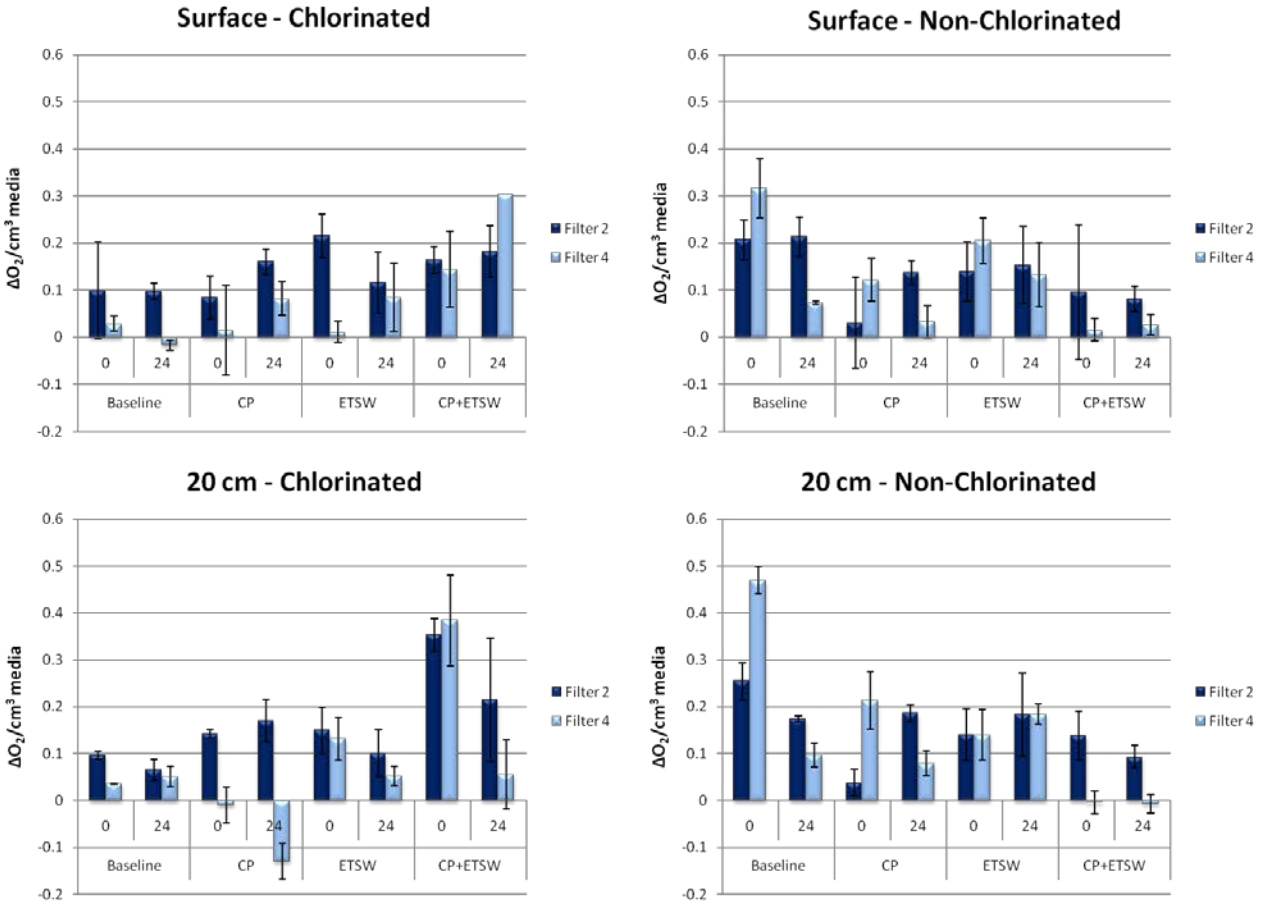

Figure 4.6: Pilot Factorial BRP Results for filter 2 (large GAC) and filter 3 (large anthracite) (mean +/- standard deviation)

4.4 Filter Run Time

Filter effluent turbidities at the MWTP are consistently below 0.1 NTU, as they were during all of the experimental periods at the MWTP. Accordingly, no discernible differences between the filters' particle removal capacities were observed. For this reason, filter run time was selected as a key parameter for assessing traditional performance. The drawback to this method is that this does not account for whether filters are being backwashed primarily due to headloss, turbidity breakthrough, or time. This is easily accounted for however, by assessing what factors trigger backwash. Using this method, both turbidity and terminal head loss accumulation are accounted for. When a significant effect on filter run time is detected, the cause of filter cycle termination can then be determined directly from the data.

The run time factorial data for all three experiments were analyzed using single replicates of a 2^4 factorial. Normal probability plots were used to check for significance of higher order interactions, and when none were found these interactions were used as an internal estimate of error. Using this estimate, results were then analyzed using ANOVA. The results of the ANOVA are presented in table 4.14. Significant results from each experiment are described individually in tables 4.15-4.17.

Table 4.14: ANOVA results for filter run time experiments 1-3

Experiment 1				Experiment 2				Experiment 3			
Source	**DF**	**MS**	**Fo**	**Source**	**DF**	**MS**	**Fo**	**Source**	**DF**	**MS**	**Fo**
A	**1**	**1142.7**	**103.8[a]**	**A**	**1**	**281.2**	**5.8[d]**	**A**	**1**	**1316.6**	**488.7[a]**
B	1	32.4	2.9	C	1	114.8	2.4	B	1	9.1	3.4
AB	1	1.7	0.2	**AC**	**1**	**363.9**	**7.6[c]**	AB	1	9.1	3.4
C	**1**	**1142.7**	**103.8[a]**	**D**	**1**	**919.1**	**19.1[b]**	**C**	**1**	**1316.6**	**488.7[a]**
AC	**1**	**1459.4**	**132.5[a]**	AD	1	160.9	3.3	**AC**	**1**	**1316.6**	**488.7[a]**
BC	1	1.7	0.2	**CD**	**1**	**351.1**	**7.3[c]**	BC	1	9.1	3.4
D	1	3.4	0.3	**ACD**	**1**	**348.6**	**7.2[d]**	D	1	0.0	0.0
AD	1	6.6	0.6	Error	8	48.2		AD	1	0.0	0.0
BD	1	2.2	0.2					BD	1	1.5	0.5
CD	1	6.6	0.6					CD	1	0.0	0.0
error	5	11.0						error	5	2.7	

a - $F_{obs} > F_{(0.01)}$, b – $F_{obs} > F_{(0.025)}$, c – $F_{obs} > F_{(0.05)}$

Table 4.15: Summary of significant effects on filter run time for pilot factorial experiment 1, comparing GAC to anthracite

Factor/Interaction (+/-)	Significance (%)	Description
A (-)	0.1	Collapsed pulse when used with chlorinated wash water significantly reduces filter run time.
C (+)	0.1	Non-chlorinated wash water is associated with longer filter run times.
AC (+)	0.1	Collapsed pulse, when used with non-chlorinated wash water significantly increases filter run time.

Significant effects detected in experiment 1 were chlorinated collapsed pulse, which had a significant negative impact on filter run time, non-chlorinated collapsed pulse, which had a significant positive effect on filter run time, and non-chlorinated wash water, which impacted filter run time positively. In each case, the effects were determined to be significant with a p-value < 0.001. Experiment 2 followed a similar trend.

Table 4.16: Summary of significant effects on filter run time for pilot factorial experiment 2, comparing large GAC to small GAC

Factor/Interaction (+/-)	P-Value	Description
A (-)	<0.05	Collapsed pulse when used with chlorinated wash water reduces filter run time.
AC (+)	<0.025	Collapsed pulse, when used with non-chlorinated wash water increases filter run time.
D (+)	<0.001	Media size significantly increases filter run time.
CD (+)	<0.025	Non-chlorinated wash water significantly increased filter run time for the large media filter.
ACD (+)	<0.05	Collapsed pulse, when used with non-chlorinated wash water increases filter run time significantly increased filter run time for the large media filter.

In experiment 2, comparing the large GAC to the small GAC, the same trend observed in experiment 1 of chlorinated collapsed pulsing shortening filter run time and non-chlorinated collapsed pulsing extending filter run time is observed, although at a lower level of significance (0.05 and 0.025 vs 0.001 and 0.001 respectively). Experiment 2 also serves to highlight the positive impact of larger filter media size on filter run time.

As expected, larger media size was also found to have a significant positive impact on filter run time (D). The extended run length of the larger filters indicates that it is possible to prevent terminal head loss build up by increasing media size, without compromising turbidity removal. This effect was particularly evident for the larger media in filter 2 (CD, ACD).

Table 4.17: Summary of significant effects on filter run time for pilot factorial experiment 3, comparing high U.C. anthracite to low U.C. anthracite

Factor/Interaction (+/-)	P-Value	Description
A (-)	<0.001	Collapsed pulse, when used with chlorinated wash water significantly reduces filter run time.
C (+)	<0.001	Non-chlorinated wash water is associated with longer filter run times.
AC (+)	<0.001	Collapsed pulse, when used with non-chlorinated wash water significantly increases filter run time.

Experiment 3 again highlighted the significant impact collapsed pulsing can have on filter run length. Uniformity coefficient played no role in determining filter run length. Figure 4.7 illustrates the dichotomy between the effect of chlorinated and non-chlorinated collapsed pulse backwashes. In the normal probability plot in figure 4.7, the ordered effect estimates are plotted on normal probability paper, negligible effects are normally distributed, with a mean of zero and variance σ^2 and will tend to fall along a straight line. Significant effects however, will have a non-zero mean and will not lie along a straight line. The collapsed pulse-absence of chlorine interaction (AC) lies far to the right of the straight line, and the chlorinated collapsed pulse effect (A) lies far to the left.

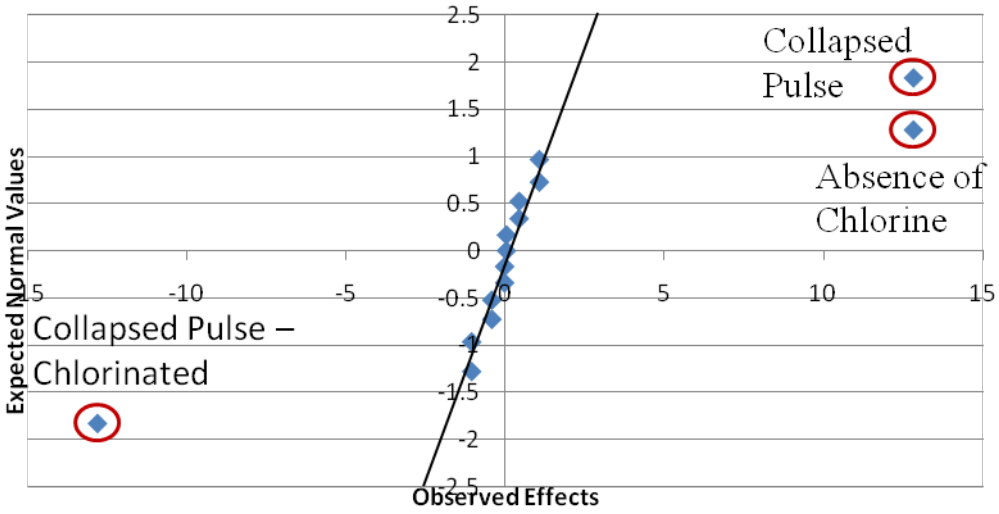

Figure 4.7: Normal probability plot of effects of backwash and media type on filter run time in experiment 3, highlighting the impact of collapsed pulse on filter run time

Each of the three pilot-scale experiments highlights the significant impact that collapsed pulse backwashing can exert on filter run time, both positive and negative. When collapsed pulse is chlorinated the filter runs are significantly attenuated. When non-chlorinated wash water is used however the filter runs are significantly extended. This result was unexpected, collapsed pulse was perhaps expected to impact one or more of the biological performance parameters, but was expected to either extend or have no effect on filter run time, regardless of whether chlorinated wash water was used. This effect is not unprecedented in the literature however, Emelko et al. (2006) reported a significant and immediate decrease in filter run time as a result of collapsed pulse backwashing, which reversed immediately upon removing collapsed pulse from the backwash procedure (figure 4.7). A delayed increase in particle passage following chlorinated backwash without collapsed pulse has also been reported (Goldgrabe et al., 1993). In that study, a filter backwashed with chlorinated water was run in parallel with a filter backwashed with non-chlorinated water. Effluent total particle counts (TPC) in the filter backwashed with chlorinated water precipitously increased from 446 to 1310 TPC/mL 48 hours following filter

backwash, while TPC in the filter backwashed with non-chlorinated water remained relatively constant.

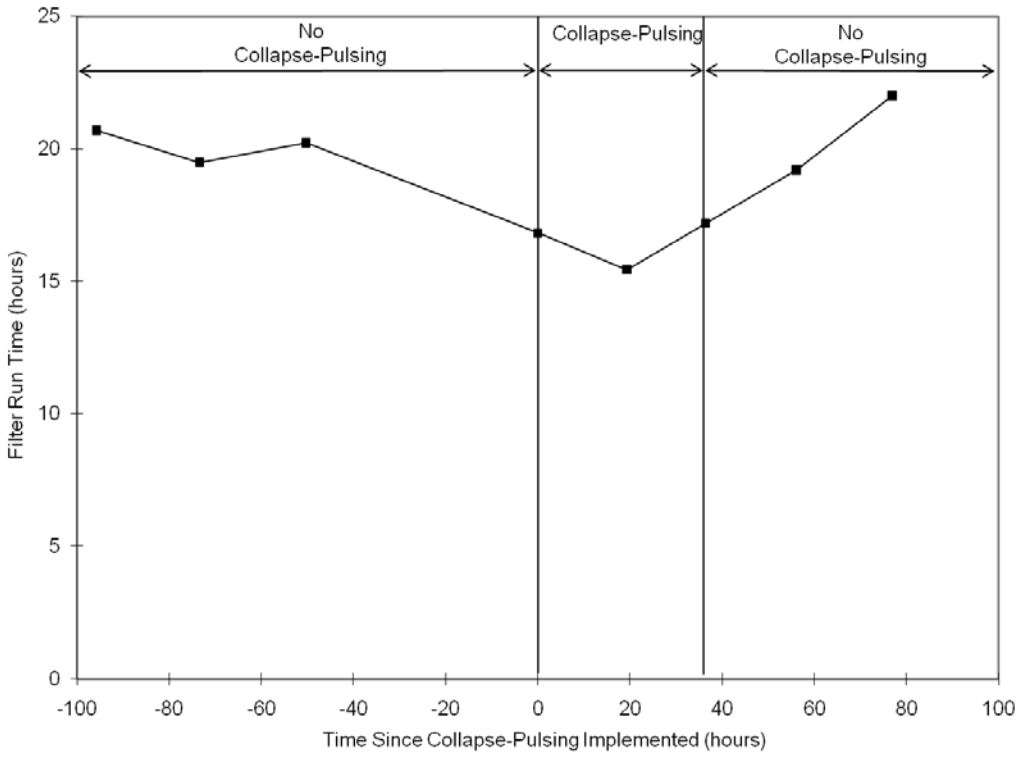

Figure 4.8: Impact of chlorinated collapsed pulse on filter run time at full-scale at the MWTP. Reproduced from Emelko et al. (2006)

In this study, each case of truncated filter run times following chlorinated collapsed pulse, the cause was a result of a sudden increase in turbidity that did not dissipate until the filter was backwashed. Figure 4.9 is an example of this sudden failure of turbidity removal. This is an example two runs of pilot filter two (GAC, ES=1.3), one backwashed with non-chlorinated collapsed pulse, the other chlorinated. Approximately 16 hours following backwash, turbidity begins to increase and causes the run to terminate by surpassing 0.1ntu. The non-chlorinated collapsed pulse backwashed filter however continues well below 0.1 ntu until it is backwashed at 48 hours.

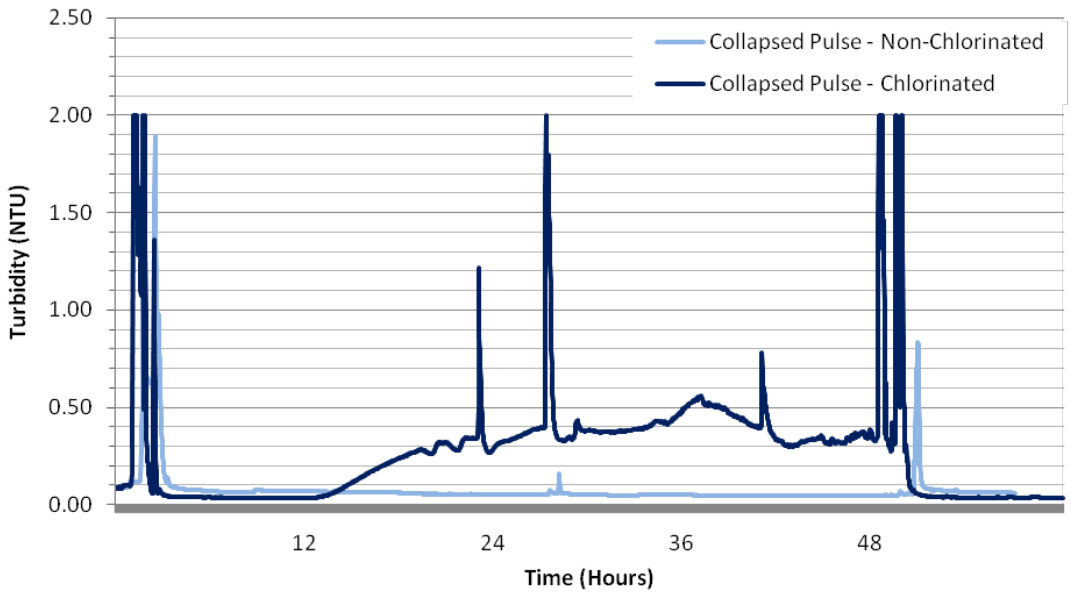

Figure 4.9: Characteristic turbidity removal failure following chlorinated collapsed pulse backwashing of a GAC filter. The large spikes in turbidity at 2 and 48 hours represent ripening spikes following filter backwash. The same effect was observed in anthracite filters

It is possible that this surge in effluent turbidity is the result of biofilm, damaged during the course of chlorinated collapsed pulsing, sloughing from the media as dead bacteria fail to produce further EPS and adhesion molecules. The fact that the breakdown in turbidity removal is associated with collapsed pulse and chlorine together, but not chlorine alone, suggests the vigorous inter-particle scouring of the collapsed pulse backwash enhances chlorine penetrance in to the biofilm. This could be verified by monitoring filter effluent EPS concentrations, a surge in EPS associated with turbidity breakthrough following chlorinated collapsed pulse would be indicative of biofilm sloughing.

Summary of Factors Affecting Filter Run Time

Factors that tended to increase filter run time were media size, non-chlorinated collapsed pulse, and the absence of chlorine from the wash water. As was expected, increasing media size shifted the primary backwash trigger from terminal head loss to

turbidity breakthrough, and due to the ability of the larger media to remove turbidity as efficiently as the smaller media, this led to significantly longer filter run times. Longer filter run times were also associated with non-chlorinated collapsed pulse backwashing. The efficient particle removal, and vigorous inter-particle scouring was likely able to prevent media clumping and preferential flow pathways, allowing the filter to maintain performance longer in to the filter cycle.

While collapsed pulse was able to significantly extend filter run time when non-chlorinated, chlorinated collapsed pulse had the opposite effect. Filter runs were siginicantly truncated due to a sudden breakdown of turbidity removal approximately fifteen to twenty-five hours into the filter cycle that did not dissipate until the subsequent backwash. It is speculated that this sudden surge in turbidity may be associated with sections of the biofilm, damaged during the course of chlorinated collapsed pulse backwasing, sloughing from the media.

Table 4.18: Summary of impacts of backwash technique and media characteristics on filter run time in biological filters

Parameter	Effect	Notes
CP	↓/↑	Negative – Effluent turbidity surges associated with chlorinated CP Positive – Non-Chlorinated
ETSW	None	Reduced ripening time technically extends filter run time however.
Absence of Chlorine	↑	Chlorine interacted with collapsed pulse shorten filter run times
GAC vs Anthracite	None	
Media Size	↑	Larger media allow for longer filter run times
Uniformity Coefficient	None	

4.5 Filter Ripening Time

The ripening time factorial data for all three experiments was analyzed using single replicates of a 2^4 factorial. Normal probability plots were used to check for significance of higher order interactions (appendix A). In all three cases, no higher order interactions were found and these were therefore used to generate an internal estimate of error. Using this estimate, results were then analyzed using ANOVA. The results of the ANOVA are presented in table 4.20.

Table 4.19: ANOVA results for filter run time experiments 1-3

Experiment 1				Experiment 2				Experiment 3			
Source	**DF**	**MS**	**Fo**	**Source**	**DF**	**MS**	**Fo**	**Source**	**DF**	**MS**	**Fo**
A	**1**	**722.0**	**6.9**	**A**	**1**	**2312.0**	**13.2**	A	1	112.5	2.5
B	**1**	**3960.5**	**37.9**	**B**	**1**	**2244.5**	**12.8**	**B**	**1**	**3698.0**	**83.7**
AB	1	144.5	1.4	AB	1	180.5	1.0	AB	1	12.5	0.3
C	1	512.0	4.9	C	1	200.0	1.1	C	1	60.5	1.4
AC	1	2.0	0.0	AC	1	2.0	0.0	AC	1	32.0	0.7
BC	1	364.5	3.5	BC	1	84.5	0.5	**BC**	**1**	**684.5**	**15.5**
D	1	18.0	0.2	D	1	18.0	0.1	D	1	200.0	4.5
AD	1	578.0	5.5	AD	1	8.0	0.0	AD	1	60.5	1.4
BD	1	60.5	0.6	BD	1	544.5	3.1	BD	1	32.0	0.7
CD	1	0.0	0.0	CD	1	72.0	0.4	CD	1	220.5	5.0
Error	5	104.4		Error	5	174.8		error	5	44.2	

a - $F_{obs} > F_{(0.01)}$, b – $F_{obs} > F_{(0.025)}$, c – $F_{obs} > F_{(0.05)}$

The one common significant factor for each of three experiments was factor B, ETSW. In all cases, ETSW was found to significantly lower, and often fully eliminate filter ripening. Significant effects from the individual experiments are summarized in tables 4.21 to 4.23.

Table 4.20: Summary of significant effects for pilot factorial experiment 1

Factor/Interaction (+/-)	P-Value	Description
A (+)	<0.05	Collapsed pulse, when used with chlorinated water may increase filter ripening time.
B (-)	<0.01	ETSW significantly shortens filter ripening time.

In experiment 1, comparing GAC to anthracite, chlorinated collapsed pulse backwashing and ETSW were the only factors found to significantly affect filter ripening time. ETSW was associated with significantly reduced, and often eliminated filter ripening spikes, allowing the filter to be put into service immediately following backwash. Conversely, chlorinated collapsed pulsing was associated with an increase in filter ripening time. The same trend was observed in experiment 2.

Table 4.21: Summary of significant effects on ripening for pilot factorial experiment 2

Factor/Interaction (+/-)	P-Value	Description
A (+)	<0.025	Collapsed pulse, when used with chlorinated water may increase filter ripening time.
B (-)	<0.025	ETSW, when used with chlorinated wash water significantly shortens filter ripening time.

In experiment 2, which compared small GAC to large GAC, collapsed pulse was found to significantly increase filter ripening time, and ETSW was found to significantly decrease it. Figure 4.10 shows the drastic impact ETSW can have on ripening time, particularly when non-chlorinated wash water is used.

Table 4.22: Summary of significant effects for pilot factorial experiment 1

Factor/Interaction (+/-)	P-Value	Description
B (-)	<0.001	ETSW, when used with chlorinated wash water significantly shortens filter ripening time.
BC (-)	<0.025	ETSW, when used with non-chlorinated wash water significantly shortens filter ripening time.

As expected, ETSW successfully reduced, and often eliminated filter ripening in each of the three factorial experiments. This was the result of the ETSW successfully sweeping away backwash remnant particles, without detaching further entrained particles from the media, and has been documented several times in the literature (Amburgey et al., 2003; Amburgey, 2005; Amburgey and Amirtharajah, 2005). While ETSW was unable to fully eliminate ripening when chlorinated collapsed pulse was used, it succeeded in significantly reducing it.

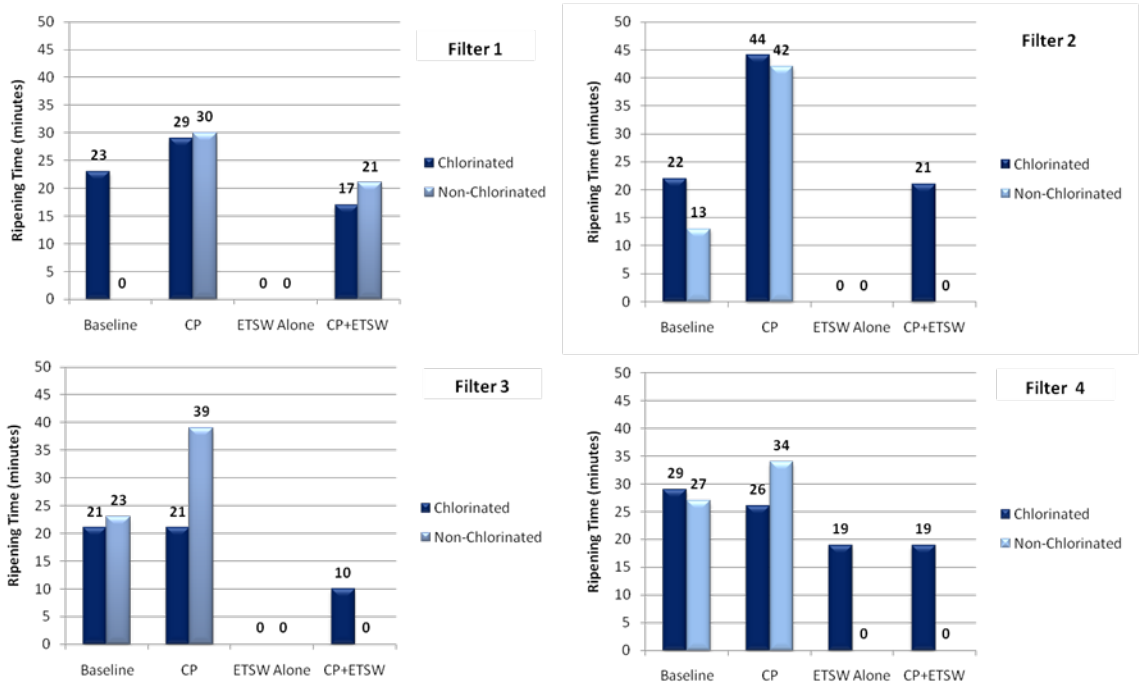

Figure 4.10: Impact of backwash on ripening time in pilot-scale biological filters

With the exception of the small GAC in filter 1, ETSW was able to eliminate filter ripening following collapsed pulse when non-chlorinated water was used. This interaction is consistent with previous research that has shown chlorinated water to be capable of increasing the intensity and the duration of filter ripening (Amburgey et al., 2004). In that study it was found that chlorine increased filter ripening in pilot-scale filters that had not previously been backwashed with chlorinated water; but not at full-scale where filters had been acclimated to chlorinated backwashing. In this study, pilot filters had been backwashed with chlorinated water only for two years in the lead up to the factorial experiments, and so were well acclimated to the presence of chlorine in the wash water. Additionally, ETSW did not fully eliminate filter ripening in filter four when chlorinated wash water was used. This was due to the fact that at that time there was a slight leak in the air scour valve that caused small quantities of air to bubble through the filter when the air compressor tank was full. This disturbance would easily detach particles from the media, artificially extending filter ripening. This mechanical defect was detected and remedied prior to the next filter run and affected only the chlorinated ETSW backwashes of filter 4.

Figure 4.11 contrasts two ETSW backwashes, one that included a collapsed pulse, one that did not, highlighting the negative impact of collapsed pulse on filter ripening, even when ETSW is used. the negative impact of collapsed pulse on filter ripening, and the ability of ETSW to mitigate this effect. The increase in filter ripening associated with the use of collapsed pulse backwashing was also an expected result. Collapsed pulse is the most efficient backwash method for particle removal, and while ETSW is able to wash away backwash remnant particles, it is likely that insufficient numbers of collectors remain on the

media to remove subsequent particles entering the filter.

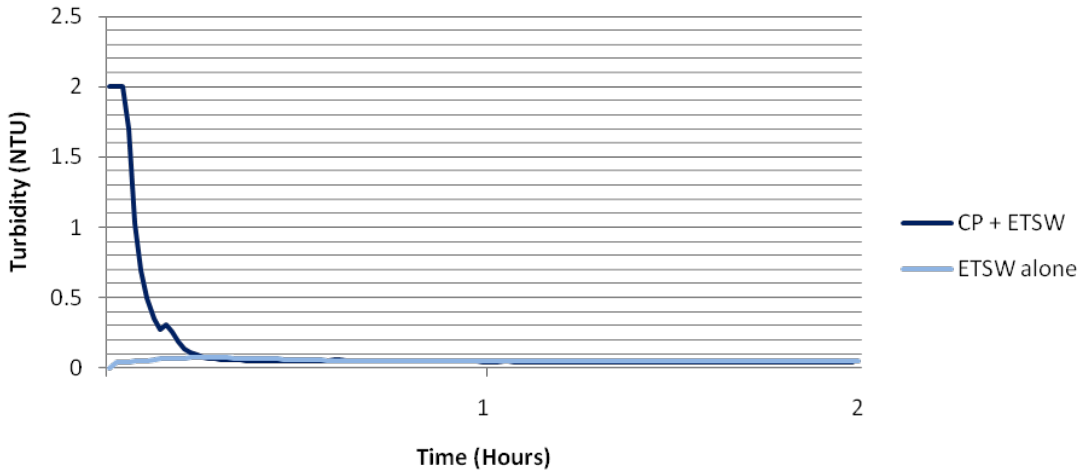

Figure 4.11 The impact of CP on ripening time when ETSW is used.

Although ETSW is unable to fully eliminate filter ripening when chlorinated collapsed pulse is used, it is clear from figure 4.12 that ripening is significantly reduced. Figure 4.12 shows filter two effluent turbidity over the course of a week of operation. Early in the week, two collapsed pulse backwashes result in traditional ripening spikes that reach the turbidimeter upper detection limit of 2 ntu, and do not achieve turbidity below 0.2 ntu for approximately 40 minutes. Later in the week, when the same backwash procedure is used with the addition of an ETSW step, peak ripening is reduced to 0.5 ntu and the filter is able to be put back in to service after 6 minutes of ripening. The ability of non-chlorinated collapsed pulse to extend filter run time may compensate for this brief ripening spike when non-chlorinated wash water is used.

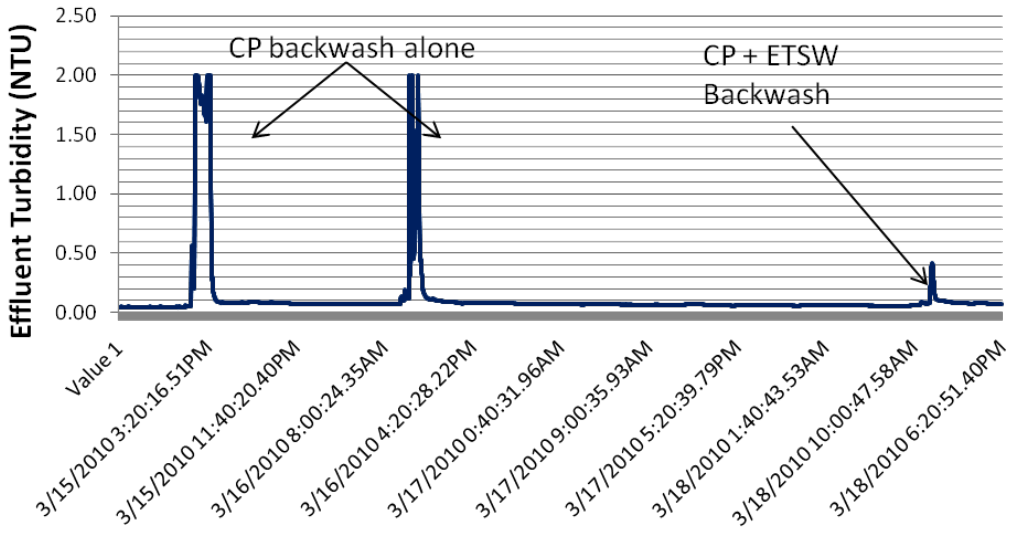

Figure 4.12: Effectiveness of ETSW backwash in mitigating the negative effect of collapsed pulse on filter run time

Summary of Factors Affecting Filter Ripening Time

Like filter run time, ripening time was unaffected by media characteristics. Larger media offered no advantage over smaller media, low uniformity coefficient offered no advantage over high uniformity coefficient, and GAC offered no advantage over anthracite. The only factors found to have a significant effect on ripening were collapsed pulse and ETSW.

Collapsed pulse, as expected, was found to increase filter run time. The efficiency with which collapsed pulse removes particles from the filter media also leads to a deficiency of collectors remaining at the start of the subsequent filter run. This effect was significantly mitigated by the use of an ETSW wash step.

ETSW had the expected effect of significantly reducing filter ripening time. In many runs, ETSW was able to eliminate filter ripening entirely. When chlorinated collapsed pulse was used however, filter ripening time was extended slightly. However, ETSW was able to mitigate this effect, reducing filter ripening to the range of 5-20 minutes from 20-24 minutes when no ETSW is used. Overall, ETSW was highly effective at reducing ripening for all

media configurations, regardless of backwash procedure or the presence of chlorine in the wash water.

Table 4.23: Summary of impacts of backwash technique and media characteristics on filter run time in biological filters

Parameter	Effect	Notes
CP	↑	Collapsed pulse increases ripening time (mitigated by ETSW)
ETSW	↑	ETSW significantly shortens, and often eliminates filter ripening entirely.
Absence of Chlorine	None	Chlorine interacted with collapsed pulse shorten filter run times
GAC vs Anthracite	None	
Media Size	None	
Uniformity Coefficient	None	

4.6 Summary of Pilot Study Findings

The goal of the pilot factorial experiments was to identify the impact of backwash techniques, media characteristics, and interactions between the two on biological filter performance. Several factors of all three categories capable of both improving, and impairing biological performance were identified. Collapsed pulse in particular was identified as impairing each of the biological and traditional measures of performance used in this study.

Table 4.24: Summary of pilot study findings

	DOC	BDOC	BRP	Run Time	Ripening Time
CP	↓/↑[a]	↓/↑[a]	↓/↑	↓/↑[b]	↑
ETSW	↑	↑	↓/↑	N.E.	↓
Chlorine	↓	↓	↓	↓	N.E
GAC vs Anth	↑	↑	↑	N.E	N.E

	DOC	BDOC	BRP	Run Time	Ripening Time
Size	N.E	N.E	N.E	↑	N.E
U.C.	N.E	N.E	N.E	N.E	N.E

a – Negative immediately following backwash, but positive after 24 hours.

b – Chlorinated results in shortened filter run times, non-chlorinated extended.

N.E. – No Effect

4.6.1 Collapsed Pulse

Collapsed pulse was found to have the most significant effect on biological filter performance of all design and operational parameters studied. The majority of the negative impacts of collapsed pulse were the result of interaction with chlorine. Chlorinated collapsed pulse significantly shortens filter run time due to a sudden precipitous breakdown in turbidity removal at approximately 15-25 hours in to the filter cycle. This phenomenon is observed in both anthracite and GAC filters, and is thought to be the result of sections of biofilm, damaged during the course of collapsed pulse backwashing sloughing from the media. A follow up study in which effluent EPS concentrations in parallel filters backwashed with chlorinated and non-chlorinated water respectively would reveal whether or not biofilm sloughing is indeed the cause of this surge in turbidity.

Collapsed pulse backwashing also impaired biological performance, specifically immediately following backwash of anthracite filters. This effect was not detected after 24 hours however, and further study is necessary to determine the duration of impaired performance. In some cases, collapsed pulse backwashing actually had a positive effect on DOC and BDOC removal, as well as BRP after 24 hours of filter operation; but only when non-chlorinated wash water was used. A proposed mechanism for this effect is increased distribution of bacteria, and concurrent increase in biofilm surface area after sufficient time has passed.

In addition to biological performance and filter run time, collapsed pulse extended filter ripening time as well. This result was expected due to the efficient particle removal

associated with collapsed pulsing. The duration of ripening was significantly reduced when collapsed pulse was used in conjunction with ETSW.

These findings indicate that the use of collapsed pulse to backwash biological filters must be re-evaluated, particularly when chlorinated wash-water is used. That is not to say that the collapsed pulse backwash does not hold promise however, biological performance in GAC filters is highly resilient to the effects of collapsed pulse. There is also some indication that collapsed pulse may actually lead to improved biological performance later in the filter cycle as a result of the re-distribution of biomass and a consequently larger biofilm surface area. Further study is necessary however, to determine the duration of impaired biological performance immediately following backwash of anthracite filters in cold water, and to determine if this effect is observed in warm water.

4.6.2 ETSW

While collapsed pulse was found to have some unexpected negative impacts on biological filter performance, ETSW was found to have several unexpected positive impacts. ETSW had the expected effect of significantly reducing, and sometimes eliminating filter ripening as well as mitigating the negative effects of collapsed pulse on filter ripening time. This indirectly extended filter run time by converting previously wasted water during filter ripening into production water.

The unexpected result however, was the positive effect associated with ETSW on DOC and BDOC removal and increased BRP both immediately following backwash and 24 hours in to the filter cycle. In particular, ETSW was associated with partially mitigating the negative effects of the collapsed pulse backwash on biological performance. No previous study of the effect of ETSW on biological filter performance exists. It is speculated that the gentle, prolonged ETSW wash step may be associated with re-distribution and increased re-attachment of microbes following backwash, leading to a more homogenous and widely distributed biofilm later in the filter cycle. Further study, potentially of phospholipid biomass distribution in the filter is necessary to determine whether this is the case.

4.6.3 Presence of Chlorine in Wash Water

The impact of chlorine on biological filter performance is complex, and typically the result of interaction with backwash technique or media type. The most dramatic impact chlorine had on biological performance was the breakdown in turbidity removal associated with chlorinated collapsed pulse. This effect seemingly precludes the use of collapsed pulse in combination with chlorinated wash water on biological filters.

Chlorine was also found to directly impair DOC removal in anthracite filters, as well as to hinder the ability of filters to increase their DOC and BDOC removal rates as the filter cycle progressed. Chlorine also hindered the ability of anthracite filters to recover from the damaging effects of collapsed pulse backwashing.

When all of the direct and indirect effects of chlorinated wash water on biological filters are summed it becomes evident that chlorine should no longer be considered an adequate biomass control measure. If a conversion to non-chlorinated wash water is infeasible, GAC would likely be the best choice of filtration media, as it tended to be more resistant to the negative effects of chlorinated backwashing. Collapsed pulse backwashing would also have to be ruled out in the case of chlorinated backwashing due to the associated breakdown in turbidity removal. Due to the limitations imposed by chlorinated backwashing, a provision should be made for non-chlorinated wash water in future biological filter design.

4.6.4 GAC vs Anthracite

Like chlorinated wash water, the advantage of GAC over anthracite is complex, the main advantage of GAC is its resilience to vigorous backwashing. GAC was found to have a positive effect on each of the three biological performance parameters quantified. Upon closer inspection of the data in the case of DOC and BDOC however, this was largely due to the resilience of GAC to collapsed pulse backwashing and chlorinated wash water. Further study is necessary to determine the period of impaired DOC and BDOC removal following non-chlorinated collapsed pulse of anthracite. If this impaired performance extends

significantly beyond the filter ripening period this would preclude the use of collapsed pulse backwashing on anthracite filters.

Overall GAC was shown to be a more flexible choice for support media, allowing for a more vigorous backwash, however GAC filters were equally susceptible to a the breakdown in turbidity removal associated with chlorinated collapsed pulse backwashing. GAC was also found to have a higher BRP than anthracite in parallel runs with a variety of backwash techniques. Although the impact of surges in organic loading was not tested in this study, it is possible that the excess respiration potential possessed by GAC may provide an advantage in responding to such surges.

It remains to be seen whether this flexibility provides performance benefits commensurate with the increased cost associated with GAC. One unit of GAC can cost up to four times more than a comparable unit of anthracite in start up costs, and is then subject to increased operations and maintenance cost as a result of media attrition due to the friability of GAC. At full-scale, and in many runs at pilot-scale anthracite was found to perform as well as GAC, and as discussed in chapter 2, much of the advantage of GAC over anthracite reported in the literature is the result of the fact that GAC is much more resistant to chlorinated backwashing (Krasner et al., 1993; Liu et al., 2001; Emelko et al., 2006), cold operating temperatures (Emelko et al., 2006; Liu et al., 2001), and the presence of oxidants in the influent (Niquette et al., 1998, Wobma et al., 2000). This suggests that the differences in BOM removal observed between GAC and anthracite can be minimized through optimized filter operation.

4.6.5 Effective Size

One of the key research goals of this study was to determine whether or not filter throughput could be increased by increasing media size, and to determine what impact this would have on DOC and BDOC removal. The results of the pilot study indicate that media size does extend filter run time by preventing terminal headloss accumulation while providing similar turbidity removal performance to the smaller filter media used in this study.

It was also found that neither DOC nor BDOC removal are compromised by the associated decrease in EBCT, even in the cold water conditions under which this study was carried out. The implications of this finding are that filter throughput of biological filters can be increased substantially without compromising turbidity removal or biological performance. Further study is necessary however to determine the ability of larger media filters to respond to both hydraulic surges and surges in organic loading.

4.6.6 Uniformity Coefficient

The uniformity of the grain size distribution of a media can have significant cost implications. The findings of this study however, indicate that the associated performance implications are not proportional. Uniformity coefficient was not found to impact any of the traditional or biological measures of performance examined in this study.

Chapter 5 – Full-scale Analysis

The impact of media type (GAC and anthracite) on both traditional and biological filter performance was studied at full-scale at the MWTP. Traditional performance was monitored by the SCADA system, which provided minute by minute turbidity, head loss, and flow rate data throughout the course of the three year study period. Biological performance was assessed during three sampling periods, which are summarized in table 5.1. To assess the impact of backwash on biological filter performance, samples were collected immediately following backwash and after 24 hours of operation during sampling periods 1 and 2.

Table 5.1: Sampling periods for full-scale biological performance assessment

Sampling Period	Dates (dd/mm/yyyy)	Parameters Sampled
1	16/10/2008 – 07/05/2009	BDOC
2	03/06/2009 – 18/06/2009	BDOC, phospholipid biomass, BRP
3	15/03/2010 – 23/03/2010	chlorine demand, THM-FP

5.1 Phospholipid Biomass

As discussed in chapter 2, phospholipid biomass (PLB) frequently does not correlate to biological filter performance (i.e. BOM removal). Nonetheless, top of filter PLB was measured in parallel with BRP and BDOC removal during sampling period 2 in order to generate a comprehensive assessment of the impact of backwash, media type, and media size on the quantity, state, and activity of supported filter biomass. The results of PLB sampling are presented in figure 5.1.

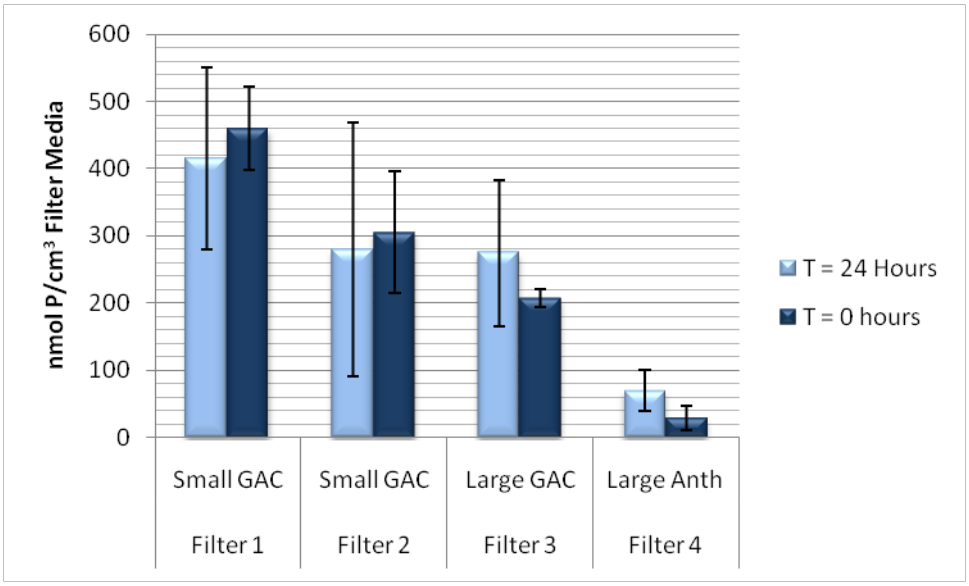

Figure 5.1: Top of filter phospholipid biomass in MWTP full-scale biological filters (mean ± standard deviation)

PLB concentrations in the all of the GAC filters were generally comparable, regardless of media size and time of sampling (i.e., immediately after backwash or after 24 hours of operation). The highest measured PLB concentration was on the small GAC in filter 1 of 460 nmol p/cm^3 of GAC. Consistent with previous literature detailing a different full-scale investigation at the MWTP (Emelko et al., 2006), the anthracite filter supported the lowest concentration of PLB. This is consistent with previous studies, which have also noted lower concentrations of PLB supported by anthracite relative to other filter media (Wang et al., 1995, Huck et al., 2000, Miltner et al., 1995). The mean measured PLB concentration on the anthracite filter studied herein was approximately 20 nmol p/cm^3 immediately following backwash, and approximately 60 nmol p/cm^3 twenty-four hours in to the filter cycle. This concentration falls within the range of the cold water, anthracite associated PLB concentrations of approximately 5 – 30 nmol p/cm^3 previously reported at the MWTP under a different filtration regime (Emelko et al., 2006). It also falls within the range of other reported PLB concentrations on anthracite filters of 64.8 ± 6.4 nmol p/cm^3 (Wang et al., 1995), and 80 ± 20 nmol p/cm^3 (Urfer et al., 2001). The mean measured anthracite associated

PLB concentration at the MWTP of 60 nmol p/cm^3 was lower than the concentration range of 75 – 125 nmol p/cm^3 reported by Huck et al. (1998), but this would be expected given that those measurements were made at warm water conditions.

In contrast to the anthracite associated PLB concentrations that were consistent with those previously measured at the MWTP, the mean GAC associated biomass concentrations, were approximately 460 nmol p/cm^3 in filter 1 and 200 nmol p/cm^3 in filter 2 (figure 5.1). These values are substantially higher than those previously reported for MWTP filters 1 and 2, which ranged from 2.5 to 15 nmol p/cm^3 for the filters, which had the same filter configurations and media characteristics, but different operating conditions (Emelko et al., 2006). The anthracite associated PLB concentrations in that study were also significantly higher than those observed on the GAC media, indicating that some other factor may have influenced PLB concentrations on the GAC filters. The range of 200 - 460 nmol p/cm^3 measured on the GAC filters in the present study is in agreement with other reported PLB concentrations on GAC filter media: 305 – 465 nmol p/cm^3 (Wang et al., 1995) and 175 nmol p/cm^3 (Persson et al., 2006).

It should also be noted that filters 1 and 2 and the MWTP contain essentially the same media, are configured in the same manner, and receive the same sample influent water; in essence, these filters represent full-scale replicates. Nonetheless, the PLB concentrations on top of filters 2 and 3 were quite similar, whereas the amount of PLB accumulated on the top of filter 1 was notably higher than on either of the other GAC filters (figure 5.1). The ongoing QA/QC program associated with this work confirmed that the PLB concentrations were correctly associated with the appropriate filters. While the necessity of sampling at 24 hours in to the filter cycle and immediately following backwashes precluded concurrent PLB (as well as BRP and BDOC) sampling because it would necessitate all four of the MWTP's full-scale filters to be out of service simultaneously, filters 3 (large GAC) and 4 (large anthracite) were sampled concurrently. Temporal variability in filter influent water quality may explain the differences in PLB observed in filters 1 and 2; however, the MWTP's source water is held in a reservoir that tends to equalize (or prevent by closing the intake) sudden

changes in source water quality. It is also possible that some other operational (e.g., difference in coagulant dosing) or sampling (e.g., spatially variability of biomass distribution on filter media surface) factor influenced the observed differences in top of filter PLB on filters 1 and 2. These differences underscores the uncertainty (e.g., due to temporal differences in water quality, spatial variability of biomass distribution, etc.) inherent to PLB sampling and analysis of full-scale biological treatment process performance; results should therefore be interpreted with consideration of this uncertainty.

Top of filter PLB remained relatively unaffected by backwash (figure 5.1). Previous studies of top of filter biomass at the MWTP reported a similar trend, with PLB on both anthracite and GAC filters being relatively unaffected by backwash when the impact of air scour was investigated (Emelko et al., 2006). Huck et al. (1998) observed a similar trend in anthracite and GAC PLB on filtration media at the Metropolitan Water District oxidation demonstration plant. Others have reported significant loss of PLB subsequent to chlorinated backwash (Lu and Huck, 1993; Miltner et al., 1995), thereby underscoring that multiple factors should be investigated to better understand the quantity, state, and activity of supported filter biomass in biological filtration applications. In the present study, backwash did not remove a significant fraction of attached top of filter PLB on either GAC or anthracite filters.

Not surprisingly, the PLB concentration supported on the GAC media in the present investigation significantly exceeded that which was supported on the anthracite media. In final consideration, however, the quantity of biomass supported by the different media types and the impact of backwash on PLB concentrations has little operation significance unless it is associated with a filter performance metric such as BOM removal, head loss accumulation, or turbidity breakthrough. It has been frequently observed in the past however, that differences in filter PLB concentrations do not necessarily correlate to differences in filter BOM removal; for this reason, biological respiration potential, , and BDOC removal were quantified in parallel with PLB throughout sampling period 2. These results are discussed in sections 5.2 and 5.3 respectively.

5.2 Biological Respiration Potential (BRP)

While PLB provided an assessment the quantity of biomass supported by the different filter media configurations, BRP was simultaneously assessed to provide an assessment of the state of the biomass on the filter media by assessing the potential of the media to degrade BOM compounds. The results of the BRP data are presented in figure 5.2.

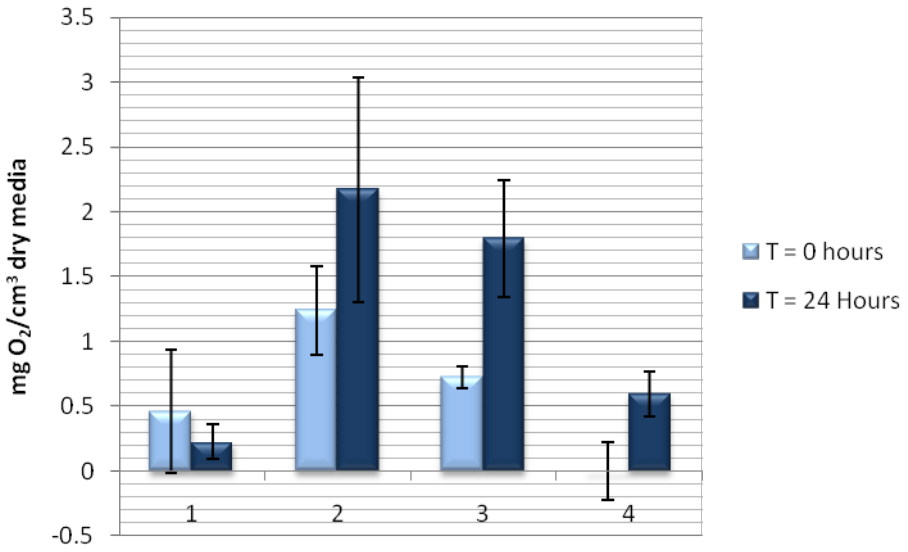

Figure 5.2: Top of filter BRP in full-scale MWTP filters (mean ± standard deviation)

The BRP results all fall within the previously reported range of 0.2 – 1.94 mg O_2/L cm^3 for anthracite filters (Urfer et al., 1997). While Persson et al. (2006) evaluated GAC associated BRP, testing was discontinued due to control samples, inactivated with sodium azide or formaldehyde exhibiting significant oxygen consumption. In the present investigation, controls were inactivated by autoclave. No discernable difference between the media types was detected. Mean detected oxygen consumptions of inactivated samples were 0.05 and 0.07 mg O_2/L cm^3 for GAC and anthracite associated biomass respectively

As with the pilot-scale study, GAC media supported a higher BRP than anthracite media. This finding is consistent with the higher quantity of PLB supported in the GAC filters (figure 5.1). Although higher concentrations of PLB do not necessarily correlate to higher BOM removals, a higher BRP would be expected in association with higher levels of PLB (i.e. higher numbers of cells). The negative impact of backwash on BRP is evident in figure 5.2 in which BRPs immediately after backwashing (T = 0) are generally quite low, particularly in the anthracite filter. This outcome is not consistent with the PLB data, which

showed only a minimal impact of backwash on attached top of filter biomass. Consideration of both the BRP and the PLB data indicates that while the quantity of biomass detached during backwash may not have been significant, the state of the attached biomass was negatively impacted by backwashing. This finding is consistent with the findings of the pilot factorial, which indicated a negative impact of chlorine on biological activity (table 4.13, figure 4.16). The operational impact of this reduced activity levels on BOM removal by the filters is discussed below in sections 5.3 and 5.4.

Like with the PLB data presented above in section 5.1 and despite filters 1 and 2 representing replication filtration conditions, it should be noted that the BRPs of filters 2 and 3 were quite similar, whereas the filter 1 BRP was notably the lowest of the filters studied (generally comparable to that on top of filter 4; figure 5.2). Moreover, the filter 1 BRP had the lowest level of measured BRP (figure 5.2) despite the highest level of measured PLB (figure 5.1); this result is counter-intuitive as a higher BRP would be expected in conjunction with higher PLB (i.e., numbers of cells). As discussed in section 5.1, it also underscores the possible uncertainties inherent to PLB and BRP sampling and analysis of full-scale biological treatment process performance; alternatively it may be indicative of the influence of an unaccounted factor in either filter 1 or 2.

5.3 BDOC Removal

Analysis of the PLB and BRP data provided a picture of the quantity and the state of the biomass on the different media configurations, but neither parameter necessarily correlates to filter BOM removal, which is the critical performance goal of biological filtration. BDOC removal represents the full extent of biologically mediated DOC removal occurring through the filter, and as such is a good parameter for quantifying the ability of the filters to remove BOM. BDOC sampling was conducted in two phases. Cold water BDOC removal was assessed for each filter via monthly sampling from October 2008 to May 2009 during the first sampling phase. The second phase of BDOC sampling was conducted in parallel with BRP and PLB sampling to generate a comprehensive assessment of the quantity, state, and activity of the biomass supported on each of the filter media

configurations. Data from sampling periods 2 and 1 are presented in figures 5.3 and 5.4 respectively.

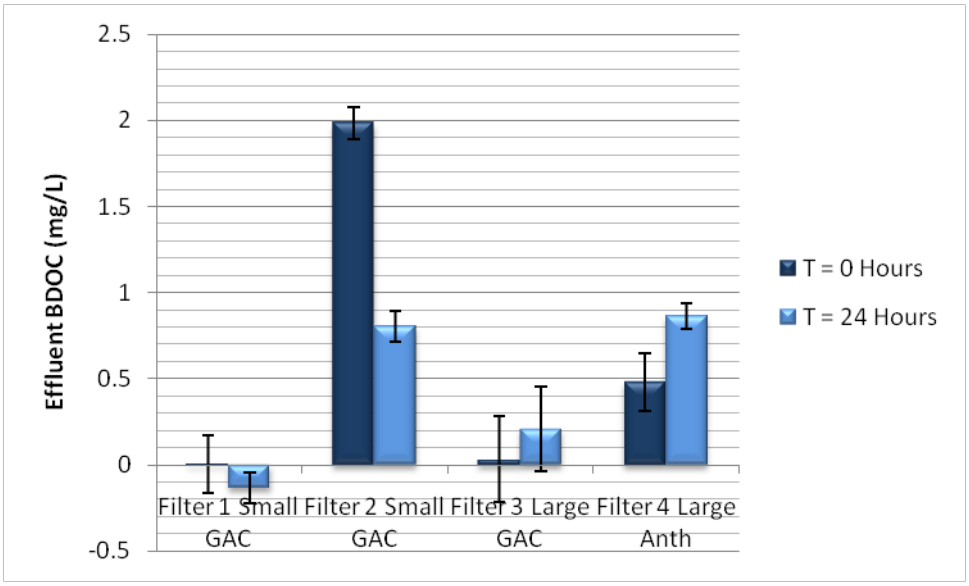

Figure 5.3: Effluent BDOC concentrations during sampling period 2 (mean ± standard deviation)

Consistent with the results presented in section 5.1 and 5.2, figure 5.4 highlights the uncertainty associated with BDOC sampling and analysis of full-scale treatment processes. Despite identical media configurations in filters 1 and 2 (small GAC), the filters yielded significantly different effluent BDOC concentrations (figure 5.3). Due to the necessity of sampling immediately after backwash, it was not possible to sample all four filters simultaneously, as this would result in all full-scale filters being taken simultaneously offline. For the purpose of GAC and anthracite comparison however, filters 3 (large GAC) and 4 (large anthracite) were sampled simultaneously, and can therefore be compared. The observed results are consistent with the BRP data, which indicated higher concentrations of active biomass on filter 2 relative to filter 1 (figure 5.2). Although the BRP data during sampling period 2 indicated a negative impact of backwash that improved as the filter cycle progressed (figure 5.2), the opposite trend was observed in BDOC removal (figure 5.3). In

both filters three and four, BDOC removal was highest immediately following backwash and decreased by 24 hours in to the filter cycle. This effect is consistent with the pilot data, which showed that BDOC removal performance decreased 24 hours in to the filter cycle following chlorinated backwash (figure 4.4, table 4.8). Reduced BOM removal by filtration in later portions of the filter cycle has been observed in other biological filtration studies as well (Carlson et al., 1996; Prevost et al., 1995). Carlson et al., (1996) attributed this effect to reduced contact time due to solids and floc loading on the filter. Prevost et al. (1995) hypothesized that backwashing improved BOM removal by removing inhibitory substances (such as potentially toxic aluminum flocs or solids inhibiting oxygen and nutrient diffusion into the biofilm). The findings of the present study support a third hypothesis: increased BOM passage late in the filter cycle may be associated with sudden increases in turbidity passage 24 hours after chlorinated backwash (as observed in the pilot factorial experiments, figures 4.7-4.9, tables 4.15-4.17). If this turbidity passage is the result of sloughing and/or erosion of biofilm sections damaged by chlorinated backwash, a concurrent increase in BOM passage would be expected. This hypothesis, though far from incontrovertible, is supported not only by the observed decrease in BDOC removal at full-scale, but also by the observed decrease in BDOC removal during the pilot experiments after 24 hours of filter operation following chlorinated backwash.

In addition to supporting a higher BRP and phospholipid biomass concentration, GAC appeared to also remove a slightly larger quantity of BDOC during both sampling period two (figure 5.3), and sampling period one (figure 5.4). These findings are consistent with a previous study of full-scale BDOC removal at the MWTP under a different filtration regime, where effluent concentrations ranged from 0.25 – 1.3 mg/L for a GAC filter and 0.2 – 1.4 mg/L for an anthracite filter (Huck et al., 2000). The results of sampling period one however, carried out from October 2008 through March 2009, must be interpreted with caution. During that period of time the University of Waterloo TOC analyzer was out of service and all BDOC samples were processed at a commercial laboratory. Control samples analyzed at the laboratory indicated carbon contamination, likely due to the use of cellulose

acetate filters, which have been found to contribute organic carbon to filtrate (Khan and Subramania-Pillai, 2007). Milli-Q^{TM} water blank samples yielded an average concentration of 0.23 (±0.15) mg/L (n = 3), and 3.0 mg/L standard samples yielded an average concentration of 3.54 (±0.15) mg/L (n=7). Given that measured DOC concentrations during sampling period 2, during which time reliable BDOC data was obtained by analyses at University of Waterloo laboratories, were typically below 0.5 mg/L, the degree of contamination at the commercially laboratory renders BDOC measurements during that sampling period highly questionable. Regardless of the contamination however, a similar trend in BDOC removal was observed, with GAC removing a slightly higher percentage of BDOC than anthracite. In all cases however, the difference in removal between the GAC and anthracite filters was less than 1 mg/L.

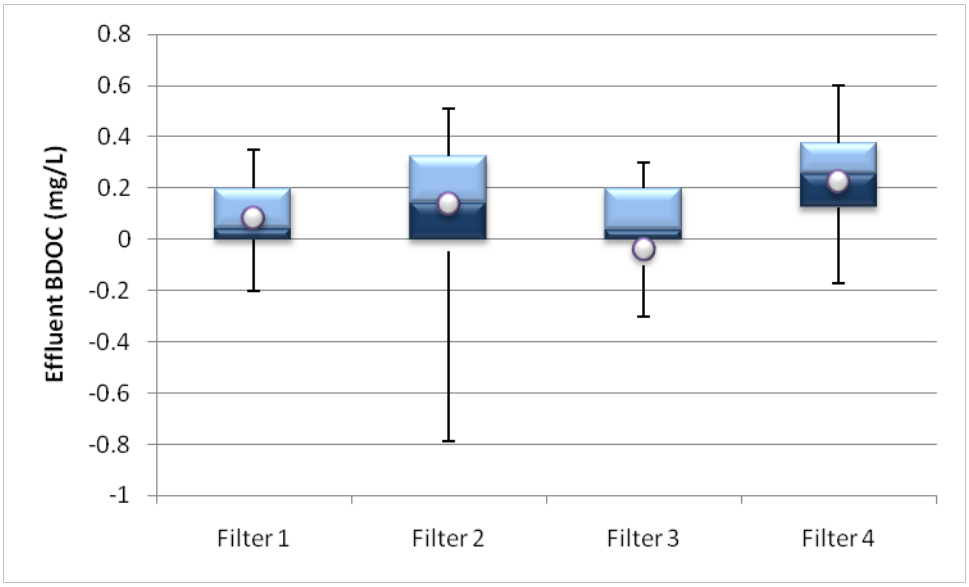

Figure 5.4: Boxplot of filter effluent BDOC through sampling period 1, winter 2009. The light blue box represents BDOC removal values from the median to 75th percentile, the dark blue box from the 25th percentile to the median, and the whiskers the maximum and minimum values observed.

The slightly higher BDOC removals observed in the full-scale, larger media GAC filter are consistent with the pilot results, which indicated that the use of chlorinated backwash in anthracite filters had a negative effect on BDOC removal, while the GAC filters remained resilient. This observation is consistent with others that have demonstrated that GAC filtration tends to remove a higher percentage of BOM than anthracite when chlorinated backwash water is used, particularly at cold water conditions (Liu et al., 2001; Emelko et al., 2006; Miltner et al., 1995). The significance of this difference in BOM removal between GAC and anthracite is unclear, however. To further explore the impact of the relatively small observed difference in BDOC removal between the media types, the removal of chlorine demand and THM precursor molecules was studied during sampling period three.

5.4 Total Tri-Halomethane Formation Potential and Chlorine Demand Removal

Measurement of BDOC removal allowed for an assessment of the full extent biologically mediated BOM removal in each filter. GAC was found to support both a higher concentration of PLB, as well as a higher BRP. This higher quantity of more active biomass was also associated with slightly higher BDOC removal in the GAC filters relative to the anthracite. BDOC removal in the larger filter media filters also decrease after 24 hours of filter operation. In these cases; however, the observed differences in BDOC removal were typically less than 0.5 mg/L, and not statistically significant. Moreover, the operational significance of this difference was unclear. To more exhaustively assess the impact of these small differences in BDOC removal, chlorine demand and THM-FP removal were quantified during sampling period 3. THM-FP removal data are presented in figure 5.5. Chlorine demand removal data are presented in figure 5.6.

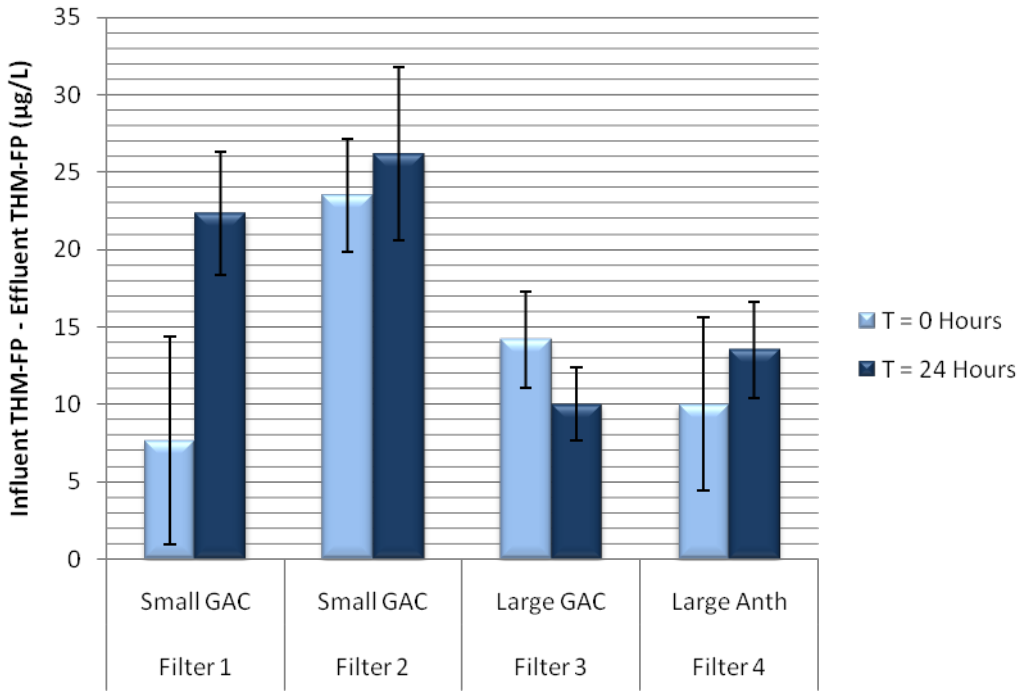

Figure 5.5: Removal of THM-FP through MWTP full-scale biological filters (mean ± standard deviation)

THM-FP removal through the full-scale biological filters ranged from approximately 5 – 14%. These findings are lower than ranges that have been reported in other studies of THM-FP removal of 27-40% (Wang et al., 1995) and 17.5-19% (Huck et al., 2000); however, those studies reported THM-FP in filters backwashed with non-chlorinated water and operated at warm water conditions . No performance differences in THM-FP removal were observed between the large size GAC and anthracite filters. This observation is consistent with Wang et al. (1995) and Huck et al. (1998), indicating that small differences in mean BDOC removal between the large GAC and anthracite filter media were not necessarily indicative of any discernible differences in the removal of THM precursor molecules. Although, as the pilot-scale data indicated that chlorinated backwash can impair BOM removal by biological filtration for a period of up to 24 hours in to the filter cycle

(figure 4.4, table 4.8), the impact of chlorine on full-scale THM-FP potential could not be assessed because a non-chlorinated backwash stream is unavailable at full-scale at the MWTP.

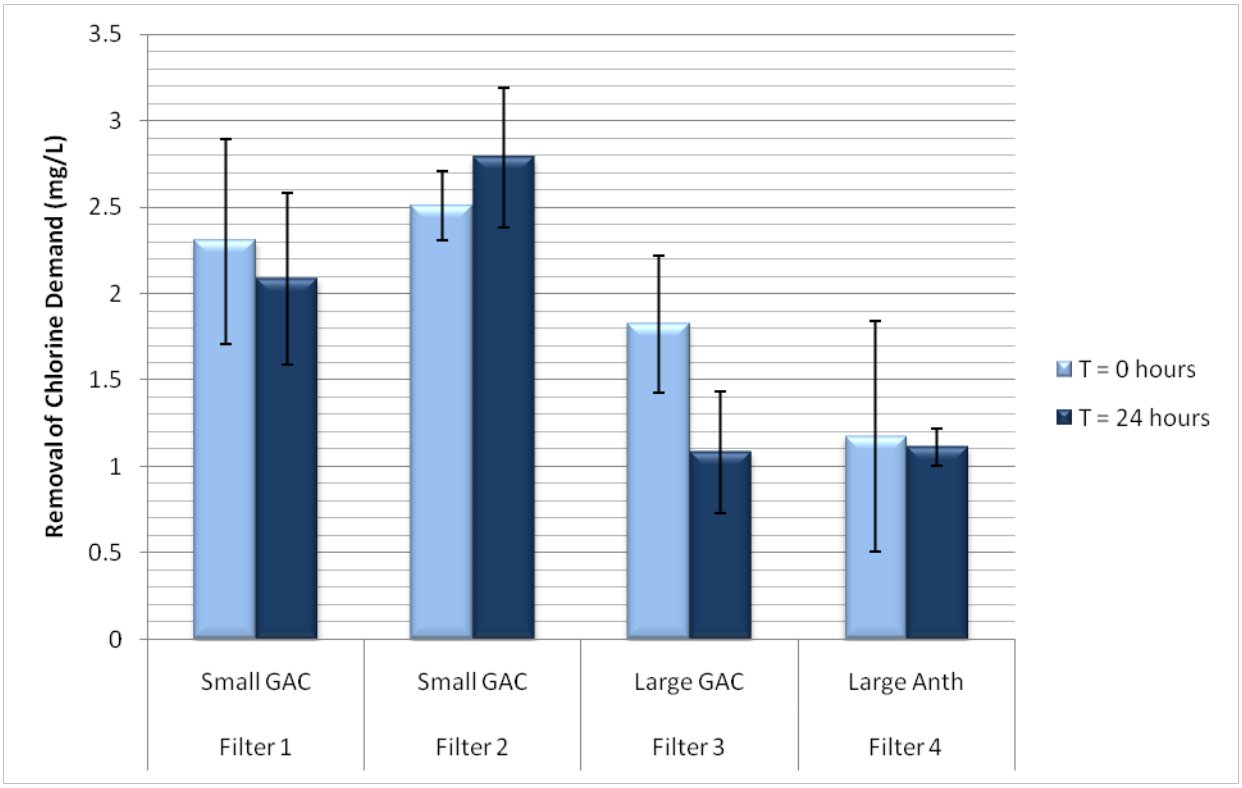

Figure 5.6: Reduction of chlorine demand through full-scale MWTP biological filters (mean ± standard deviation)

Removal of chlorine demand through the MWTP filters ranged from 7 to 25%, which slightly lower than the range reported by Huck et al. (1998) of 25-30%; this is likely due to the fact that Huck et al. (1998) measured chlorine demand removal under warm water, non-chlorinated backwash conditions. Removal of chlorine demand by the MWTP full-scale filters followed a similar pattern to THM-FP removal. No differences were observed between GAC and anthracite, which is also consistent with Huck et al. (1998). With the exception of

Huck et al. (1998) very limited data on comparisons of the removal of chlorine demand in GAC and anthracite filters

The impact of backwash on the removal of chlorine demand has also not been previously reported. Similar to THM-FP removal, the removal of chlorine demand through the full-scale MWTP filters was unaffected by backwash. This indicates that the slight reduction in BDOC removal observed 24 hours in to the filter cycle does not affect the removal of chlorine demand.

5.5 Filter Ripening Time

Operational data for each of the MWTP full-scale filters was provided by RMOW staff. Filters three and four were put in to operation in the spring of 2007. Figure 5.7 summarizes the data for the ripening period following every backwash carried out from the installation date until October 2010. At full-scale, filter ripening is defined as the period of time from the moment the filter effluent valve is opened to the time at which ten consecutive minutes of water turbidity below 0.2 ntu are recorded by the SCADA system. This definition results in ripening times at full-scale being approximately ten minutes longer than pilot-scale due to the fact that ripening at pilot-scale was determined by manual inspection of the data. The data is split in to cold water ripening (below 7^{o}C) and warm water ripening (above 7^{o}C).

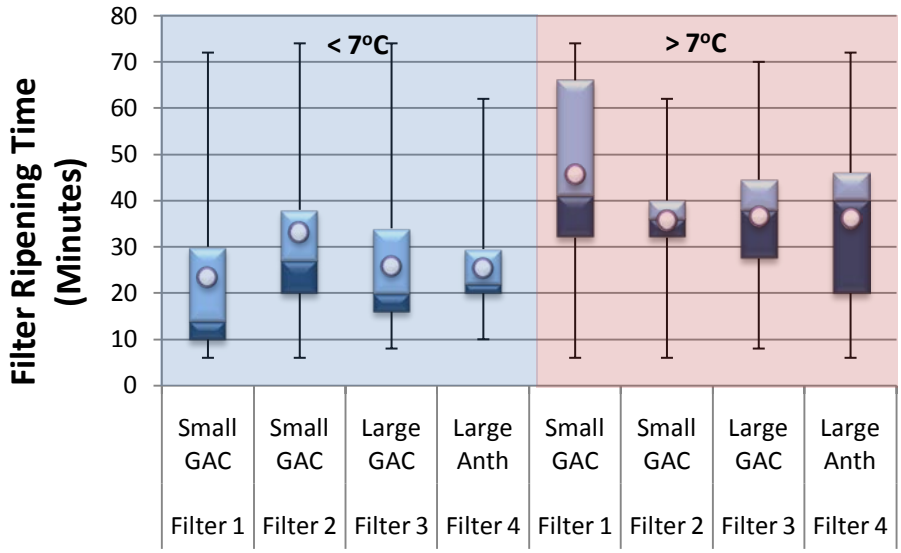

Figure 5.7: Boxplot of full-scale filter ripening throughout the study period (2007-2010). The light blue box represents BDOC removal values from the median to 75th percentile, the dark blue box from the 25th percentile to the median, and the whiskers the maximum and minimum values observed.

Ripening times were significantly longer during warm water conditions. This could be the result of different influent water quality, coagulant dosing, or some combination of the two. In both warm and cold water conditions, no difference in ripening time was observed between the GAC and anthracite filters. The significant differences between filters one and two give some indication of the variability of ripening time, both filters contain an identical media configuration. Media type was not found to have a significant impact on filter ripening time.

The impact of backwash on filter ripening time was not evaluated at full-scale, due to potential impacts on production water quality. Given the promise of ETSW at pilot-scale however, future work on implementing this backwash at full-scale is currently underway.

5.6 Filter Run Time

Similar to figure 5.7, figure 5.8 contains filter run time data for each filter run carried out between Spring 2007 and Fall 2010. Filter run time at the MWTP is dictated by the four

factors outlined in table 5.1. If any of the critical parameters in table 5.1 are exceeded, a backwash is initiated. Backwashes are also often initiated when none of the above parameters are exceeded to avoid the necessity of having several filters out of service simultaneously, a practice called staggering.

Table 5.2: Critical parameters for filter cycle termination

Factor	Termination Criteria
Head Loss	75%
Turbidity	> 0.1 ntu
Time	60 hours

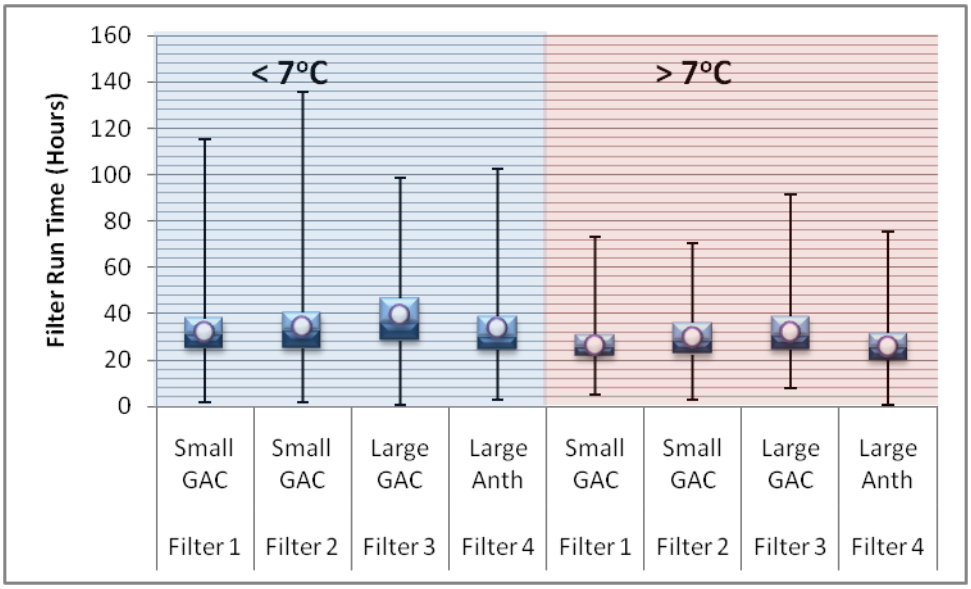

Figure 5.8: Boxplot of MWTP filter run time from Spring 2007 to Fall 2010. The light blue box represents BDOC removal values from the median to 75th percentile, the dark blue box from the 25th percentile to the median, and the whiskers the maximum and minimum values observed.

The data in figure 5.8 indicates that the majority of filter runs are the same for each of the filter media configurations. The mean, and upper 75th percentile runs of filter three, the large GAC filter, however, were approximately eight hours longer than those of the comparable anthracite filter. This was an unexpected result, given that the media effective sizes and uniformity coefficients are roughly matched. Because filter run time is such a broad parameter, figure 5.8 does not provide information regarding whether head loss or turbidity passage is responsible for the abbreviated filter run times of filter 4. An analysis of backwash terminators, presented in table 5.2 however, does.

Table 5.3: Primary initiators of backwash during study period

Filter	Media Type	Effective Size	Head Loss	Turbidity	Other
Filter 1	GAC	1.06	63	17	20
Filter 2	GAC	1.05	62	12	26
Filter 3	GAC	1.3	8	70	22
Filter 4	ANTH	1.3	40	39	20

From table 5.2, it can be seen that the abbreviated filter runs in the anthracite filter are the result of a greater susceptibility to head loss. 8% of the large GAC filter runs were terminated due to terminal head loss while 40% of anthracite filter runs were terminated due to head loss. There are two potential mechanisms for this discrepancy: increased EPS production by microorganism growing on the anthracite filter in response to larger hydraulic shear forces, or as a result of media shape. Media shape is unlikely to contribute to increased head loss in the anthracite filter. The more friable GAC filter media would be expected to become increasingly spherical after undergoing extensive grain on grain abrasion through filter backwashing. The harder anthracite however would be expected to maintain its shape longer than the GAC filter. Given that filter bed porosity decreases in response to increasing sphericity (Crittenden et al., 2005), the GAC filter would be expected to be subject to increased head loss if shape were a factor. The more likely hypothesis, is that biofilm microorganism on the anthracite filter are producing a higher quantity of EPS in response to the increased hydraulic shear forces experienced by biofilm bacteria growing on the much smoother anthracite media. Overall however, the vast majority of filter runs in the large anthracite and large GAC filters are in the same range, which is consistent with the findings of the pilot study, where media type was not found to play a significant role in filter run length.

Another key finding was that the larger media sizes were capable of equal levels of turbidity removal to the smaller media, which is consistent with the findings of the pilot factorial. This indicates that media size can be increased to extend filter run times by

decreasing filter head loss without sacrificing filter turbidity, THM-FP, chlorine demand, and BDOC removal.

Chapter 6 – Conclusions

The overall goal of this research was to determine the impact of media characteristics, backwash technique, and the interaction between the two on biological filter performance. Three pilot-scale factorial experiments were conducted. Three full-scale experiments were also conducted during a three-year period of traditional full-scale performance evaluation. The following conclusions can be drawn (for the conditions studied) from those investigations:

6.1 Effect of Media Type on Biological Filter Performance

1. Changing media effective sizes from 1.0mm and 1.3mm did not deleteriously impact either turbidity or BOM removal by biological filtration; however, the larger media virtually eliminated terminal head loss accumulation.
2. A decrease in anthracite uniformity coefficient from 1.6 to 1.3 did not impact either traditional or biological filtration performance at pilot-scale.
3. Although GAC and anthracite filters were capable of comparable levels of BDOC removal, GAC supported a larger quantity of active biomass and was more resilient to vigorous backwashing than anthracite. Relative to the anthracite filters, the GAC filters were capable of greater BDOC removal when collapsed pulse or chlorinated backwash was used.
4. Media type (GAC or anthracite) did not impact turbidity removal, head loss accumulation, or filter run time at either full- or pilot-scale.

6.2 Effect of Backwash Regime on Biological Filter Performance

1. Collapsed pulse backwashing in the absence of chlorine led to extended filter run times by extending the time to turbidity breakthrough.
2. Collapsed pulse backwashing of both GAC and anthracite with chlorinated wash water resulted in substantially decreased filter run times due to sudden surges in turbidity approximately 15 – 25 hours in to the filter cycle. A proposed mechanism for this phenomenon is the biofilm sloughing from the media due to damage during backwashing.
3. Collapsed pulse backwashing impaired BDOC removal in anthracite filters immediately following backwash.
4. When non-chlorinated wash water was used, collapsed pulse backwashing exerted a positive effect on BDOC removal and BRP in GAC filters 24 hours in to the filter cycle, particularly when used in conjunction with ETSW. A proposed mechanism for this effect is increased dispersion of bacteria leading to an increased biofilm surface area.
5. Collapsed pulse backwashing increased filter ripening time of both GAC and anthracite filters, but this was ameliorated by ETSW.
6. ETSW reduced, and often entirely eliminated, ripening of GAC and anthracite filters.
7. Chlorinated ETSW did not impact either BRP or BDOC removal in anthracite and GAC filters.

Chapter 7- Implications and Recommendations

Several of the outcomes of this investigation have implications for the optimization of biological filter design and operation during drinking water treatment. These include:

1. GAC filters offer a greater degree of operational flexibility in terms of backwash due to increased resilience to vigorous backwashing techniques such as chlorinated wash water and collapsed pulse.
2. The differences in performance observed in anthracite and GAC filters can be minimized through backwash optimization. The relative benefits of each filter media type should be balanced against capital cost.
3. Collapsed pulse backwashing should not be applied to biological filters in combination with chlorinated wash water.
4. The effective size of media can be significantly increased (from 1.0 mm to 1.3mm) to reduce head loss accumulation without compromising turbidity or BOM removal.
5. Increasing the uniformity of a media grain size distribution from 1.6 to 1.3 had no detectable impact on any filter performance metric investigated herein, suggesting that the cost of highly uniform media may not be commensurate with performance benefits.
6. ETSW showed promise in significantly reducing, and often eliminating the filter ripening sequence. The extended period of contact time with chlorine did not appear to impact biological filter performance of GAC and anthracite filters.
7. Given the various direct and indirect effects of chlorinated wash water on biological performance, chlorinated backwash water should be avoided wherever possible in biological filtration systems.

Other research outcomes warrant further investigation. These include:

1. Determination of the period of impaired BDOC removal following collapsed pulse backwashing in anthracite filters. If this period extends significantly in to

the filter cycle, it may preclude the effective use of collapsed pulse backwashing of anthracite biological filters.

2. Investigation of the sudden turbidity passage associated with chlorinated collapsed pulse backwash and its potential relationship to biofilm sloughing from the filtration media.

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Appendix A– Pilot Factorial Analyses

Table A 1: BRP ANOVA Table

A	1	0.000118	0.250858	A	1	0.000118	0.250858	A	1	0.021387	9.796607
B	1	0.011149	23.69741	B	1	0.011149	23.69741	B	1	0.013824	6.332099
AB	1	0.007659	16.28065	AB	1	0.007659	16.28065	AB	1	0.042195	19.32798
C	1	0.004576	9.727175	C	1	0.004576	9.727175	C	1	0.041944	19.21309
AC	1	0.074356	158.0493	AC	1	0.074356	158.0493	AC	1	0.1869	85.61181
BC	1	0.039532	84.02896	BC	1	0.039532	84.02896	BC	1	0.217467	99.613
ABC	1	5.96E-05	0.126609	ABC	1	5.96E-05	0.126609	ABC	1	0.05922	27.1265
D	1	0.003974	8.44659	D	1	0.003974	8.44659	D	1	0.008623	3.949898
AD	1	0.000145	0.307886	AD	1	0.000145	0.307886	AD	1	0.000503	0.2303
BD	1	0.000101	0.215391	BD	1	0.000101	0.215391	BD	1	0.000334	0.153136
ABD	1	0.00133	2.83325	ABD	1	0.00133	2.83325	ABD	1	0.00232	1.06452

		3	8			3	8			4	
CD	1	0.000212	0.451153	CD	1	0.000212	0.451153	CD	1	0.000184	0.084167
ACD	1	0.001296	2.75574	ACD	1	0.001296	2.75574	ACD	1	0.001004	0.459672
BCD	1	1.68E-05	0.035616	BCD	1	1.68E-05	0.035616	BCD	1	0.000117	0.053376
ABCD	1	6.3E-05	0.133947	ABCD	1	6.3E-05	0.133947	ABCD	1	0.000661	0.302756
E	1	0.000975	2.07146	E	1	0.000975	2.07146	E	1	0.075256	34.47172
AE	1	0.008306	17.65485	AE	1	0.008306	17.65485	AE	1	0.007226	3.310136
BE	1	0.01305	27.73915	BE	1	0.01305	27.73915	BE	1	0.036072	16.5231
ABE	1	0.006912	14.69286	ABE	1	0.006912	14.69286	ABE	1	0.01548	7.090562
CE	1	0.005644	11.9961	CE	1	0.005644	11.9961	CE	1	0.047395	21.70975
ACE	1	1.75E-05	0.037221	ACE	1	1.75E-05	0.037221	ACE	1	0.021494	9.845474
BCE	1	0.002106	4.476131	BCE	1	0.002106	4.476131	BCE	1	0.091161	41.75749
ABCE	1	0.009426	20.03578	ABCE	1	0.009426	20.03578	ABCE	1	0.000131	0.059999

DE	1	0.000335	0.712476	DE	1	0.000335	0.712476	DE	1	3.41E-05	0.015598
ADE	1	6.26E-05	0.133125	ADE	1	6.26E-05	0.133125	ADE	1	0.0006	0.274892
BDE	1	0.001449	3.080512	BDE	1	0.001449	3.080512	BDE	1	8.4E-05	0.038484
ABDE	1	0.000279	0.592829	ABDE	1	0.000279	0.592829	ABDE	1	4.46E-07	0.000204
CDE	1	0.000681	1.448435	CDE	1	0.000681	1.448435	CDE	1	0.004562	2.089503
ACDE	1	7.5E-06	0.015935	ACDE	1	7.5E-06	0.015935	ACDE	1	6.22E-05	0.028503
BCDE	1	0.000196	0.415941	BCDE	1	0.000196	0.415941	BCDE	1	0.005358	2.454279
F	1	0.003658	7.775088	F	1	0.003658	7.775088	F	1	0.00275	1.259545
AF	1	0.00777	16.51636	AF	1	0.00777	16.51636	AF	1	0.013907	6.370051
BF	1	0.000844	1.793725	BF	1	0.000844	1.793725	BF	1	0.00019 8	0.090483
ABF	1	0.001319	2.804253	ABF	1	0.001319	2.804253	ABF	1	0.005425	2.484875
CF	1	0.00053	1.126009	CF	1	0.00053	1.126009	CF	1	0.000122	0.056073
ACF	1	0.00782	16.6413	ACF	1	0.00782	16.6413	ACF	1	0.00070	0.32311

		9	5			9	5			5	9
BCF	1	0.000963	2.046066	BCF	1	0.000963	2.046066	BCF	1	0.015592	7.142137
ABCF	1	0.001261	2.679559	ABCF	1	0.001261	2.679559	ABCF	1	9.98E-05	0.045728
DF	1	1.52E-05	0.032415	DF	1	1.52E-05	0.032415	DF	1	0.000118	0.053843
ADF	1	0.000369	0.783324	ADF	1	0.000369	0.783324	ADF	1	1.12E-05	0.005125
BDF	1	1.94E-06	0.004125	BDF	1	1.94E-06	0.004125	BDF	1	0.000422	0.193235
ABDF	1	0.000403	0.856052	ABDF	1	0.000403	0.856052	ABDF	1	0.000409	0.187382
CDF	1	0.000144	0.306793	CDF	1	0.000144	0.306793	CDF	1	0.005099	2.335851
ACDF	1	0.000258	0.547734	ACDF	1	0.000258	0.547734	ACDF	1	0.00311	1.424793
BCDF	1	0.000171	0.362915	BCDF	1	0.000171	0.362915	BCDF	1	0.000997	0.456656
EF	1	0.001371	2.915028	EF	1	0.001371	2.915028	EF	1	0.014853	6.803667
AEF	1	0.002257	4.797967	AEF	1	0.002257	4.797967	AEF	1	0.005307	2.431141
BEF	1	0.00056	1.190133	BEF	1	0.00056	1.190133	BEF	1	0.004743	2.172707

ABEF	1	0.008404	17.86245	ABEF	1	0.008404	17.86245	ABEF	1	0.021705	9.942171
CEF	1	0.002133	4.533765	CEF	1	0.002133	4.533765	CEF	1	0.016314	7.472882
ACEF	1	0.005024	10.67985	ACEF	1	0.005024	10.67985	ACEF	1	0.030839	14.12615
BCEF	1	0.000174	0.370554	BCEF	1	0.000174	0.370554	BCEF	1	0.035035	16.04831
DEF	1	0.000155	0.329057	DEF	1	0.000155	0.329057	DEF	1	0.007779	3.563454
ADEF	1	1.31E-05	0.027739	ADEF	1	1.31E-05	0.027739	ADEF	1	0.007035	3.222236
BDEF	1	2.83E-07	0.000601	BDEF	1	2.83E-07	0.000601	BDEF	1	9.6E-05	0.043968
CDEF	1	0.000407	0.865031	CDEF	1	0.000407	0.865031	CDEF	1	0.002097	0.96041
error	7	0.00047	1	error	7	0.00047	1	error	7	0.002183	1

Pilot Factorial Statistical Analysis – Experiment 1

Run Time

Table A 2: Yates table, calculated effects on filter run time in experiment 1

Treatment	Response	Yates 1	Yates 2	Yates 3	Yates 4	Divisor	Estimate	SS
-	48	73.27	140.57	332.57	659.95	16	41.24688	54441.7503
A	25.27	67.3	192	327.38	-95.61	8	-11.9513	1142.65901
B	48	96	141.6	-51.43	-16.09	8	-2.01125	32.3610125
AB	19.3	96	185.78	-44.18	-3.65	8	-0.45625	1.6653125
C	48	72.75	-51.43	-5.97	95.61	8	11.95125	1142.65901
AC	48	68.85	0	-10.12	108.05	8	13.50625	1459.35031
BC	48	96	-50.4	-5.97	3.65	8	0.45625	1.6653125
ABC	48	89.78	6.22	2.32	16.09	8	2.01125	32.3610125
D	48	-22.73	-5.97	51.43	-5.19	8	-0.64875	3.3670125
AD	24.75	-28.7	0	44.18	7.25	8	0.90625	6.5703125
BD	48	0	-3.9	51.43	-4.15	8	-0.51875	2.1528125
ABD	20.85	0	-6.22	56.62	8.29	8	1.03625	8.5905125
CD	48	-23.25	-5.97	5.97	-7.25	8	-0.90625	6.5703125
ACD	48	-27.15	0	-2.32	5.19	8	0.64875	3.3670125
BCD	41.78	0	-3.9	5.97	-8.29	8	-1.03625	8.5905125
ABCD	48	6.22	6.22	10.12	4.15	8	0.51875	2.1528125

Treatment	Estimate	J	(J-0.5)/n	Z
A	-11.9513	1	0.03333	-1.833915
B	-2.01125	2	0.1	-1.281552
BCD	-1.03625	3	0.16667	-0.967422
CD	-0.90625	4	0.23333	-0.727913
D	-0.64875	5	0.3	-0.524401
BD	-0.51875	6	0.36667	-0.340695
AB	-0.45625	7	0.43333	-0.167894
BC	0.45625	8	0.5	-1.39E-16
ABCD	0.51875	9	0.56667	0.167894
ACD	0.64875	10	0.63333	0.340695
AD	0.90625	11	0.7	0.524401
ABD	1.03625	12	0.76667	0.727913
ABC	2.01125	13	0.83333	0.967422
C	11.9513	14	0.9	1.281552
AC	13.5063	15	0.96667	1.833915

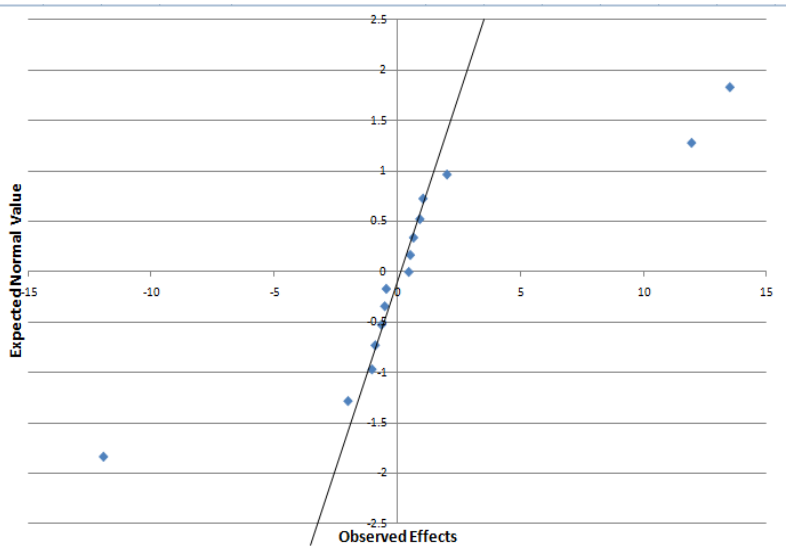

Figure A 1: Experiment 1 normal probability plot of calculated effects on filter run time

Ripening Time

Table A 3: Experiment 1 – Yates table, calculated effects on filter ripening time in experiment 1

Treatment	Response (min)	Yates 1	Yates 2	Yates 3	Yates 4	Divisor	Estimate	Sum Squares
-	29	55	93	154	296	16	18.5	10952
A	26	38	61	142	76	8	9.5	722
B	19	61	87	4	-178	8	-22.25	3960.5
AB	19	0	55	72	-34	8	-4.25	144.5
C	27	66	-3	-78	-64	8	-8	512
AC	34	21	7	-100	-4	8	-0.5	2
BC	0	55	43	-4	-54	8	-6.75	364.5
ABC	0	0	29	-30	-38	8	-4.75	180.5
D	22	-3	-17	-32	-12	8	-1.5	18
AD	44	0	-61	-32	68	8	8.5	578
BD	0	7	-45	10	-22	8	-2.75	60.5
ABD	21	0	-55	-14	-26	8	-3.25	84.5
CD	13	22	3	-44	0	8	0	0
ACD	42	21	-7	-10	-24	8	-3	72
BCD	0	29	-1	-10	34	8	4.25	144.5
ABCD	0	0	-29	-28	-18	8	-2.25	40.5

Treatment	Estimate	J	(J-0.5)/n	Z
B	-22.25	1	0.03333	-1.833915
C	-8	2	0.1	-1.281552
BC	-6.75	3	0.16667	-0.967422
ABC	-4.75	4	0.23333	-0.727913
AB	-4.25	5	0.3	-0.524401
ABD	-3.25	6	0.36667	-0.340695
ACD	-3	7	0.43333	-0.167894
BD	-2.75	8	0.5	-1.39E-16
ABCD	-2.25	9	0.56667	0.167894
D	-1.5	10	0.63333	0.340695
AC	-0.5	11	0.7	0.524401
CD	0	12	0.76667	0.727913
BCD	4.25	13	0.83333	0.967422
AD	8.5	14	0.9	1.281552
A	9.5	15	0.96667	1.833915

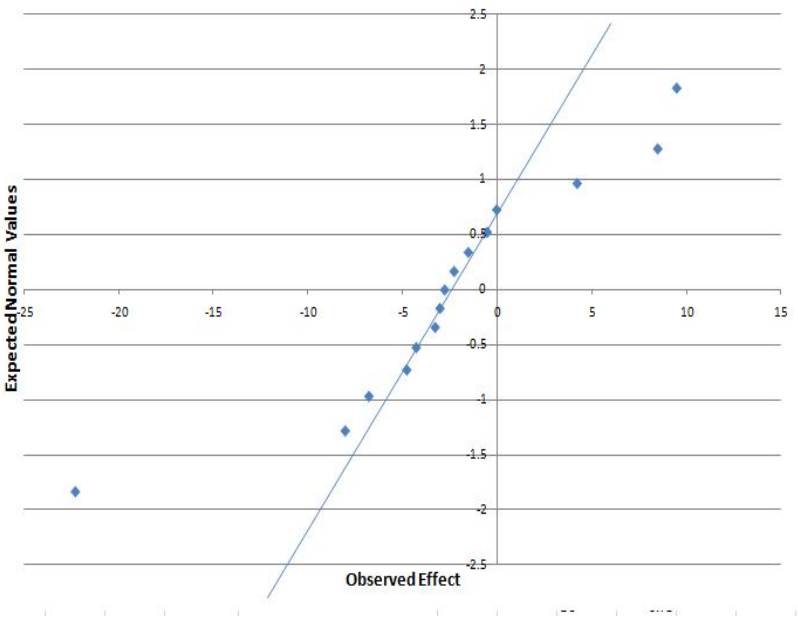

Figure A 2: Experiment 1 normal probability plot of calculated effects on filter ripening time.

DOC Removal

Table A 4: Yates table, calculated effects on DOC removal in experiment 1

Treatment	Response	Yates 1	Yates 2	Yates 3	Yates 4	Yates 5	Divisor	Estimate	Sum of Squares
I	0.136	0.399	1.639	4.216	9.600	18.688			10.914
A	0.263	1.240	2.577	5.384	9.088	-3.950	16	-0.247	0.488
B	0.386	0.753	2.261	3.839	-2.781	6.411	16	0.401	1.284
AB	0.854	1.824	3.123	5.249	-1.169	-2.889	16	-0.181	0.261
C	0.718	0.698	1.307	-1.877	3.142	4.500	16	0.281	0.633
AC	0.035	1.563	2.532	-0.903	3.269	-7.018	16	-0.439	1.539
BC	1.807	1.379	1.887	-0.433	-0.823	3.067	16	0.192	0.294
ABC	0.017	1.744	3.362	-0.736	-2.066	-5.129	16	-0.321	0.822
D	0.300	0.562	0.595	1.912	1.800	2.578	16	0.161	0.208
AD	0.398	0.745	-2.472	1.230	2.700	0.671	16	0.042	0.014
BD	0.502	0.470	0.657	1.775	-5.285	-0.962	16	-0.060	0.029
ABD	1.061	2.062	-1.560	1.495	-1.733	0.419	16	0.026	0.005
CD	0.950	1.052	0.247	-0.765	-0.270	0.174	16	0.011	0.001
ACD	0.429	0.835	-0.680	-0.057	3.337	0.971	16	0.061	0.029
BCD	1.391	0.825	0.035	-0.889	-2.427	-0.210	16	-0.013	0.001
ABCD	0.352	2.537	-0.771	-1.178	-2.702	0.499	16	0.031	0.008
E	0.279	0.127	0.841	0.938	1.168	-0.512	16	-0.032	0.008
AE	0.283	0.468	1.071	0.862	1.410	1.611	16	0.101	0.081
BE	0.251	-0.683	0.865	1.225	0.974	0.127	16	0.008	0.001
ABE	0.494	-1.789	0.365	1.475	-0.304	-1.244	16	-0.078	0.048

CE	0.123	0.098	0.183	-3.067	-0.682	0.900	16	0.056	0.025
ACE	0.347	0.559	1.592	-2.217	-0.280	3.551	16	0.222	0.394
BCE	1.483	-0.521	-0.217	-0.927	0.708	3.607	16	0.225	0.407
ABCE	0.579	-1.039	1.712	-0.806	-0.289	-0.276	16	-0.017	0.002
DE	0.537	0.004	0.341	0.230	-0.076	0.243	16	0.015	0.002
ADE	0.515	0.243	-1.106	-0.500	0.250	-1.278	16	-0.080	0.051
BDE	0.389	0.224	0.461	1.409	0.850	0.402	16	0.025	0.005
ABDE	0.446	-0.904	-0.518	1.929	0.120	-0.998	16	-0.062	0.031
CDE	0.291	-0.022	0.239	-1.447	-0.730	0.327	16	0.020	0.003
ACDE	0.534	0.057	-1.128	-0.979	0.520	-0.730	16	-0.046	0.017
BCDE	1.776	0.243	0.079	-1.367	0.468	1.250	16	0.078	0.049
ABCDE	0.761	-1.014	-1.257	-1.336	0.031	-0.438	16	-0.027	0.006

Table A 5: Ordered Effects on DOC Removal in Experiment 1

	Estimate	J	(j-0.5)/n	Z
AC	-0.464	1.000	0.016	-2.141
ABC	-0.328	2.000	0.048	-1.661
A	-0.235	3.000	0.081	-1.401
AB	-0.192	4.000	0.113	-1.211
E	-0.056	5.000	0.145	-1.057
ABCD	-0.040	6.000	0.177	-0.925
AD	-0.038	7.000	0.210	-0.808
ABD	-0.030	8.000	0.242	-0.700
ABE	-0.025	9.000	0.274	-0.600
BCD	-0.019	10.000	0.306	-0.506
ACD	-0.019	11.000	0.339	-0.416
DE	-0.006	12.000	0.371	-0.329
BE	-0.004	13.000	0.403	-0.245
BCDE	-0.004	14.000	0.435	-0.162
ABCE	-0.001	15.000	0.468	-0.081
CDE	0.001	16.000	0.500	0.000
BDE	0.003	17.000	0.532	0.081
D	0.018	18.000	0.565	0.162
CE	0.019	19.000	0.597	0.245
ACDE	0.024	20.000	0.629	0.329
ABDE	0.025	21.000	0.661	0.416
ABCDE	0.027	22.000	0.694	0.506
BD	0.035	23.000	0.726	0.600
CD	0.056	24.000	0.758	0.700
ADE	0.079	25.000	0.790	0.808
AE	0.086	26.000	0.823	0.925
BCE	0.167	27.000	0.855	1.057
BC	0.209	28.000	0.887	1.211
ACE	0.228	29.000	0.919	1.401
C	0.230	30.000	0.952	1.661
B	0.410	31.000	0.984	2.141

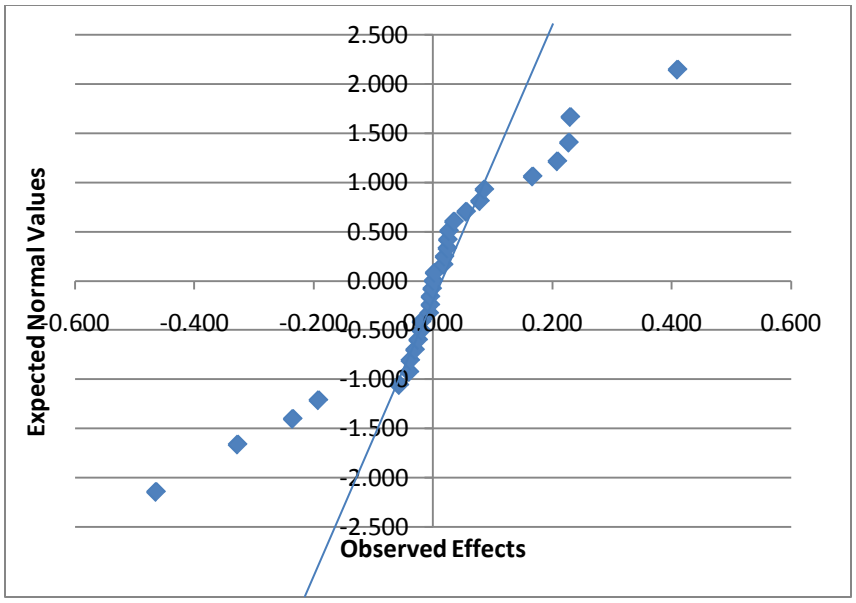

Figure A 3: Experiment 1 normal probability plot of calculated effects on filter DOC removal.

BDOC Removal

Table A 6: Yates table, calculated effects on filter BDOC Removal in experiment 1

Treatment	Response	Yates 1	Yates 2	Yates 3	Yates 4	Yates 5	Divisor	Estimate	Sum of Squares
I	0.245	0.289	1.316	2.712	7.416	11.986			4.489
A	0.044	1.027	1.396	4.705	4.570	-2.697	16.000	-0.169	0.227
B	0.286	0.648	2.096	1.194	-0.921	2.640	16.000	0.165	0.218
AB	0.741	0.748	2.609	3.376	-1.776	3.680	16.000	0.230	0.423
C	0.641	0.755	0.742	-1.100	1.575	0.954	16.000	0.060	0.028
AC	0.007	1.341	0.452	0.178	1.065	-3.361	16.000	-0.210	0.353
BC	0.734	1.229	1.362	-0.749	1.938	0.568	16.000	0.036	0.010
ABC	0.014	1.380	2.014	-1.027	1.742	3.232	16.000	0.202	0.326
D	0.218	0.384	0.254	0.838	0.592	4.175	16.000	0.261	0.545
AD	0.537	0.358	-1.354	0.738	0.362	1.001	16.000	0.063	0.031
BD	0.601	0.220	0.458	-0.014	-2.345	0.992	16.000	0.062	0.031
ABD	0.740	0.232	-0.280	1.079	-1.016	-0.128	16.000	-0.008	0.001
CD	1.071	0.812	-0.048	0.570	-1.073	1.375	16.000	0.086	0.059
ACD	0.158	0.550	-0.701	1.367	1.641	1.161	16.000	0.073	0.042
BCD	0.373	0.337	-0.332	1.334	0.986	1.768	16.000	0.111	0.098
ABCD	1.007	1.677	-0.695	0.408	2.246	1.728	16.000	0.108	0.093
E	0.184	-0.201	0.738	0.080	1.993	-2.847	16.000	-0.178	0.253
AE	0.200	0.455	0.100	0.513	2.182	-0.854	16.000	-0.053	0.023
BE	0.211	-0.634	0.586	-0.290	1.278	-0.510	16.000	-0.032	0.008
ABE	0.147	-0.720	0.152	0.652	-0.278	-0.196	16.000	-0.012	0.001

CE	0.639	0.319	-0.026	-1.608	-0.100	-0.231	16.000	-0.014	0.002
ACE	-0.419	0.139	0.012	-0.738	1.092	1.330	16.000	0.083	0.055
BCE	-0.062	-0.913	-0.262	-0.653	0.797	2.714	16.000	0.170	0.230
ABCE	0.294	0.634	1.341	-0.363	-0.925	1.260	16.000	0.079	0.050
DE	0.446	0.016	0.656	-0.638	0.433	0.189	16.000	0.012	0.001
ADE	0.366	-0.064	-0.086	-0.434	0.942	-1.556	16.000	-0.097	0.076
BDE	0.401	-1.057	-0.180	0.038	0.870	1.192	16.000	0.075	0.044
ABDE	0.149	0.356	1.547	1.603	0.290	-1.722	16.000	-0.108	0.093
CDE	0.487	-0.080	-0.080	-0.742	0.204	0.509	16.000	0.032	0.008
ACDE	-0.150	-0.252	1.414	1.727	1.564	-0.580	16.000	-0.036	0.010
BCDE	0.867	-0.637	-0.172	1.494	2.469	1.360	16.000	0.085	0.058
ABCDE	0.810	-0.057	0.580	0.752	-0.741	-3.210	16.000	-0.201	0.322

Table A 7: Ordered Effects on BDOC Removal in Experiment 1

	Estimate	J	(j-0.5)/n	Z
AC	-0.210	1.000	0.016	-2.141
ABCDE	-0.201	2.000	0.048	-1.661
E	-0.178	3.000	0.081	-1.401
A	-0.169	4.000	0.113	-1.211
ABDE	-0.108	5.000	0.145	-1.057
ADE	-0.097	6.000	0.177	-0.925
AE	-0.053	7.000	0.210	-0.808
ACDE	-0.036	8.000	0.242	-0.700
BE	-0.032	9.000	0.274	-0.600
CE	-0.014	10.000	0.306	-0.506
ABE	-0.012	11.000	0.339	-0.416
ABD	-0.008	12.000	0.371	-0.329
DE	0.012	13.000	0.403	-0.245
CDE	0.032	14.000	0.435	-0.162
BC	0.036	15.000	0.468	-0.081
C	0.060	16.000	0.500	0.000
BD	0.062	17.000	0.532	0.081
AD	0.063	18.000	0.565	0.162
ACD	0.073	19.000	0.597	0.245
BDE	0.075	20.000	0.629	0.329
ABCE	0.079	21.000	0.661	0.416
ACE	0.083	22.000	0.694	0.506
BCDE	0.085	23.000	0.726	0.600
CD	0.086	24.000	0.758	0.700
ABCD	0.108	25.000	0.790	0.808
BCD	0.111	26.000	0.823	0.925
B	0.165	27.000	0.855	1.057
BCE	0.170	28.000	0.887	1.211
ABC	0.202	29.000	0.919	1.401
AB	0.230	30.000	0.952	1.661
D	0.261	31.000	0.984	2.141

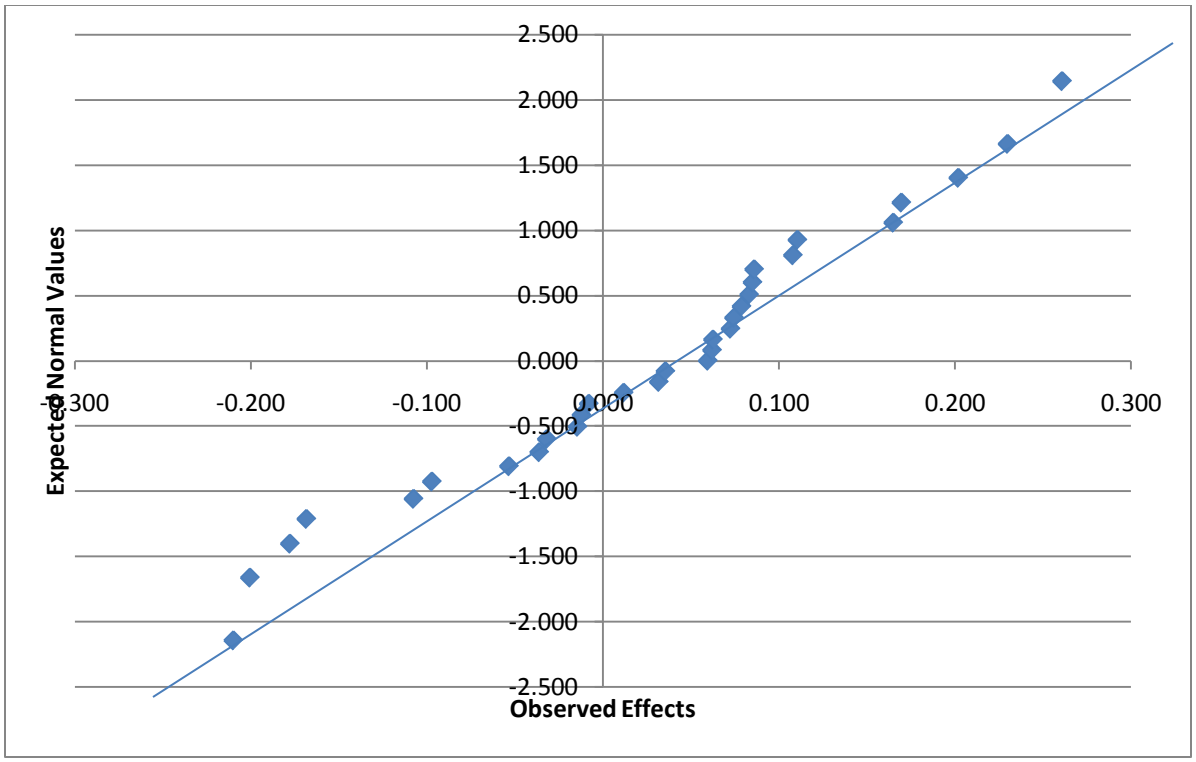

Figure A 4: Experiment 1 normal probability plot of calculated effects on filter BDOC removal.

BRP

Table A 8: Yates table, calculated effects on filter BRP in experiment 1

Treatment	Response	Yates 1	Yates 2	Yates 3	Yates 4	Yates 5	Yates 6	Divisor	Estimate	SS
1	0.029	0.044	0.199	0.857	1.941	3.815	8.054			
A	0.015	0.155	0.658	1.084	1.874	4.239	-0.766	32	-0.024	0.009
B	0.011	0.438	0.612	0.717	2.672	-0.355	0.779	32	0.024	0.009
AB	0.144	0.220	0.472	1.157	1.567	-0.411	1.125	32	0.035	0.020
C	0.316	0.183	0.452	1.361	-0.538	0.403	0.661	32	0.021	0.007
AC	0.122	0.429	0.265	1.311	0.183	0.376	-2.837	32	-0.089	0.126
BC	0.205	0.237	0.573	0.380	-0.164	0.423	-2.634	32	-0.082	0.108
ABC	0.016	0.235	0.584	1.187	-0.247	0.702	-0.870	32	-0.027	0.012
D	0.091	0.066	0.542	-0.265	0.138	0.144	1.423	32	0.044	0.032
AD	0.092	0.387	0.819	-0.273	0.265	0.518	0.583	32	0.018	0.005
BD	0.242	0.106	0.740	0.170	0.194	-1.438	-0.015	32	0.000	0.000
ABD	0.187	0.159	0.571	0.013	0.181	-1.399	-0.133	32	-0.004	0.000
CD	0.206	0.283	0.028	-0.194	0.229	-0.968	-1.086	32	-0.034	0.018
ACD	0.031	0.290	0.352	0.029	0.194	-1.666	0.368	32	0.011	0.002
BCD	0.139	0.350	0.551	-0.386	0.780	-0.269	1.010	32	0.032	0.016
ABCD	0.096	0.234	0.636	0.140	-0.078	-0.601	0.871	32	0.027	0.012
E	-0.016	0.025	0.119	-0.106	0.320	0.666	-1.172	32	-0.037	0.021
AE	0.082	0.516	-0.384	0.244	-0.177	0.757	0.638	32	0.020	0.006
BE	0.084	0.684	-0.054	0.374	0.108	-0.166	0.114	32	0.004	0.000
ABE	0.302	0.135	-0.219	-0.109	0.409	0.749	-0.894	32	-0.028	0.012
CE	0.073	0.238	0.316	-0.057	-0.668	-0.132	-0.196	32	-0.006	0.001

ACE	0.033	0.502	-0.146	0.251	-0.771	0.117	0.646	32	0.020	0.007
BCE	0.132	0.292	0.160	0.186	-1.074	0.008	1.154	32	0.036	0.021
ABCE	0.026	0.279	-0.148	-0.005	-0.325	-0.141	-0.699	32	-0.022	0.008
DE	0.135	-0.079	0.207	0.153	-0.577	-0.400	1.069	32	0.033	0.018
ADE	0.148	0.107	-0.400	0.076	-0.391	-0.686	0.154	32	0.005	0.000
BDE	0.071	0.176	0.248	0.054	-1.317	0.493	-1.332	32	-0.042	0.028
ABDE	0.219	0.177	-0.219	0.139	-0.349	-0.126	0.096	32	0.003	0.000
CDE	0.213	0.236	-0.178	0.409	0.046	0.228	1.006	32	0.031	0.016
ACDE	0.137	0.315	-0.208	0.372	-0.315	0.782	-0.590	32	-0.018	0.005
BCDE	0.153	0.360	0.218	0.013	-0.131	0.385	-0.675	32	-0.021	0.007
ABCDE	0.081	0.276	-0.078	-0.091	-0.469	0.486	-0.270	32	-0.008	0.001
F	0.035	-0.014	0.112	0.460	0.226	-0.067	0.424	32	0.013	0.003
AF	-0.010	0.133	-0.218	-0.139	0.440	-1.105	-0.056	32	-0.002	0.000
BF	0.132	-0.195	0.246	-0.188	-0.049	0.720	-0.028	32	-0.001	0.000
ABF	0.384	-0.189	-0.002	0.011	0.807	-0.082	0.279	32	0.009	0.001
CF	0.470	0.001	0.321	0.277	-0.009	0.127	0.374	32	0.012	0.002
ACF	0.214	-0.055	0.052	-0.169	-0.157	-0.013	0.040	32	0.001	0.000
BCF	0.140	-0.175	0.007	0.324	0.223	-0.036	-0.698	32	-0.022	0.008
ABCF	-0.005	-0.043	-0.115	0.085	0.526	-0.859	-0.332	32	-0.010	0.002
DF	0.097	0.098	0.491	-0.503	0.350	-0.497	0.091	32	0.003	0.000
ADF	0.141	0.218	-0.548	-0.164	-0.482	0.301	0.914	32	0.029	0.013
BDF	0.150	-0.040	0.264	-0.462	0.308	-0.103	0.249	32	0.008	0.001
ABDF	0.353	-0.106	-0.013	-0.308	-0.191	0.748	-0.149	32	-0.005	0.000
CDF	0.254	0.013	0.185	-0.607	-0.077	0.186	-0.286	32	-0.009	0.001
ACDF	0.038	0.148	0.001	-0.467	0.085	0.968	-0.619	32	-0.019	0.006
BCDF	0.141	-0.076	0.080	-0.030	-0.037	-0.362	0.554	32	0.017	0.005

ABCDF	0.138	-0.071	-0.085	-0.295	-0.104	-0.338	0.101	32	0.003	0.000
EF	0.051	-0.045	0.148	-0.329	-0.599	0.213	-1.038	32	-0.032	0.017
AEF	-0.130	0.252	0.006	-0.248	0.199	0.856	-0.802	32	-0.025	0.010
BEF	0.052	-0.256	-0.056	-0.269	-0.447	-0.149	-0.140	32	-0.004	0.000
ABEF	0.055	-0.145	0.132	-0.122	-0.239	0.303	-0.823	32	-0.026	0.011
CEF	0.097	0.045	0.120	-1.039	0.339	-0.833	0.798	32	0.025	0.010
ACEF	0.079	0.203	-0.066	-0.278	0.154	-0.500	0.851	32	0.027	0.011
BCEF	0.183	-0.216	0.135	-0.185	0.140	0.162	0.782	32	0.024	0.010
ABCEF	-0.007	-0.003	0.005	-0.164	-0.266	-0.067	0.024	32	0.001	0.000
DEF	0.066	-0.182	0.297	-0.142	0.081	0.798	0.642	32	0.020	0.006
ADEF	0.170	0.003	0.111	0.188	0.147	0.208	0.451	32	0.014	0.003
BDEF	0.101	-0.018	0.158	-0.185	0.762	-0.185	0.333	32	0.010	0.002
ABDEF	0.215	-0.190	0.213	-0.130	0.021	-0.406	-0.229	32	-0.007	0.001
CDEF	0.174	0.104	0.185	-0.186	0.330	0.066	-0.590	32	-0.018	0.005
ACDEF	0.186	0.114	-0.172	0.055	0.055	-0.741	-0.221	32	-0.007	0.001
BCDEF	0.183	0.012	0.010	-0.357	0.241	-0.274	-0.807	32	-0.025	0.010
ABCDEF	0.093	-0.090	-0.101	-0.112	0.245	0.005	0.279	32	0.009	0.001

Table A 9: Ordered Effects on BRP in Experiment 1

Treatment	Estimate	J	(j-0.5)/n	Z
AC	-0.08867	1.000	0.008	-2.412
BC	-0.08231	2.000	0.024	-1.981
BDE	-0.04163	3.000	0.040	-1.754
E	-0.03663	4.000	0.056	-1.593
CD	-0.03393	5.000	0.071	-1.465
EF	-0.03244	6.000	0.087	-1.358
ABE	-0.02795	7.000	0.103	-1.264
ABC	-0.02718	8.000	0.119	-1.180
ABEF	-0.02572	9.000	0.135	-1.103
BCDEF	-0.02521	10.000	0.151	-1.033
AEF	-0.02508	11.000	0.167	-0.967
A	-0.02395	12.000	0.183	-0.906
ABCE	-0.02185	13.000	0.198	-0.847
BCF	-0.0218	14.000	0.214	-0.792
BCDE	-0.02111	15.000	0.230	-0.738
ACDF	-0.01935	16.000	0.246	-0.687
ACDE	-0.01845	17.000	0.262	-0.637
CDEF	-0.01844	18.000	0.278	-0.589
ABCF	-0.01036	19.000	0.294	-0.543
CDF	-0.00893	20.000	0.310	-0.497
ABCDE	-0.00843	21.000	0.325	-0.453
ABDEF	-0.00716	22.000	0.341	-0.409
ACDEF	-0.00691	23.000	0.357	-0.366
CE	-0.00612	24.000	0.373	-0.324
ABDF	-0.00465	25.000	0.389	-0.282
BEF	-0.00436	26.000	0.405	-0.241
ABD	-0.00415	27.000	0.421	-0.200
AF	-0.00174	28.000	0.437	-0.160
BF	-0.00087	29.000	0.452	-0.120
BD	-0.00047	30.000	0.468	-0.080
ABCEF	0.000747	31.000	0.484	-0.040
ACF	0.001238	32.000	0.500	0.000
DF	0.002849	33.000	0.516	0.040
ABDE	0.002988	34.000	0.532	0.080
ABCDF	0.003162	35.000	0.548	0.120
BE	0.003557	36.000	0.563	0.160

ADE	0.004817	37.000	0.579	0.200
BDF	0.007781	38.000	0.595	0.241
ABF	0.00871	39.000	0.611	0.282
ABCDEF	0.008721	40.000	0.627	0.324
BDEF	0.010409	41.000	0.643	0.366
ACD	0.011494	42.000	0.659	0.409
CF	0.011685	43.000	0.675	0.453
F	0.013248	44.000	0.690	0.497
ADEF	0.014103	45.000	0.706	0.543
BCDF	0.01731	46.000	0.722	0.589
AD	0.018221	47.000	0.738	0.637
AE	0.019945	48.000	0.754	0.687
DEF	0.020075	49.000	0.770	0.738
ACE	0.020177	50.000	0.786	0.792
C	0.02066	51.000	0.802	0.847
B	0.024351	52.000	0.817	0.906
BCEF	0.024443	53.000	0.833	0.967
CEF	0.024941	54.000	0.849	1.033
ACEF	0.026595	55.000	0.865	1.103
ABCD	0.027233	56.000	0.881	1.180
ADF	0.028577	57.000	0.897	1.264
CDE	0.031424	58.000	0.913	1.358
BCD	0.031577	59.000	0.929	1.465
DE	0.033407	60.000	0.944	1.593
AB	0.035154	61.000	0.960	1.754
BCE	0.036076	62.000	0.976	1.981
D	0.044484	63.000	0.992	2.412

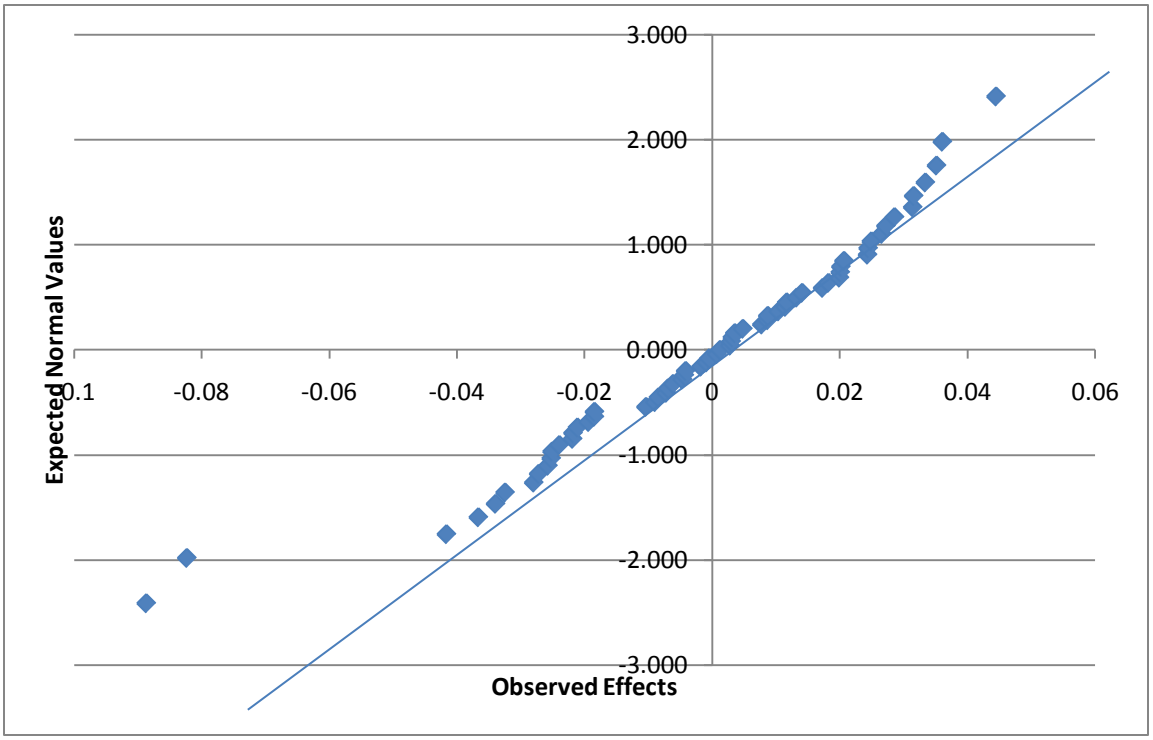

Figure A 5: Experiment 1 normal probability plot of calculated effects BRP.

Experiment 2

Run Time

Table A 10: Yates table, calculated effects on filter run time in experiment 2

Treatment	Response	Yates 1	Yates 2	Yates 3	Yates 4	Divisor	Estimate	ss
-	25.4	61.4	124.2	236.4	563.8	16	35.2	35320.2
A	36.0	62.8	112.2	327.4	-50.3	8	-6.3	281.2
B	38.4	62.1	141.6	-6.1	-20.8	8	-2.6	47.8
AB	24.4	50.1	185.8	-44.2	-19.4	8	-2.4	41.9
C	32.5	72.8	-3.4	-10.6	32.2	8	4.0	114.8
AC	29.7	68.9	-2.8	-10.1	57.2	8	7.2	363.9
BC	25.0	96.0	-50.4	-21.8	-15.8	8	-2.0	27.8
ABC	25.1	89.8	6.2	2.3	37.5	8	4.7	156.3
D	48.0	10.6	1.4	-12.0	91.0	8	11.4	919.1
AD	24.8	-14.0	-12.1	44.2	-38.1	8	-4.8	160.9

BD	48.0	-2.8	-3.9	0.6	0.5	8	0.1	0.0
ABD	20.9	0.0	-6.2	56.6	24.1	8	3.0	64.4
CD	48.0	-23.3	-24.6	-13.5	56.2	8	7.0	351.1
ACD	48.0	-27.2	2.8	-2.3	56.0	8	7.0	348.6
BCD	41.8	0.0	-3.9	27.4	11.2	8	1.4	13.9
ABCD	48.0	6.2	6.2	10.1	-17.3	8	-2.2	33.1

Treatment	Estimate	J		
A	-6.28875	1	0.03333	-1.833915
AD	-4.75625	2	0.1	-1.281552
B	-2.59375	3	0.16667	-0.967422
AB	-2.42875	4	0.23333	-0.727913
ABCD	-2.15875	5	0.3	-0.524401
BC	-1.97625	6	0.36667	-0.340695
BD	0.06375	7	0.43333	-0.167894
BCD	1.39625	8	0.5	-1.39E-16
ABD	3.00875	9	0.56667	0.167894
C	4.01875	10	0.63333	0.340695
ABC	4.68875	11	0.7	0.524401
ACD	7.00125	12	0.76667	0.727913
CD	7.02625	13	0.83333	0.967422
AC	7.15375	14	0.9	1.281552
D	11.3688	15	0.96667	1.833915

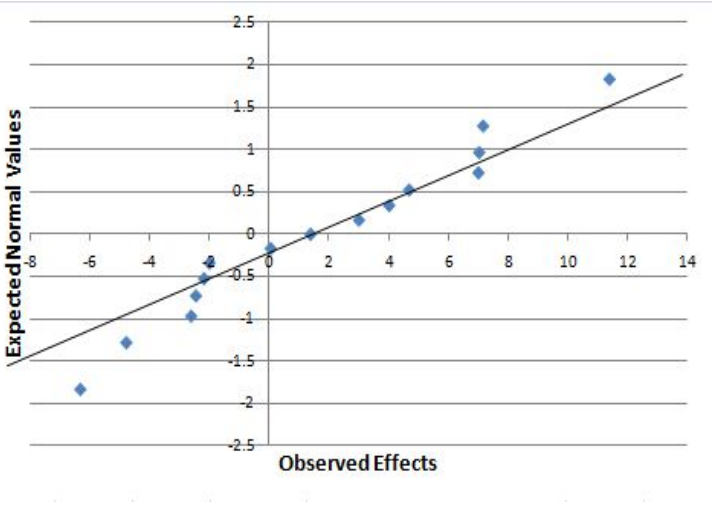

Figure A 6: Experiment 2 normal probability plot of calculated effects on filter run time.

Ripening Time

Table A 11: Yates table, calculated effects on filter ripening time in experiment 2

Treatment	Response	Yates 1	Yates 2	Yates 3	Yates 4	Divisor	Estimate	SS
-	23	52	69	130	272	16	17	9248
A	29	17	61	142	136	8	17	2312
B	0	30	87	64	-134	8	-16.75	2244.5
AB	17	31	55	72	-38	8	-4.75	180.5
C	0	66	23	-34	-40	8	-5	200
AC	30	21	41	-100	4	8	0.5	2
BC	10	55	43	-8	26	8	3.25	84.5
ABC	21	0	29	-30	-58	8	-7.25	420.5
D	22	6	-35	-8	12	8	1.5	18
AD	44	17	1	-32	8	8	1	8
BD	0	30	-45	18	-66	8	-8.25	544.5

ABD	21	11	-55	-14	-22	8	-2.75	60.5
CD	13	22	11	36	-24	8	-3	72
ACD	42	21	-19	-10	-32	8	-4	128
BCD	0	29	-1	-30	-46	8	-5.75	264.5
ABCD	0	0	-29	-28	2	8	0.25	0.5

Treatment	Estimate	J		
B	-16.75	1	0.03333	-1.833915
BD	-8.25	2	0.1	-1.281552
ABC	-7.25	3	0.16667	-0.967422
BCD	-5.75	4	0.23333	-0.727913
C	-5	5	0.3	-0.524401
AB	-4.75	6	0.36667	-0.340695
ACD	-4	7	0.43333	-0.167894
CD	-3	8	0.5	-1.39E-16
ABD	-2.75	9	0.56667	0.167894
ABCD	0.25	10	0.63333	0.340695
AC	0.5	11	0.7	0.524401
AD	1	12	0.76667	0.727913
D	1.5	13	0.83333	0.967422
BC	3.25	14	0.9	1.281552
A	17	15	0.96667	1.833915

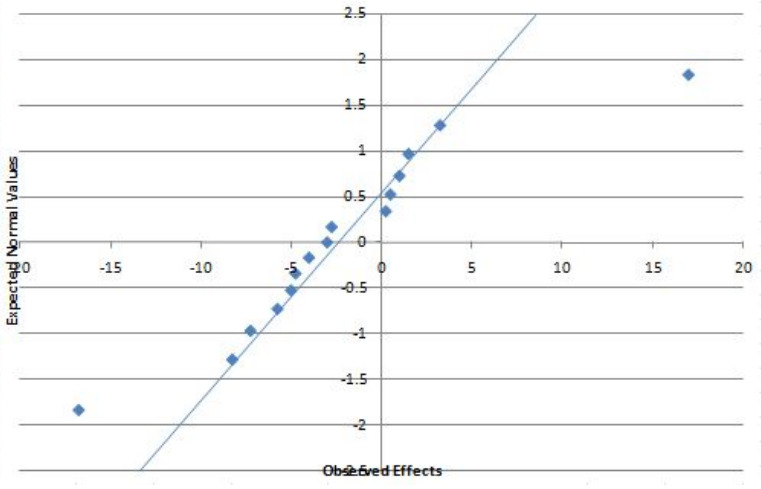

Figure A 7: Experiment 2 normal probability plot of calculated effects on filter ripening time.

DOC Removal

Table A 12: Yates table, calculated effects on DOC removal in experiment 2

Treatment	Response	Yates 1	Yates 2	Yates 3	Yates 4	Yates 5	Divisor	Estimate	Sum of Squares
I	0.522	0.914	2.202	4.346	9.730	18.301			10.466
A	0.392	1.288	2.144	5.384	8.571	-0.978	16	-0.061	0.030
B	0.396	1.416	2.261	3.322	-1.012	2.628	16	0.164	0.216
AB	0.892	0.728	3.123	5.249	0.033	-0.629	16	-0.039	0.012
C	0.883	0.698	1.633	-0.108	0.916	2.334	16	0.146	0.170
AC	0.533	1.563	1.689	-0.903	1.712	-4.080	16	-0.255	0.520
BC	0.426	1.379	1.887	0.770	0.794	0.210	16	0.013	0.001
ABC	0.302	1.744	3.362	-0.736	-1.423	-2.407	16	-0.150	0.181
D	0.300	0.723	0.366	-0.314	0.804	2.965	16	0.185	0.275
AD	0.398	0.910	-0.474	1.230	1.531	-2.301	16	-0.144	0.165
BD	0.502	0.829	0.657	0.217	-3.058	2.821	16	0.176	0.249
ABD	1.061	0.860	-1.560	1.495	-1.022	-1.841	16	-0.115	0.106
CD	0.950	1.052	0.493	0.852	-1.562	2.339	16	0.146	0.171
ACD	0.429	0.835	0.277	-0.057	1.773	-1.968	16	-0.123	0.121
BCD	1.391	0.825	0.035	-0.245	-1.380	2.647	16	0.165	0.219
ABCD	0.352	2.537	-0.771	-1.178	-1.028	-2.223	16	-0.139	0.154
E	0.169	-0.130	0.374	-0.058	1.038	-1.158	16	-0.072	0.042
AE	0.554	0.496	-0.688	0.862	1.927	1.045	16	0.065	0.034
BE	0.401	-0.350	0.865	0.056	-0.795	0.796	16	0.050	0.020
ABE	0.509	-0.124	0.365	1.475	-1.506	-2.218	16	-0.139	0.154

CE	0.353	0.098	0.187	-0.840	1.544	0.727	16	0.045	0.017
ACE	0.476	0.559	0.031	-2.217	1.277	2.036	16	0.127	0.129
BCE	0.353	-0.521	-0.217	-0.216	-0.909	3.335	16	0.208	0.348
ABCE	0.507	-1.039	1.712	-0.806	-0.933	0.352	16	0.022	0.004
DE	0.537	0.385	0.626	-1.062	0.920	0.889	16	0.056	0.025
ADE	0.515	0.108	0.226	-0.500	1.420	-0.711	16	-0.044	0.016
BDE	0.389	0.123	0.461	-0.156	-1.377	-0.267	16	-0.017	0.002
ABDE	0.446	0.154	-0.518	1.929	-0.591	-0.024	16	-0.001	0.000
CDE	0.291	-0.022	-0.277	-0.400	0.562	0.500	16	0.031	0.008
ACDE	0.534	0.057	0.031	-0.979	2.085	0.786	16	0.049	0.019
BCDE	1.776	0.243	0.079	0.308	-0.579	1.523	16	0.095	0.072
ABCDE	0.761	-1.014	-1.257	-1.336	-1.644	-1.065	16	-0.067	0.035

Table A 13: Ordered Effects on DOC removal in Experiment 2

	Estimate	J	(j-0.5)/n	Z
AC	-0.255	1.000	0.016	-2.141
ABC	-0.150	2.000	0.048	-1.661
AD	-0.144	3.000	0.081	-1.401
ABCD	-0.139	4.000	0.113	-1.211
ABE	-0.139	5.000	0.145	-1.057
ACD	-0.123	6.000	0.177	-0.925
ABD	-0.115	7.000	0.210	-0.808
E	-0.072	8.000	0.242	-0.700
ABCDE	-0.067	9.000	0.274	-0.600
A	-0.061	10.000	0.306	-0.506
ADE	-0.044	11.000	0.339	-0.416
AB	-0.039	12.000	0.371	-0.329
BDE	-0.017	13.000	0.403	-0.245
ABDE	-0.001	14.000	0.435	-0.162
BC	0.013	15.000	0.468	-0.081
ABCE	0.022	16.000	0.500	0.000
CDE	0.031	17.000	0.532	0.081
CE	0.045	18.000	0.565	0.162
ACDE	0.049	19.000	0.597	0.245
BE	0.050	20.000	0.629	0.329
DE	0.056	21.000	0.661	0.416
AE	0.065	22.000	0.694	0.506
BCDE	0.095	23.000	0.726	0.600
ACE	0.127	24.000	0.758	0.700
C	0.146	25.000	0.790	0.808
CD	0.146	26.000	0.823	0.925
B	0.164	27.000	0.855	1.057
BCD	0.165	28.000	0.887	1.211
BD	0.176	29.000	0.919	1.401
D	0.185	30.000	0.952	1.661
BCE	0.208	31.000	0.984	2.141

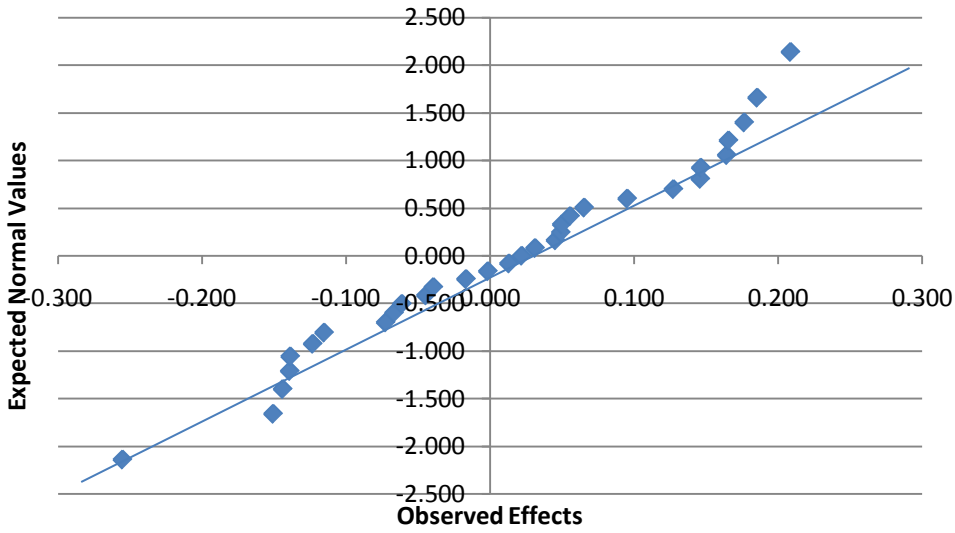

Figure A 8: Experiment 2 normal probability plot of calculated effects on DOC removal.

BDOC Removal

Table A 14: Yates table, calculated effects on BDOC removal in experiment 2

Treatment	Response	Yates 1	Yates 2	Yates 3	Yates 4	Yates 5	Divisor	Estimate	Sum of Squares
I	0.516	1.348	2.296	4.262	8.966	14.851			6.891886
A	0.832	0.948	1.966	4.705	5.884	-1.342	16	-0.084	0.056263
B	0.644	1.311	2.096	1.885	0.355	1.160	16	0.073	0.042079
AB	0.304	0.655	2.609	4.000	-1.697	-0.399	16	-0.025	0.004978
C	0.633	0.755	0.905	0.455	-0.319	0.536	16	0.034	0.008984
AC	0.678	1.341	0.980	-0.100	1.479	-0.028	16	-0.002	2.47E-05
BC	0.111	1.229	1.861	-0.482	-0.447	-1.949	16	-0.122	0.118689
ABC	0.544	1.380	2.139	-1.216	0.047	2.454	16	0.153	0.188258
D	0.218	0.331	-0.024	-1.056	0.183	2.558	16	0.160	0.204467
AD	0.537	0.574	0.479	0.738	0.353	-1.288	16	-0.081	0.051878
BD	0.601	0.546	0.180	0.354	0.962	1.023	16	0.064	0.032733
ABD	0.740	0.435	-0.280	1.125	-0.990	0.697	16	0.044	0.015168
CD	1.071	0.976	-0.051	-0.268	-0.691	1.044	16	0.065	0.034083
ACD	0.158	0.885	-0.431	-0.178	-1.257	0.947	16	0.059	0.028048
BCD	0.373	0.461	-0.303	-0.280	1.782	-1.170	16	-0.073	0.042748
ABCD	1.007	1.677	-0.913	0.327	0.672	1.559	16	0.097	0.075964
E	0.218	0.316	-0.400	-0.330	0.443	-3.082	16	-0.193	0.296847
AE	0.113	-0.340	-0.657	0.513	2.115	-2.053	16	-0.128	0.131662
BE	0.260	0.045	0.586	0.076	-0.555	1.798	16	0.112	0.101012
ABE	0.314	0.433	0.152	0.278	-0.734	0.494	16	0.031	0.007629

CE	0.271	0.319	0.243	0.502	1.794	0.170	16	0.011	0.000905
ACE	0.275	-0.139	0.111	0.460	-0.771	1.952	16	0.122	0.119084
BCE	-0.255	-0.913	-0.091	-0.381	0.090	-0.566	16	-0.035	0.010013
ABCE	0.690	0.634	1.216	-0.610	0.607	-1.110	16	-0.069	0.038479
DE	0.446	-0.105	-0.656	-0.257	0.843	1.672	16	0.104	0.087317
ADE	0.530	0.054	0.388	-0.434	0.202	-0.179	16	-0.011	0.001003
BDE	0.636	0.004	-0.458	-0.133	-0.043	-2.565	16	-0.160	0.205524
ABDE	0.249	-0.435	0.280	-1.125	0.990	-0.697	16	-0.044	0.015168
CDE	0.658	0.084	0.159	1.044	-0.177	-0.641	16	-0.040	0.012828
ACDE	-0.197	-0.387	-0.438	0.738	-0.992	1.033	16	0.065	0.033338
BCDE	0.867	-0.855	-0.471	-0.597	-0.307	-0.815	16	-0.051	0.02075
ABCDE	0.810	-0.057	0.798	1.269	1.866	2.173	16	0.136	0.147523

Table A 15: Ordered Effects on BDOC removal in Experiment 2

	Estimate	J	(j-0.5)/n	Z
E	-0.19263	1	0.016129	-2.1412
BDE	-0.16028	2	0.048387	-1.6607
AE	-0.12829	3	0.080645	-1.40075
BC	-0.1218	4	0.112903	-1.21123
A	-0.08386	5	0.145161	-1.05741
AD	-0.08053	6	0.177419	-0.92524
BCD	-0.0731	7	0.209677	-0.80754
ABCE	-0.06935	8	0.241935	-0.70009
BCDE	-0.05093	9	0.274194	-0.60018
ABDE	-0.04354	10	0.306452	-0.50593
CDE	-0.04004	11	0.33871	-0.41599
BCE	-0.03538	12	0.370968	-0.32929
AB	-0.02495	13	0.403226	-0.24501
ADE	-0.0112	14	0.435484	-0.16243
AC	-0.00176	15	0.467742	-0.08095
CE	0.010637	16	0.5	-1.4E-16
ABE	0.030881	17	0.532258	0.080947
C	0.033511	18	0.564516	0.162429
ABD	0.043543	19	0.596774	0.245006
ACD	0.059211	20	0.629032	0.329291
BD	0.063965	21	0.66129	0.415987
ACDE	0.064554	22	0.693548	0.505934
CD	0.065272	23	0.725806	0.600179
B	0.072525	24	0.758065	0.70009
ABCD	0.097445	25	0.790323	0.807541
DE	0.104473	26	0.822581	0.925245
BE	0.112368	27	0.854839	1.057414
ACE	0.122006	28	0.887097	1.211232
ABCDE	0.135795	29	0.919355	1.400745
ABC	0.153402	30	0.951613	1.660698
D	0.15987	31	0.983871	2.141198

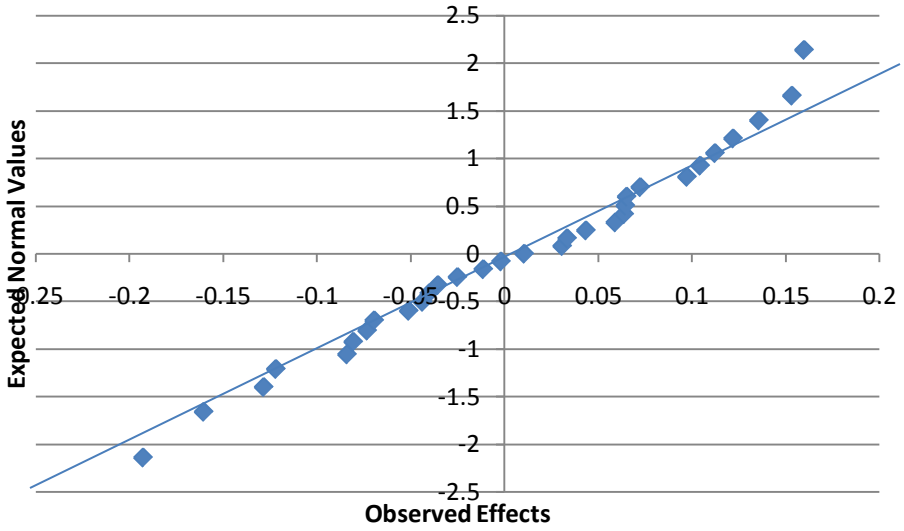

Figure A 9: Experiment 2 normal probability plot of calculated effects on BDOC removal.

BRP

Table A 16: Yates table, calculated effects on BRP removal in experiment 2

Treatment	Response	Yates 1	Yates 2	Yates 3	Yates 4	Yates 5	Yates 6	Divisor	Estimate	SS
1	0.099719	0.184133	0.563453	1.026996	2.110756	4.244772	8.973386			
A	0.084414	0.37932	0.463543	1.083761	2.134016	4.728614	-0.08691	32	-0.00272	0.000118
B	0.214898	0.23711	0.61153	0.977125	2.500807	-0.39605	0.844698	32	0.026397	0.011149
AB	0.164421	0.226433	0.472231	1.156891	2.227807	0.309142	0.700143	32	0.021879	0.007659
C	0.163744	0.182692	0.555854	1.189327	-0.47532	0.306151	-0.54118	32	-0.01691	0.004576
AC	0.073365	0.428838	0.421271	1.311481	0.079269	0.538547	-2.18146	32	-0.06817	0.074356
BC	0.136199	0.237005	0.572877	1.041094	0.067319	0.204784	-1.59062	32	-0.04971	0.039532
ABC	0.090234	0.235225	0.584014	1.186713	0.241823	0.495359	-0.06174	32	-0.00193	5.96E-05
D	0.090947	0.257594	0.675945	-0.20213	0.428875	-0.36266	0.504303	32	0.015759	0.003974
AD	0.091745	0.29826	0.513381	-0.27319	-0.12272	-0.17853	-0.09628	32	-0.00301	0.000145
BD	0.242006	0.238012	0.740398	0.066565	0.450423	-0.7368	-0.08053	32	-0.00252	0.000101
ABD	0.186832	0.183258	0.571082	0.012704	0.088124	-1.44466	0.292075	32	0.009127	0.001333
CD	0.206222	0.283098	0.486452	0.038024	0.085332	-0.67121	0.11655	32	0.003642	0.000212
ACD	0.030783	0.289779	0.554643	0.029295	0.119452	-0.91941	-0.28805	32	-0.009	0.001296
BCD	0.139302	0.349665	0.550775	0.102224	0.597303	0.11115	-0.03275	32	-0.00102	1.68E-05
ABCD	0.095923	0.234349	0.635937	0.1396	-0.10194	-0.17289	0.063506	32	0.001985	6.3E-05
E	0.09707	0.226338	-0.06578	0.18451	-0.23921	0.236531	-0.24974	32	-0.0078	0.000975
AE	0.160524	0.449608	-0.13634	0.244365	-0.12345	0.267772	0.729093	32	0.022784	0.008306
BE	0.115785	0.268759	-0.05438	-0.01409	-0.33188	-0.12493	-0.9139	32	-0.02856	0.01305
ABE	0.182475	0.244623	-0.21882	-0.10864	0.153353	0.028647	-0.66513	32	-0.02079	0.006912
CE	0.12909	0.237959	0.130143	0.199134	-0.235	-0.03469	0.600995	32	0.018781	0.005644

ACE	0.108922	0.502439	-0.06358	0.251289	-0.5018	-0.04584	0.033477	32	0.001046	1.75E-05
BCE	0.113334	0.292137	0.16039	0.093059	-0.87246	0.226311	0.367115	32	0.011472	0.002106
ABCE	0.069924	0.278946	-0.14769	-0.00493	-0.57219	0.065764	-0.7767	32	-0.02427	0.009426
DE	0.135133	0.1624	0.221854	0.009242	-0.45379	0.106331	0.146466	32	0.004577	0.000335
ADE	0.147965	0.324052	-0.18383	0.076089	-0.21742	0.010219	0.063311	32	0.001978	6.26E-05
BDE	0.071111	0.311618	0.248038	-0.02001	-0.52508	-0.20824	-0.30455	32	-0.00952	0.001449
ABDE	0.218668	0.243024	-0.21874	0.139458	-0.39433	-0.07982	-0.1336	32	-0.00418	0.000279
CDE	0.212937	0.235599	0.189481	0.225656	0.267619	-0.06864	0.208834	32	0.006526	0.000681
ACDE	0.136728	0.315176	-0.08726	0.371648	-0.15647	0.035893	0.021904	32	0.000684	7.5E-06
BCDE	0.152913	0.360224	0.217528	-0.01086	0.089861	0.004932	0.111909	32	0.003497	0.000196
ABCDE	0.081436	0.275713	-0.07793	-0.09109	-0.26275	0.058574	-0.19243	32	-0.00601	0.000579
F	0.081516	-0.01531	0.195186	-0.09991	0.056765	0.023259	0.483842	32	0.01512	0.003658
AF	0.144822	-0.05048	-0.01068	-0.1393	0.179766	-0.273	0.705193	32	0.022037	0.00777
BF	0.14553	-0.09038	0.246145	-0.13458	0.122154	0.554588	0.232396	32	0.007262	0.000844
ABF	0.304078	-0.04596	-0.00178	0.011137	0.145618	0.174504	0.290576	32	0.00908	0.001319
CF	0.21294	0.000798	0.040666	-0.16256	-0.07107	-0.5516	0.184129	32	0.005754	0.00053
ACF	0.055819	-0.05517	-0.05475	-0.16932	-0.05386	-0.3623	-0.70786	32	-0.02212	0.007829
BCF	0.135665	-0.17544	0.006681	0.068191	-0.00873	0.03412	-0.24821	32	-0.00776	0.000963
ABCF	0.108957	-0.04338	-0.11532	0.085162	0.037376	-0.69925	-0.28404	32	-0.00888	0.001261
DF	0.09659	0.063454	0.22327	-0.07056	0.059855	0.115762	0.031241	32	0.000976	1.52E-05
ADF	0.141369	0.066689	-0.02414	-0.16444	-0.09455	0.485233	0.153575	32	0.004799	0.000369
BDF	0.14959	-0.02017	0.26448	-0.19372	0.052155	-0.26679	-0.01114	32	-0.00035	1.94E-06
ABDF	0.352849	-0.04341	-0.01319	-0.30808	-0.09799	0.300271	-0.16055	32	-0.00502	0.000403
CDF	0.254046	0.012833	0.161652	-0.40568	0.066847	0.236371	-0.09611	32	-0.003	0.000144
ACDF	0.03809	0.147558	-0.06859	-0.46678	0.159464	0.130744	0.128421	32	0.004013	0.000258
BCDF	0.140866	-0.07621	0.079576	-0.27674	0.145992	-0.42409	0.104533	32	0.003267	0.000171

ABCDF	0.138079	-0.07148	-0.08451	-0.29546	-0.08023	-0.35261	0.053642	32	0.001676	4.5E-05
EF	0.051336	0.063306	-0.03517	-0.20586	-0.03939	0.123002	-0.29626	32	-0.00926	0.001371
AEF	0.111064	0.158548	0.044414	-0.24793	0.14572	0.023464	-0.38008	32	-0.01188	0.002257
BEF	0.09715	-0.15712	-0.05597	-0.09542	-0.00675	0.017206	0.189299	32	0.005916	0.00056
ABEF	0.226902	-0.02671	0.132061	-0.122	0.016972	0.046106	-0.73337	32	-0.02292	0.008404
CEF	0.157403	0.044779	0.003235	-0.24741	-0.09388	-0.1544	0.369471	32	0.011546	0.002133
ACEF	0.154215	0.203258	-0.02324	-0.27767	-0.11436	-0.15015	0.567066	32	0.017721	0.005024
BCEF	0.163547	-0.21596	0.134725	-0.23025	-0.0611	0.092617	-0.10563	32	-0.0033	0.000174
ABCEF	0.079478	-0.00279	0.004733	-0.16409	-0.01872	-0.22622	0.071474	32	0.002234	7.98E-05
DEF	0.066013	0.059729	0.095242	0.079585	-0.04206	0.18511	-0.09954	32	-0.00311	0.000155
ADEF	0.169586	0.129752	0.130413	0.188033	-0.02658	0.023724	0.0289	32	0.000903	1.31E-05
BDEF	0.100611	-0.00319	0.158479	-0.02648	-0.03027	-0.02048	0.004255	32	0.000133	2.83E-07
ABDEF	0.214565	-0.08407	0.213168	-0.12999	0.066159	0.042379	-0.31884	32	-0.00996	0.001588
CDEF	0.174227	0.103573	0.070023	0.035171	0.108448	0.015485	-0.16139	32	-0.00504	0.000407
ACDEF	0.185997	0.113954	-0.08088	0.054689	-0.10352	0.096424	0.062854	32	0.001964	6.17E-05
BCDEF	0.182705	0.01177	0.010381	-0.1509	0.019518	-0.21196	0.080939	32	0.002529	0.000102
ABCDEF	0.093008	-0.0897	-0.10147	-0.11185	0.039056	0.019538	0.231502	32	0.007234	0.000837

Table A 17: Ordered Effects on BDOC removal in Experiment 2

Treatment	Estimate	J	(j-0.5)/n	Z
AC	-0.06817	1.000	0.008	-2.412
BC	-0.04971	2.000	0.024	-1.981
BE	-0.02856	3.000	0.040	-1.754
ABCE	-0.02427	4.000	0.056	-1.593
ABEF	-0.02292	5.000	0.071	-1.465
ACF	-0.02212	6.000	0.087	-1.358
ABE	-0.02079	7.000	0.103	-1.264
C	-0.01691	8.000	0.119	-1.180
AEF	-0.01188	9.000	0.135	-1.103
ABDEF	-0.00996	10.000	0.151	-1.033
BDE	-0.00952	11.000	0.167	-0.967
EF	-0.00926	12.000	0.183	-0.906
ACD	-0.009	13.000	0.198	-0.847
ABCF	-0.00888	14.000	0.214	-0.792
E	-0.0078	15.000	0.230	-0.738
BCF	-0.00776	16.000	0.246	-0.687
ABCDE	-0.00601	17.000	0.262	-0.637
CDEF	-0.00504	18.000	0.278	-0.589
ABDF	-0.00502	19.000	0.294	-0.543
ABDE	-0.00418	20.000	0.310	-0.497
BCEF	-0.0033	21.000	0.325	-0.453
DEF	-0.00311	22.000	0.341	-0.409
AD	-0.00301	23.000	0.357	-0.366
CDF	-0.003	24.000	0.373	-0.324
A	-0.00272	25.000	0.389	-0.282
BD	-0.00252	26.000	0.405	-0.241
ABC	-0.00193	27.000	0.421	-0.200
BCD	-0.00102	28.000	0.437	-0.160
BDF	-0.00035	29.000	0.452	-0.120
BDEF	0.000133	30.000	0.468	-0.080
ACDE	0.000684	31.000	0.484	-0.040
ADEF	0.000903	32.000	0.500	0.000
DF	0.000976	33.000	0.516	0.040
ACE	0.001046	34.000	0.532	0.080
ABCDF	0.001676	35.000	0.548	0.120
ACDEF	0.001964	36.000	0.563	0.160

ADE	0.001978	37.000	0.579	0.200
ABCD	0.001985	38.000	0.595	0.241
ABCEF	0.002234	39.000	0.611	0.282
BCDEF	0.002529	40.000	0.627	0.324
BCDF	0.003267	41.000	0.643	0.366
BCDE	0.003497	42.000	0.659	0.409
CD	0.003642	43.000	0.675	0.453
ACDF	0.004013	44.000	0.690	0.497
DE	0.004577	45.000	0.706	0.543
ADF	0.004799	46.000	0.722	0.589
CF	0.005754	47.000	0.738	0.637
BEF	0.005916	48.000	0.754	0.687
CDE	0.006526	49.000	0.770	0.738
ABCDEF	0.007234	50.000	0.786	0.792
BF	0.007262	51.000	0.802	0.847
ABF	0.00908	52.000	0.817	0.906
ABD	0.009127	53.000	0.833	0.967
BCE	0.011472	54.000	0.849	1.033
CEF	0.011546	55.000	0.865	1.103
F	0.01512	56.000	0.881	1.180
D	0.015759	57.000	0.897	1.264
ACEF	0.017721	58.000	0.913	1.358
CE	0.018781	59.000	0.929	1.465
AB	0.021879	60.000	0.944	1.593
AF	0.022037	61.000	0.960	1.754
AE	0.022784	62.000	0.976	1.981
B	0.026397	63.000	0.992	2.412

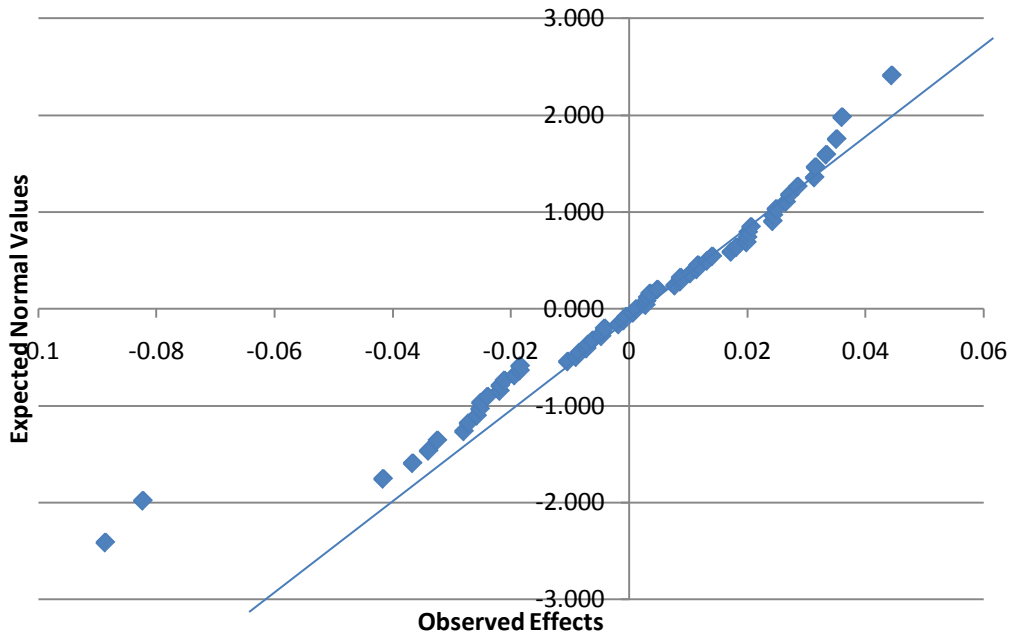

Figure A 10: Experiment 2 normal probability plot of calculated effects on BRP.

Experiment 3

Run Time

Table A 18: Yates table, calculated effects on filter run time in experiment 3

Treatment	Response	Yates 1	Yates 2	Yates 3	Yates 4	Divisor	Estimate	SS
-	48	71.67	140.8	332.8	665.37	16	41.58563	
A	23.67	69.13	192	332.57	-102.63	8	-12.8288	1316.61461
B	48	96	140.57	-51.2	-8.51	8	-1.06375	9.0525125
AB	21.13	96	192	-51.43	-8.51	8	-1.06375	9.0525125
C	48	73.27	-51.2	-2.54	102.63	8	12.82875	1316.61461
AC	48	67.3	0	-5.97	102.63	8	12.82875	1316.61461
BC	48	96	-51.43	-2.54	8.51	8	1.06375	9.0525125
ABC	48	96	0	-5.97	8.51	8	1.06375	9.0525125
D	48	-24.33	-2.54	51.2	-0.23	8	-0.02875	0.0066125
AD	25.27	-26.87	0	51.43	-0.23	8	-0.02875	0.0066125
BD	48	0	-5.97	51.2	-3.43	8	-0.42875	1.4706125
ABD	19.3	0	0	51.43	-3.43	8	-0.42875	1.4706125

CD	48	-22.73	-2.54	2.54	0.23	8	0.02875	0.0066125
ACD	48	-28.7	0	5.97	0.23	8	0.02875	0.0066125
BCD	48	0	-5.97	2.54	3.43	8	0.42875	1.4706125
ABCD	48	0	0	5.97	3.43	8	0.42875	1.4706125

Treatment	Estimate	J	(j-0.5)/n	Z
A	-12.8288	1	0.03333	-1.833915
B	-1.06375	2	0.1	-1.281552
AB	-1.06375	3	0.16667	-0.967422
ABD	-0.42875	4	0.23333	-0.727913
BD	-0.42875	5	0.3	-0.524401
D	-0.02875	6	0.36667	-0.340695
AD	-0.02875	7	0.43333	-0.167894
ACD	0.02875	8	0.5	-1.39E-16
CD	0.02875	9	0.56667	0.167894
BCD	0.42875	10	0.63333	0.340695
ABCD	0.42875	11	0.7	0.524401
ABC	1.06375	12	0.76667	0.727913
BC	1.06375	13	0.83333	0.967422
C	12.8288	14	0.9	1.281552
AC	12.8288	15	0.96667	1.833915

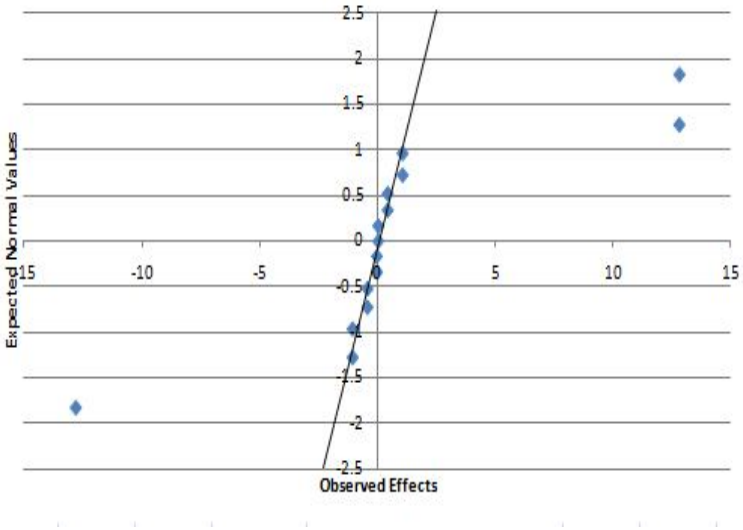

Figure A 11: Experiment 3 normal probability plot of calculated effects on filter run time.

Ripening Time

Table A 19: Yates table, calculated effects on filter ripening time in experiment 3

Treatment	Response	Yates 1	Yates 2	Yates 3	Yates 4	Divisor	Estimate	SS
-	21	42	52	114	268	16	16.75	8978
A	21	10	62	154	30	8	3.75	112.5
B	0	62	93	26	-172	8	-21.5	3698
AB	10	0	61	4	-10	8	-1.25	12.5
C	23	55	10	-94	-22	8	-2.75	60.5
AC	39	38	16	-78	16	8	2	32
BC	0	61	-3	-6	-74	8	-9.25	684.5
ABC	0	0	7	-4	-36	8	-4.5	162
D	29	0	-32	10	40	8	5	200
AD	26	10	-62	-32	-22	8	-2.75	60.5
BD	19	16	-17	6	16	8	2	32

ABD	19	0	-61	10	2	8	0.25	0.5
CD	27	-3	10	-30	-42	8	-5.25	220.5
ACD	34	0	-16	-44	4	8	0.5	2
BCD	0	7	3	-26	-14	8	-1.75	24.5
ABCD	0	0	-7	-10	16	8	2	32

Treatment	Estimate	J	(J-0.5)/n	Z
B	-21.5	1	0.03333	-1.833915
BC	-9.25	2	0.1	-1.281552
CD	-5.25	3	0.16667	-0.967422
ABC	-4.5	4	0.23333	-0.727913
C	-2.75	5	0.3	-0.524401
AD	-2.75	6	0.36667	-0.340695
BCD	-1.75	7	0.43333	-0.167894
AB	-1.25	8	0.5	-1.39E-16
ABD	0.25	9	0.56667	0.167894
ACD	0.5	10	0.63333	0.340695
AC	2	11	0.7	0.524401
BD	2	12	0.76667	0.727913
ABCD	2	13	0.83333	0.967422
A	3.75	14	0.9	1.281552
D	5	15	0.96667	1.833915

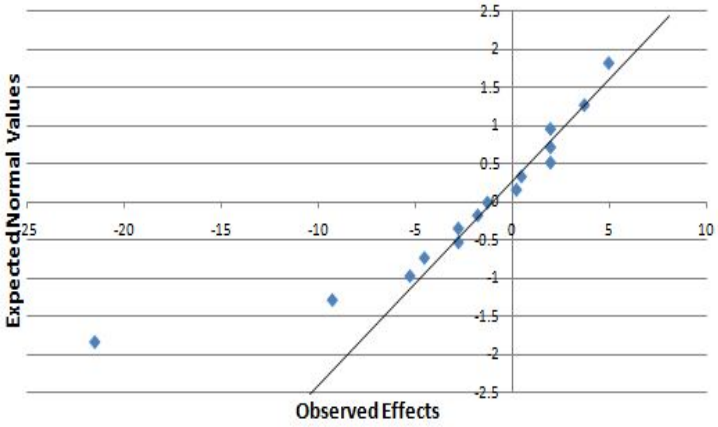

Figure A 12: Experiment 3 normal probability plot of calculated effects on filter ripening time.

DOC Removal

Table A 20: Yates table, calculated effects on DOC removal in experiment 3

Treatment	Response	Yates 1	Yates 2	Yates 3	Yates 4	Yates 5	Divisor	Estimate	Sum of Squares
I	0.129	0.555	1.761	4.147	8.488	16.073			8.073
A	0.426	1.206	2.386	4.341	7.585	-3.763	16	-0.235	0.443
B	0.307	0.755	1.639	3.746	-2.571	6.561	16	0.410	1.345
AB	0.899	1.631	2.702	3.839	-1.192	-3.075	16	-0.192	0.296
C	0.618	0.399	1.487	-0.818	3.316	3.685	16	0.230	0.424
AC	0.137	1.240	2.259	-1.753	3.245	-7.431	16	-0.464	1.726
BC	1.429	0.877	1.307	-0.759	-1.339	3.337	16	0.209	0.348
ABC	0.203	1.824	2.532	-0.433	-1.736	-5.243	16	-0.328	0.859
D	0.136	0.775	0.889	1.528	1.688	0.286	16	0.018	0.003
AD	0.263	0.712	-1.707	1.788	1.997	-0.608	16	-0.038	0.012
BD	0.386	0.363	0.595	1.471	-5.539	0.563	16	0.035	0.010
ABD	0.854	1.897	-2.348	1.775	-1.892	-0.481	16	-0.030	0.007
CD	0.718	0.562	0.103	-0.449	0.332	0.890	16	0.056	0.025
ACD	0.159	0.745	-0.862	-0.890	3.005	-0.308	16	-0.019	0.003
BCD	1.807	0.470	0.247	-0.848	-2.611	-0.309	16	-0.019	0.003
ABCD	0.017	2.062	-0.680	-0.889	-2.632	-0.633	16	-0.040	0.013
E	0.414	0.297	0.651	0.625	0.194	-0.902	16	-0.056	0.025
AE	0.361	0.592	0.877	1.063	0.093	1.379	16	0.086	0.059
BE	0.278	-0.481	0.841	0.772	-0.935	-0.070	16	-0.004	0.000
ABE	0.434	-1.226	0.947	1.225	0.326	-0.397	16	-0.025	0.005

CE	0.133	0.127	-0.063	-2.596	0.260	0.310	16	0.019	0.003
ACE	0.230	0.468	1.534	-2.943	0.304	3.647	16	0.228	0.416
BCE	1.428	-0.559	0.183	-0.965	-0.441	2.674	16	0.167	0.223
ABCE	0.468	-1.789	1.592	-0.927	-0.041	-0.021	16	-0.001	0.000
DE	0.279	-0.053	0.295	0.226	0.438	-0.101	16	-0.006	0.000
ADE	0.283	0.156	-0.744	0.106	0.453	1.261	16	0.079	0.050
BDE	0.251	0.097	0.341	1.597	-0.347	0.044	16	0.003	0.000
ABDE	0.494	-0.960	-1.231	1.409	0.038	0.400	16	0.025	0.005
CDE	0.123	0.004	0.209	-1.039	-0.120	0.015	16	0.001	0.000
ACDE	0.347	0.243	-1.057	-1.572	-0.188	0.385	16	0.024	0.005
BCDE	1.483	0.224	0.239	-1.266	-0.533	-0.068	16	-0.004	0.000
ABCDE	0.579	-0.904	-1.128	-1.367	-0.101	0.432	16	0.027	0.006

Table A 21: Ordered Effects on DOC removal in Experiment 3

Treatment	Estimate	J	(j-0.5)/n	Z
AC	-0.464	1.000	0.016	-2.141
ABC	-0.328	2.000	0.048	-1.661
A	-0.235	3.000	0.081	-1.401
AB	-0.192	4.000	0.113	-1.211
E	-0.056	5.000	0.145	-1.057
ABCD	-0.040	6.000	0.177	-0.925
AD	-0.038	7.000	0.210	-0.808
ABD	-0.030	8.000	0.242	-0.700
ABE	-0.025	9.000	0.274	-0.600
BCD	-0.019	10.000	0.306	-0.506
ACD	-0.019	11.000	0.339	-0.416
DE	-0.006	12.000	0.371	-0.329
BE	-0.004	13.000	0.403	-0.245
BCDE	-0.004	14.000	0.435	-0.162
ABCE	-0.001	15.000	0.468	-0.081
CDE	0.001	16.000	0.500	0.000
BDE	0.003	17.000	0.532	0.081
D	0.018	18.000	0.565	0.162
CE	0.019	19.000	0.597	0.245
ACDE	0.024	20.000	0.629	0.329
ABDE	0.025	21.000	0.661	0.416
ABCDE	0.027	22.000	0.694	0.506
BD	0.035	23.000	0.726	0.600
CD	0.056	24.000	0.758	0.700
ADE	0.079	25.000	0.790	0.808
AE	0.086	26.000	0.823	0.925
BCE	0.167	27.000	0.855	1.057
BC	0.209	28.000	0.887	1.211
ACE	0.228	29.000	0.919	1.401
C	0.230	30.000	0.952	1.661
B	0.410	31.000	0.984	2.141

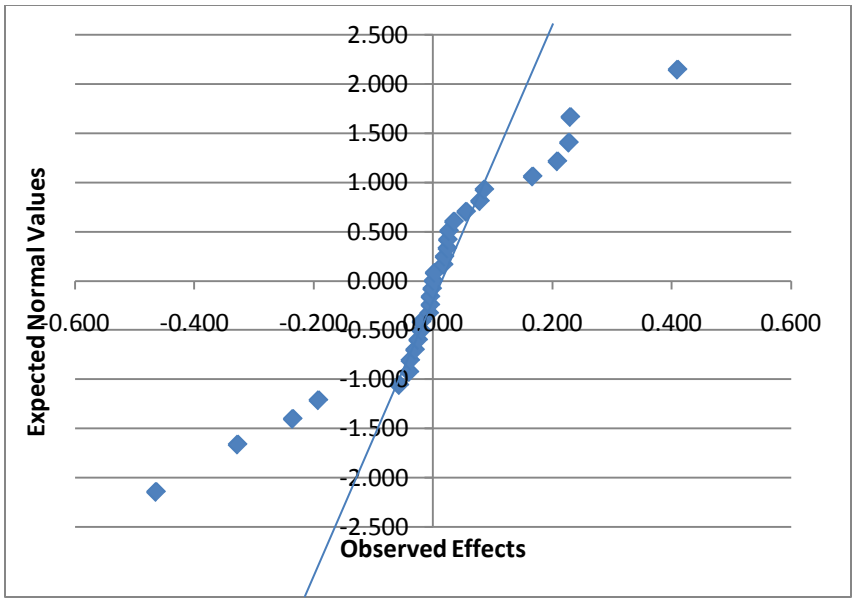

Figure A 13: Experiment 3 normal probability plot of calculated effects on DOC.

BDOC Removal

Table A 22: Yates table, calculated effects on BDOC removal in experiment 3

Treatment	Response	Yates 1	Yates 2	Yates 3	Yates 4	Yates 5	Divisor	Estimate	Sum of Squares
I	0.217	0.351	1.617	2.863	5.574	10.037			3.148051
A	0.134	1.266	1.246	2.712	4.463	-5.513	16	-0.345	0.949801
B	0.432	0.759	1.316	3.269	-1.889	3.352	16	0.210	0.351181
AB	0.834	0.487	1.396	1.194	-3.624	1.686	16	0.105	0.088876
C	0.650	0.289	1.809	-0.789	1.481	-0.931	16	-0.058	0.027097
AC	0.109	1.027	1.460	-1.100	1.871	-5.397	16	-0.337	0.910252
BC	0.527	0.648	0.742	-2.875	1.030	-0.960	16	-0.060	0.028783
ABC	-0.040	0.748	0.452	-0.749	0.657	0.994	16	0.062	0.030902
D	0.245	0.640	0.319	0.643	-0.292	-2.226	16	-0.139	0.154825
AD	0.044	1.169	-1.108	0.838	-0.639	1.815	16	0.113	0.102995
BD	0.286	0.052	0.254	1.885	-3.035	-1.705	16	-0.107	0.090793
ABD	0.741	1.408	-1.354	-0.014	-2.362	2.122	16	0.133	0.140659
CD	0.641	0.384	-0.583	0.459	-1.825	0.510	16	0.032	0.008133
ACD	0.007	0.358	-2.292	0.570	0.865	0.875	16	0.055	0.02395
BCD	0.734	0.220	-0.048	-0.677	-1.252	-0.241	16	-0.015	0.001808
ABCD	0.014	0.232	-0.701	1.334	2.247	0.510	16	0.032	0.008115
E	0.287	-0.083	0.915	-0.371	-0.151	-1.112	16	-0.069	0.038613
AE	0.353	0.402	-0.272	0.080	-2.075	-1.735	16	-0.108	0.094041

BE	0.909	-0.541	0.738	-0.349	-0.310	0.391	16	0.024	0.004767
ABE	0.260	-0.567	0.100	-0.290	2.126	-0.373	16	-0.023	0.004339
CE	0.608	-0.201	0.529	-1.427	0.194	-0.348	16	-0.022	0.003775
ACE	-0.557	0.455	1.356	-1.608	-1.899	0.673	16	0.042	0.014165
BCE	1.267	-0.634	-0.026	-1.709	0.111	2.691	16	0.168	0.226222
ABCE	0.141	-0.720	0.012	-0.653	2.010	3.499	16	0.219	0.382679
DE	0.184	0.066	0.485	-1.187	0.451	-1.924	16	-0.120	0.115647
ADE	0.200	-0.649	-0.026	-0.638	0.059	2.436	16	0.152	0.185424
BDE	0.211	-1.165	0.656	0.827	-0.180	-2.093	16	-0.131	0.136902
ABDE	0.147	-1.127	-0.086	0.038	1.056	1.899	16	0.119	0.112736
CDE	0.639	0.016	-0.715	-0.511	0.548	-0.392	16	-0.024	0.004795
ACDE	-0.419	-0.064	0.038	-0.742	-0.789	1.236	16	0.077	0.047732
BCDE	-0.062	-1.057	-0.080	0.753	-0.231	-1.337	16	-0.084	0.055866
ABCDE	0.294	0.356	1.414	1.494	0.740	0.971	16	0.061	0.029486

Table A 23: Ordered Effects on DOC removal in Experiment 3

Treatment	Estimate	J	(j-0.5)/n	Z
A	-0.34457	1	0.016129	-2.1412
AC	-0.33732	2	0.048387	-1.6607
D	-0.13912	3	0.080645	-1.40075
BDE	-0.13082	4	0.112903	-1.21123
DE	-0.12023	5	0.145161	-1.05741
AE	-0.10842	6	0.177419	-0.92524
BD	-0.10653	7	0.209677	-0.80754
BCDE	-0.08357	8	0.241935	-0.70009
E	-0.06947	9	0.274194	-0.60018
BC	-0.05998	10	0.306452	-0.50593
C	-0.0582	11	0.33871	-0.41599
CDE	-0.02448	12	0.370968	-0.32929
ABE	-0.02329	13	0.403226	-0.24501
CE	-0.02172	14	0.435484	-0.16243
BCD	-0.01503	15	0.467742	-0.08095
BE	0.02441	16	0.5	-1.4E-16
ABCD	0.031849	17	0.532258	0.080947
CD	0.031885	18	0.564516	0.162429
ACE	0.042079	19	0.596774	0.245006
ACD	0.054715	20	0.629032	0.329291
ABCDE	0.06071	21	0.66129	0.415987
ABC	0.062151	22	0.693548	0.505934
ACDE	0.077243	23	0.725806	0.600179
AB	0.105401	24	0.758065	0.70009
AD	0.113465	25	0.790323	0.807541
ABDE	0.11871	26	0.822581	0.925245
ABD	0.132599	27	0.854839	1.057414
ADE	0.152243	28	0.887097	1.211232
BCE	0.16816	29	0.919355	1.400745
B	0.209518	30	0.951613	1.660698
ABCE	0.218712	31	0.983871	2.141198

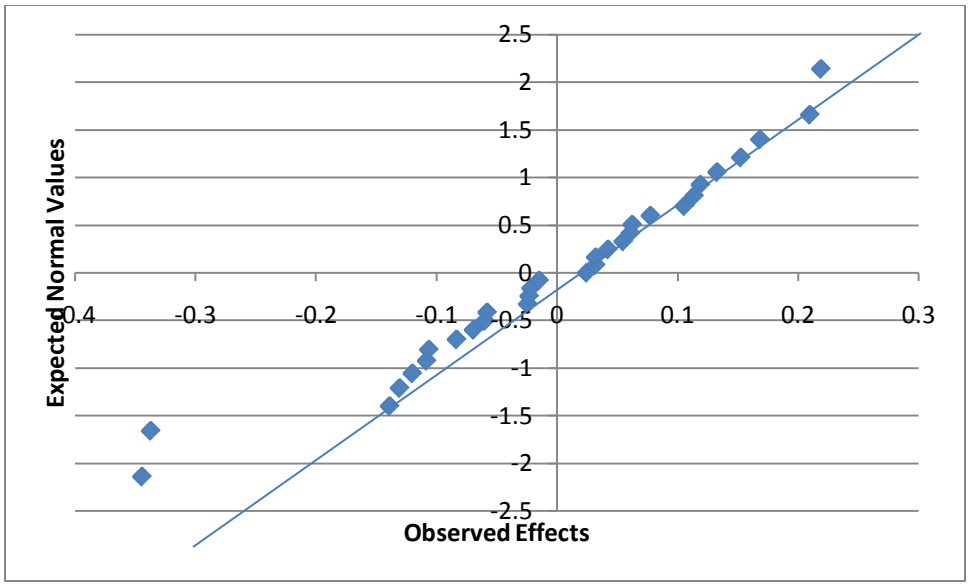

Figure A 14: Figure A 15: Experiment 3 normal probability plot of calculated effects on BDOC removal.

BRP

Table A 24: Yates table, calculated effects on DOC removal in experiment 3

Treatment	Response	Yates 1	Yates 2	Yates 3	Yates 4	Yates 5	Yates 6	Divisor	Estimate	SS
1	0.05	0.03	0.28	1.19	2.04	3.48	7.37			
A	-0.01	0.25	0.90	0.86	1.43	3.90	-1.17	32	-0.04	0.02
B	0.03	0.63	0.20	0.72	2.74	-0.11	0.94	32	0.03	0.01
AB	0.22	0.27	0.66	0.72	1.16	-1.06	1.64	32	0.05	0.04
C	0.33	0.04	0.42	1.38	-0.37	0.53	1.64	32	0.05	0.04
AC	0.30	0.16	0.30	1.36	0.26	0.41	-3.46	32	-0.11	0.19
BC	0.24	0.44	0.45	0.78	-0.55	0.53	-3.73	32	-0.12	0.22
ABC	0.03	0.22	0.26	0.38	-0.50	1.12	-1.95	32	-0.06	0.06
D	0.03	0.06	0.52	-0.11	-0.25	0.77	-0.74	32	-0.02	0.01
AD	0.01	0.36	0.86	-0.26	0.78	0.86	-0.18	32	-0.01	0.00
BD	0.01	0.09	0.54	0.09	-0.04	-1.84	-0.15	32	0.00	0.00
ABD	0.14	0.21	0.82	0.17	0.45	-1.62	-0.39	32	-0.01	0.00
CD	0.32	0.07	0.42	-0.36	0.22	-1.37	0.11	32	0.00	0.00
ACD	0.12	0.39	0.35	-0.19	0.31	-2.36	0.25	32	0.01	0.00
BCD	0.20	0.11	0.03	-0.12	1.10	-1.01	0.09	32	0.00	0.00
ABCD	0.02	0.16	0.35	-0.39	0.01	-0.93	0.21	32	0.01	0.00
E	0.02	-0.02	0.12	-0.15	1.08	-0.33	-2.19	32	-0.07	0.08
AE	0.04	0.55	-0.23	-0.11	-0.30	-0.41	0.68	32	0.02	0.01

BE	0.04	0.70	0.12	0.41	0.61	-0.08	1.52	32	0.05	0.04
ABE	0.32	0.15	-0.38	0.37	0.25	-0.10	-1.00	32	-0.03	0.02
CE	0.10	0.03	0.30	0.02	-0.86	0.01	-1.74	32	-0.05	0.05
ACE	-0.01	0.52	-0.21	-0.06	-0.98	-0.16	1.17	32	0.04	0.02
BCE	0.16	0.68	0.32	0.26	-1.46	-0.11	2.42	32	0.08	0.09
ABCE	0.05	0.14	-0.15	0.19	-0.17	-0.27	-0.09	32	0.00	0.00
DE	-0.02	0.14	0.25	0.06	-0.91	-0.23	-0.05	32	0.00	0.00
ADE	0.08	0.28	-0.60	0.15	-0.45	0.34	-0.20	32	-0.01	0.00
BDE	0.08	0.12	0.21	0.26	-2.16	-0.10	-0.07	32	0.00	0.00
ABDE	0.30	0.24	-0.40	0.05	-0.20	0.35	0.01	32	0.00	0.00
CDE	0.07	-0.08	0.01	0.69	-0.57	0.17	0.54	32	0.02	0.00
ACDE	0.03	0.11	-0.13	0.41	-0.44	-0.08	0.06	32	0.00	0.00
BCDE	0.13	0.18	-0.18	0.00	-0.35	0.36	-0.59	32	-0.02	0.01
ABCDE	0.03	0.18	-0.21	0.01	-0.58	-0.15	-0.34	32	-0.01	0.00
F	0.03	-0.06	0.22	0.62	-0.33	-0.61	0.42	32	0.01	0.00
AF	-0.06	0.19	-0.37	0.46	0.00	-1.58	-0.94	32	-0.03	0.01
BF	0.10	-0.02	0.11	-0.12	-0.02	0.63	-0.11	32	0.00	0.00
ABF	0.44	-0.21	-0.22	-0.19	-0.40	0.05	0.59	32	0.02	0.01
CF	0.57	-0.01	0.30	0.33	-0.16	1.04	0.09	32	0.00	0.00
ACF	0.14	0.13	0.11	0.28	0.08	0.48	0.21	32	0.01	0.00
BCF	0.16	-0.19	0.32	-0.07	0.17	0.09	-1.00	32	-0.03	0.02
ABCF	-0.01	-0.19	0.05	0.32	-0.27	-1.09	0.08	32	0.00	0.00
DF	0.04	0.02	0.57	-0.36	0.04	-1.38	-0.09	32	0.00	0.00

ADF	-0.01	0.28	-0.55	-0.50	-0.03	-0.36	-0.03	32	0.00	0.00
BDF	0.13	-0.11	0.49	-0.51	-0.08	-0.12	-0.16	32	-0.01	0.00
ABDF	0.38	-0.11	-0.55	-0.46	-0.08	1.29	-0.16	32	-0.01	0.00
CDF	0.47	0.10	0.14	-0.85	0.09	0.46	0.57	32	0.02	0.01
ACDF	0.21	0.22	0.12	-0.61	-0.20	1.96	0.45	32	0.01	0.00
BCDF	0.14	-0.04	0.19	-0.14	-0.28	0.13	-0.25	32	-0.01	0.00
ABCDF	0.00	-0.11	0.00	-0.03	0.01	-0.23	-0.51	32	-0.02	0.00
EF	0.10	-0.09	0.25	-0.58	-0.16	0.33	-0.97	32	-0.03	0.01
AEF	0.05	0.34	-0.18	-0.33	-0.07	-0.38	-0.58	32	-0.02	0.01
BEF	0.11	-0.43	0.15	-0.18	-0.06	0.24	-0.55	32	-0.02	0.00
ABEF	0.17	-0.17	0.01	-0.27	0.40	-0.43	-1.18	32	-0.04	0.02
CEF	0.06	-0.05	0.25	-1.12	-0.15	-0.08	1.02	32	0.03	0.02
ACEF	0.05	0.25	0.00	-1.04	0.05	0.00	1.40	32	0.04	0.03
BCEF	0.18	-0.26	0.12	-0.02	0.24	-0.29	1.50	32	0.05	0.04
ABCEF	0.06	-0.14	-0.07	-0.18	0.11	0.29	-0.36	32	-0.01	0.00
DEF	0.05	-0.05	0.43	-0.43	0.25	0.09	-0.71	32	-0.02	0.01
ADEF	-0.13	0.06	0.26	-0.14	-0.08	0.45	-0.67	32	-0.02	0.01
BDEF	0.05	-0.01	0.30	-0.25	0.08	0.20	0.08	32	0.00	0.00
ABDEF	0.06	-0.12	0.11	-0.19	-0.17	-0.13	0.58	32	0.02	0.01
CDEF	0.10	-0.18	0.11	-0.17	0.29	-0.34	0.37	32	0.01	0.00
ACDEF	0.08	0.00	-0.11	-0.19	0.07	-0.25	-0.33	32	-0.01	0.00
BCDEF	0.18	-0.02	0.18	-0.22	-0.02	-0.22	0.09	32	0.00	0.00
ABCDEF	-0.01	-0.19	-0.17	-0.36	-0.14	-0.12	0.10	32	0.00	0.00

Table A 25: Ordered Effects on BRP in Experiment 3

Treatment	Estimate	J	(j-0.5)/n	Z
BC	-0.11658	1.000	0.008	-2.412
AC	-0.10808	2.000	0.024	-1.981
E	-0.06858	3.000	0.040	-1.754
ABC	-0.06084	4.000	0.056	-1.593
CE	-0.05443	5.000	0.071	-1.465
ABEF	-0.03683	6.000	0.087	-1.358
A	-0.03656	7.000	0.103	-1.264
BCF	-0.03122	8.000	0.119	-1.180
ABE	-0.0311	9.000	0.135	-1.103
EF	-0.03047	10.000	0.151	-1.033
AF	-0.02948	11.000	0.167	-0.967
D	-0.02322	12.000	0.183	-0.906
DEF	-0.02205	13.000	0.198	-0.847
ADEF	-0.02097	14.000	0.214	-0.792
BCDE	-0.0183	15.000	0.230	-0.738
AEF	-0.01821	16.000	0.246	-0.687
BEF	-0.01722	17.000	0.262	-0.637
ABCDF	-0.01602	18.000	0.278	-0.589
ABD	-0.01205	19.000	0.294	-0.543
ABCEF	-0.01123	20.000	0.310	-0.497
ABCDE	-0.01056	21.000	0.325	-0.453
ACDEF	-0.0104	22.000	0.341	-0.409
BCDF	-0.00789	23.000	0.357	-0.366
ADE	-0.00612	24.000	0.373	-0.324
AD	-0.00561	25.000	0.389	-0.282
BDF	-0.00513	26.000	0.405	-0.241
ABDF	-0.00506	27.000	0.421	-0.200
BD	-0.00457	28.000	0.437	-0.160
BF	-0.00351	29.000	0.452	-0.120

ABCE	-0.00286	30.000	0.468	-0.080
DF	-0.00271	31.000	0.484	-0.040
BDE	-0.00229	32.000	0.500	0.000
DE	-0.00146	33.000	0.516	0.040
ADF	-0.00084	34.000	0.532	0.080
ABDE	0.000167	35.000	0.548	0.120
ACDE	0.001972	36.000	0.563	0.160
BDEF	0.002449	37.000	0.579	0.200
ABCF	0.002498	38.000	0.595	0.241
BCD	0.002699	39.000	0.611	0.282
CF	0.002766	40.000	0.627	0.324
BCDEF	0.002864	41.000	0.643	0.366
ABCDEF	0.003258	42.000	0.659	0.409
CD	0.003389	43.000	0.675	0.453
ABCD	0.006427	44.000	0.690	0.497
ACF	0.00664	45.000	0.706	0.543
ACD	0.00792	46.000	0.722	0.589
CDEF	0.011447	47.000	0.738	0.637
F	0.013109	48.000	0.754	0.687
ACDF	0.013943	49.000	0.770	0.738
CDE	0.016885	50.000	0.786	0.792
CDF	0.017853	51.000	0.802	0.847
ABDEF	0.018269	52.000	0.817	0.906
ABF	0.018413	53.000	0.833	0.967
AE	0.021252	54.000	0.849	1.033
B	0.029394	55.000	0.865	1.103
CEF	0.031932	56.000	0.881	1.180
ACE	0.036652	57.000	0.897	1.264
ACEF	0.043903	58.000	0.913	1.358
BCEF	0.046794	59.000	0.929	1.465
BE	0.047481	60.000	0.944	1.593
C	0.051201	61.000	0.960	1.754
AB	0.051354	62.000	0.976	1.981
BCE	0.075482	63.000	0.992	2.412

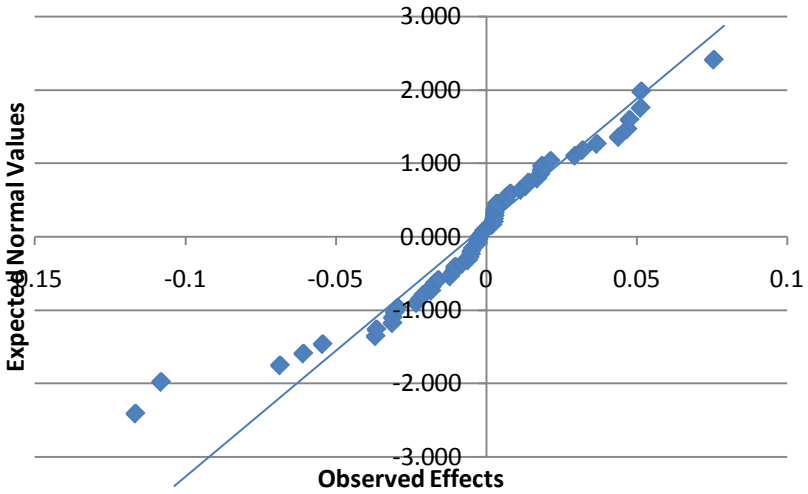

Figure A 16: Experiment 3 normal probability plot of calculated effects on BRP

Appendix B – Pilot-scale Factorial Raw Data

Table B 1: DOC Removal Raw Data

Run ID	T (hours)	Date	Filter	DOCi	DOCi (SDEV)	DOC Removal	DOC Removal (SD)
		09/02/2010	Influent	4.98	0.05		
		09/02/2010	1	4.36	0.06	0.62	0.12
		09/02/2010	2	4.39	0.04	0.59	0.12
		09/02/2010	3	4.49	0.05	0.49	0.12
	0	09/02/2010	4	4.53	0.07	0.45	0.13
		10/02/2010	Influent	4.43	0.11		
		10/02/2010	1	4.69	0.07	-0.26	0.13
		10/02/2010	2	4.49	0.07	-0.06	0.13
		10/02/2010	3	4.81	0.06	-0.38	0.12
1a	24	10/02/2010	4	4.45	0.15	-0.02	0.18
		11/02/2010	Influent	4.79	0.12		
		11/02/2010	1	4.53	0.08	0.26	0.13
		11/02/2010	2	4.49	0.09	0.30	0.14
		11/02/2010	3	4.66	0.04	0.13	0.12
	0	11/02/2010	4	4.65	0.08	0.14	0.13
		12/02/2010	Influent	5.06	0.06		
		12/02/2010	1	absent			
		12/02/2010	2	4.52	0.05	0.54	0.12
		12/02/2010	3	4.64	0.11	0.41	0.16
1b	24	12/02/2010	4	4.78	0.06	0.28	0.12
		16/02/2010	Influent	5.04	0.09		
		16/02/2010	1	4.55	0.06	0.49	0.12
		16/02/2010	2	4.42	0.07	0.61	0.13
		16/02/2010	3	4.59	0.05	0.44	0.12
	0	16/02/2010	4	4.54	0.04	0.50	0.12
		17/02/2010	Influent	4.66	0.16		
		17/02/2010	1	4.51		0.14	0.11
		17/02/2010	2	4.41	0.08	0.24	0.13
		17/02/2010	3	4.80	0.11	-0.15	0.16
2a	24	17/02/2010	4	4.53	0.11	0.12	0.15
		18/02/2010	Influent	4.87	0.04		
		18/02/2010	1	4.47		0.39	0.11
		18/02/2010	2	4.47	0.06	0.40	0.12
2b	0	18/02/2010	3	4.44		0.43	0.11

		18/02/2010	4	4.60	0.04	0.26	0.11
		19/02/2010	Influent	4.93	0.06		
		19/02/2010	1	4.38		0.55	0.11
		19/02/2010	2	4.42	0.05	0.52	0.12
		19/02/2010	3	4.57	0.03	0.36	0.11
	24	19/02/2010	4	4.65	0.03	0.28	0.11
		23/02/2010	Influent	4.70	0.12		
		23/02/2010	1	4.33		0.37	0.11
		23/02/2010	2	4.43	0.07	0.27	0.13
		23/02/2010	3	4.39	0.16	0.31	0.20
	0	23/02/2010	4	4.44	0.11	0.26	0.15
		24/02/2010	Influent	5.09			
		24/02/2010	1	4.59	0.04	0.51	0.12
		24/02/2010	2	4.59	0.13	0.51	0.17
		24/02/2010	3	4.79	0.14	0.30	0.18
4a	24	24/02/2010	4	4.26	0.22	0.84	0.24
		25/02/2010	Influent	5.50	0.05		
		25/02/2010	1	4.61		0.89	0.11
		25/02/2010	2	4.44	0.10	1.06	0.15
		25/02/2010	3	4.60	0.11	0.90	0.15
	0	25/02/2010	4	4.65	0.09	0.85	0.14
		26/02/2010	Influent	5.12	0.10		
		26/02/2010	1	4.61		0.51	0.11
		26/02/2010	2	4.68	0.13	0.45	0.17
		26/02/2010	3	4.69	0.04	0.43	0.12
4b	24	26/02/2010	4	4.63	0.16	0.49	0.19
		02/03/2010	Influent	4.33	0.04		
		02/03/2010	1	3.62		0.70	0.11
		02/03/2010	2	3.57	0.03	0.76	0.11
		02/03/2010	3	3.59	0.02	0.73	0.11
	0	02/03/2010	4	3.71	0.05	0.62	0.12
		03/03/2010	Influent	4.79	0.05		
		03/03/2010	1	4.26		0.54	0.11
		03/03/2010	2	4.25	0.13	0.54	0.17
		03/03/2010	3	4.36	0.08	0.43	0.14
3a	24	03/03/2010	4	4.34	0.04	0.45	0.11
		04/03/2010	Influent	4.52	0.07		
		04/03/2010	1	4.12	0.11	0.40	0.15
		04/03/2010	2	4.02	0.06	0.50	0.13
		04/03/2010	3	4.21	0.05	0.31	0.12
	0	04/03/2010	4	4.13	0.16	0.39	0.19
3b	24	05/03/2010	Influent	4.41	0.16		

		05/03/2010	1	4.01	0.10	0.40	0.15
		05/03/2010	2	4.02	0.08	0.39	0.13
		05/03/2010	3	4.13	0.06	0.28	0.13
		05/03/2010	4	4.16	0.15	0.25	0.19
		09/03/2010	Influent	4.59	0.06		
		09/03/2010	1	4.27		0.32	0.11
		09/03/2010	2	4.15	0.06	0.44	0.13
		09/03/2010	3	4.38	0.03	0.21	0.11
	0	09/03/2010	4	4.37	0.01	0.22	0.11
		10/03/2010	Influent	4.61	0.05		
		10/03/2010	1	4.42		0.19	0.11
		10/03/2010	2	4.20	0.02	0.42	0.11
		10/03/2010	3	4.48	0.06	0.13	0.12
5a	24	10/03/2010	4	4.41	0.03	0.20	0.11
		11/03/2010	Influent	5.16	0.10		
		11/03/2010	1	4.27	0.02	0.88	0.11
		11/03/2010	2	4.21	0.01	0.95	0.11
		11/03/2010	3	4.54	0.13	0.62	0.17
	0	11/03/2010	4	4.44	0.02	0.72	0.11
		12/03/2010	Influent	4.24	0.05		
		12/03/2010	1	3.88		0.35	0.11
		12/03/2010	2	3.95	0.02	0.29	0.11
		12/03/2010	3	4.10	0.03	0.13	0.11
5b	24	12/03/2010	4	4.11	0.02	0.12	0.11
		16/03/2010	Influent	4.00	0.10		
		16/03/2010	1	3.57		0.43	0.11
		16/03/2010	2	3.65	0.04	0.35	0.11
		16/03/2010	3	3.86	0.06	0.14	0.13
	0	16/03/2010	4	3.84	0.21	0.16	0.23
		17/03/2010	Influent	4.04	0.04		
		17/03/2010	1	3.99	0.07	0.05	0.13
		17/03/2010	2	3.88	0.13	0.00	0.17
		17/03/2010	3	4.34	0.09	-0.30	0.14
6a	24	17/03/2010	4	4.24	0.09	-0.20	0.14
		18/03/2010	Influent	4.63	0.03		
		18/03/2010	1	3.74		0.89	0.11
		18/03/2010	2	3.85	0.02	0.79	0.11
		18/03/2010	3	4.28	0.10	0.36	0.15
	0	18/03/2010	4	4.24	0.06	0.39	0.12
		19/03/2010	Influent	4.76	0.12		
		19/03/2010	1	4.29		0.48	0.11
6b	24	19/03/2010	2	4.23	0.03	0.53	0.11

		19/03/2010	3	4.53	0.04	0.23	0.11
		19/03/2010	4	4.41	0.09	0.35	0.14
		23/03/2010	Influent	5.02	0.21		
		23/03/2010	1	4.19	0.10	0.83	0.14
		23/03/2010	2	4.37	0.07	0.65	0.13
		23/03/2010	3	4.90	0.26	0.13	0.28
	0	23/03/2010	4	4.63	0.10	0.39	0.15
		24/03/2010	Influent	4.55	0.12		
		24/03/2010	1	3.87	0.12	0.68	0.16
		24/03/2010	2	3.86	0.10	0.69	0.15
		24/03/2010	3	4.11	0.08	0.44	0.14
8a	24	24/03/2010	4	4.15	0.10	0.40	0.14
		30/03/2010	Influent	3.91			
		30/03/2010	1	3.37		0.54	0.11
		30/03/2010	2	3.56		0.35	0.11
		30/03/2010	3	3.71		0.20	0.11
	0	30/03/2010	4	3.89		0.02	0.11
		31/03/2010	Influent	4.63			
		31/03/2010	1	4.12		0.51	0.11
		31/03/2010	2	3.86		0.76	0.11
		31/03/2010	3	4.16		0.47	0.11
8b	24	31/03/2010	4	4.05		0.58	0.11
		04/04/2010	Influent	5.33			
		04/04/2010	1	3.45		1.88	0.11
		04/04/2010	2	3.94		1.39	0.11
		04/04/2010	3	3.90		1.43	0.11
	0	04/04/2010	4	3.53		1.81	0.11
		05/04/2010	Influent	5.22			
		05/04/2010	1	xxx			
		05/04/2010	2	3.44		1.78	0.11
		05/04/2010	3	3.79		1.43	0.11
7a	24	05/04/2010	4	3.73		1.48	0.11
		06/04/2010	Influent	4.05			
		06/04/2010	1	3.75		0.30	0.11
		06/04/2010	2	3.78		0.27	0.11
		06/04/2010	3	3.84		0.21	0.11
	0	06/04/2010	4	4.25		-0.20	0.11
		07/04/2010	Influent	3.98			
		07/04/2010	1	3.63		0.35	0.11
		07/04/2010	2	4.07		-0.08	0.11
		07/04/2010	3	xxx			0.11
7b	24	07/04/2010	4	3.77		0.21	0.11

		10/04/2010	Influent	3.75			
		10/04/2010	1	4.18		-0.43	0.11
		10/04/2010	2	3.59		0.16	0.11
		10/04/2010	3	4.11		-0.36	0.11
	0	10/04/2010	4	3.84		-0.09	0.11
		11/04/2010	Influent	4.22			
		11/04/2010	1	3.51		0.71	0.11
		11/04/2010	2	3.76		0.46	0.11
		11/04/2010	3	3.96		0.27	0.11
1a	24	11/04/2010	4	4.06		0.16	0.11
		12/04/2010	Influent	4.36			
		12/04/2010	1	3.84		0.52	0.11
		12/04/2010	2	4.06		0.31	0.11
		12/04/2010	3	4.29		0.07	0.11
	0	12/04/2010	4	4.38		-0.02	0.11
		13/04/2010	Influent	4.52			
		13/04/2010	1	4.35		0.17	0.11
		13/04/2010	2	4.20		0.32	0.11
		13/04/2010	3	3.70		0.82	0.11
1b	24	13/04/2010	4	5.15		-0.63	0.11
		14/04/2010	Influent	7.12			
		14/04/2010	1	3.79		3.32	0.11
		14/04/2010	2	4.06		3.06	0.11
		14/04/2010	3	5.05		2.07	0.11
	0	14/04/2010	4	4.28		2.83	0.11
		15/04/2010	Influent	3.53			
		15/04/2010	1	3.48		0.05	0.11
		15/04/2010	2	3.80		-0.27	0.11
		15/04/2010	3	3.82		-0.29	0.11
4a	24	15/04/2010	4	3.82		-0.29	0.11

Table B 2: Raw BDOC Data

Run ID	T (hours)	Date	Filter	DOCi	DOCi (SDEV)	DOCe	DOCe (SDEV)	BDOC	(SD)	BDOC Removal	SD
		09/02/2010	Influent	4.98	0.05	4.07	0.01	0.91	0.06		
		09/02/2010	1	4.36	0.06	4.04	0.05	0.32	0.08	0.59	0.10
		09/02/2010	2	4.39	0.04	4.07	0.02	0.32	0.05	0.59	0.07
		09/02/2010	3	4.49	0.05	4.16	0.05	0.33	0.07	0.58	0.09
	0	09/02/2010	4	4.53	0.07	4.20	0.02	0.34	0.07	0.57	0.09
		10/02/2010	Influent	4.43	0.11	3.87	0.10	0.56	0.15		
		10/02/2010	1	4.69	0.07	3.69	0.14	1.01	0.16	-0.45	0.22
		10/02/2010	2	4.49	0.07	3.82	0.14	0.67	0.15	-0.11	0.22
		10/02/2010	3	4.81	0.06	3.88	0.17	0.93	0.18	-0.37	0.23
1a	24	10/02/2010	4	4.45	0.15	3.56	0.04	0.89	0.15	-0.34	0.21
		11/02/2010	Influent	4.79	0.12	3.97	0.12	0.82	0.17		
		11/02/2010	1	4.53	0.08	3.91	0.04	0.63	0.09	0.19	0.19
		11/02/2010	2	4.49	0.09	3.89	0.10	0.60	0.14	0.22	0.22
		11/02/2010	3	4.66	0.04	4.06	0.06	0.60	0.07	0.22	0.18
	0	11/02/2010	4	4.65	0.08	4.08	0.05	0.57	0.09	0.25	0.19
		12/02/2010	Influent	5.06	0.06	3.97	0.13	1.08	0.14		
		12/02/2010	1	absent		3.77	0.15				
		12/02/2010	2	4.52	0.05	3.88	0.11	0.64	0.12	0.45	0.19
		12/02/2010	3	4.64	0.11	3.85	0.06	0.80	0.13	0.29	0.19
1b	24	12/02/2010	4	4.78	0.06	3.88	0.07	0.90	0.09	0.18	0.17
2a	0	16/02/2010	Influent	5.04	0.09	4.01	0.09	1.03	0.13		

		16/02/2010	1	4.55	0.06	4.35	0.08	0.19	0.10	0.83	0.16
		16/02/2010	2	4.42	0.07	4.09	0.04	0.34	0.09	0.69	0.15
		16/02/2010	3	4.59	0.05	4.20	0.06	0.39	0.08	0.63	0.15
		16/02/2010	4	4.54	0.04	4.14	0.05	0.40	0.07	0.62	0.15
		17/02/2010	Influent	4.66	0.16	3.69	0.10	0.97	0.19		
		17/02/2010	1	4.51		3.66	0.10	0.85		0.11	0.19
		17/02/2010	2	4.41	0.08	3.81	0.16	0.60	0.17	0.37	0.26
		17/02/2010	3	4.80	0.11	4.44	0.04	0.36	0.12	0.61	0.23
	24	17/02/2010	4	4.53	0.11	3.69	0.10	0.85	0.15	0.12	0.24
		18/02/2010	Influent	4.87	0.04	3.97	0.07	0.90	0.08		
		18/02/2010	1	4.47		1.24	0.07	3.24		-2.34	0.08
		18/02/2010	2	4.47	0.06	4.11	0.06	0.36	0.08	0.54	0.11
		18/02/2010	3	4.44		3.86	0.06	0.58		0.32	0.08
	0	18/02/2010	4	4.60	0.04	3.75	0.08	0.86	0.09	0.04	0.12
		19/02/2010	Influent	4.93	0.06	3.84	0.01	1.10	0.06		
		19/02/2010	1	4.38		3.71	0.05	0.67	0.05	0.43	0.08
		19/02/2010	2	4.42	0.05	3.85	0.04	0.57	0.07	0.53	0.09
		19/02/2010	3	4.57	0.03	3.83	0.08	0.74	0.09	0.35	0.11
2b	24	19/02/2010	4	4.65	0.03	3.76	0.03	0.90	0.04	0.20	0.08
		23/02/2010	Influent	4.70	0.12	3.98	0.06	0.72	0.14		
		23/02/2010	1	4.33		3.92		0.42	0.00	0.30	0.14
		23/02/2010	2	4.43	0.07	3.95	0.12	0.48	0.14	0.24	0.19
		23/02/2010	3	4.39	0.16	3.95	0.15	0.44	0.22	0.28	0.26
	0	23/02/2010	4	4.44	0.11	4.05	0.20	0.39	0.23	0.34	0.27
4a	24	24/02/2010	Influent	5.09		3.83	0.12	1.27	0.12		

		24/02/2010	1	4.59	0.04	3.84	0.09	0.75	0.10	0.52	0.15
		24/02/2010	2	4.59	0.13	3.67	0.12	0.91	0.18	0.36	0.21
		24/02/2010	3	4.79	0.14	4.48	0.16	0.31	0.22	0.95	0.25
		24/02/2010	4	4.26	0.22	4.00	0.26	0.26	0.34	1.01	0.36
		25/02/2010	Influent	5.50	0.05	4.07	0.14	1.43	0.15		
		25/02/2010	1	4.61		3.70	0.10	0.91	0.10	0.52	0.18
		25/02/2010	2	4.44	0.10	3.75	0.11	0.69	0.14	0.74	0.20
		25/02/2010	3	4.60	0.11	4.01	0.18	0.60	0.21	0.83	0.26
	0	25/02/2010	4	4.65	0.09	3.96	0.10	0.69	0.13	0.74	0.20
		26/02/2010	Influent	5.12	0.10	4.11	0.08	1.01	0.13		
		26/02/2010	1	4.61		3.73	0.09	0.88	0.09	0.13	0.16
		26/02/2010	2	4.68	0.13	3.91	0.10	0.76	0.16	0.25	0.21
		26/02/2010	3	4.69	0.04	3.94	0.06	0.75	0.07	0.26	0.15
4b	24	26/02/2010	4	4.63	0.16	3.76	0.10	0.86	0.18	0.15	0.22
		02/03/2010	Influent	4.33	0.04	3.45	0.04	0.88	0.06		
		02/03/2010	1	3.62		3.39	0.01	0.23	0.01	0.64	0.06
		02/03/2010	2	3.57	0.03	3.61	0.05	-0.04	0.06	0.92	0.08
		02/03/2010	3	3.59	0.02	3.45	0.04	0.14	0.05	0.74	0.07
	0	02/03/2010	4	3.71	0.05	3.67	0.03	0.04	0.06	0.84	0.08
		03/03/2010	Influent	4.79	0.05	3.32	0.04	1.47	0.06		
		03/03/2010	1	4.26		3.04	0.02	1.21	0.02	0.26	0.07
		03/03/2010	2	4.25	0.13	3.18	0.07	1.07	0.15	0.40	0.16
		03/03/2010	3	4.36	0.08	3.50	0.08	0.86	0.12	0.61	0.13
3a	24	03/03/2010	4	4.34	0.04	3.40	0.06	0.94	0.07	0.53	0.10
3b	0	04/03/2010	Influent	4.52	0.07	3.35	0.14	1.17	0.16		

		04/03/2010	1	4.12	0.11	3.26	0.09	0.86	0.14	0.31	0.21
		04/03/2010	2	4.02	0.06	3.45	0.07	0.57	0.10	0.60	0.18
		04/03/2010	3	4.21	0.05	3.48	0.11	0.74	0.12	0.43	0.20
		04/03/2010	4	4.13	0.16	3.25	0.12	0.88	0.20	0.29	0.26
		05/03/2010	Influent	4.41	0.16	3.24	0.15	1.17	0.22		
		05/03/2010	1	4.01	0.10	3.12	0.08	0.89	0.13	0.28	0.26
		05/03/2010	2	4.02	0.08	3.49	0.09	0.53	0.12	0.64	0.25
		05/03/2010	3	4.13	0.06	3.87	0.04	0.26	0.08	0.91	0.23
	24	05/03/2010	4	4.16	0.15	3.20	0.11	0.95	0.19	0.21	0.29
		09/03/2010	Influent	4.59	0.06	3.31	0.17	1.28	0.17		
		09/03/2010	1	4.27		3.63	0.02	0.65	0.02	0.63	0.17
		09/03/2010	2	4.15	0.06	3.71	0.05	0.44	0.08	0.84	0.19
		09/03/2010	3	4.38	0.03	3.38	0.04	1.01	0.05	0.27	0.18
	0	09/03/2010	4	4.37	0.01	3.47	0.04	0.90	0.04	0.38	0.18
		10/03/2010	Influent	4.61	0.05	3.50	0.05	1.12	0.07		
		10/03/2010	1	4.42		3.58	0.03	0.85	0.03	0.27	0.08
		10/03/2010	2	4.20	0.02	3.57	0.06	0.63	0.07	0.49	0.10
		10/03/2010	3	4.48	0.06	3.56	0.04	0.92	0.07	0.20	0.10
5a	24	10/03/2010	4	4.41	0.03	3.54	0.01	0.87	0.03	0.25	0.08
		11/03/2010	Influent	5.16	0.10	3.43		1.72	0.10		
		11/03/2010	1	4.27	0.02	3.31	0.03	0.96	0.03	0.76	0.11
		11/03/2010	2	4.21	0.01	3.55	0.02	0.65	0.03	1.07	0.10
		11/03/2010	3	4.54	0.13	3.47	0.02	1.07	0.13	0.65	0.16
	0	11/03/2010	4	4.44	0.02	3.36	0.04	1.08	0.04	0.64	0.11
5b	24	12/03/2010	Influent	4.24	0.05	3.02		1.22	0.05		

		12/03/2010	1	3.88		3.17	0.03	0.71	0.03	0.50	0.06
		12/03/2010	2	3.95	0.02	3.13	0.05	0.82	0.05	0.40	0.07
		12/03/2010	3	4.10	0.03	3.24	0.13	0.87	0.13	0.35	0.14
		12/03/2010	4	4.11	0.02	3.28	0.01	0.84	0.02	0.38	0.06
		16/03/2010	Influent	4.00	0.10	3.24	0.07	0.76	0.12		
		16/03/2010	1	3.57		3.49	0.05	0.08	0.05	0.68	0.13
		16/03/2010	2	3.65	0.04	3.04	0.06	0.60	0.07	0.16	0.14
		16/03/2010	3	3.86	0.06	3.21	0.02	0.65	0.07	0.11	0.14
	0	16/03/2010	4	3.84	0.21	3.08	0.08	0.75	0.22	0.01	0.25
		17/03/2010	Influent	4.04	0.04	3.11	0.03	0.93	0.06		
		17/03/2010	1	3.99	0.07	3.33	0.04	0.66	0.08	0.27	0.10
		17/03/2010	2	3.88	0.13	2.80	0.01	1.08	0.13	-0.15	0.14
		17/03/2010	3	4.34	0.09	2.85	0.07	1.49	0.11	-0.56	0.12
6a	24	17/03/2010	4	4.24	0.09	2.89	0.06	1.35	0.11	-0.42	0.12
		18/03/2010	Influent	4.63	0.03	2.78	0.08	1.85	0.08		
		18/03/2010	1	3.74		3.13	0.02	0.61	0.02	1.24	0.09
		18/03/2010	2	3.85	0.02	3.62	0.00	0.22	0.02	1.63	0.09
		18/03/2010	3	4.28	0.10	2.99	0.03	1.28	0.10	0.57	0.13
	0	18/03/2010	4	4.24	0.06	2.88	0.06	1.36	0.08	0.49	0.12
		19/03/2010	Influent	4.76	0.12	2.94	0.05	1.82	0.13		
		19/03/2010	1	4.29		3.81	0.00	0.48	0.00	1.35	0.13
		19/03/2010	2	4.23	0.03	3.08	0.02	1.15	0.03	0.67	0.14
		19/03/2010	3	4.53	0.04	3.23	0.03	1.30	0.05	0.52	0.14
6b	24	19/03/2010	4	4.41	0.09	3.19	0.06	1.22	0.11	0.60	0.17
8a	0	23/03/2010	Influent	5.02	0.21	3.65	0.03	1.37	0.21		

		23/03/2010	1	4.19	0.10	3.37	0.06	0.82	0.11	0.54	0.24
		23/03/2010	2	4.37	0.07	4.01	0.17	0.36	0.19	1.01	0.28
		23/03/2010	3	4.90	0.26	3.37	0.06	1.52	0.27	-0.16	0.34
		23/03/2010	4	4.63	0.10	3.20	0.03	1.43	0.11	-0.06	0.24
		24/03/2010	Influent	4.55	0.12	3.22	0.08	1.33	0.15		
		24/03/2010	1	3.87	0.12	3.23	0.04	0.64	0.13	0.69	0.19
		24/03/2010	2	3.86	0.10	3.34	0.13	0.52	0.16	0.81	0.22
		24/03/2010	3	4.11	0.08	3.33	0.04	0.78	0.09	0.55	0.17
	24	24/03/2010	4	4.15	0.10	3.13	0.07	1.02	0.12	0.31	0.19
		30/03/2010	Influent	3.91		3.16		0.75	0.00		
		30/03/2010	1	3.37		2.82		0.55	0.00	0.20	0.00
		30/03/2010	2	3.56		3.45		0.11	0.00	0.64	0.00
		30/03/2010	3	3.71		2.92		0.79	0.00	-0.04	0.00
	0	30/03/2010	4	3.89		3.16		0.73	0.00	0.01	0.00
		31/03/2010	Influent	4.63		3.80		0.82	0.00		
		31/03/2010	1	4.12		2.92		1.20	0.00	-0.38	0.00
		31/03/2010	2	3.86		2.89		0.98	0.00	-0.15	0.00
		31/03/2010	3	4.16		3.55		0.61	0.00	0.22	0.00
8b	24	31/03/2010	4	4.05		3.59		0.45	0.00	0.37	0.00
		04/04/2010	Influent	5.33		3.91		1.43	0.00		
		04/04/2010	1	3.45		2.44		1.02	0.00	0.41	0.00
		04/04/2010	2	3.94		2.89		1.05	0.00	0.37	0.00
		04/04/2010	3	3.90		3.00		0.90	0.00	0.53	0.00
	0	04/04/2010	4	3.53		2.83		0.69	0.00	0.73	0.00
7a	24	05/04/2010	Influent	5.22		4.37		0.85	0.00		

		05/04/2010	1	xxx		2.80			0.00		
		05/04/2010	2	3.44		2.89		0.55	0.00	0.29	0.00
		05/04/2010	3	3.79		3.64		0.15	0.00	0.69	0.00
		05/04/2010	4	3.73		2.95		0.78	0.00	0.06	0.00
		06/04/2010	Influent	4.05		3.14		0.91	0.00		
		06/04/2010	1	3.75		2.95		0.80	0.00	0.11	0.00
		06/04/2010	2	3.78		3.08		0.69	0.00	0.22	0.00
		06/04/2010	3	3.84		3.28		0.56	0.00	0.35	0.00
	0	06/04/2010	4	4.25		2.99		1.26	0.00	-0.35	0.00
		07/04/2010	Influent	3.98		3.52		0.46	0.00		
		07/04/2010	1	3.63		2.91		0.71	0.00	-0.26	0.00
		07/04/2010	2	4.07		2.91		1.16	0.00	-0.70	0.00
		07/04/2010	3	xxx		2.89			0.00		0.00
7b	24	07/04/2010	4	3.77		2.92		0.86	0.00	-0.40	0.00
		10/04/2010	Influent	3.75		3.16		0.60	0.00		
		10/04/2010	1	4.18		2.18		2.00	0.00	-1.40	0.00
		10/04/2010	2	3.59		2.59		1.00	0.00	-0.40	0.00
		10/04/2010	3	4.11		2.42		1.68	0.00	-1.09	0.00
	0	10/04/2010	4	3.84		2.74		1.10	0.00	-0.51	0.00
		11/04/2010	Influent	4.22		3.25		0.97	0.00		
		11/04/2010	1	3.51		2.76		0.76	0.00	0.22	0.00
		11/04/2010	2	3.76		2.76		1.00	0.00	-0.03	0.00
		11/04/2010	3	3.96		2.93		1.03	0.00	-0.05	0.00
1a	24	11/04/2010	4	4.06		2.60		1.47	0.00	-0.49	0.00
1b	0	12/04/2010	Influent	4.36		2.92		1.44			

		12/04/2010	1	3.84		2.91		0.92	0.00	0.52	
		12/04/2010	2	4.06		3.00		1.05	0.00	0.39	
		12/04/2010	3	4.29		3.23		1.06	0.00	0.38	
		12/04/2010	4	4.38		3.55		0.84	0.00	0.61	
		13/04/2010	Influent	4.52		3.53		0.99	0.00		
		13/04/2010	1	4.35		3.31		1.04	0.00	-0.05	0.00
		13/04/2010	2	4.20		3.06		1.14	0.00	-0.15	0.00
		13/04/2010	3	3.70		3.62		0.08	0.00	0.91	0.00
	24	13/04/2010	4	5.15		3.46		1.70	0.00	-0.71	0.00
		14/04/2010	Influent	7.12		2.87		4.24			
		14/04/2010	1	3.79		3.12		0.67	0.00	3.57	
		14/04/2010	2	4.06		3.53		0.52	0.00	3.72	
		14/04/2010	3	5.05		3.53		1.52	0.00	2.73	
	0	14/04/2010	4	4.28		3.26		1.03	0.00	3.22	
		15/04/2010	Influent	3.53		3.24		0.29	0.00		
		15/04/2010	1	3.48		xxx			0.00		0.00
		15/04/2010	2	3.80		3.04		0.76	0.00	-0.47	0.00
		15/04/2010	3	3.82		3.14		0.68	0.00	-0.39	0.00
4a	24	15/04/2010	4	3.82		3.14		0.68	0.00	-0.39	0.00

Appendix C – Full-scale Raw Data

Phospholipid Biomass

Pre Backwash

	A610			nmol P			Dry Weight			nmol P/g			Avg	SDEV
Filter	Rep 1	Rep 2	Rep 3	Rep 1	Rep 2	Rep 3	Rep 1	Rep 2	Rep 3	Rep 1	Rep 2	Rep 3		
1	1.54	1.95	1.82	35.74	45.33	42.40	0.04	0.09	0.10	569.86	357.79	316.81	**414.82**	135.82
2	1.06	0.52	0.96	24.59	12.20	22.23	0.04	0.09	0.06	473.44	95.73	270.00	**279.73**	189.04
3	1.96	0.74	1.11	45.62	17.25	25.70	0.21	0.04	0.05	155.63	299.01	369.39	**274.68**	108.94
4	0.20	0.11	0.17	4.70	2.65	3.91	0.03	0.02	0.07	94.70	78.80	36.61	**70.04**	30.02

cm3 conversion

0.06	0.13	0.13
0.05	0.13	0.08
0.29	0.06	0.07
0.05	0.03	0.11

Post Backwash Light

	A610			nmol P			Dry Weight			nmol P/cm3			Avg	SDEV
Filter	Rep 1	Rep 2	Rep 3	Rep 1	Rep 2	Rep 3	Rep 1	Rep 2	Rep 3	Rep 1	Rep 2	Rep 3		
1	1.24	1.10	1.55	28.79	25.69	36.06	0.04	0.05	0.05	491.90	387.96	498.24	**459.37**	61.92
2	0.60	0.74	0.52	13.89	17.11	12.05	0.03	0.06	0.03	390.63	209.98	314.09	**304.90**	90.67

3	1.58	0.72	0.94	36.83	16.76	21.86	0.12	0.06	0.08	219.57	193.68	207.64	**206.96**	12.95
4	0.12	0.13	0.25	2.87	2.98	5.72	0.07	0.15	0.07	25.56	12.39	48.10	**28.69**	18.06

Cubic CM		
0.06	0.07	0.07
0.04	0.08	0.04
0.17	0.09	0.11
0.11	0.24	0.12

Biological Respiration Potential

Pre Backwash

	DO initial	DO Final			DO Consumed			Dry Weight			DO Consumed/cm3				
Filter		Rep 1	Rep 2	Rep 3	Rep 1	Rep 2	Rep 3	Rep 1	Rep 2	Rep 3	Rep 1	Rep 2	Rep 3	**Average**	STDEV
1.00	8.28	4.99	5.65	6.60	3.29	2.63	1.68	6.49	8.73	12.61	0.36	0.22	0.10	**0.22**	0.13
2.00	7.79	5.56	5.05	5.78	2.23	2.74	2.01	0.50	1.19	0.85	3.17	1.64	1.69	**2.17**	0.87
3.00	8.06	5.63	4.53	5.35	2.43	3.53	2.71	1.00	1.11	1.40	1.73	2.27	1.38	**1.79**	0.45
4.00	8.76	6.34	7.83	7.00	2.42	0.93	1.76	2.19	1.48	1.56	0.69	0.39	0.70	**0.60**	0.18

cm3 conversion

9.08	12.23	17.66
0.70	1.67	1.19
1.40	1.55	1.97
3.50	2.37	2.50

Post Backwash

	DO initial	DO Final			DO Consumed			Dry Weight			DO Consumed/cm3				
Filter		Rep 1	Rep 2	Rep 3	Rep 1	Rep 2	Rep 3	Rep 1	Rep 2	Rep 3	Rep 1	Rep 2	Rep 3	**Average**	STDEV
1.00	8.28	4.81	5.05	6.01	3.47	3.23	2.27	2.45	12.61	8.39	1.01	0.18	0.19	**0.46**	0.47
2.00	7.79	6.37	6.26	6.29	1.42	1.53	1.50	0.63	1.17	0.91	1.60	0.94	1.18	**1.24**	0.34
3.00	8.06	6.34	6.59	6.38	1.72	1.47	1.68	1.52	1.45	1.88	0.81	0.72	0.64	**0.72**	0.08
4.00	8.76	8.25	9.54	8.63	0.51	-0.78	0.13	1.69	1.98	1.72	0.19	-0.25	0.05	**0.00**	0.22

cm3 conversion

3.44	17.66	11.75
0.89	1.64	1.28

2.13	2.03	2.63
2.70	3.17	2.74

BDOC

Sampling Period 1

Date	Filter	DOCi	DOCe	BDOC	BDOC Removal	DOC Removal
09/02/2010	Influent	4.100	3.600	0.500		
09/02/2010	1	3.300	3.500	-0.200	-0.200	0.800
09/02/2010	2	3.000	3.000	0.000	0.000	1.100
09/02/2010	3	3.300	3.600	-0.300	-0.300	0.800
09/02/2010	4	3.400	3.100	0.300	0.300	0.700

Date	Filter	DOCi	DOCe	BDOC	BDOC Removal	DOC Removal
28/02/2009	Influent	3.900	3.000	0.900		
28/02/2009	1	3.600	3.400	0.200	0.200	0.300
28/02/2009	2	3.400	3.200	0.200	0.200	0.500
28/02/2009	3	3.300	3.100	0.200	0.200	0.600
28/02/2009	4	3.600	3.500	0.100	0.100	0.300

Date	Filter	DOCi	DOCe	BDOC	BDOC Removal	DOC Removal
16/03/2009	Influent	3.300	2.900	0.400		
16/03/2009	1	3.000	3.000	0.000	0.000	0.300
16/03/2009	2	4.700	2.700	2.000	2.000	-1.400
16/03/2009	3	2.800	3.000	-0.200	-0.200	0.500

16/03/2009	4	3.100	2.500	0.600	0.600	0.200

Date	Filter	DOCi	DOCe	BDOC	BDOC Removal	DOC Removal
09/02/2010	Influent	3.700	3.400	0.300		
09/02/2010	1	x	x			
09/02/2010	2	3.200	3.100	0.100	0.100	0.500
09/02/2010	3	3.200	5.600	-2.400	-2.400	0.500
09/02/2010	4	3.300	3.100	0.200	0.200	0.400

Date	Filter	DOCi	DOCe	BDOC	BDOC Removal	DOC Removal
09/02/2010	Influent	5.420	4.165	1.255		
09/02/2010	1	5.096	5.055	0.041	0.041	0.324
09/02/2010	2	5.088	4.995	0.093	0.093	0.332
09/02/2010	3	4.904	4.964	-0.060	-0.060	0.516
09/02/2010	4	4.865	5.037	-0.172	-0.172	0.555

Date	Filter	DOCi	DOCe	BDOC	BDOC Removal	DOC Removal
04/11/2008	Influent	x	x			
04/11/2008	1	4.796	4.448	0.348		
04/11/2008	2	4.934	4.519	0.415		
04/11/2008	3	4.593	4.660	-0.067		
04/11/2008	4	5.070	4.874	0.196		

Date	Filter	DOCi	DOCe	BDOC	BDOC Removal	DOC Removal
16/10/2008	Influent	x	x			
16/10/2008	1	4.369	4.332	0.037		

16/10/2008	2	3.517	4.380	-0.863		
16/10/2008	3	4.357	4.392	-0.035		
16/10/2008	4	4.384	5.528	-1.144		

Date	Filter	DOCi	DOCe	BDOC	BDOC Removal	DOC Removal
18/12/2008	Influent	5.584	3.931	1.653		
18/12/2008	1	4.472	4.534	-0.062	-0.062	1.112
18/12/2008	2	4.454	5.208	-0.754	-0.754	1.130
18/12/2008	3	4.561	4.417	0.144	0.144	1.023
18/12/2008	4	4.834	4.398	0.436	0.436	0.750

Date	Filter	DOCi	DOCe	BDOC	BDOC Removal	DOC Removal
01/05/2009	Influent	4.600	2.800	1.800		
01/05/2009	1	3.300	3.100	0.200	0.200	1.300
01/05/2009	2	3.300	3.000	0.300	0.300	1.300
01/05/2009	3	3.300	3.400	-0.100	-0.100	1.300
01/05/2009	4	3.600	3.200	0.400	0.400	1.000

Date	Filter	DOCi	DOCe	BDOC	BDOC Removal	DOC Removal
07/05/2009	Influent	4.300	3.000	1.300		
07/05/2009	1	3.300	3.100	0.200	0.200	1.000
07/05/2009	2	3.300	4.200	-0.900	-0.900	1.000
07/05/2009	3	3.300	3.200	0.100	0.100	1.000
07/05/2009	4	3.500	3.200	0.300	0.300	0.800

Sampling Period 2

F1 Pre Backwash						
Sample ID	DOC	S.D.	Column	BDOC	S.D.	DOC Removal(%)
F1i	4.412	0.075004	F1	-0.132	0.089285725	-2.991840435
F1e	4.544	0.048439	I1	1.543	0.305911729	28.29635063
I1i	5.453	0.155411	I2	1.691	0.182871535	31.01045296
Ie1a	3.91	0.263495				
Ie1b	3.762	0.096382				
			Effluent DOC	S.D.	DOC removal(%)	S.D.
			4.412	0.075004	19.09040895	0.173832
			Effluent BDOC	S.D.	Effluent BDOC Removal	S.D.
			-0.132	0.089286	107.8060319	0.007108605

F2 Pre Backwash						
Sample ID	DOC	SD (%)	Column	BDOC	S.D.	DOC Removal(%)
F1i	4.663	0.069805	F1	0.804	0.088935268	17.24211881
F1e	3.859	0.055107	I1	2.171	0.157326782	34.29158111
I1i	6.331	0.113895	I2	2.423	0.15113142	38.27199495
Ie1a	4.16	0.108534				
Ie1b	3.908	0.099341				
			Effluent DOC	S.D.	DOC removal(%)	S.D.
			4.663	0.069805	26.34654873	0.088830865
			Effluent BDOC	S.D.	Effluent BDOC Removal	S.D.
			0.804	0.088935	41.15368731	0.005189771

F3 Pre Backwash						
Sample ID	DOC	SD (%)	Column	BDOC	S.D.	DOC Removal(%)
F1i	4.256	0.161941	F1	0.21	0.246401191	4.934210526
F1e	4.046	0.185711	I3	1.417	0.180832781	28.90656875
I2	4.902	0.133629				
Ie3	3.485	0.121836				
			Effluent DOC	S.D.	DOC removal(%)	S.D.
			4.256	0.161941	13.17829457	0.355183733
			Effluent BDOC	S.D.	Effluent BDOC Removal	S.D.
			0.21	0.246401	85.17995766	0.013856606

F4 Pre Backwash						
Sample ID	DOC	SD (%)	Column	BDOC	S.D.	DOC Removal(%)
F1i	4.69	0.089954	F1	0.861	0.107432662	18.35820896
F1e	3.829	0.058737	I3	0.89	0.185782567	19.22661482
I2	4.629	0.134843				0
Ie3	3.739	0.127799				
			Effluent DOC	S.D.	DOC removal(%)	S.D.
			4.69	0.089954	-1.317779218	-2.646676017
			Effluent BDOC	S.D.	Effluent BDOC Removal	S.D.
			0.861	0.107433	3.258426966	0.074635482

F1 Post Backwash						
Sample ID	DOC	SD (%)	Column	BDOC	S.D.	DOC Removal(%)
F1i	4.345	0.13652	F1	0.004	0.17063239	0.092059839
F1e	4.341	0.102361	I1	1.486	0.578000718	30.41965217
I1i	4.885	0.124323	I2	1.129	0.193656373	23.11566023
Ie1a	3.399	0.564472				
Ie1b	3.756	0.148362				
			Effluent DOC	S.D.	DOC removal(%)	S.D.
			4.345	0.136519	11.05424477	0.365779124
			Effluent BDOC	S.D.	Effluent BDOC Removal	S.D.
			0.004	0.170632	11.84766823	3.602806883

F2 Post Backwash						
Sample ID	DOC	SD (%)	Column	BDOC	S.D.	DOC Removal(%)
F1i	6.201	0.078133	F1	1.983	0.092548518	31.97871311
F1e	4.218	0.049604	I1	1.295	0.173993367	24.71845772
I1i	5.239	0.127203	I2	1.614	0.157863008	30.80740599
Ie1a	3.944	0.118714				
Ie1b	3.625	0.093489				
			Effluent DOC	S.D.	DOC removal(%)	S.D.
			6.201	0.078133	-18.36228288	-0.148972097
			Effluent BDOC	S.D.	Effluent BDOC Removal	S.D.
			1.983	0.092548	41.37415747	0.007864645

F3 Post Backwash						
Sample ID	DOC	SD (%)	Column	BDOC	S.D.	DOC Removal(%)
F1i	4.275	0.18639	F1	0.032	0.250858794	0.748538012
F1e	4.243	0.167896	I3	0.59	0.211519482	13.58208
I2	4.342	0.171031				
Ie3	3.752	0.124454				
			Effluent DOC	S.D.	DOC removal(%)	S.D.
			4.275	0.18639	1.540367711	3.807885482
			Effluent BDOC	S.D.	Effluent BDOC Removal	S.D.
			0.032	0.250858794	94.57627119	0.082975682

F4 Post Backwash						
Sample ID	DOC	SD (%)	Column	BDOC	S.D.	DOC Removal(%)
F1i	4.645	0.134148	F1	0.48	0.169765291	10.33369214
F1e	4.165	0.104042	I3	0.956	0.160611256	20.41862452
I2	4.682	0.125618				
Ie3	3.726	0.10008				
			Effluent DOC	S.D.	DOC removal(%)	S.D.
			4.645	0.134476	0.790260572	0.049888175
			Effluent BDOC	S.D.	Effluent BDOC Removal	S.D.
			0.48	0.169765291	49.79079498	0.007863946

Total Trihalomethane Formation Potential and Chlorine Demand

		Influent					Effluent					% removals
Date Sampled	Filter	Rep 1	Rep 2	Rep 3	Average	S.D.	Rep 1	Rep 2	Rep 3	Average	S.D.	
15/03/2010	3	12.82	12.38	12.98	12.72667	0.310698	11.05	11.04	10.62	10.90333	0.245425	14.32687
16/03/2010	3	12.82	12.38	12.98	12.72667	0.310698	11.82	11.49	11.62	11.64333	0.166233	8.51231
16/03/2010	4	12.82	12.38	12.98	12.72667	0.310698	11.99	10.24	12.63	11.62	1.237215	8.695652
17/03/2010	4	12.18	11.39	11.32	11.63	0.477598	10.32	11.16	10.94	10.80667	0.435584	7.079392
17/03/2010	2	11.38	11.02	11.07	11.15667	0.195021	8.68	8.62		8.65	0.042426	22.46788
18/03/2010	2		10.9	11.37	11.135	0.33234	8.38	8.56	8.11	8.35	0.226495	25.01123
22/03/2010	1	10.43	11.15	10.05	10.54333	0.558689	8.18	8.08	8.46	8.24	0.196977	21.84635
23/03/2010	1	11.65	10.75	10.85	11.08333	0.493288	9.02	9.02	8.95	8.996667	0.040415	18.82707
06/04/2010	4	11.28	10.4		10.84	0.622254	9.63	9.45	9.92	9.666667	0.237136	10.82411
07/04/2010	4	10.85	10.78	10.67	10.76667	0.090738	9.72	9.62	9.62	9.653333	0.057735	10.34056
07/04/2010	3	10.85	10.78	10.67	10.76667	0.090738	8.53	9.63		9.08	0.777817	15.66563
08/04/2010	3	11.08	9.87	10.47	10.47333	0.605007	10.08	10.32	7.32	9.24	1.667093	11.77594
14/04/2010	2	11.38	10.38	11.48	11.08	0.608276	7.96	8.06	7.78	7.933333	0.141892	28.39952
13/04/2010	2	10.75	10.44	10.07	10.42	0.340441	7.76	8.25	7.95	7.986667	0.247049	23.35253