BOM removal by biofiltration- Developing a quantitative basis for comparison

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

Biological filtration (Biofiltration) processes have been used first in Europe and then in North America for decades, however currently there is not a good overall parameter to guide biofiltration design and operation except adopting parameters from traditional particle-removal filtration process. On the basis of the biofilm model developed by Rittmann and McCarty (1980a) and the pseudo-analytical solution for the model, Zhang and Huck (1996a) obtained an analytical solution for PF (plug flow) reactors (which can be used for biofilters approximately) after demonstrating that axial dispersion could be reasonably ignored and developed a new parameter, X*, which incorporates considerations of physical contact time, filter media particle size, kinetics, etc. A small-scale application on peers' engineering/research data by Huck (1999) demonstrated it was a better indicator than other parameters for biofiltration performance. By collecting, screening and investigating literature on AOC, BDOC and odorous compounds removal by biofiltration process, this thesis applied the X* concept to the collected investigations to assess process performances among different target parameters, different filters and different investigations. To the author's knowledge, this is the first such attempted comprehensive comparison of literature studies, interpreted in terms of a common parameter (X*). The wide ranges of particle sizes, EBCTs, temperatures and high diversity of pre-treatment and operation conditions for the collected cases were considered to be able to well represent biofiltration practices for studied removal targets.

No significant relationship between EBCTs and removal percentages were found, indicating that EBCT alone is not able to guide biofiltration design and operation. Based on kinetics parameter comparison, BDOC removal-X* relationship was established. A new parameter, $\theta\alpha$, was developed in this thesis to refer to estimated X* values only considering EBCT and particle size. $\theta\alpha$ parameter values were estimated by comparison of ratios of $\theta\alpha$ products (($\theta\alpha$)') based on the properly chosen calculation bases. Distribution of the $\theta\alpha$ values for temperature-favored (i.e. temperature ≥15°C) AOC and BDOC removal biofiltration

processes matched the established removal-X* relationship reasonably. Given the exploratory nature of this research and the complexity of attempting quantitations, fits were assessed based on visual comparison. With the assistance of supporting information and by adopting available temperature activity coefficients, temperature-adjustment coefficients for $\theta\alpha$ values were determined for the different temperature ranges. Temperature-adjusted AOC and BDOC removal-$\theta\alpha$ relationships were developed and temperature-adjusted $\theta\alpha$ parameter values for AOC and BDOC removal were also estimated. Comparisons were conducted, showing fair matches based on visual examinations, for most of the temperature ranges. No relationships were found between ozone dosages and AOC/BDOC removal percentages and the statistical analysis indicated there was significant difference of removal efficiencies between ozonated and non-ozonated influents for biofilters, suggesting ozonation may not only increase the amount of BOM for following biofilter and increase the biodegradability of bulk water; it may also increase the biodegradability of AOC and BDOC themselves.

It may not be realistic to obtain the estimated $\theta\alpha$ values for MIB and geosmin removal by biofiltration. However, plotting $\theta\alpha$ product vs. removal percentage for the collected MIB and geosmin removal cases shows more positive co-relationships than EBCT-removal percentage relationships visually. A utilization factor η was proposed to guide biofilter design and operation and to assess “over-design” and “under-operated”. Biofilter over-design or under-operated is common for the collected cases.

In general, examining X* (or $\theta\alpha$, a parameter incorporating the physical components of X*) provided useful information in terms of evaluation and prediction of biodegradable organic compounds removal by biofiltration, which confirms that X* is a better parameter for biofiltration design and operation than other parameters, such as EBCT.

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Dedication

To my wife, Aijuan

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List of Abbreviations

AOC	Assimilable Organic Carbon
BAC	Biologically Active Carbon
BDOC	Biodegradable Dissolved Organic Carbon
BF	Biofiltration Factor
BOM	Biodegradable Organic Matter
DBP	Disinfection By-Product
DOC	Dissolved Organic Carbon
EBCT	Empty Bed Contact Time
EC	Expanded Clay
GAC	Granular Activated Carbon
HAA	Haloacetic Acid
HLR	Hydraulic Loading Rate
MAP	Microbial Available Phosphorus
MIB	2-methylisoborneol
NDMA	N-nitrosodimethylamine
NOM	Natural Organic Matter
NPOC	Non-Purgeable Organic Carbon
NPOXFP	Non-Purgeable Organic Halide Formation Potential
NVOC	Non-Volatile Organic Carbon
OBP	Ozonation By-Product
PPCP	Pharmaceuticals and Personal Care Product
THM	Trihalomethanes
THMFP	Trihalomethanes Formation Potential
TOC	Total Organic Carbon
TOXFP	Total Organic Halide Formation Potential
UV	Ultraviolet

Chapter 1

Introduction

1.1 Background

Biological filtration (Biofiltration) processes have been used in Europe and in North America for decades (Graham, 1999), however currently there is not a good overall parameter to guide biofiltration design and operation except adopting parameters from traditional particle- removal filtration process.

On the basis of the biofilm model developed by Rittmann and McCarty (1980a) and the pseudo-analytical solution for the model, Zhang and Huck (1996a) obtained an analytical solution for PF (plug flow) reactors (which could be applicable for the case of biofilters) after demonstrating that axial dispersion could be reasonably ignored. They further developed a new parameter, X*, which incorporates considerations of physical contact time, filter media particle size, kinetics and etc. A small-scale application on real data by Huck (1999) demonstrated it was a better indicator than other parameters for biofiltration performance.

1.2 Objective

The objective of this research is described as follows:

1. To collect literature of AOC (Assimilable Organic Carbon) and BDOC (Biodegradable Dissolved Organic Carbon) removal by biofiltration and extract necessary information from the literature for the X* analysis;
2. To apply the X* concept to the collected AOC and BDOC removal cases (Appendix A and B) and estimate the X* values from real data and compare them with the removal-X* relationship derived from modeling work to test the applicability of X* concept;
3. To examine the effect of various affecting factors, such as temperature and pre-ozonation, on the X* values and biofiltration process treatment efficiencies;

4. To explore the appropriateness of extending the X* concept to other biofiltration applications, such as secondary utilization.

1.3 Structure of Thesis

Chapter 2 presents a literature review of the relevant background information to this research.

Chapter 3 and Chapter 4 present the X* analysis for AOC and BDOC removal by biofiltration respectively and study the factors affecting X* values.

Chapter 5 presents the X* analysis for secondary utilization in biofilters (MIB and geosmin as examples) and discusses the appropriateness of extending the X* concept to other biofiltration applications.

The final chapter (Chapter 6) addresses the overall conclusions and recommendations for future research and implications for practice.

Chapter 2

Literature Review

The objective of drinking water treatment is to provide safe and clean potable water. In a typical conventional plant treating surface water, the treatment train comprises chemical coagulation, flocculation, sedimentation, filtration (rapid sand filtration) and disinfection. Slow sand filtration and bank filtration are other biological processes, however they are typically preceded by no or limited pretreatment. Rapid biological filtration (or biofiltration) has been applied in drinking water treatment practice, first in Europe and since the 1980s in North American water works (Prévost et al., 2005).

In this review, introduction of biofiltration process, factors affecting biofilter performance will be reviewed firstly. The review will then focus on drinking water biostability, quantifying parameters and applications of biofiltration process to achieve biostability. Other biofiltration applications will also be summarized. Biofilm modeling and a more recently developed parameter X* will be reviewed in the following sections. The research goal of this thesis will be discussed in the last section of this review.

2.1 Introduction and Process Description

Two types of microbial aggregates have been seen applications in environmental biotechnology: suspended flocs and attached biofilm. All of the biological processes for drinking water treatment are biofilm processes, in which a biologically active matrix of microorganisms and extracellular polymeric substances accumulate on the outer surface of a solid to form a layer of microorganisms, their waste products and sorbed/attached materials (Rittmann and McCarty, 2001). Microbial growth is achieved by passing substrate-containing liquid flow past the biofilm, which is immobilized on the solid surface. Figure 2-1 is a sketch of biofilm and biofilter.

The biofilm on the solid surface undergoes a process of initial colonization, cell accumulation and biofilm formation. In this stage, the substrate in the liquid undergoes both adsorption and degradation, which will be discussed later in this chapter.

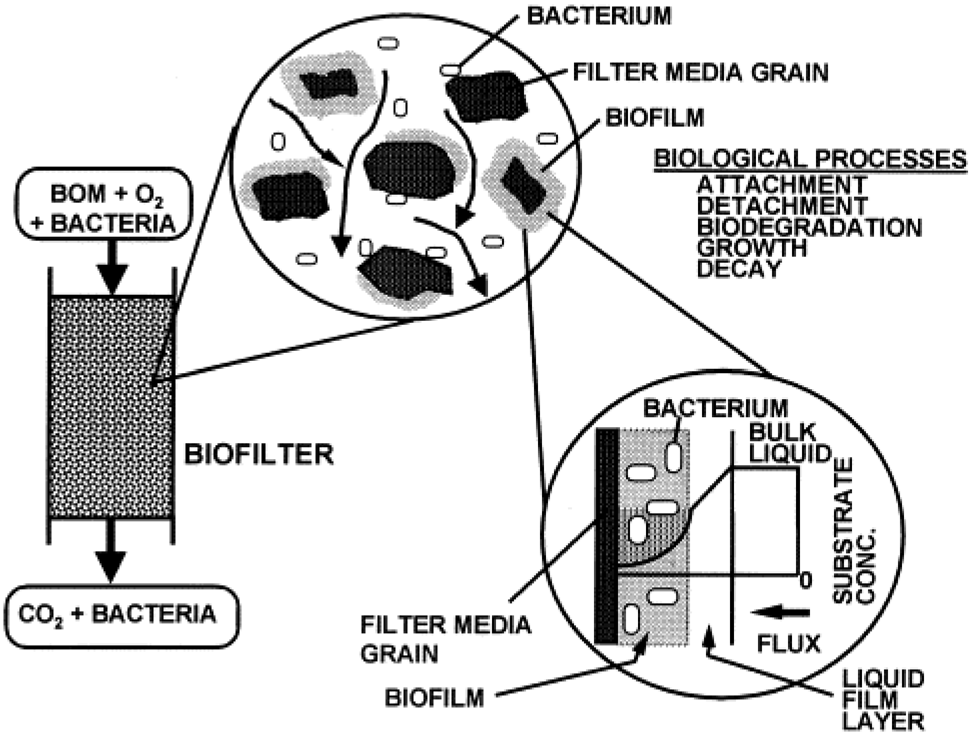

Figure 2-1 Biofilm in Biological filters (Hozalski et al., Water Research, 2001a)

The maintenance of a biofilm in the biofilter involves a number of sophisticated physical-chemical and biological processes, including attachment, detachment, microbial growth and endogenous decay. These processes occur dynamically, simultaneously and will reach a steady-state after a certain amount of time (Rittmann and McCarty, 2001).

Sand, anthracite and granular activated carbon (GAC) are three most applied filter media, although applications of bio-ceramic, zeolite, expanded clay (EC), coke and etc. were also reported in water treatment research and practice (Sang, et al., 2003; Persson et al., 2006). The physico-chemical properties of the filter media affect parameters such as biomass amount, and in turn affect biofiltration performance greatly (Huck, 1999), although performance is not necessarily linearly related to biomass (e.g. Huck and Sozański, 2008).

In some cases, when a filter is placed ahead of the whole drinking water treatment process train and receives raw water, it is called a “roughing filter”. Filter media in roughing filters normally consist of coarse materials.

2.2 Drinking Water Biostability

2.2.1 Introduction

A biodegradable organic matter (BOM) residual in finished water can provide a nutrient and energy source for growth of microorganisms in distribution system (Prévost et al., 2005). Regrowth occurs when large numbers of bacteria are observed in a distribution system, which contains only very limited bacteria as a result of disinfection processes. The levels of coliform bacteria are regulated in many jurisdictions, and ooccurrences of coliforms have been shown to be a function of temperature, disinfectant residual, and BOM levels (Volk and LeChevallier, 2000).

A disinfectant residual can suppress regrowth in distribution systems, however, it was found in some studies to have the ability to oxidize natural organic matter in treated water and produce more substrate for heterotrophic and nitrifying bacteria (Liu et al., 2002; Harrington et al., 2003). Srinivasan and Harrington (2007) thus pointed out that there was a trade-off between maintaining a high disinfectant residual and achieving a low substrate concentration for controlling microbial regrowth. It also should be noted that unlike North American practice, secondary disinfectant residuals are not required in some European countries (Haas, 1999).

The ability to limit regrowth in a treated drinking water is referred to as biological stability or biostability (Rittman and Snoeyink, 1984). It is a concept that addresses an overall tendency of the water to promote or suppress microbial proliferation, and can be viewed as an assessment of overall water quality in the distribution system, in terms of its microbial aspect. The greater biostability of a treated water, the less concern for microorganism regrowth.

Srinivasan and Harrington (2007) developed a standardized graphical algorithm for assessing biostability of a treated water and enabled water utilities to adjust operational parameters to achieve a more biostable water on the basis of two independent parameters: inactivation due to the presence of a disinfectant and growth due to the presence of a substrate.

2.2.2 Concepts, Measurement and Criteria of AOC and BDOC

Since organic carbon has been widely considered to be a limiting nutrient for bacterial growth in distribution systems (Prévost et al., 2005), biostability evaluation often relies on BOM levels in finished water, instead of using direct bacteria counting. There are several methods to quantify the potential of bacterial regrowth in water samples. AOC and BDOC are the two most used parameters to quantify BOM and thus characterize biostability, individually or in conjunction (Huck, 1990).

The AOC concept was developed by van der Kooij (e.g. van der Kooij et al., 1982). It is used to describe the fraction of labile dissolved organic carbon that is readily assimilated by microorganisms in water, resulting in growth. AOC consists of a broad range of low molecular weight organic matter such as sugars, organic acid and amino acids which only represents 0.1-9% of the total organic carbon (Hammes and Egli, 2005). Studies showed that heterotrophic density in the water distribution systems significantly correlated to the AOC levels (van der Kooij et al., 1982; LeChevallier et al., 1987).

The initial AOC measurement method was developed by van der Kooij (e.g. van der Kooij et al., 1982) and later modifications of the AOC analysis have been proposed by other studies, such as Kaplan et al. (1993). The methods basically use a bioassay to quantify the bacterial batch growth as the number of colony forming units (cfu) in a water sample from inoculation until stationary phase. *Pseudomonas fluorescens* strain P-17 and *Spirillum* strain NOX are used as test organisms; the final result is expressed as acetate-C equivalents.

The BDOC concept and measurement was firstly developed by Servais et al. (1987), assuming organic carbon is the limiting nutrient for microorganism regrowth in distribution systems.

Unlike AOC analysis, which measures the quantity of biomass in the sample, the BDOC analysis directly measures dissolve organic carbon (DOC) levels during a period of re-incubation of bacteria in a water sample and considers the difference of the initial and final DOC concentrations as BDOC. BDOC is that portion of the organic carbon in water than can be mineralized by heterotrophic microorganisms (Huck, 1990).

Huck (1990) provided a comprehensive comparison of the concepts, measurement and applications of AOC and BDOC. He suggested that "if the concern is with bacterial regrowth generally or growth of coliforms specifically, the appropriate parameter is AOC, if the concern is the reduction in chlorine demand or disinfection by-product formation potential through a biological treatment process, the appropriate parameter would be BDOC.", since AOC measures bacterial amount directly and BDOC measures organic carbon difference.

Previous research has shown that when there was no disinfectant residual and the AOC level was below 10 µg/L, heterotrophic bacteria did not increase, indicating a relatively stable condition (van der Kooij et al., 1982). LeChevallier et al. (1987) also suggested coliform regrowth may be limited by AOC levels less than 50-100 µg/L when there was a 3-6 mg/L chlorine residual remaining in the system.

Volk et al. (1994) found that biostability is temperature-dependent and suggested 0.30 mg/L (15°C) and 0.15 mg/L (20°C) of BDOC being the threshold values above which the water is not biologically stable. A biostability criterion of 0.16 mg/L BDOC was also proposed by Servais et al. (1993) for a sample with or without disinfectant residual. Volk and LeChevallier (2000) showed that the probability of coliform occurrences greatly increased when threshold values were exceeded: (1). 15°C, (2). AOC > 100 µg/L and (3). dead-end disinfectant levels <0.5 mg/L for free chlorine or 1.0 mg/L for chloramines. When all three criteria were exceeded, the likelihood of finding coliforms in distribution system water was eight times as high as when none of the criteria was exceeded.

Different AOC and BDOC threshold concentrations summarized above suggest that biostability of a water is multi-variable dependent and quantifying biostability using BOM parameters may need the assistance of other parameters.

2.2.3 Occurrences and Relationship of AOC and BDOC

Liu et al. (2002) investigated the variation of AOC concentrations in water from conventional water treatment plants and distribution systems in a northern city of China and showed AOC levels in most of the product waters could not meet LeChevallier et al. (1987)'s biostability criterion of 50-100 µg/L (only 4% of the measured samples were less than 100 µg/L).Volk and LeChevallier (2000) summarized full scale plants' data from a US-wide survey which represented a broad range of plant sizes, raw water and treatment trains. Yearly average AOC and BDOC concentration in the finished water at the sites studied ranged from 18 to 214 µg/L and from 0.03 to 1.03 mg/L, with the geometric mean values of almost 100 µg/L and 0.32 mg/L, respectively. Sixteen to nineteen percent of the sites had AOC concentrations of greater than 150 µg/L and BDOC concentrations of greater than 0.6 mg/L, which might have cased low biostability in their distribution systems. In the same article, a weak but still significant correlation between AOC and BDOC was shown for their surveyed data. In another survey (12 months, 64 plants) conducted by the same research group (Volk and LeChevallier, 2002), the mean effluent AOC levels were 123 µg/L for river water as raw water source, 41 µg/L for river and groundwater/infiltration gallery as influent source.

AOC values in Norwegian waters reported by Charnock and Kionno (2000) were much lower compared to those reported for both raw water and treated water in the United States and China. The average AOC (*P17*) level in this study was 27.3 µg/L in raw waters and 23.8 µg/L in drinking waters (average reduction of about 14%). However, the mean BDOC was 0.91 mg/L in raw waters and 0.45 mg/L in drinking waters, which is comparable to the values reported by the US counterparts (Kaplan et al., 1994).

Besides comparison of concepts, measurements and applications of AOC and BDOC, their co-occurrences and relationship in raw water and treated water have been discussed in several studies. Huck (1990) investigated the relationship of AOC/DOC and BDOC/DOC and did not find any overall relationship between the two ratios since they highly depend on the type of organic matter present in raw water. The author also observed a seasonal variation for AOC: DOC but not for BDOC: DOC.

Charnock and Kionno (2000) also examined AOC and BDOC concentrations in raw waters and after different treatment processes and could not find relationships of occurrences as well as removals of AOC and BDOC. The removal results suggested that bulk BDOC is typically of higher molecular weight than AOC and correlations between AOC and BDOC in raw waters and drinking waters were not significant. They concluded it was likely that the AOC and BDOC analyses target different fractions of BOM.

Escobar and Randall (2001) also conducted a study to evaluate the necessity of measuring both AOC and BDOC as indicators of bacterial regrowth-potential. They proposed that both AOC and BDOC should be used as complementary measurements of bacterial regrowth potential as well as the fate of BOM during treatment and distribution systems. This proposal was based on results in their experiments, such as that the nanofiltration unit removed over 90% of BDOC while allowing the majority of AOC through. They believed that even though AOC was a fraction of BDOC, measuring only one of these parameters cannot obtain an accurate description of the biostability condition, with AOC analysis underestimating and BDOC analysis overestimating the bacterial regrowth potential of the water.

Unlike the results mentioned above, Kaplan et al. (1994) found in a US national survey that correlations between AOC and BDOC, AOC and DOC, BDOC and DOC were statistically significant; the authors, however, realized the limited data and urged focus on individual samples. van der Kooij (1990) also suggested quotient AOC/BDOC be regarded as an indication of the relative biological stability for drinking water.

Extensive research has been conducted to study AOC and/or BDOC removal since they are water quality concerns (Urfer, 1997; Prévost, et al., 2005). AOC and BDOC concentrations after each conventional treatment units were collected by Volk and LeChevallier (2000) and the data showed conventional treatment has the ability to remove AOC and BDOC to some degree, although the contribution of each unit in a treatment train varies greatly. Other researchers also studied the effect of advanced treatment processes such as UV, adsorption and oxidation on AOC or BDOC concentration changes (Lehtola et al., 2003; Hu et al., 1999). Biofiltration, itself or in combination with other processes, has been used as an effective way to produce more biostable water.

2.2.4 Discussion on Different Limiting Nutrient and Biostability

Although the application of AOC and/or BDOC as indicators for microorganism growth has been recognized by drinking water researchers and facilities, it is based on the assumption that organic carbon is the limiting nutrient for the microorganisms. However, recent studies have shown that if water contains high amounts of organic matter, microbial growth in drinking water can be limited by phosphorus instead of organic carbon (Miettinen et al., 1997; Sathasivan et al., 1997; Keinänen et al., 2002). The reported cases of phosphorus limited water are distributed in boreal regions or highly polluted sites in Northern Europe and East Asia.

In this context, Lehtola et al. (1999) developed a new parameter, microbial available phosphorus (MAP), to replace AOC and BDOC as an indicator of microbial growth and biostability condition. The concentration of MAP is analyzed by a bioassay, where the maximum growth of strain *P17* in a sterilized water sample is related to the phosphorus concentration. The concurrences of MAP, AOC and microbial growth potential were presented by Lehtola et al. (2002) for conventional water treatment processes. Lehtola et al. (2001, 2003) also showed for the ozone and UV treated samples they analyzed, minor changes in MAP concentrations would dramatically increase the amount of heterotrophic microbes, while AOC did not limited biostability. For such phosphorus limited water, the evaluation of biological stability needs to be examined carefully.

2.3 Factors Affecting Biofiltration

Various factors have been confirmed to have the capability to influence performance of biofiltration processes. Several most important factors are summarized in this section.

2.3.1 Effect of Empty Bed Contact Time (EBCT)

Servais et al. (1992) identified EBCT as a critical design factor for the removal of BDOC in biofilters. In another study, Servais et al. (1994) operated three pilot-scale BAC filters at different filtration rates (6, 12 and 18 $m \times h^{-1}$) but at the same EBCT (10 minutes). Significant differences in BDOC removal in the different filters were not found in this study, indicating that filtration rate does not affect biological activity and therefore EBCT is the main design parameter. Carlson and Amy (1995) also compared the relative importance of HLR (Hydraulic Loading Rate, Equation (3-5)) and EBCT in biofiltration. Carlson and Amy (1998) indicated that "the biomass utilization as a function of EBCT did not depend on HLR, indicating that EBCT is an acceptable biofilter design parameter throughout a range of HLRs and filter bed depths."

LeChevallier et al. (1992) reported an increase in total organic carbon (TOC) removal from around 30% to around 50% when EBCT increased from 5 to 20 minutes. Merlet et al. (1991) observed that the removal of BDOC increased with EBCT and plateaued at 25 minutes at 12 °C, with final BDOC concentration of around 0.5 mg/L. These studies indicated that increasing EBCT could improve the organic matter removal efficiency, but the relation was less than proportional. Above a certain range of EBCT, increasing EBCT was not able to gain much improved performance in return.

The results from other research are inconsistent. Hozalski et al. (1995) studied TOC removal by biofilters and observed TOC removals were similar for all of the experiments regardless of EBCT or natural organic matter (NOM) origin and independent of EBCT in the range of 4-20 minutes.

2.3.2 Effect of Temperature

Temperature affects microbial ecology, degradation kinetics and mass transfer rate in the biofilter, thus in theory affects the organic matter removal. Low water temperatures during winter may represent a potential problem for BOM removal, especially in biofilters having a short EBCT.

Melin et al. (2002) ran a small full-scale ozonation/biofiltration process at temperatures between 4-22°C. The results showed low temperature and low phosphorus level were potential limitations for microbial growth and organic carbon removal. The calculated average rate constants increased from 0.066 d^{-1} at 4°C to 0.41 d^{-1} at 22°C.

Although previous research showed the dependence of organic matter removal on temperature, it should be noted that that scenario is usually confounded as changes in temperature usually occur simultaneously with changes in water quality related to seasonal variation in the source water (i.e. summer and winter conditions).

In a study by Emelko et al. (2006), there were not statistically significant differences in terms of NPOC (Non-purgeable Organic Carbon) removal for warm (21-24°C) and cold (1-3°C) conditions in full-scale anthracite-sand and GAC filters, although it reported that water temperature significantly affected oxalate removal.

Despite the above mentioned inconsistent observations, several studies show that EBCT should be increased to compensate the effect of temperature with decreasing temperature. For example, Nishijima et al. (1998) observed that biofilters with long EBCT of 24 minutes) can achieve a nearly constant DOC removal in the range of 5-30°C.

Temperature not only affects the performance of a steady-state biofilter, but also affects that of a non-steady state one and the time to reach steady state. Liu et al. (2001) observed start-up time requirements of ozonation by-products removal increased from 20-40 days to 20-60 days when temperature dropped from 20°C to 5°C.

2.3.3. Effect of Filter Media

Selecting filter media for a biofiltration process is not only for economic considerations, but also of technical significance. The type of biofilter media affects the amount of biomass accumulating in the biofilter. For example, Niquette et al. (1998) measured normalized biomass of 14 μg C/cm^3 in sand-GAC filters and 3μg C/cm^3 in sand-anthracite filters.

A number of researchers compared the performance of biofilters with different filter media. Najm et al. (2005) compared lignite and bituminous GAC and did not find a statistical difference in terms of DOC, AOC, or aldehyde removals for these types of filter. LeChevallier et al. (1992) compared a GAC-sand filter with an anthracite-sand filter and found better AOC removal of 86% for the GAC-sand filter and 75% for the anthracite-sand filter. Rittmann et al. (2002) also indicated GAC was better than sand for removal of DOC and color.

Most of the literature documented a better performance of GAC than sand or anthracite in biofiltration. It is believed that the greater surface area of GAC on which biomass accumulates and the ability of GAC to adsorb substrate for subsequent biodegradation biomass lead to this difference. Zhang (1996) established a quantitative method to quantify the contribution of media surface area on the organic matter removal. It should be highlighted here that all the GAC media mentioned in this section were exhausted in terms of adsorption capacity because of the scope of this thesis. Differentiation of biodegradation removal and adsorption removal of organic compounds in biofilter will be discussed in Section 2.3.7.

2.3.4 Effect of Ozonation

In drinking water practice, biofiltration is often applied with pre-ozonation. The combination is considered sometimes as a single process, i.e., O_3-Biofiltration, or O_3-BAC if activated carbon is used as filter media (Simpson, 2008).

2.3.4.1 Introduction

Ozone is unstable in water. The decay of ozone in natural waters is characterized by a fast initial decrease of ozone, followed by a second phase in which ozone decreases with first-order kinetics. Ozone follows two basic reaction paths: 1) direct oxidation, which is rather slow and selective, and 2) auto-decomposition to the hydroxyl radical. A low pH favors the slow, direct oxidation reaction path involving O_3, and a high pH or a high concentration of organic matter favor the auto-decomposition route (von Gunten, 2003a, b).

NOM can affect the ozone stability in two ways: it can either (i) directly react with ozone or (ii) indirectly affect its stability through scavenging of OH radicals. There have been various attempts to deduct both the kinetics of the direct ozone-NOM reaction and the various fractions of NOM (von Gunten, 2003a, b)

2.3.4.2 Effect of Ozonation on Organic Matter

NOM comprises refractory organic matter, slowly and easily degradable organic matter. Oxidant ozone can convert refractory organic matter into easily biodegradable matter. For drinking water treatment, this conversion can be considered as a positive effect if ozonation is applied prior to biofiltration process. The process combination makes it possible to produce water with lower organic carbon, lower THMFP (Trihalomethanes Formation Potential) and chlorine demands (Mehrnaz. 2008); while if there is no following biofiltration, the produced bioavailable organic matter would facilitate bacterial regrowth later on and increase the biological instability in distribution systems (Shu et al., 2008).

It should be noted that ozonation and biofiltration plays reverse roles in terms of biodegradable organic carbon removal. The effect of producing more degradable organic carbon by ozonation offsets the removal effect by biofiltration. One would not be surprised that the overall percentage removal of biodegradable carbon by ozonation-biofiltration is not very high, and in some cases, the percentage removal could be negative.

Directly quantifying increased NOM biodegradability after ozonation is difficult due to the varieties of biodegradability measurement. Yavich et al. (2004) developed a simple procedure to describe the kinetics of biodegradation of natural organic matter in drinking water and used this procedure to evaluate changes in the concentration of biodegradable organic matter. The proposed approach quantitatively showed, regardless of origin of source water, that ozonation of

NOM may result in increasing biodegradable components of NOM. For Lake Huron Lake water samples, when the ozone dose increased from 0 to 1.0 mg O_3/mg C, the overall initial BDOC, the slow and fast degradable BDOC concentration doubled, the corresponding maximum substrate utilization rate, which were determined by their proposed method, increased from 0.017 mg/L min to 0.11 mg/L min.

Goel et al. (1995) conducted batch scale tests to evaluate the extent of biodegradation of natural organic matter as a function of ozone dosage. Four NOM sources that might be encountered in drinking water treatment were characterized and tested. The biodegradability of all source water was enhanced as the ozone dose was increased from 0 to 7.3 mg ozone/mg Carbon. Increased ozonation resulted in consistently improved TOC removals for NOM sources having a large fraction of high-molecular-weight organics.

Nishijima et al. (2003) evaluated single-stage ozonation and multi-stage ozonation for the different types of raw water. The results showed with the same total empty bed contact time, the multi-stage ozonation/biofiltration process had a higher DOC and BDOC removal than the single stage one.

2.3.4.3 Ozonation By-Products

In addition to increasing the biodegradability of organic matter, ozonation causes formation of ozonation by-products (OBPs). These byproducts include small molecular weight compounds such as aldehydes, acetones, ketoacids and carboxylic acids. These four groups of ozone by-products were used by some studies to mimic the situation in water after ozonation (Urfer, 1998).

Griffini et al. (1999) analyzed the formation of ozonation by-products. The formation of aldehydes and ketoacids was found to be proportional to the DOC concentration and ozone dosage, and to have strong relationship with BDOC concentration. Carlson and Amy (1998) in their study also determined the relative compositions of OBPs in the tested ozonated water.

Besides the four groups of ozonation by-products mentioned above, von Gunten and Hoigne

(1994) indentified bromate formation during ozonation of bromide-containing waters.

At least some of these ozonation by-products can potentially cause adverse health problems, so they are undesirable compounds in terms of providing safe drinking water. They themselves are the treatment targets for following processes. Biofiltration was found to be an effective way to remove the ozonation by-products (Huck et al., 1990; Griffini et al., 1999; Liu et al., 2001). For example, Melin and Ødegaard (2000) studied the treatment efficiency of organic ozonation by-products in biofilters with plastic biofilm media and expanded clay under different hydraulic conditions and observed over 80% removal generally.

2.3.4.4 Effect of Ozone Residual in Biofilter

Decay of ozone/H_2O_2 and their reaction in water break down larger molecules into smaller ones and increase biodegradability of NOM, as well as bring oxygen into biofilters and facilitate the living environment of microorganisms. However, the oxidants used in the drinking water industry are normally disinfectants too; therefore in theory, the presence of an ozone residual in the biofilter influent was detrimental to the microorganisms in the biofilm and should be avoided in practice.

Huck et al. (1991) reported a presence of 0.1-0.3 mg/L ozone in the influent of a dual media filter (anthracite/sand) and the performance of the biofilter was affected by the ozone residual.

Urfer (1998) studied the periodic presence of 0.1-0.2 mg/L ozone (1-2 hours per week) and did not observe any significant negative effect on organic matter removal. The biomass and biomass activity was similar in the filters with and without ozone residual. He concluded that the ozone residual was likely to be eliminated in the very top layer of the biofilters, or to react with media if the media was anthracite.

A detailed review of the effect of other oxidants on the biofiltration was provided by Urfer et al. (1997) where the effects of Cl_2, H_2O_2 and chloramine were summarized. Urfer (1998) also compared the negative effects of residuals of these oxidants and pointed out that free Cl_2 had the strongest negative impact on biofilm and biofiltration process.

2.3.5 Effect of Backwashing

Backwashing, which may include air scour, and use chlorinated or non-chlorinated water, is a necessary operation to remove biological and non-biological particles accumulated on filter beds. An effective backwashing strategy involves removing excess particles and biomass while maintaining enough attached biomass in the filter for the next run. (Ahmad et al., 1998a)

Filter backwashing was reported to improve the efficiency of first-stage sand biofilters (Niquette et al.1998), while there is concern for whether or not backwashing can impair the performance of a biofilter at least during the early filter cycle by losing biomass or inhibiting bioactivity (Ahmad et al., 1998b).

Miltner et al. (1995), Liu et al. (2001) and Emelko et al. (2006) observed no effect of air scour and water backwashing conditions on organic matter removal at different scales of practice. These reports were supported by models developed by Hozalski and Bouwer (2001a, b) that less than 60% loss of biomass would not impair biofilter performance.

2.3.6 Effect of Biofilter Start-up

Biofilm growth in newly operated biofilters is a slow process. When a biological filter is first put into use, it needs a certain period of time to reach a steady-state (or at least a pseudo-steady state); this period is referred to as “start-up”.

“Steady state” here has two definitions, depending on which factor is used for steady state evaluation. New biofilters go through a colonization phase to achieve full biodegradation activity. Normally this is a long process. Wang et al. (1995) measured the amount of phospholipid-P for the top layer of a dual-media filter and GAC filter. After four to five months, a stable level of 80 nmol Lipid-P/g media was reached and they claimed this was “steady-state”.

However, if one evaluates the start-up time of a biofilter using stable removal performance evaluation, the time to reach "steady state" is different as mentioned above. Achieving steady-state biological removal of organic matter does not take as long a time as steady-state biofilm formation. This may be attributable to two reasons: one is that physical-chemical processes such as adsorption occur in the filter, the second is the parameter of biomass amount is not necessarily related to the removal of substrate, or the activity of the working microorganisms, although essentially a high amount of biomass in the filter represents a more facilitating environment for substrate removal (Liu et al., 2001 and Wang et al., 1995).

In the same study mentioned above (Wang et al., 1995), stable removals of BOM parameters were observed in an anthracite-sand filter within 100 days (much less than 4-5 months), although the biomass continued to increase after this period.

Liu et al. (2001) studied the start-up performance for typical ozonation products in the anthracite and GAC biofilters. More than 20 days were needed to achieve steady-state in all filters and more time (20-60 days) was required to reach stable BOM removal at low temperatures than at high temperatures (20-40 days).

Modeling work, which will be discussed later, predicted a shorter time to reach steady state (Hozalski, 1996) for biofiltration by assuming several ideal conditions including no oxidant residual in the biofilter.

2.3.7 Discrimination between Biodegradation and Adsorption

Start-up time allows initial attachment and growth of biofilm on the surface of filter media. Since some media used for biofiltration, such as GAC, are also widely used sorbents, extensive research has been done for adsorption of natural organic matter (Snoeyink, 1990; Newcombe et al., 1997) from water by GAC. For virgin or non-exhausted filter media, it is apparent that adsorption and biodegradation occur simultaneously in the filter during the start-up period, thus overall observed substrate removal by a filter comprises not only biological removal but also physicochemical removal until the adsorption capability of the filter media is exhausted.

Persson et al. (2007) compared biofilters' performances to remove geosmin and MIB with adsorptive media (GAC) and non-adsorptive media (crushed Expanded Clay, EC). At lower temperatures and when microbial activity was suppressed by chemicals, BOM removal by EC filters was significantly lower than by GAC filters. A methylene blue adsorption test confirmed that the GAC particles, even after almost four years of operation receiving surface water, had the capacity to remove geosmin and MIB by adsorption.

Wang et al., (2007) used a sterile GAC column, a conventional GAC column and a sand column to study biofiltration of microcystin toxins. The results demonstrated that biodegradation was efficient once it commenced, although adsorption of microcystins was prevalent during initial stages of the GAC columns. Up to 70% removal of microcystin was still observed after 6 months of operation of the sterile GAC column, indicating that adsorption still played a vital role in removal of this toxin. This study also noted there was a competitive effect on the occurrence of adsorption and biodegradation because the applied model showed an active biofilm hindered the adsorption of microcystin.

Erlanson et al., (1997) developed an equilibrium model for simultaneous biodegradation and adsorption and verified it with existing data. The results from modeling over 300 hypothetical situations identified that biodegradation significantly extended the service life of GAC adsorption columns.

Liang et al., (2007) also developed a non-steady-state numerical model to differentiate adsorption and biodegradation (with considerations of convection and diffusion) in BAC (Biologically Active Carbon) columns. Simulation results showed that the Freundlich isotherm constant, together with maximum specific substrate utilization rate and diffusion coefficient, was the most sensitive variable affecting BAC performance; decreasing particle size resulted in more substrate diffusion across the biofilm, and increased the ratio of adsorption in relation to biodegradation.

2.4 Applications of Biofiltration

2.4.1 Removal of AOC and BDOC

The biofiltration process has been applied extensively to produce more bio-stable water in drinking water treatment practice. Urfer et al. (1997), Carlson and Amy (1998), Graham (1999) and Huck and Sozański (2008) provided comprehensive reviews for the removal of BOM-related parameters (including AOC and BDOC) by biofiltration.

Servais et al. (1994) studied BDOC removals by pilot-scale filters. The biomass amount in the filters reached steady-state after three months of colonization, and biodegradation processes took a dominant role for BDOC removal.

LeChevallier et al. (1992) evaluated the performance of pilot-scale biologically active rapid filters with different filter media, EBCTs, temperatures as well as pre-oxidations and found in addition to satisfactory turbidity removals, that all filters could achieve effluent AOC levels <100 µg/L, with the treatment efficiencies ranging from around 50% to as high as around 90%.

2.4.2 Removal of Chlorine Demand and DBP Formation Potential

Chlorine demand of a treated water is the difference between the amount of chlorine applied in the water and the amount of free chlorine residual remaining at the end of the contact period. Chlorine demand is of removal concern for drinking water treatment because of two reasons. It affects the amount of chlorine that must be added in finished water to maintain the regulated residual in some countries. Chlorine and chloramine in the water also react with some organic compounds in the water and produce stable chlorinated organics, such as trihalomethanes (THM), haloacetic acids (HAAs), nitrosamines and N-nitrosodimethylamine (NDMA), many of which may be of health concern. These chlorinated organics are called disinfection by-products (DBPs) and are driving forces in changing disinfection practices.

Pilot-scale and full-scale GAC and anthracite biofiltration tests were conducted by Mehrnaz (2008) to assess removal performance of chlorine demands and various DBPs. Despite seasonal changes, around 10%-30% removal of chlorine demand, precursors of THMs, HAAs and NDMA were observed in this study.

Other authors also reported significant removals of various DBP formation potentials by biological processes (Miltner et al., 1992; Shukairy et al., 1992a; Huck et al., 1994; Wu and Xie 2005). For example, biofilters following ozonation processes in Shukairy et al. (1992b) reduced POXFP (Purgeable Organic Halide Formation Potential) level from 234 µg/L to 47 µg/L; NPOXFP (Non-Purgeable Organic Halide Formation Potential) level from 762 µg/L to 210 µg/L and TOXFP (Total Organic Halide Formation Potential) level from 996 µg/L to 257 µg/L.

2.4.3 Secondary Utilization

Secondary utilization refers to the degradation of a growth substrate (secondary substrate), but at a concentration lower than that required for sustaining bacterial growth (e.g. below $S_{\min}$), and therefore, this process occurs in the obligate presence of another growth substrate (primary substrate) that sustains biomass, or occurs while the active biomass degrades itself to satisfy the cell energy demand (Rittmann et al., 1995; Aranda et al., 2003).

For drinking water biofiltration processes, removing taste/odour substances, algal toxins and emerging contaminants (such as pharmaceuticals and personal care products, PPCPs) are main applications of secondary utilization (Elhadi, 2004; Ho et al., 2006; Hallé et al., 2009).

2.4.3.1 Removal of Taste and Odor Secondary Utilization

Taste and odour problems are common for water utilities. MIB (2-methylisoborneol) and geosmin are the two most common odorants, which are associated with blue-green algae and *actinomycetes* (Pei, 2003). The human threshold concentration for odour detection ranges from 0.004 to 0.020 µg/L for geosmin and from 0.009 to 0.042 µg/L for MIB.

Although previous study indicated that MIB and geosmin were poorly removed by conventional drinking water treatment processes (Bruce et al., 2002), Elhadi (2004) summarized that advanced treatment processes, including biological process, oxidation and adsorption, had been successfully applied to remove MIB and geosmin, among which biofiltration is one of the most promising processes.

Ho et al. (2007) showed that sand filters were effective for nearly completely removing MIB and geosmin and confirmed that the removals were predominantly through biodegradation. Elhadi et al. (2004) conducted bench experiments for removing MIB and geosmin by fresh and exhausted GAC filters. The exhausted GAC filters, in which only biodegradation processes could occur, still presented overall significant removing efficiencies. Their results also presented good performances of the biofilters for the start-up period and transient presences of MIB and geosmin.

Pei (2003) showed in his study that GAC filtration processes after ozonation could achieve 40-60% removals of MIB; geosmin could be almost completely removed by biofilters. Surveys of utilities in the US conducted in the same study also showed 60-90% removals of MIB and 60-100% removals of geosmin by ozonation-biofiltration processes. Most utilities had MIB and geosmin concentrations under 2 ng/L (the detection limits) in ozone-biofiltration treated waters.

2.4.3.2 Removal of Pharmaceuticals and Personal Care Products (PPCPs)

Over the last decade, interest in the presence of PPCP as well as endocrine disruptors and illicit drugs in the environment has increased significantly. Although PPCPs have been found to be present at very trace concentrations in the aquatic environment (ng-µg/L levels), they are considered as potentially hazardous compounds (Snyder et al., 2003). The levels of their concentrations and fates in the aqueous environment vary greatly and depend on several factors such as geographical location, effectiveness of wastewater treatment, proximity to wastewater treatment facilities and meteorological conditions (Boyd et al., 2003; Kasprzyk-Hordern et al., 2008).

Various treatment processes, conventional and advanced, have been studied to evaluate performance of removing PPCPs from raw water (Westerhoff et al., 2005; Yu, et al., 2009).

Depending on individual biodegradability and other chemical properties, PPCPs are expected to demonstrate various biodegradation rates and removals in biofilter. Huck (2008) and his co-workers (Guay et al., 2007; Hallé et al., 2009) reported from zero (e.g., atrazine) to satisfactory removals (e.g., ibuprofen) of several selected PPCPs by biofiltration processes. More work needs to be done for evaluating and predicting individual emerging contaminant removal by various treatment processes, including biofiltration.

2.4.4 Biofiltration Application for Other Objectives

In addition to the applications discussed in the earlier sections, biofiltration processes have been applied for other drinking water treatment objectives.

Urfer et al. (1997) summarized previous studies and concluded there was no difference in turbidity removal between biological and non-biological filters, while non-biological filters were reported to outperform biological filters for particle removal, probably because of different particle detachment mechanisms.

Biofiltration applications were also reported to remove pathogens by Betancourt and Rose (2004), TOC by Hozalski et al. (1995), DOC by Carlson and Amy (1998), bromate and ammonia by Wert et al. (2008) and Griffini et al. (1999).

Fouling is a major limitation for application of membrane technology in drinking water treatment (Flemming, 2002). Recently, a biofilter was placed prior to membrane units as a pre-treatment process. Meesters et al. (2003) found that biofiltration significantly reduced AOC levels and biofouling for a recirculation cooling system on the assumption that AOC is the limiting substrate for the growth of biofoulants. Hallé et al. (2009) showed that biofiltration was effective to reduce fouling for low pressure membranes because it was able to remove biopolymer, which are highly likely to be responsible for organic fouling.

2.5 Modeling Practices of Biofiltration

2.5.1 Early Modeling Attempts

For the past three decades, a number of modeling developments have been proposed and practiced in an attempt to describe biofiltration processes quantitatively for research. Among these developed models, some are for drinking water treatment; some are for wastewater and waste gas treatment.

Considering microbial kinetics (Monod equation) and mass transport (Fick's law), Rittmann and McCarty (1980a) developed a steady-state biofilm model, which was described by Urfer et al. (1997) as "seminal". This model was first developed in a wastewater context, while later it was applied for drinking water biological processes successfully. Almost all the later model developments were based on this model, in which a major contribution was the introduction of a concept of $S_{\min}$. S is the only rate-limiting substrate and $S_{\min}$ is defined as the bulk substrate concentration below which a steady-state biofilm cannot exist.

Basic equations (Equations (2-1), (2-2) and (2-3)) for the model are presented below in dimensionless form:

$$S_b^* = S_s^* + J^* L^* / D^* \tag{2--1}$$

$$S_{\min}^* = b / (Yk - b) \tag{2--2}$$

$$L_f^* = \frac{1 + S_{\min}^* J^*}{S_{\min}^*} = J^* kY / b \tag{2--3}$$

where

$S_b^* = S_b / K_s \qquad S_s^* = S_s / K_s \qquad S_{\min}^* = S_{\min} / K_s$

$J^* = J\tau / K_s D_f \qquad L_f^* = L_f / \tau \qquad L^* = L / \tau$

$D^* = D / D_f \qquad (\tau = [K_s D_f / kX_f]^{1/2}, L)$

Where S_b is the bulk liquid concentration of the substrate (M_sL^{-3}); D is the free liquid diffusivity (L^2T^{-1}); D_f (L^2T^{-1}) is the diffusivity in the biofilm; K_s is the half velocity constant in the Monod expression (M_sL^{-3}); k is the maximum specific rate of substrate utilization (T^{-1}); Y is the yield coefficient, M_X / M_S; b denotes the overall biofilm decay rate coefficient (T^{-1}); X_f is the biofilm density (M_XL^{-3}); L_f is the biofilm thickness (L) and L denotes thickness of the effective diffusion layer (L).

Rittmann and McCarty (1980b) evaluated this model and obtained values of the biofilm kinetics using a bench-scale biofilm reactor. The study verified the existence of $S_{\min}$ (e.g. 0.66 mg/liter for acetate as substrate) and indicated that this biofilm model can be applied both to wastewater and drinking water treatment. Saez and Rittmann (1988, 1990, and 1992) provided pseudo-analytical solutions to the model equations since analytical solutions for this model are not possible unless simplifying assumptions are made.

By ignoring the axial dispersion effect and making several assumptions, Zhang and Huck (1996a) later obtained an analytical solution and developed the new concept of X* (dimensionless contact time). Later, Huck and Sozański (2008) developed the concept of a practically-oriented biofiltration factor (BF), which is linearly related to X*. Both of these will be discussed in later sections.

2.5.2 Biofiltration Modeling Practices

Urfer et al. (1997), Chaudhary et al. (2007) and Huck and Sozański (2008) provided comprehensive reviews on biofilm modeling advancements. This section will summarize the typical and most influential biofilm models.

1) Chang and Rittmann's BFAC model for Activated Carbon (1987)

Chang and Rittmann (1987a) developed a model (BFAC model) for biofilm on activated carbon. The model incorporated film transfer, biodegradation and adsorption of a substrate, as well as biofilm growth. They verified this model using lab-scale activated carbon columns and found that it underestimated effluent biomass, compared to experimental data (Chang and Rittmann, 1987b).

2) CHABROL model (Billen et al., 1992)

Billen et al. (1992) developed the CHABROL model specifically for BDOC removal from water by biofiltration, which related BOM consumption to biomass densities in order to predict BOM removal profiles. Application of this model needs a careful division of BDOC into non-degradable, slowly degradable and rapidly degradable subdivisions. One disadvantage for the CHABROL model, as Urfer et al. (1997) pointed out, is that it requires considerable data collection in the filters being modeled, since undetermined constants in this model vary from case to case and it is difficult to assign a value for them.

Laurent et al. (1999) assessed the CHABROL model using a large database from pilot- and full-scale filters located in two Canadian cities (Laval and Montreal). This database includes data from two different water sources and three biofilter configurations; nearly half of the data was sampled at very low temperature ($\leq 1°C$). The assessment showed overall satisfactory results for this model in an ability to predict BDOC removals after a major modification to adjust for the effect of low temperature. In another study by Cauchi et al. (1993), BDOC removals by GAC contactors also verified the CHABROL model predictions very well.

3) Uhl's model (2001)

According to the concept of $S_{\min}$ and the models based on $S_{\min}$, the very low substrate concentrations of a few μg acetate-carbon/L (as AOC) commonly observed in the effluent of filters for drinking water treatment should not be achievable. Noticing this inconsistency, Uhl et al. (2001) assumed in a new model that bacteria were capable of degrading the substrate at any low substrate concentration, in another words, no physiological threshold concentration exists.

The model development also made assumptions that kinetics of attached bacteria and suspended bacteria were the same and mass transfer of the substrate from the bulk liquid to the media surface was not limiting.

This model specifically described attachment and detachment of suspended bacteria to and away from filter media. The model simulation results gave a good description for the observations of AOC and DOC removals. The authors also noted that it might be applicable for dynamic condition as well as steady-state conditions.

4) Wang's model

Wang (1995) proposed a heterogeneous biofiltration model, which was based on a modification of a homogeneous model in the same research. The heterogeneous model also divided DOC into non-biodegradable, easily and slowly biodegradable fractions. Monod-type kinetics was used to describe utilization of the quickly biodegradable DOC fraction and first-order kinetics were used to simulate utilization of the slowly biodegradable DOC. The model assumed the utilization kinetics of each NOM fraction was uniformly distributed along filter depth, however varied with the filter depth. The DOC concentration and biomass versus filter depth were able to be predicted by this model. Model validation research using lab- and pilot-scale biofilter performance data with externally determined parameters indicated a good fit for the simulations.

5) Model developed by Huck and Huck's co-workers

A first-order reaction model was first developed by Huck and Anderson (1992) and Huck et al. (1994). New concepts of dimensionless contact time (X*) and biofiltration factor (BF) were developed and applied for practical use in Zhang and Huck (1996a, b) and Huck (1999). This will be discussed in the later section.

2.5.3 Multi-Species Modeling Practices

For drinking water treatment biological process, when the treatment goal of BOM is expressed as one surrogate parameter, such as AOC and BDOC, it is reasonable to treat biofilm microorganisms as a uniform population (mono-species) although the ecology in the biofilm system is sophisticated in which a large number of species co-exist (Zhang, 1996). Models considering various specifies (multi-species) also have been developed (Wanner and Guier, 1986; Rittmann and Manem, 1992; Rittmann et al., 2002; Rittmann and Stilwell, 2002). Typical divisions of biofilm microorganisms are heterotrophs, autotrophs and inert biomass. Rittmann et al. (2002a) specified ammonia oxidizers and nitrite oxidizers and seven chemical species, eight reactions as well as diffusion effects and biomass dynamics in their TSMPB model. A major improvement for this model, comparing with the previous discussed models, is a detailed description of constant and periodic detachment of biofilm ($b_{\det}$).

Multi-specifies model prediction clearly showed there was inhibitory effect among different specifies and their substrate utilizations (Rittmann and Manem, 1986). Rittamnn et al. (2002b) used pilot-scale experiments to remove DOC from raw water and provided interpretation for the TSMPB model, which demonstrated existence of the interactions, e.g., a smaller detachment rate would increase soluble microbial products and build up inert biomass.

2.5.4 Non-Steady State Modeling

Biofilters can achieve a steady-state operation condition when temperature and raw water quality do not fluctuate very much. Almost all the developed models are for steady-state, or at least for pseudo-steady state conditions.

Hozalski and Bouwer (2001a) developed a “BIOFILT” model for biofilter non-steady state conditions, which simulated biomass (both attached and suspended) and substrate profiles as functions of time. One of the innovative features of BIOFILT compared to previous biofilm models was an ability to simulate the effect of a sudden and/or severe loss of attached biomass due to filter backwash on substrate removal. In a subsequent article, Hozalski and Bouwer (2001b) also tested this model for a full scale treatment plant and found that real data validated the model output satisfactorily.

2.6 Development of X*

2.6.1 Deriving the Concept of X*

On the basis of the biofilm model developed by Rittmann and McCarty (1980) and the pseudo-analytical solution provided by Rittmann and co-workers (Saez and Rittmann, 1988, 1990 and 1992), Zhang (1996) and Zhang and Huck (1996a) obtained an analytical solution for PF (plug flow) reactors (which is applicable for the case of biofilters) after demonstrating that axial dispersion could be reasonably ignored. Based on their solution, they then obtained Equation (2-4), which showed there was a relationship between depth of filter (X) and substrate concentration (at the biofilm outer surface):

$$X = \frac{v\tau}{\alpha D_f}\int_{S_{SX}^*}^{S_{so}^*}\frac{dS_s^*}{J^*} + \frac{vL}{\alpha D} In\frac{J_O^*}{J_X^*} \tag{2--4}$$

Notations were specified in Section 2.5.1.

Equations (2-5) and (2-6) can be transformed by arranging the equation (4) and making some substitutions:

$$X^* = \theta\frac{\alpha D_f}{\tau}\int_{S_{SX}^*}^{S_{so}^*}\frac{dS_s^*}{J^*} + \frac{L^*}{D^*} In\frac{J_O^*}{J_X^*} \tag{2--5}$$

$$X^* = \theta\alpha D_f^{1/2}(kX_f / K_s)^{1/2} \tag{2--6}$$

where $\theta = X / v$, which is the empty bed contact time for a filter.

Equations (2-5) and (2-6) introduce a new concept of an index for contact time, X*. X* is also referred to as dimensionless empty bed contact time, is a major contribution by Zhang (1996) and Zhang and Huck (1996a). X*, which is derived directly from the model by Rittmann and McCarty (1980), theoretically relates to the substrate concentration and in turn BOM removal efficiency. From Equation (2-6), it is apparent that the X* parameter is determined by actual contact time, specific surface area of the attached biofilm (which depends on the specific surface area of the filter media), BOM diffusivity in the biofilm, biofilm density and biodegradation kinetics. Unlike other parameters such as EBCT and L/d, X* incorporates consideration of influent water quality, filter media and filter dimension, and therefore can be considered as a more advanced parameter in terms of describing BOM removal by biofiltration.

For unknown parameters in Equations (2-5) and (2-6), Zhang and Huck (1996b) presented an approach to estimate their values for AOC removal through a bench-scale experiment and statistical method. Estimated values of k, $S_{\min}$ and average AOC diffusivity (D) were in the expected range, while estimated K_s were higher than values obtained from a previous pilot-scale study by Zhang and Huck (1996a).

2.6.2 First-order Reaction Approach for Biofiltration

Huck and Anderson (1992) and Huck et al. (1994) plotted removal rates of several BOM parameters (e.g. THMFP, AOC and chlorine demand) *vs.* EBCTs and influent concentrations of the BOM parameters *vs.* removal rates (normalized by EBCT) and concluded that removal of these parameters could be approximated by a first-order relationship. Gagnon et al. (1997) also confirmed statistically that removal of carboxylic acids during biological drinking water treatment can be considered as a first-order process.

Another pilot study of ozonation by-products removal (Melin and Ødegaard, 2000) also tested the validity of the first-order reaction model (Equation (2-7)) by plotting removal rates and substrate loading rates. The $S_{\min}$ values for different ozonation by-products were also calculated and compared, from which the authors recommended that in a steady-state situation, $S_{\min}$ was probably the determining factor in biofilter performance, providing that capacity of the biofilter to accumulate biomass was not exceeded.

$$S_e = S_0 e^{-k\theta} + S_{\min} \quad (2\text{--}7)$$

Theoretical modeling also leads to the same conclusion. By using experiment data from Huck et al. (1991), Zhang (1996) and Zhang and Huck (1996a) demonstrated a linear relationship between substrate removal and influent substrate concentration (i.e. a constant percentage removal) and thus an essentially first-order reaction from modeling analysis. They further provided extensive interpretation related to the X* parameter, since the slope of this linear relationship (essentially the percentage removal) initially increased linearly with increasing X* and then reached a plateau at which the maximum removal was achieved. Based on this observation, Huck (1999) and Huck and Sozański (2008) developed a table and a figure for the AOC removal-X* relationship.

Urfer et al. (1997) examined the data in Billen et al., (1992) and found that BDOC removal was essentially directly proportional to influent BDOC at a specific EBCT. This conclusion and an integrated simulation of a drinking water treatment train by Rietveld et al., (2008) strongly supported the modeling analysis of Zhang (1996) and Zhang and Huck (1996a, b).

2.6.3 Consideration of Mass Transfer

Both mass transfer processes and biodegradation processes theoretically play a role in BOM removal by biofiltration. Transfer of substrate from the bulk liquid to the biofilm outer surface through a diffusion layer with effective thickness L, is called external mass transfer; the diffusion of substrate within the biofilm is called internal mass transfer. If the mass transfer rate is slower than the biodegradation rate, the overall substrate degradation will be controlled by mass transfer processes.

Booth et al., (2004) evaluated the importance of both external and internal mass transfer in a study using pilot-scale biofilters to remove ozonation by-products by adopting external mass transfer rate determination approach (Karel et al., 1985) and internal mass transfer rate determination approach (Weisz, 1973). They found that it was biodegradation kinetics, rather than any mass transfer rate, that controlled the rate of substrate removal; in another words, mass transfer here could be ignored reasonably. Other researchers, such as Wang (1995), also reported removal of BDOC by biofiltration was governed by reaction rate. Gagnon and Huck (2001) obtained the same conclusion for a distribution system simulator.

The effect of external mass transfer, however, was considered when Zhang and Huck (1996a) calculated specific surface area (α) of filter media , since Roberts et al., (1985) found that by assuming spherical particles, the external mass transfer rate was over-measured (around 1.5-2.0 times) for irregular GAC particles and thus an adjustment was necessary.

2.6.4 Further Simplification and Interpretation of X*

As discussed above that BOM removal can be approximated by first-order reaction and biodegradation is the limiting step, Huck and Sozański (2008) simplified the X* expression as shown below, since the term D_f (for substrate diffusivity) and K_s (for Monod relationship) are not important.

$$X^* = \theta \alpha K (kX_f)^{1/2} \tag{2--8}$$

where K is a constant.

Notations were specified in Section 2.5.1. In practice, the filter parameter (θ) and media parameter (α) can be easily calculated, kX_f can be obtained by experiments, the constant K can be derived from the established removal-X* relationship figure and table (Huck, 1999 and Huck and Sozański, 2008). For a given substrate the X* value is then essentially proportional to the product of θ and α. Huck (1999) applied this simplified practical method to estimate relative X* values for different values of organic matter removal and compare treatment efficiency among different process conditions on this platform.

Huck and Sozański (2008) further observed that approximately 50% AOC removal would be obtained at an X* value of 0.5. They proposed another new parameter, biofiltration factor (BF) and assigned a BF value of 0.5 to a filter achieving 50% AOC removal at approximately 20 °C. BF values required for other treatment objectives or at other temperatures can then be directly proportional to this scenario. Similar with G or GT values for coagulation, CT value for disinfection as Huck and Sozański (2008) suggested and SRT value for activated sludge, BF can be considered as a key and comprehensive parameter for biofiltration, and has the potential to be a critical design criterion for drinking water biofiltration process.

2.7 The Goals for This Research

For traditional filter for which the primary goal is to remove particles and microorganisms, MWH (2005) described various factors (such as effluent water quality and length of a filter run) affecting rapid filtration and demonstrated design criteria and procedures. Tchio et al., (2001) identified filtration rate, media effective size, media depth and water head were the parameters that most impacted filter performance. In terms of selecting a major parameter to guide practical filter design, Kawamura (1999) recommended L/d (where L is media depth and d is the inner diameter of the column).

Huck and Sozański (2008) stated that for biological filtration, L/d was directly proportional to the product of θ and α, and in turn, to the dimensionless contact time, X*. They compared similarity of using X* for biofiltration with using L/d for particles removal for filter design and operation.

Although the relationship between AOC removal and X* were established by Zhang and Huck (1996a), Huck (1999) and Huck and Sozański (2008), only with a small-scale application on real data by Huck (1999), no comprehensive investigation of X* has been done for research or engineering practice.

The goal of this research is to apply X* to interpret biofiltration results reported in the literature, to assess the applicability of this parameter for biofiltration practice. Searching previous literature/technical reports and recording various BOM (such as AOC, BDOC and secondary utilization) removal data and their relevant parameters will be the first step.

Chapter 3

X* Analysis for AOC Removal by Biofiltration

Biological filtration has been used in drinking water treatment for multiple objectives in recent years, which include removing BOM parameters such AOC and BDOC to produce more biologically stable water. As discussed in the literature review chapter, Zhang (1996) and Zhang and Huck (1996a) developed a new parameter, X* for biofiltration performance. A simplified X* expression (Huck and Sozański, 2008) is essentially determined by physical EBCT, specific surface area of attached biofilm (or filter media), biofilm density and biodegradation kinetics. Using the concept and expression of X*, the objective of this chapter is to analyze AOC removal by biofiltration from the collected literature, examine how the above mentioned factors affect biofiltration performance and X* values as well as how X* values (or approximated values) estimated from biofilter configurations fit the model-based theoretical AOC removal-X* relationship. Analysis results and discussion are presented here. Calculation results are presented in Appendix A.

3.1 Data Collection

The first step for analysis was collecting filter media, operation and other information for AOC removal cases from journals and conference proceedings. The collected cases are summarized in Table 3-1. Published or presented years for these studies are from 1992 to current. Appendix A-1 lists detailed reference information.

Table 3-1 Summary of the collected cases for AOC removal by biofiltration processes

Source/Scale[(1)]	Media Type	Pre-treatment and Operation Conditions	AOC Removal (%)
Ahmad, et al., 1998 (P)	Dual-Anthracite/Sand	Ozonation Different backwashing strategies	From 21 to 80

Chien et al., 2007 (P)*	GAC	Ozonation	60
Chien et al., 2008 (B)	GAC	Ozonation	86, 17
Daniel and Teefy, 1995 (P)	Dual-GAC/Sand; Dual-Anthracite/Sand	Ozonation	70
Heinicke et al., 2006 (P)	GAC	No-ozonation, receiving raw water	19
Hijnen and van der Kooij, 1992 (B)	Sand	Simulated influent water with acetate	72, 76
Hu et al., 1999 (B)	Bio-ceramic	Highly-polluted, receiving raw water	44
Hu et al., 2005 (P)	Zeolite	Chlorination, aerated	4, 45, 53
Huck et al., 1991(P)	GAC	Ozonation Ozonation and Sand	From 33 to 88
Krasner et al., 1993 (P)	GAC	Ozonation, various dosage	72, 74
Lai et al., 2002 (P)*	GAC	Two stage ozonation and one filter ahead	76, 88
LeChevallier et al., 1992a,b (P)	Mixed-Anthracite/Sand/Garnet; GAC; Dual-GAC/ Sand	No ozonation; Ozonation, various dosage; Chlorination; Chloramination	From 75 to 91
Moll et al., 1999 (B)	Sand	Special treatment for source water Ozonation Upflow mode	44, 55, 57
Najm et al., 2005 (F)	Dual-GAC/Sand	Ozonation, various dosage	75
Persson et al., 2006 (P)	GAC; Expanded clay	No ozonation, receiving raw water	22, 35, 41
Shu et al., 2008 (P)	GAC	Ozonation	35
Vahala et al., 1998a (P)	GAC	Ozonation, various dosage, nutrient addition	From 27 to 79
Vahala et al., 1998b (P)	GAC	Ozonation, dosage NA	73, 76
van der Aa et al., 2003 (P)	GAC	Ozonation, various dosage	55, 64, 71
van der Hoek et al., 2000 (P)	GAC	Ozonation	64, 75,76, 80
Wang, et al., 1995 (P)	Sand; Dual-GAC/Sand; Dual-Anthracite/Sand	Ozonation	From 37 to 51
Wang and Summers, 1996(B)	Sand	Special treatment for source water Ozonation	From 59 to 76

Wert et al., 2008 (P)*	Dual-Sand/Anthracite	Ozonation, dosage NA, pre-chlorination	60
Zappia et al., 2007 (P)	Dual-Sand/Anthracite; Dual-GAC/Sand	Enhanced coagulation; MIEX clarifier; Chlorination; Different backwashing	From 42 to 79
Zhang et al., 2004 (P)	Dual-GAC/Sand	No ozonation	84

Note: (1) B-Bench; P-Pilot; F-Full scale;

*Contacted author for information.

Media types include GAC, sand, anthracite, expanded clay (EC), zeolite, garnet, bio-ceramic as well as dual-media or multi-media, such as GAC-sand, anthracite-sand and Anthracite-Sand-Garnet. GAC was the most used biofilter medium for the collected cases (only exhausted GAC biofilters were chosen for the analysis). Sand and anthracite were also commonly used for biofilters. Dual media were used in a significant number of biofiltration processes. Particle sizes of filter media for the collected biofiltration cases are summarized and calculated in Section 3.1.1 and 3.2 respectively.

Appendix A-3 lists EBCT values and AOC percentage removals. Recorded EBCT values range from 0.4 minutes to 60 minutes. AOC removal efficiencies range from poor removal of only 4% to nearly complete removal of 91%. Detailed summaries and discussions are presented in Section 3.1.2, 3.1.3 and 3.1.4.

Water temperature information is listed in Table 3-7 and Appendix A-7, with a lowest temperature of 3.5°C and a highest temperature of 35°C. The detailed information for temperature and its effect on biofiltration and X* are presented in Section 3.5.

Pre-ozonation information is listed in Table 3-12. Appendix A-9 lists calculation results of carbon (TOC)-normalized ozone dosages for the chosen AOC removal cases by biofiltration, which ranges from 0 (without ozonation) to 1.30 mg O_3/mg TOC.

Table 3-1 also summarizes pre-treatment and operation conditions for the collected cases with conventional processes considered as default pre-treatment prior to the application of biofiltration.

The most applied pre-treatment process was single-stage or multi-stage ozonation, for it is able to convert refractory organic matter into easily biodegradable matter and facilitate biodegradation processes in biofilters, as discussed in Chapter 2. Ozonation dosage varied greatly and was expressed in different ways for the collected cases. Chlorination, chloramination, enhanced coagulation and aeration were also used as pre-treatment processes. For some cases, the biofilter directly received raw water without pre-treatment; or received water from conventional processes (default) without ozonation. The detailed information for pre-ozonation and its effect on biofiltration and X* is presented in Section 3.6.

The wide ranges of particle sizes, EBCTs, temperatures and high diversity of pre-treatment and operation conditions summarized above are able to well represent biofiltration practices for AOC removal.

3.1.1 Collecting and Determining Representative Sizes of Filter Media

Representative sizes of filter media need to be determined or calculated for the following calculations of specific surface area of media particles. Information extracted directly from the literature, including effective sizes (with uniformity coefficients), median or mean sizes; and media information from outside commercial website source are the three main sources to determine representative sizes of filter media particles in this study. Another approach, communicating with authors, was also used to obtain media particle information. Below are descriptions of these four approaches.

(1) Effective size (ES) and Uniformity Coefficient (UC)

Distribution of particle sizes is known as “gradation”. Particle-size distribution curves are often used in North America to describe significant size characteristics of the particles by plotting data obtained by sieving. The standard procedure for conducting a sieve analysis of a filter medium is detailed in ASTM Standard Test C136-06 (ASTM International, 2003). Figure 3-1 illustrates a typical distribution curve of a sample of commercially available anthracite in Canada.

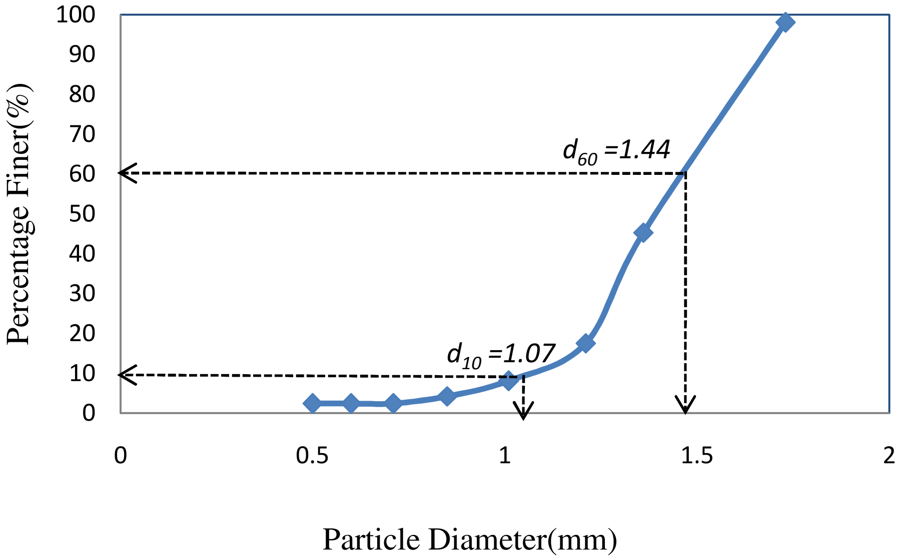

Figure 3-1 A typical filter media particle-size distribution curve
(Data provided by Anthrafilter Media & Coal Ltd., October 20th, 1992)

Effective Size (ES), also refers to as d_{10}, is particle diameter at which 10% weight of the media particle is finer (90% is coarser than the ES); d_{60} is particle diameter at which 60% weight of the media particle is finer. Uniformity Coefficient (UC) is a measure of size range of the media and defined as ratio between d_{10} and d_{60} ((Equation (3-1)). For the anthracite particle distribution in Figure 3-1, UC was 1.44/1.07=1.35. If a range of a parameter was provided in the literature, median value is adopted; if a UC parameter range was given by an inequality, limiting maximum or minimum value is then used in the calculations.

$$UC = \frac{d_{60}}{d_{10}} \qquad (3-1)$$

For filter media in the collected literature, the UC values are normally less than 2.0, except in Lai et al. (2002) the UC value was 2.3.

For slow sand filters, ES of 0.15 to 0.30 mm (3 < UC < 5) and ES of 0.30 to 0.45 mm (UC< 2.5) were recommended by different guidelines and summarized by Baruth (2005); ES of 0.15 to 0.40

mm (1.5 <UC< 3.6) was also cited in Letterman (1999). For rapid-sand filters, ES of 0.35 to 0.60 mm (1.3 < UC < 1.8) was recommended (Baruth, 2005). Anthracite beds have been used in the same basic configuration as rapid sand beds, however anthracite is more angular than sand and parameters such as bed porosity and specific gravity are different, and anthracite does not perform exactly in the same manner as sand of equivalent size consequently in terms of percentage removal efficiencies (Prévost et al., 2005).

Typical UC for GAC produced in the United States is less than or equal to 1.9. The maximum UC recommended by AWWA is 2.1 (Baruth, 2005), while higher UCs were reported for some dual-media retrofitted filters (≤2.4). Typically ES of 0.5 to 0.65 mm (Baruth, 2005) and 0.55 to 0.65 mm (Letterman, 1999) for GAC were summarized from practices of conventional filters and advanced filters to remove taste and odor.

Zhang and Huck (1996a) stated that d_{60} could be used as particle representative size to calculate thickness of effective liquid diffusion layer, while they suggested that the average of d_{10} and d_{60}, instead of d_{10} or d_{60} itself, should be used to calculate specific surface area (Equation 3-2).

$$\bar{d} = \frac{d_{60} + d_{10}}{2} = \frac{UC \times d_{10} + d_{10}}{2} \qquad (3-2)$$

It should be noted here that the grain size distribution curve is not linear. Theoretically neither any of d_{50}, d_{60}, d_{10} nor the average of d_{10} and d_{60} is able to represent particle size of a whole community in filters. The average of d_{60} and d_{10}, was used in this thesis as a compromise of the non-linear distribution of filter particles.

Langenbach et al. (2009) used Equation (3-3) to calculate specific surface area (Equation (3-4), M^2/M^3 or M^{-1}) of media particles in slow sand filter because sieve analysis plots for most filter materials behave in a linear manner on Log probability papers.

$$\bar{d} = d_{10}(1 + 2\log UC) \qquad (3-3)$$

$$Specific\ surface\ area = \frac{Total\ grain\ surface\ area}{Grain\ volume} = \frac{\pi\,\overline{(d)}^2}{\frac{\pi}{6}\frac{1}{\overline{(d)}^3}} = \frac{6}{\overline{d}} \qquad (3-4)$$

Comparison of calculations using Equation (2) and (3) above, which was not presented in this study, showed for some cases results by Equation (3-2) were greater than those by Equation (3-3); while for other cases, results by Equation (3-2) were smaller than those by Equation (3-3).

(2) Mean or median size

For some collected cases, “median diameter” or “average diameter” was provided when size distribution information was not available. Although methods of how to determine “median diameter” or “average diameter” were not provided, these diameter values were used as representative diameter $(\overline{d})$ in the following calculations.

(3) Commercially available media

For a few cases when particle size was not directly provided, while media types (commercially available) were given in footnotes or sections of material introduction in the literature. Therefore, particle information can be obtained from manufacturers or from studies in which the specific types of media were applied. Table 3-2 summarizes particle size data for typical types of commercially available GAC media manufactured by Calgon Corporation (US), PICAUSA and Norit, which were applied in the collected biofiltration practices. Determination of representative diameter $(\overline{d})$ followed the method described in the first approach above.

Table 3-2 GAC particle Information for commercially available
Fitrasorb[1] (Calgon Company), PICABiol[2] (PICAUSA) and Norit ROW[3] (NORIT)

Model	F100	F200	F300	F400	F820	Norit ROW 0.8S	PICABiol
Effective size	0.85	0.63	0.95	0.64	1.0	1.8	1.0-1.1

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[1] http://www.calgoncarbon.com/carbon_products/water.html (2009-08-06);

[2] Baudin B., 2009. Advanced training in drinking water treatment processes, EuroAquae Coursenote;

[3] Laurent P., Kihn A., Andersson A. and Servais P. 2003. Impact of backwashing on nitrification in the biological activated carbon filters used in drinking water treatment. *Environmental Technology*, 24: 277-287.

Uniformity Coefficient	≤1.9	≤1.7	≤2.3	≤1.6	≤1.5	≤1.4	≤1.7

(4) Other approaches

When the literature did not provide any information or only provided incomplete information for media particle size, communication with authors was another way to obtain data. If particle size information was incomplete, it could be found in other reference which reported the same study (such as the corresponding thesis). For these cases, footnotes were made in Appendix A-2.

If there was no way to collect the necessary particle size information, the study was not included in this thesis then. In this thesis, the author did not assume any media particle size or any related parameters because particle size directly determines specific surface area, which is a crucial parameter for the following calculations.

Appendix A-2 lists biofilter media particle sizes for the collected AOC removal cases.

3.1.2 Collecting and Determining Empty Bed Contact Time

EBCT values have two sources: directly given in the literature and calculated by Equation (3-5).

$$EBCT = \frac{h}{HLR} = \frac{h}{Q/A} \qquad (3-5)$$

where HLR refers to as hydraulic loading rate (LT^{-1}), h is filter depth (L), Q is filter influent flow rate (L^3T^{-1}), and A is cross-section area of the filter (L^2), determined by inner diameter of the filter.

For multiple-media filters, such as GAC-sand and GAC-anthracite, even when overall EBCT for the whole filter was given in the literature, the EBCT for individual layer was calculated (Equation 3-6), because a later section involves multiplication EBCT of each layer by the specific surface area of particles in the layer.

$$EBCT_i = \frac{h_i}{h} \times EBCT_{Overall} \qquad (3-6)$$

where $EBCT_i$ is the EBCT of the i layer, h_i is the depth of the corresponding layer.

Appendix A-3 lists EBCT values for the collected AOC removal cases. Ways of obtaining these values are also indicated.

3.1.3 Determining AOC Removals

Only studies under "steady-state condition" and studies (or studying periods) of using non-adsorptive filter or exhausted adsorptive filter media were included, since biodegradation is considered the most important removal mechanism in biofilters. Because ozonation and biofiltration were used together for a large percent of the studies investigated, "removal percentage" was reported sometimes using the AOC concentration difference between the ozone contactor's influent and the biofilter's effluent, divided by the AOC concentration in the ozone contactor's influent; while it should be clarified here that the "removal" in this thesis only refers to the percentage expression of the AOC concentration difference between biofilter's influent and effluent dividing by the AOC concentration in biofilter's influent (Equation (3-7)).

$$AOC\ Removal\ Percentage = \frac{AOC_{Filter\ Influent} - AOC_{Filter\ Effluent}}{AOC_{Filter\ Influent}} \qquad (3-7)$$

where AOC value refers to the sum of AOC-P17 and AOC-NOX, if they were expressed separately.

If a value was given by an inequality, the limiting maximum or minimum value is then used in the later calculations and plotting. When AOC removal percentages were not directly given, the biofilter's influent and effluent AOC concentrations were collected or measured on the article's figures by a ruler and used for calculation by Equation (3-7). Appendix A-3 lists AOC removal percentages for the collected AOC removal cases. Ways of obtaining these values are also indicated.

3.1.4 Relationship between EBCT and AOC Removal Efficiencies

Zhang (1996) and Zhang and Huck (1996a) summarized that for a specific water and biofilter and within a certain range, increasing EBCT may increase BOM removal efficiency, while it is in a less than proportional way and removal efficiency will not improve after a certain range of EBCT when AOC removal efficiency reaches a plateau.

The distribution of AOC removal percentages vs. EBCT (Figure 3-2) for the collected cases in this study shows EBCT values range from 0.4 minutes to 60 minutes with most less than 40 minutes and AOC removal percentages range from 4% to 91% with most greater than 20%. It is demonstrated clearly that in general biofiltration is an effective process to reduce AOC levels from water and be able to produce more bio-stable water.

The distribution of AOC removal percentages over EBCTs was scattered. No significant co-relationship between EBCT and AOC removal percentage could be found in Figure 3-2. It is obvious that EBCT alone is not able to describe biofiltration performance. Considering EBCT only is not adequate to guide biofiltration design and operation. This is important for drinking water treatment practice, since contact time is normally used as a main design parameter for conventional filters and biological filters. A later section will show dimensionless contact time (X*) is a better indicator than physical contact time to predict and evaluate AOC removal by biofiltration quantitatively and make comparisons among studies because it incorporates multiple factors, such as EBCT, filter media and biodegradation kinetics, into one parameter.

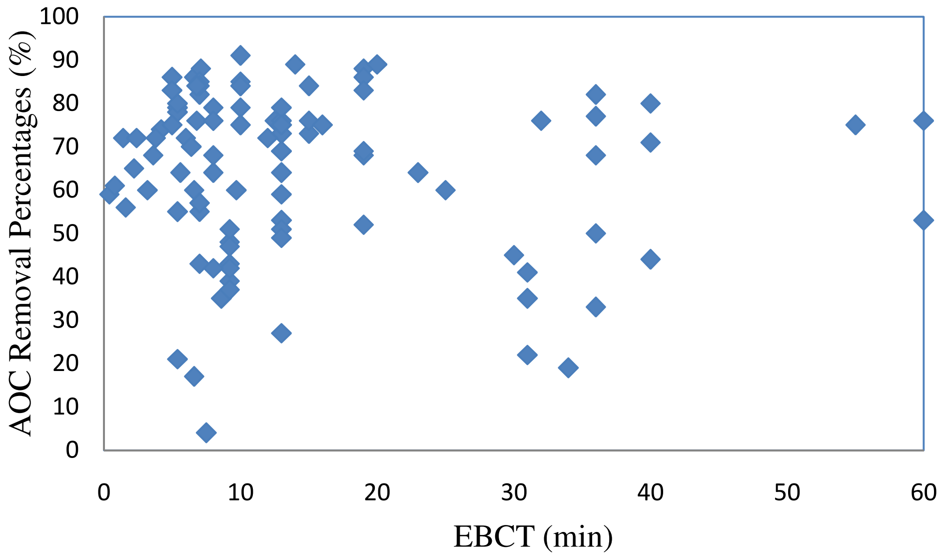

Figure 3-2 EBCTs distribution over AOC removal percentages

A normalized removal percentage by EBCT can be used to compare biofiltration operations for different filters or for different BOM parameters and can be considered as an approximate indicator of biofiltration treatment efficiencies (Equation (3-8)), if zero order reaction is assumed here for BOM removal by biofiltration processes. The author therefore introduces this approach. As far as he is aware, it has not been used elsewhere. (It is known that removals are not zero order, however Equation (3-8) is useful as an initial rough approximation for screening purposes.)

$$EBCT\ Normalized\ Percentage\ Removal\ = \frac{AOC\ Removal\ Percentage}{EBCT} \qquad (3-8)$$

Appendix A-3 presents EBCT-normalized percentage removal rates (T^{-1}), most of which are smaller than 0.2 min^{-1}. The two lowest and two highest EBCT-normalized removals, together with literature source, EBCTs and AOC removal percentages, are listed in Table 3-3.

Table 3-3 Two most inefficient and two most efficient biofiltration processes (Based on EBCT)

	Source	EBCT (min)	AOC Removal (%)	EBCT Normalized Percentage Removal (min^{-1})
Most inefficient	Persson et al. 2006	31	22	0.70
	Heinicke et al., 2006	34	19	0.56

Most efficient	Wang and Summers, 1996	0.4 0.8	59 61	100 80
	Krasner et al., 1993	1.4	72	50

The common point for the two most inefficient biofiltration processes here is that both processes received raw water and there was no ozonation as pre-treatment process to improve poor biodegradability of the raw water. This indicates that if water treatment plants want to improve AOC removal, essential pre-treatment, especially pre-ozonation is a feasible approach.

Raw water was specially treated in Wang and Summers (1996) before the ozonation/biofiltration processes by anion exchange resin, GAC adsorption, reverse osmosis and secondary GAC adsorption. The influent water was free of large molecular organics and particles, therefore AOC removal efficiencies were very high. In Krasner et al. (1993), the non-linear relationship between incubation time and measurement results for BOM (AOC and BDOC) measurement, as well as various sensitivities of different BOM components to ozonation may be the reason for high AOC removal efficiencies, because the biofilter influent contained high concentrations of glyoxal, methyl-glyoxal and formaldehyde. The detailed discussion in this regard will be presented in Section 4.7, Chapter 4. The AOC removal efficiencies reported in these two studies are significantly higher than those reported elsewhere and even the authors for the latter literature realized the fact themselves.

3.2 Filter Media Size and Actively Specific Surface Area

Appendix A-4 presents calculation results of actively specific surface area for filter media.

By adoption of the calculation method in Zhang and Huck (1996a), actively specific surface area of filter media was calculated by Equation (3-9):

$$\alpha = \frac{6 \times \beta \times \varphi \times (1 - \epsilon)}{\bar{d}} \qquad (3-9)$$

where α is actively specific surface area of media particle, β is biofilm coverage on particle surface, ε is filter bed porosity, φ is shape correction factor, $\bar{d}$ is representative size of filter media parameters determined in section 3.1.1.

Because the media particles are not necessarily fully covered by biofilm, β was assigned 90% as Zhang and Huck (1996a) suggested.

Shape and roundness of filter media are important to determine actual surface area of particles. Sphericity ψ of a grain quantifies surface area ratio of an equal volume sphere to the grain. Typical ψ values of 0.7-0.8, 0.46-0.60 and 0.75 for sand, anthracite and GAC respectively were cited in Letterman (1999). The relationship between sphericity ψ and shape correction factor φ in this study is as Equation (3-10):

$$\varphi = \frac{1}{\psi} \qquad (3-10)$$

Roberts et al. (1985) found that for irregular GAC particles of 300 mm diameter, the measured external mass transfer rate was 1.44-2.04 times greater than the rate when assuming an ideal spherical shape. Zhang and Huck (1996a) then suggested assigning 1.5 to the correction factor φ. This φ value corresponds to a sphericity (ψ) of 0.67 and falls into or near the typical ψ ranges of filter media mentioned above, thus was adopted in this study.

Porosity ε was also assigned to be 0.425 as Zhang and Huck (1996a) suggested, except that Persson et al. (2006) provided different porosities (38% for GAC, 49% for fine expanded clay and 45% for coarse expanded clay) in their study. This assignment of 0.425 falls into or near porosity ranges summarized in Letterman (1999), which are 0.42-0.47 for sand, 0.56-0.60 for anthracite and 0.50 for GAC respectively.

It is necessary to mention that although these correction coefficients in Equation (3-8) were applied in calculations in attempts to approximate "real conditions" on the one hand, as a matter of fact on the other hand, their applications in calculation affected all the data equally in a linear way. These values therefore do not affect the comparisons made, although they may distort the "real conditions" in some cases. For example, 90% was assigned to correct biofilm coverage uniformly for all cases, while actual biofilm coverage may differ case by case, especially for different types of media particles and for different filter depths.

The larger the representative size, the smaller the specific surface area for ideal spherical shape and actively specific surface area after the correction procedures mentioned above. Table 3-4 summaries ranges of particle sizes.

Table 3-4 Summaries of media particle sizes

Media	Representative Size(mm)	Actively Specific Surface Area (m^{-1})
GAC	0.55-1.60	8470-2910
Sand	0.55-1.60	8470-2910
Anthracite	1.18-1.62	3950-2880
Expanded Clay	1.19, 3.25	3470, 1120
Zeolite	1.50	2330
Bio-ceramic	3.50	1340
Garnet	0.48	9750

3.3 Expected X* Values

As summarized in the literature review, Huck (1999) and Huck and Sozański (2008) developed a table (Table 3-5) and a Figure (Figure 3-3) respectively for the AOC removal-X* relationship, based on the modeling work by Zhang and Huck (1996a).

Table 3-5 Approximate percentage removals as a function of X* (After Huck, 1999)

X*	Approximate AOC percentage Removal
0.25	33
0.5	47
1.0	66
1.5	80
2.0	85
3.0	95

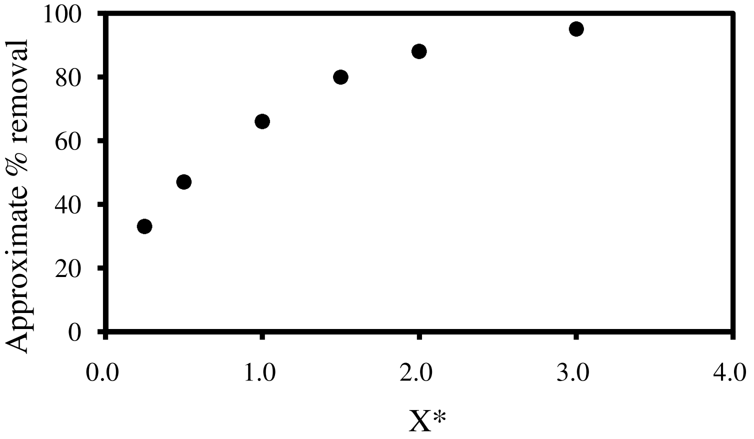

Figure 3-3 Impact of X* on percentage removal of substrate (After Huck and Sozański, 2008)

Using a ruler to measure the corresponding X* value of each AOC removal percentage (Appendix A-4) on an eight times enlarged Figure 3-3, the obtained X* is defined as "Expected X*" and listed in Appendix A-6. AOC removal efficiencies from the collected biofiltration applications range from barely removal of 4% to nearly complete removal of 91%; expected X* values range from 0.1 to 2.5 accordingly. Figure 3-4 presents the relationship between expected X* values and AOC removal percentages.

The purpose of this study is to compare expected X* values, which were derived from analytical solution of a biofilm model, with X* values estimated from real process data and to examine factors affecting X* values. In this way, a better understanding of X* and optimization of biofiltration design and operation can be obtained.

It would not be a surprise that Figure 3-4 follows the same trend as that of Figure 3-3 or Table 3-5. For a particular AOC removal goal by biofiltration process, expected X* value is considered as the minimum necessary X* value. For a specific X* value, it is associated with a basic process configuration or configuration combination, which can be "expected" to achieve a specific AOC removal efficiency under "standard" conditions discussed later in this study.

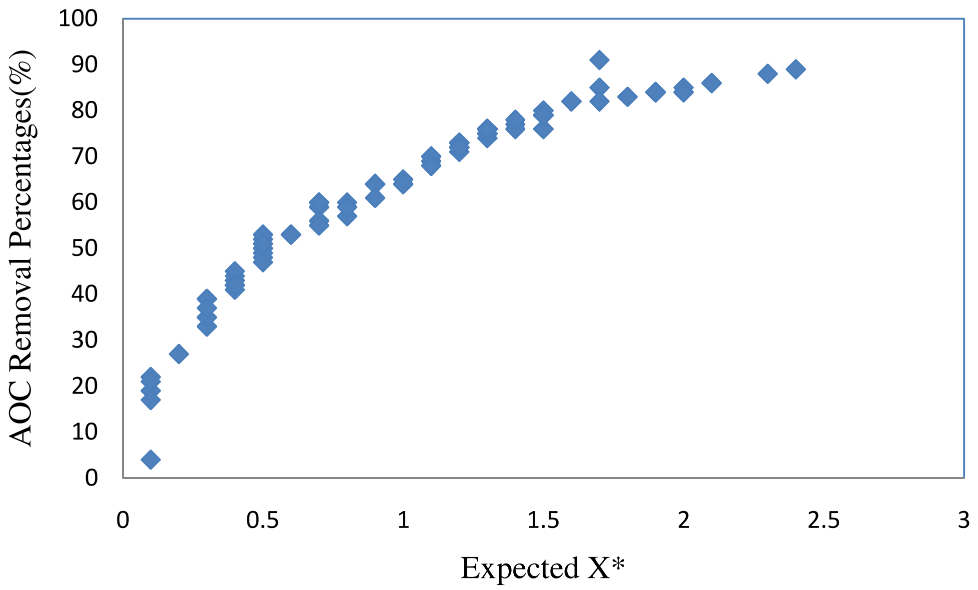

Figure 3-4 Distribution of expected X* values

3.4 *θα* Parameter

Recalling the simplified X* expression (Huck and Sozański, 2008) in the literature review, which was obtained by assuming first-order reaction and that mass transfer is not important:

$$X^* = \theta\alpha K(kX_f)^{1/2} \qquad (2-8)$$

where *K* is a constant.

Various factors such as raw water characteristics, temperature and pre-oxidation affect the biodegradation kinetic parameter *k*; X_f highly depends on factors such as media type, temperature and hydraulic conditions. In practice, they can be obtained by experiments.

If *k* and X_f are ignored here temporarily (effects of them on the X* value will be examined in later sections), one can draw a conclusion from Equation (2-8) that X* for AOC removal is essentially proportional to the product of EBCT (θ) and actively specific surface area (α). Appendix A-5 lists EBCTs, actively specific surface area and products of these two parameters (designated ($\theta\alpha$)').

Ignoring *k* temporarily assumes that all AOC has the same biodegradation kinetics, and that all literature results were obtained at approximately the same temperature (which is not the case, as will be seen). Ignoring X_f temporarily assumes that the biomass density does not vary greatly among filters. The fact that both *k* and X_f appear in Equation (2-8) to the 0.5 power lessens the effect of these assumptions.

For dual-media or multi-media filter, calculation of overall products of EBCT and actively specific surface area *((θα)')* is as Equation (3-11), which adds up each layer's *(θα)'* value. Although study has showed that most biodegradation occurred in the top of filter beds and biomass decreased sharply along biofilter depth (Wang et al., 1995), it is logical in the present study to recognize contribution of all layers of media in dual-layer or multi-layer filter beds to AOC removal and X* values because measurement or calculation of contact time is for the whole filter depth.

$$(\theta\alpha)'_{overall} = \sum(\theta\alpha)'_{each\ layer} \qquad (3-11)$$

The aim of this section is to initially estimate a parameter only based only on considerations of contact time and surface area of media particle; while X*, based on modeling work by Zhang and Huck (1996a) and detailed discussion by Huck and Sozański (2008), incorporates not only contact time and particle size, but also biodegradation kinetics and biofilm density even for the simplified X* expression.

Therefore a new parameter *(θα* parameter*)* is used here to refer to the partially estimated X* values without taking into account factors other than *θ* and *α*. The *θα* parameter, as estimated in this section, is proportional to X*, assuming no change in biodegradation kinetics parameter *k* and biofilm density X_f. The *θα* parameter will undergo adjustment in later sections based on other biofilter configuration parameters, then actual X* values can be estimated.

To obtain *θα* parameter values, one set of AOC removal data (Table 3-6) was chosen as the calculation base (the reason will be specified below, this is also the only data set that meets the requirement for a calculation base). Dual-media (20 inches GAC and 10 inches sand) were used in LeChevallier et al. (1992a, b) with actively specific surface areas of 3790 m^{-1} and 7170 m^{-1} respectively. AOC influent, effluent, removal percentages and EBCTs were provided in the literature.

Table 3-6 shows that when EBCT increased from 5 to 20 minutes, removal percentages hardly increased. Therefore, it is reasonable to deduce that when EBCT was greater than 5 minutes, the corresponding X* value reached the curve plateau in Figure 3-3; in other words, the X* value at an EBCT of 5 minutes was the minimal necessary X* value to obtain an AOC removal percentage of around 83%, which means it is equal to the expected X* values associated with 83% AOC removals.

Table 3-6 Calculation bases for $\theta\alpha$ parameter calculation (Data source: LeChevallier et al. (1992a, b))

EBCT	Removal Percentage
5	83
10	84
15	84
20	89

In LeChevallier et al. (1992a, b), when EBCT was 5 minutes and AOC removal percentage was 83%, the expected X* value was 1.8, the corresponding *$(\theta\alpha)'$* value was 2.5 (Appendix A-5). Therefore based on this ratio (1.8/2.5), the $\theta\alpha$ parameter values for other biofiltration configurations are then able to be calculated (Equation (3-12)) and listed in Appendix A-5. The $\theta\alpha$ parameter is partially estimated X* and dimensionless, and can be plotted on the same horizontal axis as X*, facilitating later discussions.

$$(\theta\alpha) = (\theta\alpha)'\frac{1.8}{2.5} \qquad (3-12)^4$$

3.5 Effect of Temperature on Biofiltration and $\theta\alpha$ Values

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[4] It should be noted that *$(\theta\alpha)'$* is multiplication product of parameter θ and parameter α ($(\theta\alpha)' = \theta \times \alpha$), while $\theta\alpha$ parameter is partially estimated X*.

3.5.1 Collecting Temperature-Related Information

As summarized in the literature review, temperature is one of the most important factors affecting performance of biofiltration processes since many drinking water plants experience significantly seasonal temperature variations. This variability in source water temperature and ambient temperature may impact microbial ecology, degradation kinetics and mass transfer rates and in turn impact removal of BOM (such as AOC) by biofilters.

Table 3-7 records temperature-related information for the collected AOC removal cases. For some cases, specific temperature values were directly given; for other cases, average/median temperatures, ranges or limits of temperatures were given. If different sets of data in the studies had different temperatures or temperature ranges in a same study, specific temperature values are all listed on the table.

For AOC removal studies in which temperature information was not available directly, the seasons in which investigations were conducted (alone or with temperature ranges) are recorded for these cases. Since the start-up and acclimation period for biofiltration is normally long and may occur in different seasons, the conducted locations are also pulled out. With the assistance of season and/or location information, one can have an approximate idea of the water temperature ranges which were not given in the literature. That groundwater was used in the treatment facility in Zappia et al. (2007) is also indicated in Table 3-7, since the temperature of groundwater is lower than that of surface water in warm seasons. AOC removal percentages are also listed in Table 3-7.

Table 3-7 Temperature related information for the collected AOC removal cases

Source	Temperature (°C) and Related Information	Conducted Location	AOC Removal Range (%)
Ahmad, et al., 1998	NA	Alberta, Canada	From 21 to 80
Chien et al., 2007 *	From 25 to 30	Southern Taiwan	60
Chien et al., 2008	From 25 to 30	Southern Taiwan	86, 17

Daniel and Teefy, 1995	Summer	California, USA	70
Heinicke et al., 2006	NA	Sweden	19
Hijnen and van der Kooij, 1992	From 10 to 18	Netherlands	72, 76
Hu et al., 1999	Spring season	Northern China	44
Hu et al., 2005	From 25 to 30	Singapore	4, 45, 53
Huck et al., 1991**	Summer (From 4 to 25) Fall-Winter (From 7 to 14) Spring (From 6 to 9)	Alberta, Canada	From 33 to 88
Krasner et al., 1993	From 15 to 25 May-September	California, USA	72, 74
Lai et al., 2002	From 19.8 to 30.4 (average 25.4)	Southern Taiwan	76, 88
LeChevallier et al., 1992a,b	3.5, 10.5, 4.7, 23.7,19.4 and NA	New Jersey, USA	From 75 to 91
Moll et al., 1999	5, 20, 35	Ohio, USA	44, 55, 57
Najm et al., 2005	Lowest of 11 in winter Highest of 28 in summer	North Carolina, USA	75
Persson et al., 2006	From 1.5 to 20.2 (median 7.2)	Sweden	22, 35, 41
Shu et al., 2008	NA	Southern China	35
Vahala et al., 1998a	From 4.0 to 13.6	Finland	From 27 to 79
Vahala et al., 1998b	From 8.2 to 9.6	Finland	73, 76
van der Aa et al., 2003	NA	Netherlands	55, 64, 71
van der Hoek et al., 2000	NA	Netherlands	64, 75,76, 80
Wang, et al., 1995	From 13 to 27	Ohio, USA	From 37 to 51
Wang and Summers, 1996	NA	Ohio, USA	From 59 to 76
Wert et al., 2008	From around 15 to 19	Nevada, USA	60
Zappia et al., 2007	Groundwater March	Australia	From 42 to 79
Zhang et al., 2004	8	Central China	84

Note: * Communication with authors;
** Raw water temperature figure provided, and temperature ranges were measured from the figure.

3.5.2 Plotting $\theta\alpha$ Values for Temperature-Favored Biofiltration Processes

The model-based relationship of AOC removal percentages vs. X* (Table 3-5 and Figure 3-3) are based on biofiltration operation at around room temperature (20°C), while a large percentage of the collected data in this chapter are extracted from biofiltration studies with influent water temperature below 15°C. Low temperature slows down biodegradation process, and the kinetic parameter k in Equation (3-10) is then smaller. It is obvious from Equation (3-10) that if k is lower, product of θ and α $((\theta\alpha)')$ must be larger to achieve the same X* (i.e. percentage removal), assuming all other factors remain the same. (For a given filter, this effectively means that contact time must be longer in the winter to achieve a given percentage removal.)

The aim of this section is to screen (in Appendix A-6) values of $\theta\alpha$ parameter which are associated with influent water temperature of around 20°C, plot these $\theta\alpha$ parameter vs. corresponding AOC removal percentages to explore if data from these biofiltration processes match the standard AOC removal-X* relationship. Screening criteria are presented below. Screening results are listed in Table 3-8.

Although high temperature, which refers to temperatures higher than 20°C and lower than 35°C in this thesis, may facilitate biodegradation processes in the biofilter, the stimulatory effect of high temperature on BOM removal by biofiltration is limited and not a concern for drinking water treatment, therefore data associated with water temperatures above 20°C are included and all treated equally (i.e. if they had been collected at 20°C).

On the other hand, the effect of low temperature on biofilter performance is of concern; the lower the temperature, the more impact it has on microbial ecology and treatment performance of biofilter (Moll et al., 1999). In this section, data associated with water temperature below 20°C but above 15°C are still chosen for analysis because it is assumed that temperature in this range would just cause a minor impairment of performance of the biofilter and by including AOC removal data associated with this temperature range, the size of the analyzed data pool is larger. Data associated with water temperatures below 15°C are not chosen and adjustment will be performed for these data in a later section.

For Wang et al. (1995), AOC measurement was only expressed as AOC-NOX, thus the data were not chosen for analysis, although it was determined that the water temperature fell into the range of greater than 15°C. Data from Hu et al. (2005) were not chosen because the biofiltration application for AOC removal was not for drinking water treatment, but for a water reclamation system. Biofiltration processes which meet the above- mentioned criteria are defined as "temperature-favored biofiltration" in this thesis.

It should be noted here that although X* and $\theta\alpha$ parameter were used as variables simultaneously on the same axes in Figures presented above, they are two different parameters. $\theta\alpha$ parameter is a component of X* without considering factors other than θ and α. $\theta\alpha$ parameter is proportional to X* quantitatively and both parameters are dimensionless.

It should also be noted here that if in a given study, such as Wert et al. (2008), the $\theta\alpha$ parameter values are greater than associated expected X*, and AOC removal percentages showed no or very little increase if EBCT increased greatly without changing other biofilter configurations, only data associated with lowest EBCT were chosen.

Table 3-8 Summaries of temperature-favored (i.e., temperature ≥15°C) biofiltration processes and associated $\theta\alpha$ parameter values

Source	**$\theta\alpha$ Parameter Values**	**EBCT (min)**	**AOC Removal (%)**	**Temperature and Conducted Season, Location[1,2]**
Chien et al., 2007	2.7	6.6	60	Southern Taiwan (25-30°C)
Chien et al., 2008	2.7	6.6	86	Southern Taiwan (25-30°C)
	1.5	6.6	17	
Daniel and Teefy, 1995	1.7	6.4	70	Summer, California, USA
	1.7	6.4	70	
Lai et al., 2002	1.5	7.1	88	Southern Taiwan (19.8-30.4°C, average 25.4°C)
	2.7	12.5	76	
LeChevallier et al., 1992a,b	3.5	10	75	New Jersey, USA (>15°C)
	3.5	10	79	
	3.5	10	84	
	2.4	7	82	
	2.4	7	84	
	1.8	5	83	
Najm et al., 2005	3.3	10	75	North Carolina, USA

				(Summer, highest of 28°C)
Krasner et al., 1993	0.4	1.4	72	May-September(California, USA) (15-25°C)
	1.1	4.2	74	
Moll et al., 1999	5.0	7.0	55	20
	5.0	7.0	57	35
Wang and Summers, 1996	0.3	0.4	59	Ohio, USA 13-27°C(3)
	0.5	0.8	61	
	1.0	1.6	56	
	1.4	2.2	65	
	2.3	3.6	68	
	2.4	3.8	72	
	3.5	5.6	64	
	4.3	6.8	76	
Wert et al., 2008	1.0	3.2	60	Nevada, USA (From around 15 to 19°C)

Note: (1). Refer to section 3.5.2 for screening criteria;
(2). Refer to Appendix A-7 for weather references;
(3). Determination of temperature range followed Wang et al. (1995).

Figure 3-5 presents the distribution of $\theta\alpha$ parameter associated with temperature-favored biofiltration. Also shown is the expected relationship of percentage removal versus $\theta\alpha$ parameter in subsequent discussion, the comparisons that can be made between the values in the figure and the expected relationship are based on visual assessment. A quantitative involving the relationship to the AOC- $\theta\alpha$ parameter points extracted from the literature was not possible within the scope of this thesis for the following reasons: a conventional (linear or non-linear) regression requires both that the equation for the relationship must be known, and that the values on the x-axis are essentially error-free. The equation for the line in Figure 3-5 is not known, because as explained earlier the relationship is estimated from other calculations. Further, the $\theta\alpha$ parameter values are not error-free, and because they are calculated/estimated from various studies, their variances would also be extremely difficult to estimate. This would be required before even an error-in-variable approach could be used.

Given this, as well as the considerable effort required to obtain the information from the literature (this is the first such effort of its kind, to the author's knowledge) and develop it to the point of being able to present in Figure 3-5, comparisons were based on visual assessment. This was considered appropriate for this exploratory research in order to provide an initial answer to the research question as to whether the X* concept could be applied to interpret and to at least some extent synthesize the various site-specific studies reported in the literature. If the relationship generally fits, further quantification would be appropriate in a subsequent investigation. If it does not generally fit even visually, the inference that can be drawn is that the expected relationship itself should be re-evaluated.

In this regard, in subsequent discussion, in this chapter (and in Chapter 4 and 5), a “reasonable or “satisfactory” fit is defined as one in which the expected relationship and the points are in general visual agreement, i.e., there are no systematic discrepancies. Thus with regard to Figure 3-5, it appears that with some exceptions where associated AOC removal efficiencies were much lower than expected, distribution of $\theta\alpha$ parameter values matches the model-based AOC removal-X* relationship reasonably, if $\theta\alpha$ parameter value is smaller than about 2. AOC removal percentage is not shown to increase with the increasing $\theta\alpha$ parameter, if $\theta\alpha$ parameter value is greater than 2.

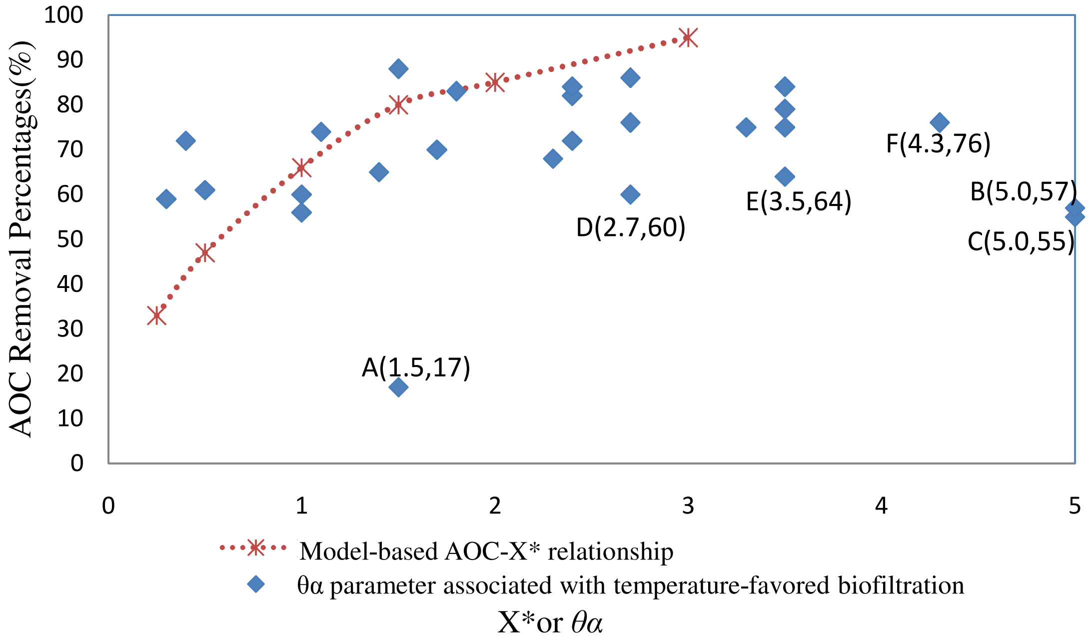

Figure 3-5 Distribution of $\theta\alpha$ associated with temperature-favored (i.e., temperature ≥15°C) biofiltration

Comparing Figure 3-5 and previous figure based on EBCT (Figure 3-2), one would have a clear idea that this comprehensive X* parameter (and its estimated value) is a better indicator than EBCT alone. “Substantially diverging ” points, identified by letters in Figure 3-5, are points far away from the model-based AOC removal-X* relationship and associated with biofiltration processes which achieved much lower removal efficiencies than expected. In each of these cases, the original literature sources were re-examined to see if there might be a reason why removals were lower than expected. (If not, it would suggest that the expected relationship might ultimately need to be reevaluated). Each of these cases is discussed briefly below.

For example, an anthracite biofilter in Chien et al. (2008) only achieved 17% removal with a $\theta\alpha$ parameter value of 1.5 (point A in Figure 3-5, corresponding EBCT of 6.6 minutes); while a GAC biofilter under the same operation conditions achieved 86% removal and the associated $\theta\alpha$ parameter was only 1.8 times larger than that for point A. Increased influent pH as a result of pellet softening in the treatment train might have impaired microbial activity in the biofilters; another reason is that an ozone residual might exist in the influent and damage anthracite media (Urfer, 1998) in this study, since the step of purging ozone residual (such as adding sodium thiosulphate or bubbling with nitrogen) from the influent before BAC was not mentioned in the study.

Only 57% and 55% AOC percentage removals were reported for sand filters with $\theta\alpha$ parameter values of 5.0 (point B and C in Figure 3-5, corresponding EBCT of 7 minutes) at 35°C and 20°C respectively in Moll et al. (1999). The biofilters sand media was first acclimated with settled Ohio River water for one month, and then the feed water switched to prepared NOM solution, which underwent nanofiltration, tap water dilution and GAC filter. That the feedwater was significantly different from the water used for initial acclimation might be one of the reasons for low removal efficiencies in this study. Also, 5-8 weeks of subsequent acclimation for the biofilters receiving prepared NOM solution was a little bit shorter than normally required time to reach steady-state, although there was evidence that biomass at the top of the biofilters no longer increased with respect to time.

It should be pointed out even although one screening criterion was set to discard $\theta\alpha$ parameter datapoints associated with over-design or under-operated (“over-design” and “under-operated” will be discussed in detail in Section 5.2, Chapter 5), it is not possible to exclude all “invisible” potential over-design and under-operated cases from Table 3-8 and Figure 3-5 unless there is a series of available $\theta\alpha$ parameter values (i.e. typically, investigations at different contact time) for a given water. “Over-design” and “under-operated” might be the reason for the other three “substantially diverging” points (points D, E and F).

On the other hand, however, as the model-based AOC removal-X* relationship predicts, when expected X* reaches 2.5, the corresponding AOC removal percentage is expected to reach 90%; while observed AOC removal percentages (Figure 3-5) hardly reached 90% (only one case in the collected AOC removal literature), no matter how great X* ($\theta\alpha$ parameter values in this section for temperature-favored biofiltration processes) were applied for biofilters configuration. Consequently, the distribution of $\theta\alpha$ parameter values associated with temperature-favored biofiltration processes has a lower and flatter tailing than the model-based AOC removal-X* curve.

This suggests that the model-based AOC removal-X* relationship may not be able to evaluate AOC removal efficiencies quantitatively at high X* range of above about 2. SMP (Soluble Microbial Products) may comprise a significant portion of the organic matter in the effluent of a biofilter and may impose a maximum organic matter removal efficiency (Rittmann and McCarty, 2001; Carlson and Amy, 2000). The biofilm model originally developed by Rittmann and McCarty (1982a), as well as solutions for the model equation, did not consider SMP. The appropriateness of applying the concept of S_{min} to be the base for the whole framework of drinking water biofilm modeling has also receive comment (Uhl et al., 2001). (S_{min} is a fundamental concept in the Rittmann and McCarty (1982a) model.) In addition, other factors may also confound X* values and $\theta\alpha$ parameter values.

3.5.3 Effect of Temperature on $\theta\alpha$ Values

Recalling Equation (2-8):

$$X^* = \theta\alpha K(kX_f)^{1/2} \qquad (2-8)$$

where K is a constant.

Equation (2-8) can be rewritten as:

$$X^* = \theta\alpha KK' = (\theta\alpha\, parameter)K_2 \times K' \qquad (3-13)$$

where K_2 is a constant and $K' = (kX_f)^{1/2}$ $(3-14)$

The effect of temperature on biological progress kinetics is usually expressed in the following form (Metcalf and Eddy, 2003):

$$k_T = k_{20}\lambda^{T-20} \qquad (3-15)$$

where k_T is the maximum utilization rate at T°C (T^{-1} or $ML^{-3}T^{-1}$), k_{20} is the maximum utilization rate at 20°C, and λ is the temperature activity coefficient (θ is often used for this coefficient, but since it has been used to refer to EBCT in this thesis, λ is then used here). In practice, values of k_T and k_{20} can be determined by experiment, and λ can then be solved using Equation (3-15).

Low temperature does not only affect the kinetic parameter (k). Evidence in Laurent et al. (1999) showed that bacterial activity decreased all the way from around 30°C down to 1°C in biofilters, suggesting low temperature would affect amount of biomass, which can be expressed as biofilm density (X_f).

In Zhang and Huck (1996a, b), X_f was combined with k and kX_f was then treated as a single parameter mathematically. When an adjustment is applied later for the effect of temperature on θα parameter values, k and X_f are adjusted simultaneously with the same square root relationship (Equation (3-10)). Arguably, when adjusting for the effect of temperature on k, it is assumed that the adjustment also include consideration of the effect of temperature on X_f, at least in an approximate way.

Given a same biofilter (same media) and the same operation condition (same EBCT and influent water), combining Equation (3-14) with (3-15) will obtain Equation (3-16).

$$\frac{K'_{20}}{K'_{T}} = \left(\frac{k_{20}}{k_{T}}\right)^{1/2} = \lambda^{10-\frac{T}{2}} \qquad (3-16)$$

where K_T' and K_{20}' are K' values at T°C and 20°C respectively.

Equation (3-16) can used to adjust for the effect of low temperature on $\theta\alpha$ parameter values (Equation (3-13)) to maintain the same X* value.

Biodegradation kinetics parameter k can be obtained from experimental data; however, no such information could be extracted from the collected literature in this study. Elhadi (2004) determined values of k_{obs} in her thesis for biofilters removing geosmin and MIB. The λ value was 1.07 for GAC biofilter and 1.22 for anthracite biofilter. Different λ values for sand/anthracite and GAC are supported by other studies that low temperature affected anthracite and sand biofilters greater than a GAC filter (Wang, 1995; Emelko et al., 2006). In this thesis, these λ values will be used.

Among the collected AOC removal cases, reported temperature ranged from 3.5°C to 35°C. In this section, four temperature divisions were created for biofilters: 0°C-5°C, 5°C-10°C, 10°C-15°C and above 15°C. The median temperature of each division was used as the representative temperature and substituted into Equation (3-16). In a similar approach to that used in Section 3.5.2, the additional stimulatory effect of temperatures warmer than 15°C on biodegradation kinetics is ignored and 1 is assigned for the corresponding adjustment coefficient for all temperatures≥15°C. Table 3-9 shows the adjustment results.

Adopting different λ values for GAC and anthracite or sand is reasonable and consistent with studies showing that at high temperatures performance of GAC biofilters may not show much difference compared to that of anthracite or sand filter (e.g. Emelko et al., 2006), whereas at low temperature GAC biofilters outperformed anthracite and sand filters.

Table 3-9 Adjustment of $\theta\alpha$ values under different temperature ranges

Media	Temperature Division	Representative Temperature	Adjustment Coefficient
GAC (λ =1.07)	≥15°C	20°C	1
	10-15°C	12.5°C	1.29
	5-10°C	7.5°C	1.53
	0-5°C	2.5°C	1.81
Sand or Anthracite (λ =1.22)	≥15°C	20°C	1
	10-15°C	12.5°C	2.11
	5-10°C	7.5°C	3.46
	0-5°C	2.5°C	5.71

3.5.4 Development of Temperature-Adjusted AOC Removal- $\theta\alpha$ Relationship

The model-based AOC removal -X* relationship (Figure 3-3 and Table 3-5) was developed for biofiltration processes with influent water temperature of near room temperature (20°C). Using the temperature-adjustment coefficients in Table 3-9, Table 3-10, Figure 3-6 and Table 3-11, Figure 3-7 were developed to show approximate relationships between AOC removal and the $\theta\alpha$ Parameter under different temperature ranges for GAC and anthracite or sand biofilters, respectively. A sand or anthracite biofilter was assumed to have the same AOC removal performance as a GAC biofilter at 20°C, so that they can share the same model-based AOC removal- $\theta\alpha$ relationship for temperature-adjustment of $\theta\alpha$ values.

Figure 3-6 and 3-7 show that in the same way as the model-based AOC-X* relationship, with increasing $\theta\alpha$, AOC removal percentage increases, but not in a proportional way. Beyond a certain $\theta\alpha$ parameter value, increasing $\theta\alpha$ will provide no or very limited increase of removal efficiency.

Table 3-10 Approximate relationship between AOC percentage removals and $\theta\alpha$ values for GAC biofilter under different temperature ranges

Approximate AOC percentage Removal	$\theta\alpha$ (≥15°C)	$\theta\alpha$ (10~15°C)	$\theta\alpha$ (5~10°C)	$\theta\alpha$ (≤5°C)

33	0.25	0.30	0.40	0.50
47	0.50	0.60	0.80	0.90
66	1.0	1.3	1.5	1.8
80	1.5	1.9	2.3	2.7
85	2.0	2.6	3.1	3.6
95	3.0	3.9	4.6	5.4

Table 3-11 Approximate relationship between AOC percentage removals and $\theta\alpha$ values for anthracite or sand biofilter under different temperature ranges

Approximate AOC percentage Removal	$\theta\alpha$ (≥15°C)	$\theta\alpha$ (10~15°C)	$\theta\alpha$ (5~10°C)	$\theta\alpha$ (≤5°C)
33	0.25	0.50	0.90	1.4
47	0.50	1.0	1.7	2.8
66	1.0	2.0	3.5	5.7
80	1.5	3.2	5.2	8.5
85	2.0	4.2	6.9	11
95	3.0	6.3	10	17

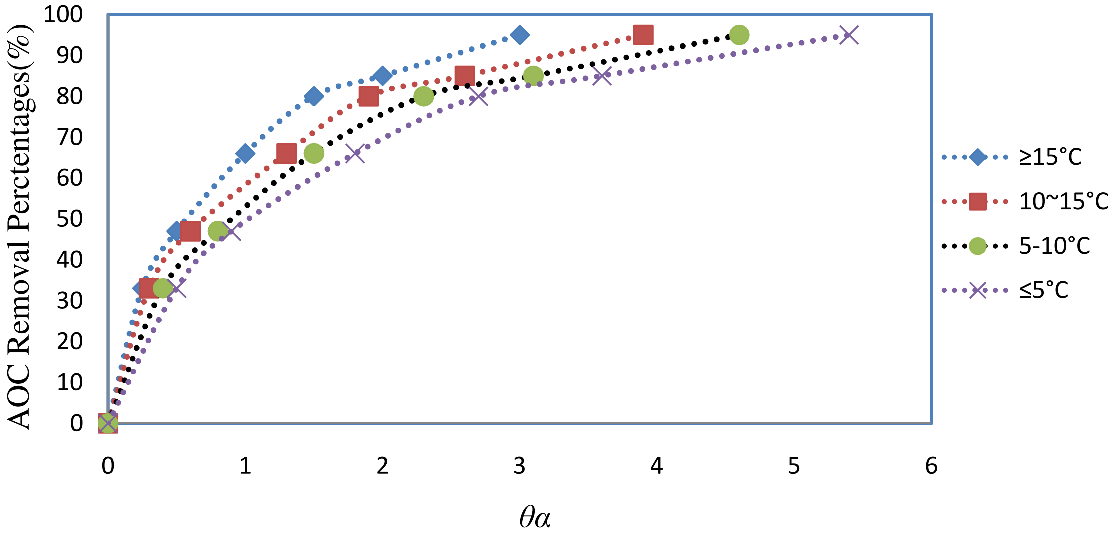

Figure 3-6 AOC removal-$\theta\alpha$ for GAC biofilter under different temperature ranges

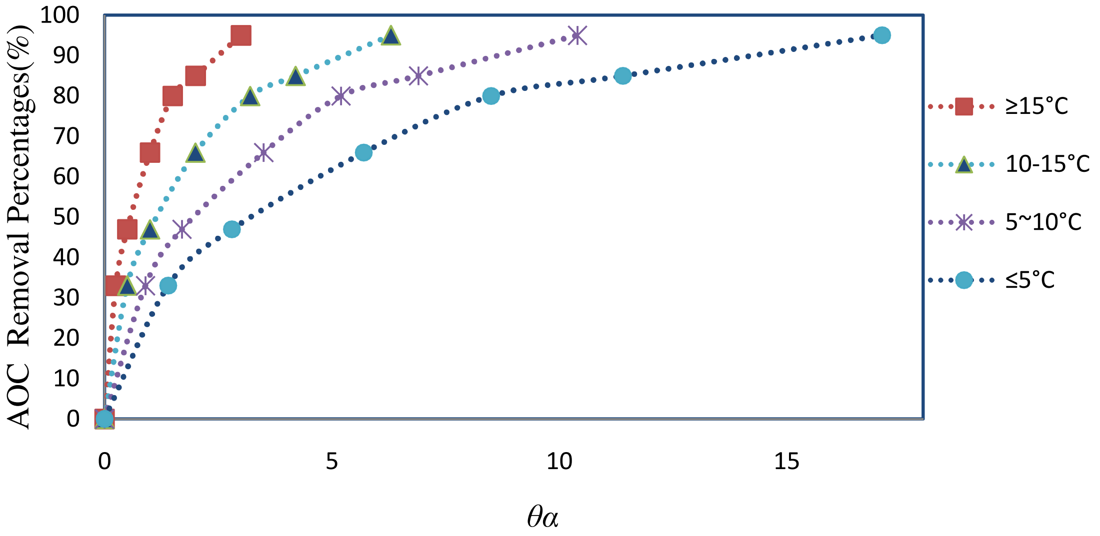

Figure 3-7 AOC removal-$\theta\alpha$ for sand or anthracite biofilter under different temperature ranges

The negative effect of low temperature on the GAC biofiltration process is less than on a sand or anthracite biofiltration process. The lower the temperature, the greater the effect for all types of biofilters, as would be expected. For example, $\theta\alpha$ parameter value needs to be greater than 0.8 and 1.7 to maintain a 50% AOC removal at temperature range of 5-10°C for a sand or anthracite filter and GAC filter respectively, compared with a $\theta\alpha$ parameter value (X* value here, as a matter of fact) of 0.5 as approximated in Huck (1999) and Huck and Sozański (2008) for biofilter operation at 20°C. $\theta\alpha$ parameter value needs to be greater than 2.7 and 8.5 to achieve 80% AOC removal in a temperature range lower than 5°C for a sand or anthracite filter and a GAC filter respectively, compared with $\theta\alpha$ parameter value (X* value here, as a matter of fact) 1.5 as approximated in Huck (1999) and Huck and Sozański (2008) for biofilter operation at around 20°C.

This set of AOC-$\theta\alpha$ parameter relationships is of practical use. For the same AOC percentage removal requirement, because kX_f is smaller at lower temperatures than at 20°C, a greater product of $\theta\alpha$ $((\theta\alpha)')$ is needed to compensate the negative effect of temperature on performance. This suggests increasing EBCT can maintain a satisfactory AOC removal in cold weather. (Although this is known from experience, the AOC-$\theta\alpha$ parameter relationship provides a means of quantifying this.)

Theoretically from the X* concept and expression, replacing filter media with finer media is another option. However finer media results in greater accumulation of filter bed head loss, especially at low temperature since the viscosity of water is greater than that at warm temperature, thus this option may not be feasible practically.

For example, the $\theta\alpha$ parameter value for a well operated anthracite biofilter (stable 85% AOC removal) in summer (influent water temperature is around 20°C) is 2; if this biofilter is required to operate in cold weather (≤5°C) achieving the same AOC removal, the corresponding $\theta\alpha$ parameter value then would necessarily be as high as 11.4. In theory, possible options include (1) increasing filter bed depth to 5.7 times of previous depth or adding a second stage biofilter; (2) decreasing hydraulic loading rate to around 18% of previous rate; (3) replacing anthracite with GAC; and (4) any combination of these four ways. For option 3 and 4, the diameter of new filter media can be calculated by Equation (3-14). In practice, water utilities are unlikely to undertake such drastic measures, however they may be able to decrease hydraulic loading somewhat, and because of slower biodegradation kinetics in the distribution system in winter, and lower biofilter efficiencies may be acceptable.

Figures 3-6 and 3-7 are also able to assess how much removal percentage will be reduced for a biofilter configured in summer when it runs in winter. For example, configuration of a GAC biofilter achieving 85% AOC removal at 20°C is associated with a $\theta\alpha$ parameter value of 2. If the configuration remains the same and the biofilter runs at 5-10°C, AOC removal is expected to decrease from 85% to 70% from Figure 3-6; if configuration remains the same and the biofilter runs below 5°C, AOC removal percentage is expected to decrease further to 60%.

3.5.5 Temperature-Adjusted $\theta\alpha$ Parameter Values

With temperature related information (temperatures, temperature ranges, conducted locations and seasons) in Table 3-7 and division ranges in Table 3-9, a representative temperature range for each AOC removal case is determined and listed in Appendix A-7. The ways of determination of the representative temperature range are specified below.

For AOC removal cases without temperatures/temperature ranges, or temperature ranges were too wide to fall within one division, season and location information are listed in Appendix A-7. With weather references, representative temperature ranges were approximated. If a seasonal raw water temperature figure was provided, the temperature range was measured.

If the provided temperature range slightly comes across two or more divisions, the division that the temperature of most operating days falls into, or the major division that the recorded temperature range falls into was chosen as the representative temperature range. If the provided temperature was just 5, 10 or 15°C, the upper division was chosen as the representative temperature range.

If the filter media was dual-layer or multi-layer, temperature-adjustment of $\theta\alpha$ parameter values followed that of the top layer media type, for most substrate removal happens in the top layer. If the filter media was bio-ceramic, the adjustment follows that of GAC; if the filter media was expanded clay or zeolite, the adjustment followed that of sand/anthracite.

If a representative temperature range was difficult to determine for a particular study, a representative temperature range was cross-referenced from another collected literature in which the raw water source was the same. The temperature of groundwater was set in the range of below 5°C.

Four biofilter configurations for AOC removal in LeChevallier et al. (1992a, b) were chosen as the estimation base of the $\theta\alpha$ parameter in section 3.4, but temperature was not provided for them. No effect of temperature was assumed for the estimation base; therefore the representative temperature ranges for them was ≥15°C.

Appendix A-7 also lists temperature-adjustment coefficients according to determined representative temperature ranges. Temperature adjusted $\theta\alpha$ parameter values, initially estimated $\theta\alpha$ parameter values and actual EBCTs are all listed in Appendix A-8.

Figure 3-8 compares temperature-adjusted $\theta\alpha$ parameter values and initially estimated $\theta\alpha$

parameter values. Temperature adjusted *θα* parameter are distributed more tightly, with more datapoints falling into the range from 0 to 5 than is the case for the initially estimated *θα* parameter values.

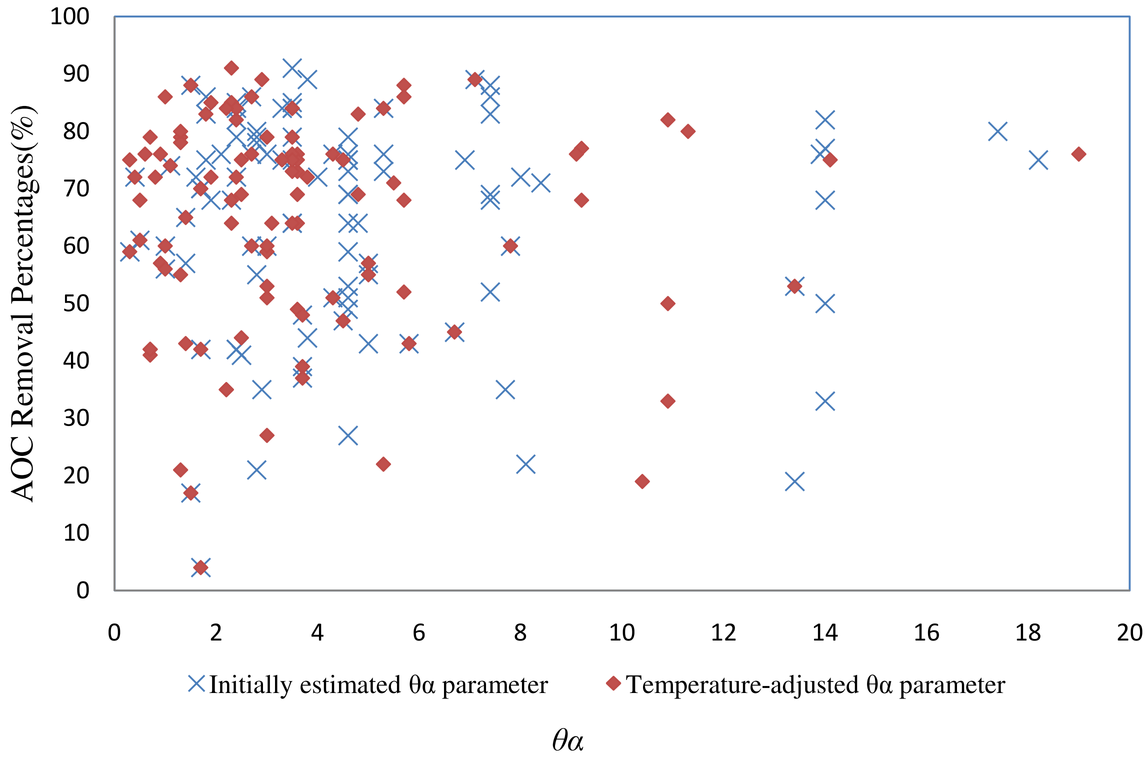

Figure 3-8 Comparison of temperature-adjusted *θα* values and initially estimated *θα* values (One datapoint of (40, 76) for initially estimated *θα* parameter was not shown in the figure)

Figure 3-9 and Figure 3-10 compare distributions of temperature-adjusted *θα* parameter values with the temperature-adjusted AOC removal-*θα* relationship for GAC biofilters (or biofilters using GAC as the top layer) for temperature ranges of 10-15°C and 5-10°C respectively.

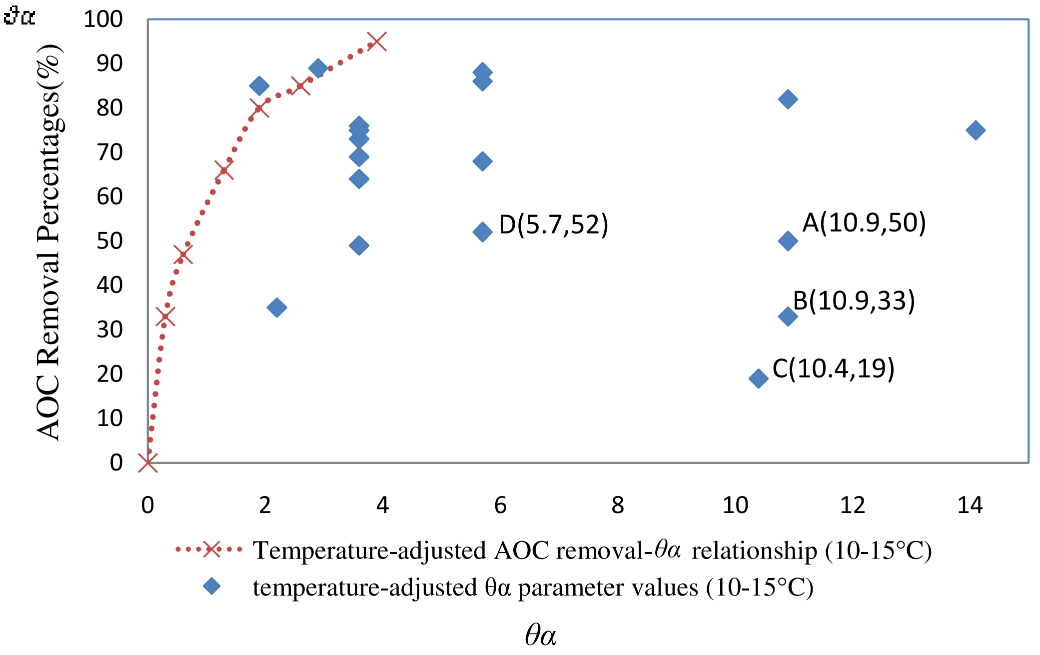

Figure 3-9 Temperature-adjusted AOC-$\theta\alpha$ relationship and distribution of temperature adjusted $\theta\alpha$ values (10-15°C range for GAC filters or filters using GAC as top layer)

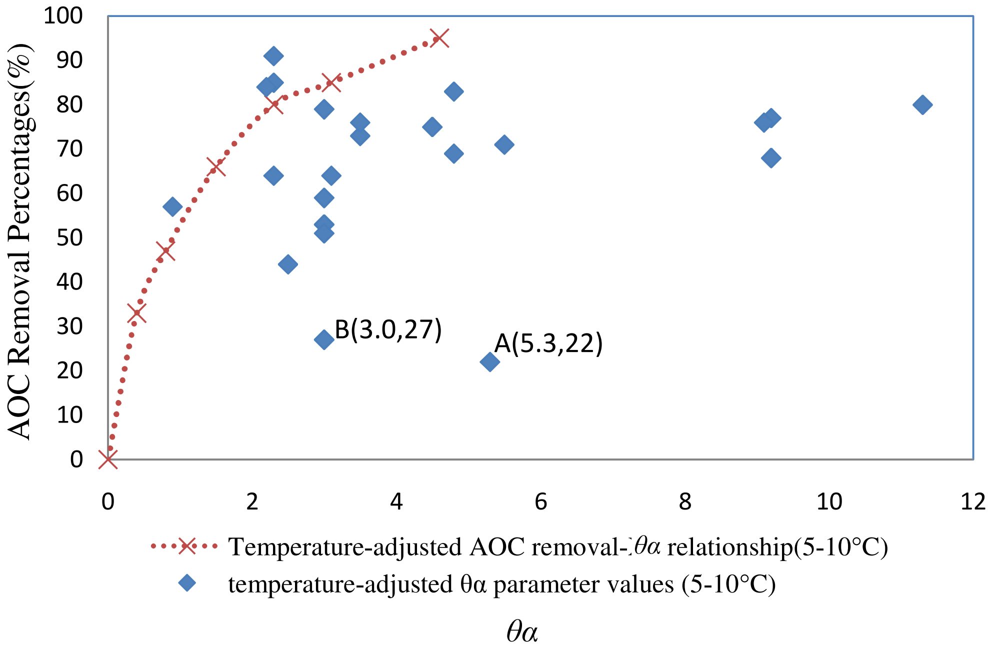

Figure 3-10 Temperature-adjusted AOC-$\theta\alpha$ relationship and distribution of temperature adjusted $\theta\alpha$ values (5-10°C range for GAC filters or filters using GAC as top layer)

Plotting of the temperature-adjusted $\theta\alpha$ parameter values does not show an overall satisfactory match with the corresponding AOC removal-$\theta\alpha$ relationship (the expected relationship) for GAC biofilters in Figure 3-9 and 3-10. “Over-design” or “under-operated”, which will be discussed in Chapter 5, is likely to be the main reason for this discrepancy. This suggests that for GAC biofilters in low temperature ranges (5-15°C), researchers and water works tend to design/operate conservatively (i.e., apply much longer EBCTs) to ensure AOC removal goals for safety reasons.

On the other hand, this may also indicates that the effect of temperature on AOC removal kinetics and in turn on removal performance may be more severe than expected in the earlier section, thus the coefficient λ adopted from Elhadi (2004) for GAC biofiltration may be too small to reflect impairment of AOC removal performances. (It should be noted that Elhadi’s coefficients were calculated for geosmin/MIB removal.)

“Substantially diverging ” points A, B and D in Figure 3-9 were associated with Huck et al. (1991); one possible reason that they are “substantially diverging” points is that the backwashing strategies used might have impaired the microbial community in the biofilters greatly and affected AOC removal efficiencies. “Substantially diverging” pointC in Figure 3-9 and point A in Figure 3-10 were associated with Heinicke et al. (2006) and Persson et al. (2006) respectively, with no pre-oxidation processes ahead of the biofilters. “Substantially diverging” point B in Figure 3-10 was associated with Vahala et al. (1998a), and possibly occurred because of an influent loading shock.

Figure 3-9 and 3-10 also show that beyond a certain $\theta\alpha$ parameter value (i.e. 4), no matter how great X* (or the temperature-adjusted $\theta\alpha$ parameter) would be applied for GAC biofilters at 5-10°C and 10-15°C, actual observed AOC removal efficiencies could hardly be improved, although they could be considered as fair removal ($\geq$ 50%). This is the same as for temperature-favored biofiltration processes in Figure 3-5; detailed discussions were specified in section 3.5.2.

Figure 3-11 and Figure 3-12 compare distributions of temperature-adjusted $\theta\alpha$ parameter values based on observed results and the expected temperature-adjusted AOC removal-$\theta\alpha$ relationship for sand or anthracite biofilters (or biofilters using sand or anthracite as the top layer) in the temperature ranges of 10-15°C and 5-10°C respectively.

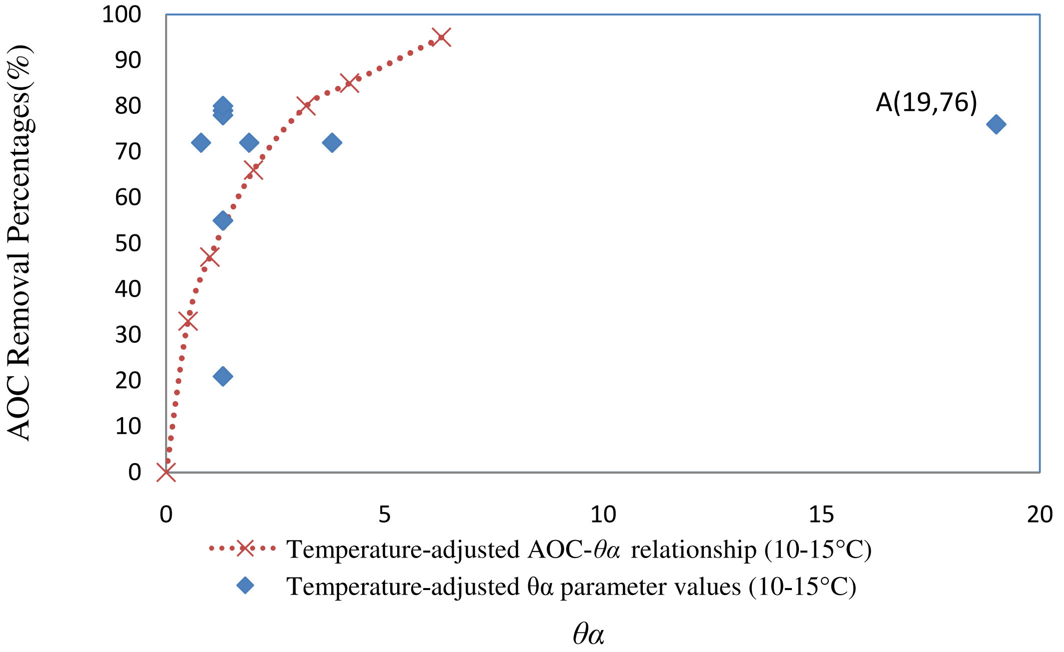

Figure 3-11 Temperature-adjusted AOC-$\theta\alpha$ relationship and distribution of temperature adjusted $\theta\alpha$ values (10-15°C range for sand or anthracite filters or filters using sand or anthracite as top layer)

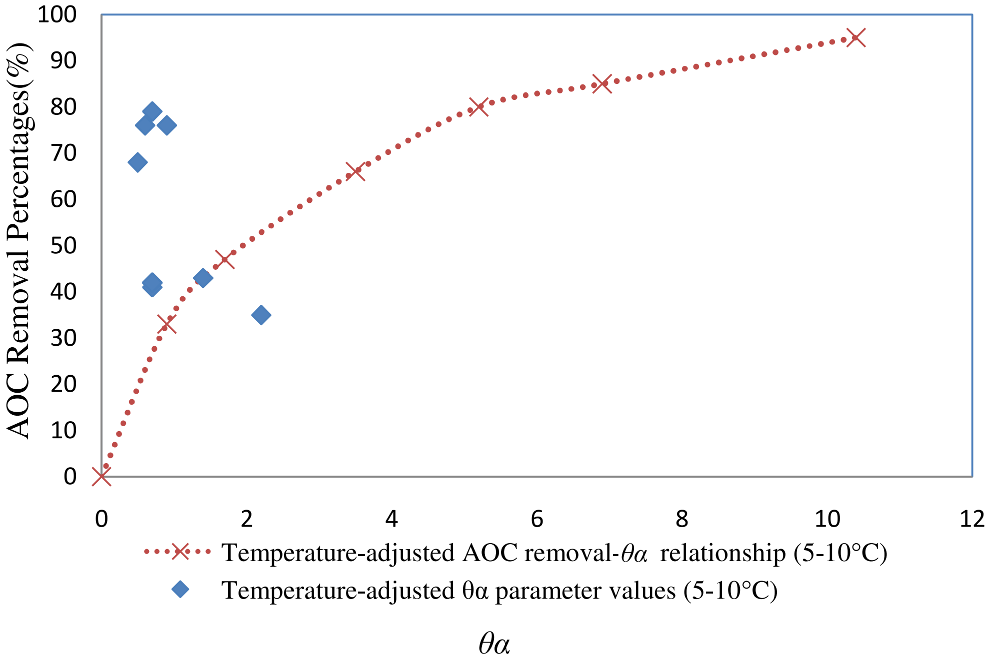

Figure 3-12 Temperature-adjusted AOC-$\theta\alpha$ relationship and distribution of temperature adjusted $\theta\alpha$ values (5-10°C range for sand or anthracite filters or filters using sand or anthracite as top layer)

“Substantially diverging” point A (19, 76) in Figure 3-11 is associated with a significant “over-design” or “under-operated” scenario in Hijnen and van der Kooij (1992), because the biofilter achieved an AOC removal of 72% at an EBCT of 2.4 minutes (temperature-adjusted $\theta\alpha$ parameter value of 0.8); while the “substantially diverging” point -associated EBCT was 60 minutes (temperature-adjusted $\theta\alpha$ parameter value of 19.0), with an AOC removal of 76%. Thus increasing EBCT by more than 20 times EBCT only obtained an additional 4 percentage points improvement in AOC removal.

In Figure 3-12, there are also a few “Substantially diverging” pointss with the temperature-adjusted $\theta\alpha$ parameter values much smaller than the necessarily expected $\theta\alpha$ parameter values at this temperature range. In other words, for these “Substantially diverging” points, the associated biofiltration processes performed much better than expected in terms of AOC removal efficiencies. Factors such as biomass acclimation to a cold environment might be the reason. Detailed discussions will be presented in Section 4.5.4.

Figure 3-13 compares distributions of temperature-adjusted $\theta\alpha$ parameter values and temperature-adjusted AOC removal-$\theta\alpha$ relationships in the temperature range below 5°C for GAC and anthracite or sand biofilters.

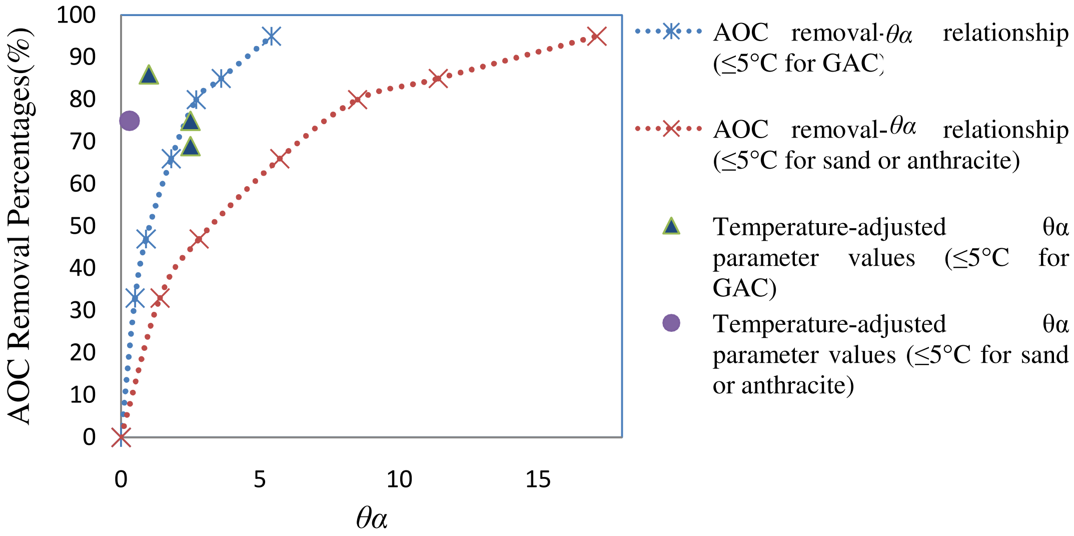

Figure 3-13 Temperature-adjusted AOC-$\theta\alpha$ relationship and distribution of temperature adjusted $\theta\alpha$ values (≤5°C)

For Figure 3-13, because the number of datapoints is limited, no firm conclusion can be drawn in terms of matching the distribution of temperature-adjusted $\theta\alpha$ parameter values with the temperature-adjusted AOC removal- $\theta\alpha$ relationship. However, the single sand/anthracite data point available shows much better performance than expected.

It should be noted here that although X* and $\theta\alpha$ parameter were used as variables simultaneously on the same axes in Figures presented above, they are two different parameters. $\theta\alpha$ parameter is a component of X* without considering factors other than θ and α. $\theta\alpha$ parameter is proportional to X* quantitatively and both parameters are dimensionless.

In practice, temperature change is invariable accompanied by seasonal fluctuation of various water quality parameters due to normal seasonal cycles in water bodies, which often requires operation adjustments, such as coagulants, ozone doses and pH; therefore the impact of temperature may not be able to be completely isolated. In addition, biofilters may exhibit a lag effect with respect to seasonal temperature changes, as demonstrated recently by Hallé (2010) for the removal of pharmaceuticals by biofilters treating a river water.

3.6 Consideration of Effect of Ozonation on Biofiltration Processes and $\theta\alpha$ Values

3.6.1 Collecting Ozonation-Related Information and Calculation of Ozone Dosages

As summarized in the literature review, pre-ozonation is also one of the most important factors affecting performance of biofiltration processes. Table 3-12 records ozonation information for the collected cases of AOC removal by biofiltration.

Table 3-12 TOC and pre-ozonation information for the collected AOC removal cases

Source	TOC(mg/L)**	Ozone Dosage(mg/L)**	Ozone Dosage (mgO_3/mg TOC)** ***
Ahmad et al., 1998			1.00$^{(1)}$ (mgO_3/mg NPOC)
Chien et al., 2007	1.2	NA	NA
Chien et al., 2008	1.2	NA	NA

Daniel and Teefy, 1995	1.8	3.0	0.60
Heinicke et al., 2006	4.7	0(2)	0
Hijnen and van der Kooij, 1992			0(3), simulated influent water(acetate)
Hu et al., 1999	7.0-13.6*	0	0(4)
Hu et al., 2005		0	0 Chlorinated
Huck et al., 1991			0.50(5) (mg O_3/mg NVOC)
			1.00(5) (mg O_3/mg NVOC)
Krasner et al., 1993	3.5	1.8(transferred) (6)	0.54
Lai et al., 2002	1.2	2.8(7)	0.43
		2.0(7)	0.60
LeChevallier et al., 1992a, b			0.50-1.00*
			0
			0(8) (with other pre-oxidation)
Moll et al., 1999			1.30
Najm et al., 2005	1.5-6.0*	4.8	0.78
Persson et al., 2006		0(9)	0
Shu et al., 2008	3.8(10)	1.5	0.39
Vahala et al., 1998a			From 0.22 to 1.23
Vahala et al., 1998b	2.6	NA	NA
van der Aa et al., 2003	4.5-6* (11) DOC	1.2(12) (together with peroxide)	
van der Hoek et al., 2000	0.75-0.90*		0.38-0.45(13) * (mg O_3/mg DOC)
Wang, et al., 1995			0.80(14) (mg transferred O_3/mg DOC)
Wang and Summers, 1996			0.35(15) (mg O_3/mg DOC)
Wert et al., 2008	2.3-3.1*	1.40-1.70*	
Zappia et al., 2007			0(16) (with other pre-oxidation)
Zhang et al., 2004			NA

Note: * Average is used for calculation;
** Refer to 3.6.1 for collecting TOC and ozonation information;
*** Refer to 3.6.1 for calculation of ozone dosages;
1. Expressed as mg O_3/mg NPOC;
NVOC=Non-purgeable Organic Carbon;
2. Receiving raw water;
3. Special treatment for raw water, data excluded in calculations;
4. Chlorinated, data excluded in calculations;
5. Expressed as mg O_3/mg NVOC;
NVOC=Non-Volatile Organic Carbon;
6. Expressed as transferred ozone concentration, transferring efficiencies are assigned as 90%;
7. Second stage ozonation;
8. Pre-oxidation other than ozonation, data excluded in calculations;
9. Receiving raw water;
10. Adding up all fractions;
11. Expressed as DOC;
12. Applied together with peroxide, data excluded in calculations;
13. Expressed as mg O_3/mg DOC;
14. Expressed as mg transferred O_3/mg DOC;
15. Expressed as mg O_3/mg DOC;
16. Other irregular pre-treatment processes, data excluded in calculations.

Two approaches are normally used to quantify ozonation dosages: (1) expressed directly as ozone concentrations or ozone residual; (2) expressed as carbon-normalized ozone concentrations.

Ozone transfer efficiency is affected by various factors, such as bubble size, concentration, temperature, and water chemistry, and is defined as the percentage ratio between transferred dosage and applied dosage. Several handbooks and reports from public works suggested 85%-95% transfer efficiencies for ozone generators were able to be achieved in treatment plants (Langlais et al., 1991; Rakness, 2005). 90% was then assigned for ozone transfer efficiency in this section. Since most the collected cases in this thesis used applied dosages, transferred dosages were then transformed to applied dosages.

Ozone reacts with NOM in water; therefore the relative concentration of ozone to organic carbon is a more reasonable indicator for intensity of ozonation dosing. When ozone dosage is expressed as carbon-normalized ozone concentration, the carbon parameters can be TOC or DOC. NPOC (Non-Purgeable Organic Carbon) and NVOC (Non-Volatile Organic Carbon) were also seen in the collected literature. TOC was used in this study to normalize aquatic ozone concentrations. DOC and other carbon parameters were used in the calculations as reasonable approximations to

TOC, since particulate organic matter is normally only a minor fraction of TOC and there were coagulation processes ahead of the biofilters (unless indicated in Table 3-1 and Table 3-12).

When ozone concentration and organic carbon concentration were given separately, normalized ozone dosage is calculated by Equation (3-17):

$$Normalized\ Ozone\ Dosage = \frac{Applied\ ozone\ dosage}{TOC} \qquad (3-17)$$

The units for normalized ozone dosage are mg O_3/mg TOC.

It should be highly mentioned here that the TOC concentration in Equation (3-19) should be the measurement right before ozone is bubbled into the water. However, when using Equation (3-19) to calculate normalized ozone dosage, the TOC concentration in influent of ozone contactor is hard to obtain, and therefore TOC concentrations were extracted from characterizations of the raw water. It is known that each process before ozonation is able to remove TOC more or less, e.g. a highest TOC removal of 38% by coagulation was reported in a summary of 46 treatment plants (Bratby, 2006). Therefore, the calculation results by Equation (3-17) are likely to underestimate normalized ozonation dosages, or can be considered as the minimum dosing values for these cases.

Unlike temperature, ozone dosage cannot be estimated if it is not provided in the literature; only cases with adequate ozone dosing information were chosen for the following discussion.

3.6.2 Effect of Ozonation on Biofiltration and $\theta\alpha$ Parameter

Appendix A-9 lists calculation results of carbon-normalized ozone dosages for the chosen AOC removal cases by biofiltration. Five rows in the top of Appendix A-9 are associated with biofilters without pre-ozonation processes. Data were excluded if there were other pre-oxidation or irregular pre-treatment units/operations before biofiltration, such as peroxide addition, chlorination and chloramination, even if normalized ozone dosages were given or could be calculated. Only immediate pre-ozonation data was included. Figure 3-14 presents plotting of

AOC removal percentages vs. ozone dosages.

For biofiltration processes with pre-ozonation examined in this study, the average ozone dosage was 0.68 O_3/mg TOC, which falls within the range of doses (0.5-1.0 O_3/mg TOC) currently applied in full-scale water treatment facilities (Langlais et al., 1991), keeping in mind that some calculated ozone dosage values may have been underestimated as concluded in Section 3.6.1.

This average value is also consistent with optimal ozone dosages suggested in several studies (0.5 mgO_3/mg DOC in Huck et al., 1989; 1 mgO_3/mg DOC in Werner and Hambsch, 1986; 1.5-2.0 mgO_3/mg DOC in van der Kooij et al., 1989) and indicated that beyond a dosage limit there is only minor effect of ozonation on NOM. This is true for both AOC studies and BDOC studies.

An ANOVA test conducted by the author of this thesis indicated there was a significant difference of AOC removal efficiencies between ozonated and non-ozonated influent for biofilters, with the average AOC percentage removals of 66% and 32% respectively (p<0.05). However, no significant difference of AOC removal efficiencies was found among different ozone dosing ranges for biofilters influent: 0-0.5 mg O_3/mg TOC, 0.5-1.0 mg O_3/mg TOC and 1.0-1.5 mg O_3/mg TOC (p<0.05).

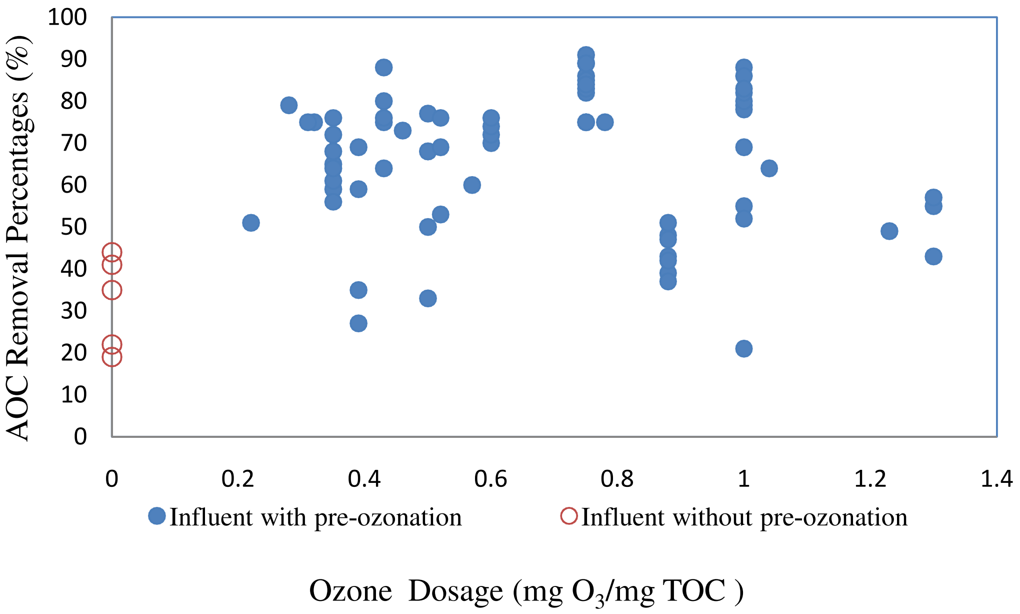

Figure 3-14 Ozone dosages and AOC removal percentages

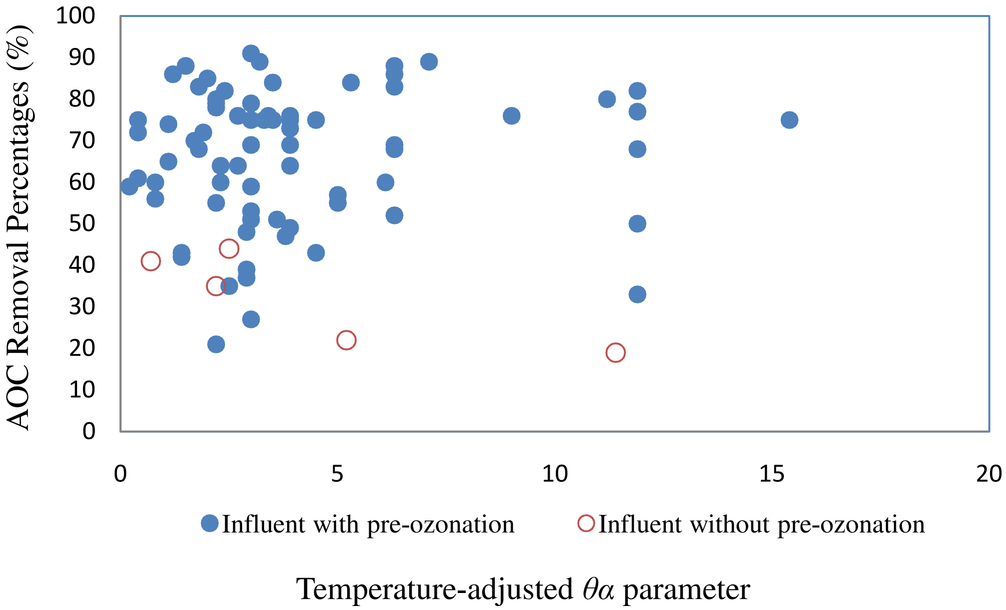

Figure 3-15 Temperature-adjusted $\theta\alpha$ values for biofilters with and without pre-ozonation

For reference, Figure 3-15 also presents AOC removal percentages vs. the corresponding temperature-adjusted $\theta\alpha$ parameter, for biofilters with and without pre-ozonation for the data pool in this section. Since temperature-adjusted $\theta\alpha$ parameter and ozonation can be considered as two independent factors, no grouping of the temperature-adjusted $\theta\alpha$ parameter was found for biofilters with and without ozonation.

Ozonation is able to convert refractory organic matter in water and increase the absolute amount of easily biodegradable matter, as measured by the parameters AOC and BDOC. As Langlais et al. (1991) summarized, numerous studies concluded that ozonation is able to increase biodegradability in drinking water treatment when measurement of AOC, BDOC or AOC/DOC, BDOC/DOC is used as the quantitative indicator for biodegradability.

For AOC or BDOC removal by biofiltration, the "visible" effect of pre-ozonation is increasing the influent concentration for the following biofilters. However, AOC or BDOC concentration in influent can be considered not to be a factor affecting biofiltration performance or AOC and BDOC removal efficiencies, since AOC and BDOC removal by biofiltration can be approximated as first-order reactions (Huck and Anderson, 1992; Huck et al., 1994; Zhang and

Huck, 1996a).

Increasing substrate level in the influent does not necessarily lead to an increasing substrate utilization rate. It is very important therefore to underline the difference between the biodegradability of bulk water (or of the whole NOM) and the biodegradability of AOC and BDOC themselves (strictly speaking, the organic compounds pool associated with AOC and BDOC measurements). Kinetics of AOC and BDOC biodegradation were seldom studied and no comparable information could be pulled out.

Despite the discussions above, there is evidence, such as reduction of specific UV_{254} absorbance (SUVA) (van der Kooij et al., 1989; Seredyńska-Sobecka, et al., 2006), indicating that ozonation is able to increase not only AOC concentration in biofilter influent (the amount of BOM), but also the biodegradability (within the contact time available in a filter; recall that the standard AOC test takes several days) of the organic compound pool associated with AOC measurement (the utilization rate of BOM), since BOM with low molecular weight normally is more biodegradable. ANOVA results mentioned above in this section also show AOC removal efficiency increases with pre-ozonation statistically, suggesting it may be a result of increasing biodegradability of AOC associated compounds pool after ozonation.

It is then believed that ozonation not only increases the absolute amount of BOM, but also changes the relative composition of various BOM components (from biodegradable to more biodegradable). The shift from P17 to NOX in the AOC bioassay following ozonation may be the case, since Zhang and Huck (1996b) pointed out AOC-NOX was much more easily biodegradable than AOC-P17.

This is also true for BDOC removal by ozonation-biofiltration. The effect of ozonation on BDOC removal by biofiltration and the effect of BOM parameter measurement on the interpretation of removal data will be presented in Section 4.7, Chapter 4.

3.7 Conclusions

X* analysis was performed for AOC removal by biofiltration in this chapter. Wide ranges of media sizes, EBCTs, temperatures and high diversity of pre-treatment and operating conditions for the collected cases were able to well represent biofiltration practices for AOC removal by biofiltration.

No significant co-relationship between EBCT and AOC removal percentage were found, indicating that EBCT alone is not able to guide biofiltration design and operation. Using the model-based AOC removal-X* relationship, expected X* values were obtained. A new parameter, the $\theta\alpha$ parameter, was developed in this chapter to refer to partially estimated X* values, considering only the physical components of X*, EBCT and media size. $\theta\alpha$ parameter values were estimated by comparison of ratios of $\theta\alpha$ products based on an appropriately chosen calculation base. The finding can be also considered as "survey" results of θ, α, $\theta\alpha$ and $\theta\alpha$ parameter value among various studies and can narrow down the scope of AOC removal biofiltration process configuration and treatability study.

The raw water temperature range for each collected case was determined with the assistance of information on temperatures, temperature ranges and conducted season and location. Distribution of $\theta\alpha$ parameter values for temperature-favored biofiltration processes (defined as processes for which influent water temperature greater than 15°C) matched the model-based AOC-X* relationship reasonably (based on visual observation, as discussed in relation to Figure 3-5), if the $\theta\alpha$ parameter value was smaller than about 2. Really high AOC percentage removals were not observed for the collected AOC removal cases. The distribution of $\theta\alpha$ parameters associated with temperature-favored biofiltration processes has a lower and flatter tailing than the model-based AOC removal-X* curve if the $\theta\alpha$ parameter value is above 2. This observation does not follow the conclusions in Zhang and Huck (1996a), suggesting that the model-based AOC removal-X* relationship may not be able to evaluate AOC removal efficiencies quantitatively in the high $\theta\alpha$ parameter range (above 2). Possible reasons include the ignoring of soluble microbial products by Zhang and Huck (1996a) and perhaps questioning the applicability of $S_{\min}$ as the base for the whole framework of drinking water biofilm modeling.

Using the temperature activity coefficient in Elhadi (2004), temperature-adjustment coefficients for the $\theta\alpha$ parameter values were determined for different temperature ranges for GAC biofilters and sand or anthracite biofilters. Temperature-adjusted AOC removal-$\theta\alpha$ relationships were developed and temperature-adjusted $\theta\alpha$ parameter values were also estimated. Comparisons were conducted between temperature adjusted AOC removal-$\theta\alpha$ relationships and temperature-adjusted $\theta\alpha$ parameter values. Points which were especially distant from the expected relationship were examined to see if there were particular experimental circumstances that might have contributed to a much different than expected result. For GAC biofilters under low temperature ranges (5-15°C), researchers and waterworks tend to design/operate conservatively. Biofilter over-design or under-operated is common for the collected AOC removal cases. Beyond a certain X* value (i.e. 4), no matter how great an X* (or temperature-adjusted $\theta\alpha$ parameter) were to be applied for GAC biofilters at 5-10°C and 10-15°C, AOC removal efficiencies could hardly be improved, although they could be considered as fair removal (≥50%).

TOC-normalized ozone dosages were calculated for part of the collected AOC removal cases. The average ozone dosage was 0.68 O_3/mg TOC for biofiltration processes with pre-ozonation and falls into the suggested ozone to TOC range reasonably. Although no relationship was found between ozone dosages and AOC removal percentages, an ANOVA test indicated there was a significant difference of AOC removal efficiencies between ozonated and non-ozonated influents for biofilters. However there was no significant difference of AOC removal efficiencies among different ozone dosing ranges for biofilter influents.

This suggests ozonation process may not only increase the absolute amount of BOM for the following biofilter and increase the biodegradability of the bulk water or the whole NOM; it may also increase the rate of biodegradability of BOM after ozonation by changing the relative composition of various BOM components.

In general, examining X* (or partially estimated X*, which is the $\theta\alpha$ parameter) provided more useful information than EBCT alone in terms of evaluation and prediction of AOC removal by biofiltration, which confirms that X* is a better parameter for biofiltration design and operation.

Chapter 4

X* Analysis for BDOC Removal by Biofiltration

Another BOM parameter, Biodegradable Dissolved Organic Carbon (BDOC), is also an indicator of biological instability for drinking water. The biological filtration process has been used in drinking water treatment to reduce BDOC level in recent years.

As summarized in the literature review, Zhang (1996) and Zhang and Huck (1996a) developed a new parameter, dimensionless contact time X*, for biofiltration performance. As discussed in Chapter 3, the simplified X* expression (Huck and Sozański, 2008) is essentially determined by physical EBCT, specific surface area of attached biofilm (or filter media), biofilm density and biodegradation kinetics by assuming first-order reaction and that mass transfer is not important.

Using the concept and expression of X*, this chapter analyzes BDOC removal by biofiltration for from the collected literature, establishes BDOC-X* relationship based on BDOC removal kinetics, estimates X* values (or approximated values) step by step and examines how the above mentioned factors affect biofiltration performance and X* values (or approximated values). BDOC is considered in this chapter as the limiting substrate for biofiltration, based on considerations provided by Zhang and Huck (1996a). Analysis results and discussions are presented here. Calculation results are presented in Appendix B.

4.1 Data Collection

Information on filter media, operation and other information for BDOC removal cases from journals and conference proceedings was collected. The collected cases are summarized in Table 4-1. Published or presented years for these studies are from 1991 to current. Appendix B-1 lists detailed reference information.

Table 4-1 Summary of the collected cases for BDOC removal by biofiltration processes

Source/Scale(1)	**Media Type**	**Pre-treatment and Operation Conditions**	**BDOC Removal (%)**
Fonseca and Summers, 2003 (B)	Sand	No ozonation; First-stage ozonation; Second-stage ozonation	From 65 to 100
Griffini, et al., 1999 (P)	GAC; Biolite	Pre-chlorination and Ozonation	From 52 to 74
Heinicke et al., 2006 (P)	GAC	No-ozonation, receiving raw water	23
Melin et al., 2002 (F)	Multiple media	Ozonation	From 36 to 68
Moll et al., 1999 (B)	Sand	Special treatment for source water, Ozonation, Upflow mode	38, 60, 60
Niquette et al., 1998 (P)	Dual-GAC/Sand; GAC	Ozonation, Different backwashing	From 15 to 48
Nishijima and Speitel, 2004 (B)	Dual-GAC/sand	Prepared ozonated fulvic acid solution as influent	63
Persson et al., 2006 (P)	GAC; Expanded clay	No ozonation, receiving raw water	28, 30, 34
Prévost et al., 1992 (P)	GAC	Ozonation	8, 41, 95
Sang et al., 2003 (P)	Bio-ceramic	Receiving raw water, Aeration, nutrient addition	From 55 to 71
Servais et al., 1991 (F)	GAC	Two stage ozonation, Sand filter ahead	From 25 to 53
Servais et al., 1994 (P)	GAC	Ozonation and Chlorination	From 43 to 56
Thiel, et al., 2006 (P)	Dual-Anthracite/Sand; Dual-GAC/Sand	Enhanced coagulation; MIEX clarifier; Chlorination; Different backwashing	From 12 to 23
Wang et al., 2000 (P)	GAC	Ozonation, For water reclamation	78
Wang and Summers, 1996 (B)	Sand	Special treatment for source water, Ozonation	From 16 to 40
Zappia et al., 2007 (P)	Dual-Sand/Anthracite; Dual-GAC/Sand	Enhanced coagulation; MIEX clarifier; Chlorination; Different backwashing	From 5 to 25

Note: (1) B-Bench; P-Pilot; F-Full scale.

Media types include GAC, sand, expanded clay, zeolite, bio-ceramic and biolite, as well as dual-media or multi-media, such as GAC-sand, anthracite-sand and GAC-phonolith-calcium carbonate. GAC and sand were the most used biofilter media for the collected BDOC removal by biofiltration cases. Dual-media was used in a significant number of biofiltration processes. Particle sizes of filter media for the collected BDOC biofiltration cases are summarized and calculated in Section 4.1.1 and 4.2 respectively (Appendix B-2 and B-3).

Appendix B-4 lists EBCT values and BDOC percentage removals. Recorded EBCT values range from 1.5 minutes to 45 minutes. BDOC removal efficiencies range from poor removal of only 5% to complete removal of 100%. Detailed summaries and discussions are presented in Section 4.1.2 and 4.1.3.

Water temperature information is listed in Table 4-7 and Appendix A-8, with lowest temperature of 0.5°C and highest temperature of 35°C. The detailed information for temperature and its effect on biofiltration and X* are presented in Section 4.5.

Pre-ozonation information is listed in Table 4-11. Appendix B-9 lists calculation results of carbon-normalized ozone dosages for the chosen BDOC removal cases by biofiltration, which ranges from 0 (without ozonation) to 1.50 mg O_3/mg TOC. The detailed information for pre-ozonation and its effect on biofiltration and X* are presented in Section 4.6.

Table 4-1 also summarizes pre-treatment and operation conditions for the collected cases with conventional processes considered as default pre-treatment prior to application of biofiltration. The most applied pre-treatment process was single-stage or multi-stage ozonation, for ozonation is able to convert refractory organic matter into easily biodegradable matter and facilitate the biodegradation process in biofilters. Ozonation dosage varied greatly and was expressed in different ways for the collected cases. Chlorination, chloramination, enhanced coagulation and aeration were also used as pre-treatment processes. For some cases, the biofilter directly received raw water without pre-treatment; or received water from conventional processes (default) without ozonation.

The wide ranges of particle sizes, EBCTs, temperatures and high diversity of pre-treatment and operation conditions summarized above are able to well represent biofiltration practices for BDOC removal.

4.1.1 Collecting and Determining Representative Sizes of Filter Media Particles

The approaches for obtaining representative sizes and actively specific surface area of the media particle are the same as in Chapter 3 for AOC removal. If there was no way to collect the necessary particle size information, a study was not included. In this study, the author did not assume any media particle size or any related parameters because particle size directly determines specific surface area, which is a crucial parameter for the following calculation. Appendix B-2 lists biofilter media particle sizes for the collected BDOC removal cases.

4.1.2 Collecting and Determining EBCTs and BDOC Removals

Appendix B-4 lists EBCT and BDOC removal values for the collected BDOC removal cases. The ways of obtaining these values are also indicated. The approaches are the same as in Chapter 3.

4.1.3 Relationship between EBCT and BDOC Removal Efficiencies

Zhang (1996) and Huck et al. (2000) summarized that for a specific water and biofilter and within a certain range, increasing EBCT may increase BOM removal efficiency, while it is in a less than proportional way and removal efficiency will not improve after a certain range of EBCT when BOM removal efficiency reaches a plateau.

Figure 4-1 shows the relationship between biofilter EBCTs and BDOC removal percentages. It is demonstrated clearly that biofiltration is an effective process to reduce BDOC levels from water and be able to produce more bio-stable water. The EBCT datapoints were scattered over the BDOC removal percentages and no significant co-relationship between EBCT and BDOC removal percentage could be found in Figure 4-1. The same conclusion can be drawn for BDOC removal as for AOC removal in Chapter 3 that EBCT alone is not able to describe biofiltration performance for BDOC removal. Considering EBCTs only is not adequate to guide biofiltration design and operation.

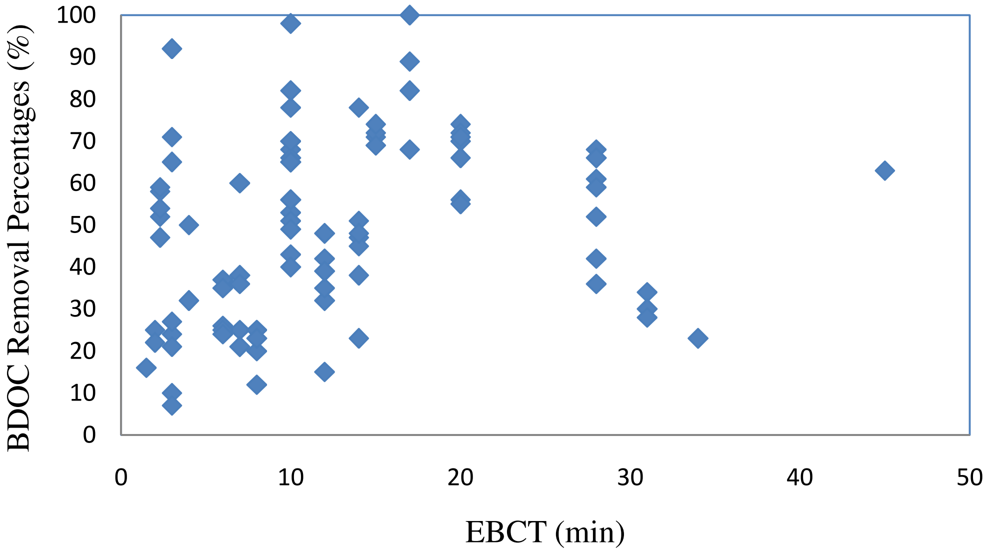

Figure 4-1 Distribution of EBCTs over BDOC removal percentages

As in Chapter 3, by assuming a zero-order reaction for BOM removal by biofiltration processes, Appendix B-4 also presents EBCT-normalized percentage removal rates (T^{-1}), which is considered as an approximate indicator for treatment efficiency. The lowest and highest EBCT-normalized percentage removals, together with literature sources, EBCTs and BDOC removal percentages, are listed in Table 4-2.

Table 4-2 Most inefficient and efficient biofiltration processes (Based on EBCT)

	Source	EBCT (min)	BDOC Removal (%)	EBCT Normalized Percentage Removal (min^{-1})
Most inefficient	Heinicke et al., 2006	34	23	0.70
Most efficient	Fonseca and Summers, 2003	3	92	30

In Heinicke et al. (2006), only 23% BDOC removal was achieved with a contact time of 34 minutes. The poor removal efficiency may be because the biofilters received raw water without pre-treatment, but could be due to other reasons as well.

In Fonseca and Summers (2003), over 90% BDOC removal was achieved with an EBCT as short as 3 minutes, which may be the highest BDOC removal efficiency reported. A modified simplified and rapid BDOC measurement method (Allgeier et al., 1995) was used in this study and might have contributed to the unusually high BDOC removal efficiency. The effect of BOM parameter measurement on the interpretation of measurement results and removal data will be presented in a later section.

4.2 Filter Media Particle Sizes and Actively Specific Surface Area

Appendix B-3 presents calculation results of actively specific surface area for filter media particles. The calculation approach was described in Chapter 3 for AOC removal analysis.

4.3 Establishing BDOC Removal-X* Relationship

As summarized in the literature review, Huck (1999) and Huck and Sozański (2008) developed a table (Table 3-5) and a Figure (Figure 3-3) for the AOC removal-X* relationship, based on the modeling work by Zhang and Huck (1996a). Recalling the simplified X* expression (Huck and Sozański, 2008):

$$X^* = \theta \alpha K (k X_f)^{1/2} \qquad (4-1)$$

where *K* is a constant,

for a given biofiltration process configuration and a given water, if both AOC and BDOC are removal targets, θ, α and X_f for them are the same. By neglecting any quantitative difference in diffusivities between the BDOC- and AOC-associated compound pools in water (then the same constant *K* values can be assumed for AOC removal and BDOC removal), Equation (4-2) can be obtained:

$$\frac{X^*_{BDOC}}{X^*_{AOC}} = \left(\frac{k_{BDOC}}{k_{AOC}}\right)^{1/2} \qquad (4-2)$$

Equation (4-2) indicates that based on several assumptions, X* values for BDOC removal can be estimated from X* values for AOC removal and the square root of the ratio of their kinetic parameter *k*.

Although AOC and BDOC removal by biofiltration have been studied for decades, there are few studies that examined AOC and BDOC removal by biofiltration at the same time, probably because both AOC and BDOC measurements are complicated, time consuming and difficult to handle. There are even fewer investigations which studied AOC or BDOC degradation kinetics. What is more, among the scarce literature in this field, reported kinetics parameter values vary greatly.

Differences between the nature of AOC and BDOC were well studied and summarized in the literature review. The fate of BDOC- and AOC- associated compounds for a given water in a given biofilter is expected to differ greatly. Comparing Figure 3-2 (for AOC removal) and Figure 4-1 (for BDOC removal), it is observed that, although there is considerable scatter in the data, biofilters were generally able to achieve higher AOC percentage removals than BDOC percentage removals in the low to moderate EBCT range (i.e. 0-15 minutes). It is believed that AOC is more readily biodegradable than BDOC because AOC-associated organic compounds are all easily biodegradable while BDOC consists of both fast degradable compounds and slowly degradable compounds (Huck, 1990; Carlson and Amy, 1997). Prévost et al. (1992) reported that up to 90% of AOC could be removed at an EBCT of only 2 minutes, while 10-20 minutes were required to remove similar quantities of BDOC; they pointed out that “using AOC to determine EBCT in BAC filters would set EBCT at much lower values than necessary for BOC (“Biodegradable Organic Carbon”-author) removal”.

Huck et al. (1994) reported average specific AOC removal rates for a pilot-scale treatment facility were 0.077-0.128 /min for anthracite-sand filters and 0.022-0.043 /min for GAC contactors. They also provided average specific AOC removal rates of 0.126 /min for dual-media filter and 0.111 /min for GAC-sand filter from another study. In the same article, an average specific removal rate of 0.032 /min for BDOC in another study was also provided for comparison, which showed that specific removal rates for AOC and BDOC were in the same order of magnitude.

Average specific removal rate (or "EBCT-normalized removal percentage" in this study) is an approximate indicator of treatment efficiency; however it is only a rough guide because it is based on the assumption of zero-order reaction and likely to underestimate degradation kinetics because the EBCT in a particular study may be unnecessarily long to achieve a specific percentage removal ("over-design" or "under-operated" in Section 5.2, Chapter 5), therefore average specific removal rate should not be used to replace reaction kinetics parameter k. Table 4-3 lists calculation results of maximum specific substrate utilization rates for BDOC and AOC in biofilters from scarce literature.

The estimated values of parameter kX_f in Zhang and Huck (1996b) varied from 290 g acetate C/m^3-d for anthracite-sand filter and 440 g acetate C/m^3-d for GAC contactor. Adopting a X_f value of 30,000 mg/L as suggested in Zhang and Huck (1996a), calculation results of maximum specific rates of substrate utilization k vary from 6.71 /min to 10.19 /min.

Yavich et al. (2004) evaluated biodegradability of NOM after ozonation, in which maximum substrate utilization rates (ML^{-3}T^{-1}) were provided for two types of raw water and under three ozonation dosing conditions. Calculation results of maximum specific rates of substrate utilization k vary from 0.01/min to 0.06/min. Because ozonation breaks large molecular organic compounds into small ones, it is likely to change biodegradability of water after ozonation. An ANOVA test performed by the author of this thesis confirmed that there was significant difference of calculated k values between non-ozonated water and ozonated water ($p<0.05$) with average values of 0.02 /min and 0.04 /min respectively. Therefore, only the calculated k values which were associated with ozonation scenarios are chosen for the following calculation.

Table 4-3 Comparison of maximum substrate utilization rates

Calculated maximum substrate utilization rate for AOC removal (Data source: Zhang and Huck, 1996b)			
	Provided kX_f ×E03 (g acetate C/m^3-d)	Provided X_f (mg/L)	Calculated in this thesis k (min^{-1})
Anthracite-sand filter1	290	30,000	6.7
Anthracite-sand filter2	340	30,000	7.9
GAC filter2	360	30,000	8.3

GAC filter3	310	30,000	7.2
GAC filter4	440	30,000	10

Average *k* value: 8.06 /min

Calculated maximum substrate utilization rate for BDOC removal
(Data source: Yavich et al., 2004)

Ozone dosages (mg O_3/mg C)	Provided maximum substrate utilization rates R_{max} (mg/L min)	Provided BDOC (mg/L)	Calculated in this thesis *k* (min^{-1})
0 (Huron River Water)	0.017	1.15	0.01
0.5 (Huron River Water)	0.088	2.31	0.04
0.75(Huron River Water)	0.087	2.55	0.03
1.0 (Huron River Water)	0.11	2.78	0.04
0 (Lake Lansing Water)	0.15	5.04	0.03
0.75(Lake Lansing Water)	0.25	6.06	0.04
1.5 (Lake Lansing Water)	0.36	6.86	0.05
3.0 (Lake Lansing Water)	0.46	7.39	0.06

Average *k* value: 0.04 /min (for ozonated influent water only)

The above calculated *k* values for BDOC are consistent with the range reported in Huck et al. (1994), but the calculated *k* values for AOC (from information provided by Zhang and Huck, 1996b), are much higher than values in Huck et al. (1994) and Zhang and Huck (1996a). The *k* value was calculated in Zhang and Huck (1996a), while it was estimated by a previous pilot-scale study in Zhang and Huck (1996b). However, in the absence of other information, the *k* values calculated above are used in the discussion below. (If different *k* values were to be obtained from new data in the future, it would not change the general nature of the following discussion.

According to Equation (4-2), X^*_{BDOC} is then equal to the square root of the ratio of calculated *k* values for BDOC and AOC ($X^*_{BDOC} \approx 1/14\ X^*_{AOC}$). This indicates that for a same biofilter and a same operation condition, dimensionless contact time X* for BDOC removal is around 7 percent of X* for AOC removal as a result of different utilization rates; on the other word, around 14 times of X* value is needed to maintain a similar removal percentage as that of AOC. Based on this ratio, a “standard” table and a figure for BDOC removal-X* relationship are established (Table 4-4 and Figure 4-2). Figure 4-2 also includes AOC removal-X* relationship for comparison.

Table 4-4 BDOC removal and X*

X*	Approximate percentage Removal
3.5	33
7	47
14	66
21	80
28	85
42	95

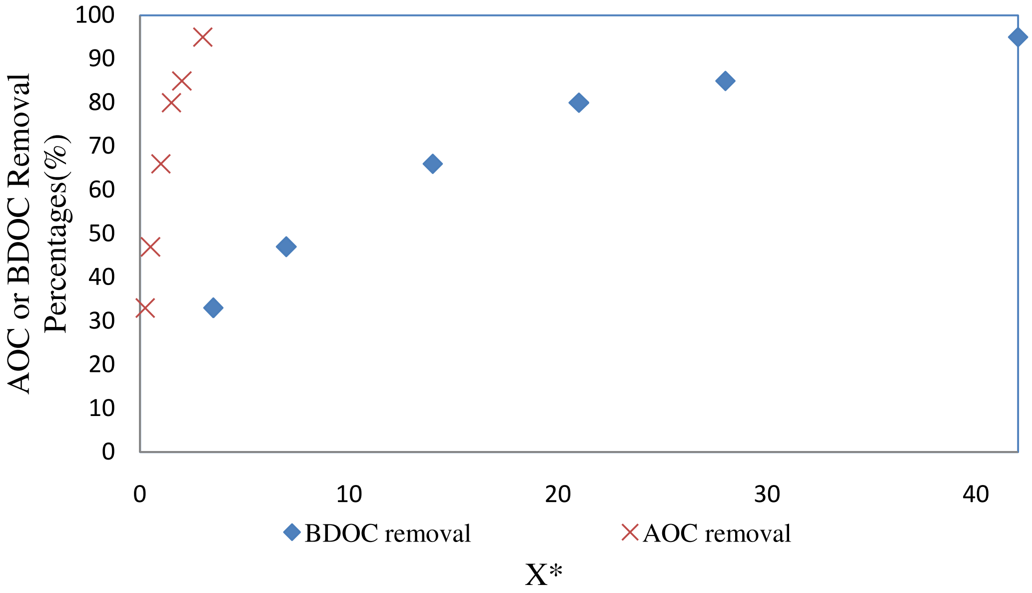

Figure 4-2 BDOC removal-X* relationship

The new established BDOC removal-X* relationship is of practical use. It suggests since BDOC degradation kinetics value is much smaller than that of AOC, more than 10 times of EBCT is then required to compensate the negative effect of low degradation kinetics and achieve a similar removal as that of AOC. This is consistent with the observation of Prévost et al. (1992) mentioned above.

Using a ruler to measure corresponding X* value of each BDOC removal percentage (Appendix B-5) on an eight times enlarged Figure 4-2, the obtained X* is defined as “Expected X*” and listed in Appendix B-6. BDOC removal efficiencies from the collected biofiltration applications range from bare removal of 5% to complete removal; expected X* values range from 1 to 43 accordingly. Removal percentage over 80% is hard to achieve for most of the collected cases. Figure 4-3 presents relationship between expected X* values and BDOC removal percentages.

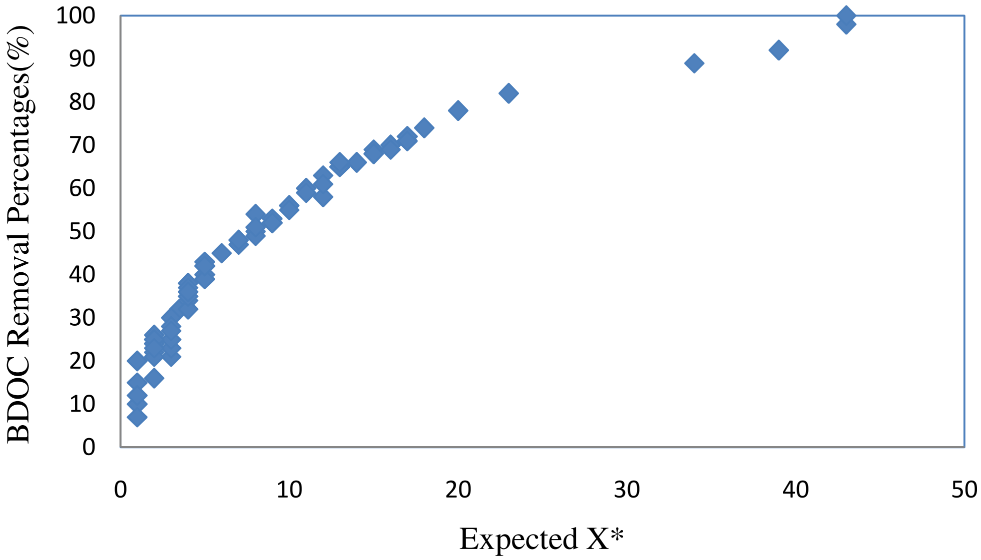

Figure 4-3 Distribution of expected X* Values

As pointed out in Chapter 3, for a particular BDOC removal goal by biofiltration, expected X* value is considered as the necessary and also lowest required X* value to achieve the removal goal. For a specific X* value, it is associated with a basic process configuration or configuration combination, which can be “expected” to achieve a specific BDOC removal efficiency under “standard” conditions discussed later in this study.

4.4 $\theta\alpha$ Parameter

The $\theta\alpha$ parameter, which was defined in Chapter 3, is used to refer to the estimated X* values without taking into account factors other than θ and α. The $\theta\alpha$ parameter is proportional to X*, assuming no change in biodegradation kinetics parameter k and biofilm density X_f. The $\theta\alpha$ parameter is estimated in this section and will undergo adjustment in later sections based on other biofilters configuration parameters, then actual X* values can be approximately by estimation.

Appendix B-5 lists calculated $\theta\alpha$ product values. Following the same procedure as in Chapter 3, $\theta\alpha$ parameter values for BDOC removal were initially estimated. To obtain $\theta\alpha$ parameter values, four sets of BDOC removal data in Griffini et al. (1999) (Table 4-5) were chosen as calculation bases. Four types of GAC were used in the study with actively specific surface area (considering factors such as porosity) of 4930 m^{-1}, 3250 m^{-1}, 3900 m^{-1} and 3870 m^{-1} respectively. Sampling depths and removal percentages were provided in the literature, EBCT values were calculated.

Table 4-5 Calculation bases for $\theta\alpha$ parameter value calculation (Data Source: Griffini et al. (1999))

GAC A filter		GAC B filter		GAC C filter		GAC D filter	
EBCT (min)	Removal Percentage	EBCT (min)	Removal Percentage	EBCT (min)	Removal Percentage	EBCT (min)	Removal Percentage
2.3	58	2.3	52	2.3	54	2.3	59
10	70	10	68	10	66	10	70
15	71	15	72	15	69	15	74
20	74	20	71	20	70	20	72

Table 4-5 shows when EBCT reached 10 minutes, removal percentages hardly increased for all biofilters. Therefore, it is reasonable to deduce that when EBCTs was greater than 10 minutes, corresponding X* value reached the curve plateau in Figure 4-3; in other words, the X* values at EBCT of 10 minutes were the minimally necessary X* values to obtain BDOC removal percentages of 70%, 68%, 66% and 70%, which means they are equal to the expected X* values associated with these removal percentages.

Table 4-6 lists $(\theta\alpha)'$ values and the corresponding expected X* values, which can be found in Appendix B-5 and B-6. The X* value based on the actual physical biofilter configuration is defined in this study as "Estimated X*". For the chosen calculation bases, when the average expected X* value was 15, the corresponding average $(\theta\alpha)'$ value was 4.0. Therefore based on this ratio (15/4.0), the $\theta\alpha$ parameter values for other BDOC biofiltration configurations are then able to be calculated (Equation (4-3)) and listed in Appendix B-5.

Table 4-6 $(\theta\alpha)'$ values and expected X* for calculation bases

$(\theta\alpha)'$	Removal Percentage	Expected X*
4.9	70	16
3.3	68	15
3.9	66	14
3.9	70	16
Average 4.0		Average 15

$$(\theta\alpha) = (\theta\alpha)'\frac{15}{4.0} \qquad (4-3)^{5}$$

4.5 Effect of Temperature on Biofiltration and $\theta\alpha$ Parameter Values

4.5.1 Collecting Temperature-Related Information

As summarized in the literature review, temperature is one of the most important factors affecting performance of biofiltration processes, including BDOC removal, since many drinking water plants experience seasonal temperature variations. Table 4-7 records temperature-related information for the collected cases of BDOC removal by biofiltration, including temperature values, ranges, conducted locations and seasons, so that one can have an approximate idea on influent water temperature of these studies. The detailed approaches for extracting temperature-related information can be found in Chapter 3. Weather references are listed in Appendix B-7.

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[5] It should be noted again that *$(\theta\alpha)'$* is multiplication product of parameter θ and parameter α ($(\theta\alpha)' = \theta \times \alpha$), while $\theta\alpha$ parameter is partially estimated X*.

Table 4-7 Temperature related information for the collected BDOC removal cases

Source	Temperature (°C) and Related Information	Conducted Location	BDOC Removal Range (%)
Fonseca and Summers, 2003	>12, 3	Colorado, USA	From 65 to 100
Griffini, et al, 1999	From 15 to 28 From 7 to 14	Florence, Italy	From 52 to 74
Heinicke et al., 2006	NA	Sweden	23
Melin et al., 2002	From 2.8 to 10	Mid-Norway	From 36 to 68
Moll et al., 1999	5, 20, 35	Ohio, USA	38, 60, 60
Niquette et al., 1998	0.5, 12, 14	Quebec, Canada	From 15 to 48
Nishijima and Speitel, 2004	25	Japan	63
Persson et al., 2006	From 1.5 to 20.2 (median 7.2)	Sweden	28, 30, 34
Prévost et al., 1992	NA January	Quebec, Canada	8, 41, 95
Sang et al., 2003	From 10 to 23	Beijing, China	From 55 to 71
Servais et al., 1991	Specific month	France	From 25 to 53
Servais et al., 1994	From 9 to 22 April or November	Paris, France	From 43 to 56
Thiel, et al., 2006	NA	Australia	From 12 to 23
Wang et al., 2000	NA May to October	Northern China	78
Wang and Summers, 1996	NA	Ohio, USA	From 16 to 40
Zappia et al., 2007	March	Australia	From 5 to 25

Note: 1. Refer to section 3.5.2 for screening criteria;
2. Refer to Appendix B-7 for weather references.

4.5.2 Effect of Temperature on $\theta\alpha$ Values

The effect of temperature on $\theta\alpha$ values, as well as the rationale to adjust $\theta\alpha$ parameter values under different temperature ranges, was discussed in detail in Section 3.5, Chapter 3. Recalling Equation (3-13) and Equation (3-16):

$$X^* = (\theta\alpha)'KK' = (\theta\alpha)K_2 \times K' \qquad (3-13)$$

$$\frac{K'_{20}}{K'_{T}} = \left(\frac{k_{20}}{k_{T}}\right)^{1/2} = \lambda^{10-\frac{T}{2}} \qquad (3-16)$$

Information on kinetics parameter k values of under different temperatures for BDOC removal by biofiltration is scarce; therefore λ values in Elhadi (2004) were directly used in this study. Unlike in Chapter 3, an average λ value of 1.15 was adopted as a uniform coefficient for all types of biofilters in this chapter in order to provide a different view on the temperature adjustment. This λ value is consistent with the mean value (1.135) of 1.02-1.25 in Metcalf and Eddy (2003, page 55 and 585) provided for wastewater.

Among the collected BDOC removal cases, reported temperature ranged from 0.5°C to 35°C. In this section, three temperature categories were established for all filters (0°C-5°C, 5°C-10°C and 10-15°C). The median temperature of each category (2.5°C, 7.5°C and 12.5°C) was used as a representative temperature and substituted into Equation (3-18). The stimulatory effect of high temperature over 15°C on biodegradation kinetics was ignored and 1 was assigned as the value for the corresponding adjustment coefficient. Table 4-8 shows the adjustment coefficients under the different temperature ranges.

Table 4-8 Adjustment of $\theta\alpha$ values under different temperature ranges (λ =1.15)

Temperature Division	Representative Temperature	Adjustment Coefficient for $\theta\alpha$ Parameter Values
>15°C	20°C	1
10-15°C	12.5°C	1.69
5-10°C	7.5°C	2.40
0-5°C	2.5°C	3.40

4.5.3 Development of Temperature-Adjusted BDOC Removal-$\theta\alpha$ Relationship

The “standard” BDOC removal-X* relationship (Figure 4-2 and Table 4-4) was developed for biofiltration processes with influent water temperature of near room temperature (20°C). Using the temperature adjustment coefficients in Table 4-8, Table 4-9 and Figure 4-4 were developed to show approximate $\theta\alpha$ relationships under different temperature ranges for all biofilters.

Table 4-9 Approximate relationship between BDOC percentage removals and $\theta\alpha$ parameter values for all biofilters under different temperature ranges

Approximate BDOC percentage Removal	$\theta\alpha$ (≥15°C)	$\theta\alpha$ (10~15°C)	$\theta\alpha$ (5~10°C)	$\theta\alpha$ (≤5°C)
33	3.5	6.0	8.0	12.0
47	7	12	17	24
66	14	24	34	48
80	21	36	50	71
85	28	47	67	95
95	42	71	100	140

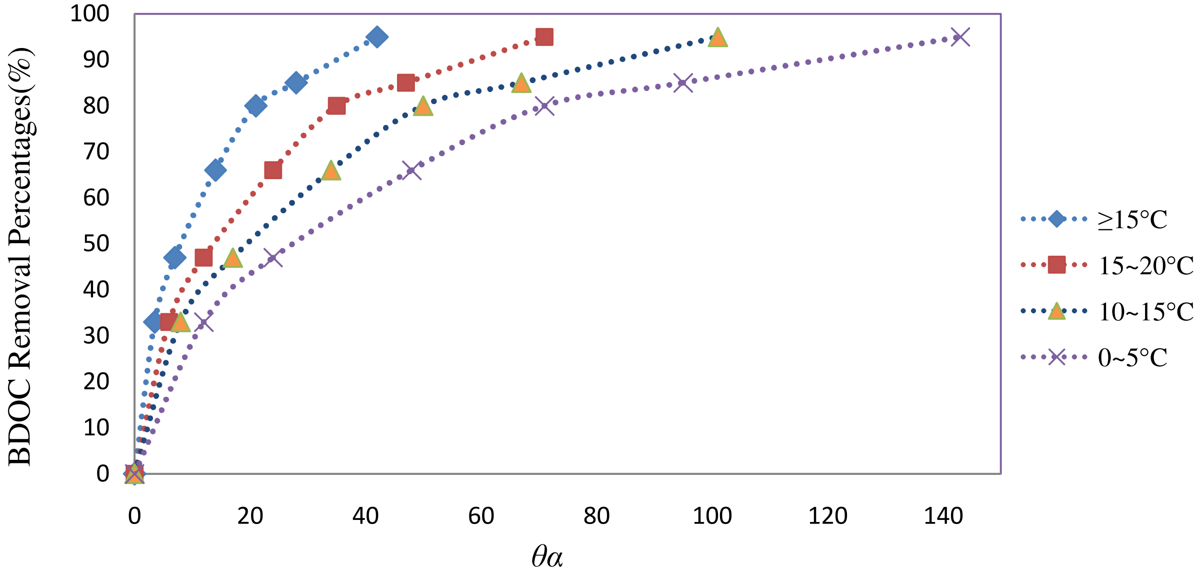

Figure 4-4 BDOC removal-$\theta\alpha$ for biofilters under different temperature ranges

The lower the temperature, the greater the negative effect is for biofilter performances and $\theta\alpha$ values. For example, $\theta\alpha$ needs to be greater than 24 to maintain an approximate 50% BDOC removal under 5°C, comparing with $\theta\alpha$ value of 35×10^{-1} at 20°C for BDOC removal and $\theta\alpha$ value of 5×10^{-1} for AOC removal at 20°C.

This set of BDOC removal-$\theta\alpha$ relationship is of practical use. For a given BDOC removal percentage requirement, because the biodegradation kinetics parameter k is smaller at low temperature than at warm temperature, a greater product of $\theta\alpha$ is needed to compensate the negative effect of temperature on performance. Increasing EBCT is expected to maintain a satisfactory BDOC removal in cold weather. Detailed interpretation of similar information on AOC comparable to that in Table 4-9 and Figure 4-4 can be found in Section 3.5.4, Chapter 3.

4.5.4 Temperature-Adjusted $\theta\alpha$ Values

With temperature related information (temperature values, temperature ranges, conducted locations and seasons) in Table 4-7 and temperature division ranges in Table 4-8 as well as the assistance of weather information (notes in Appendix B-7), a representative temperature range for each BDOC removal case was determined and listed in Appendix B-7. The approaches for determination of representative temperature range were specified in Chapter 3. With the assistance of representative temperature ranges, temperature-adjustment coefficients were determined and listed in Appendix B-7 for each BDOC removal case. Appendix B-8 presents the adjustment results of $\theta\alpha$ parameter values. Figure 4-5 compares the temperature-adjusted $\theta\alpha$ parameter values and the initially estimated $\theta\alpha$ parameter values. The temperature-adjusted $\theta\alpha$ parameters are distributed more tightly.

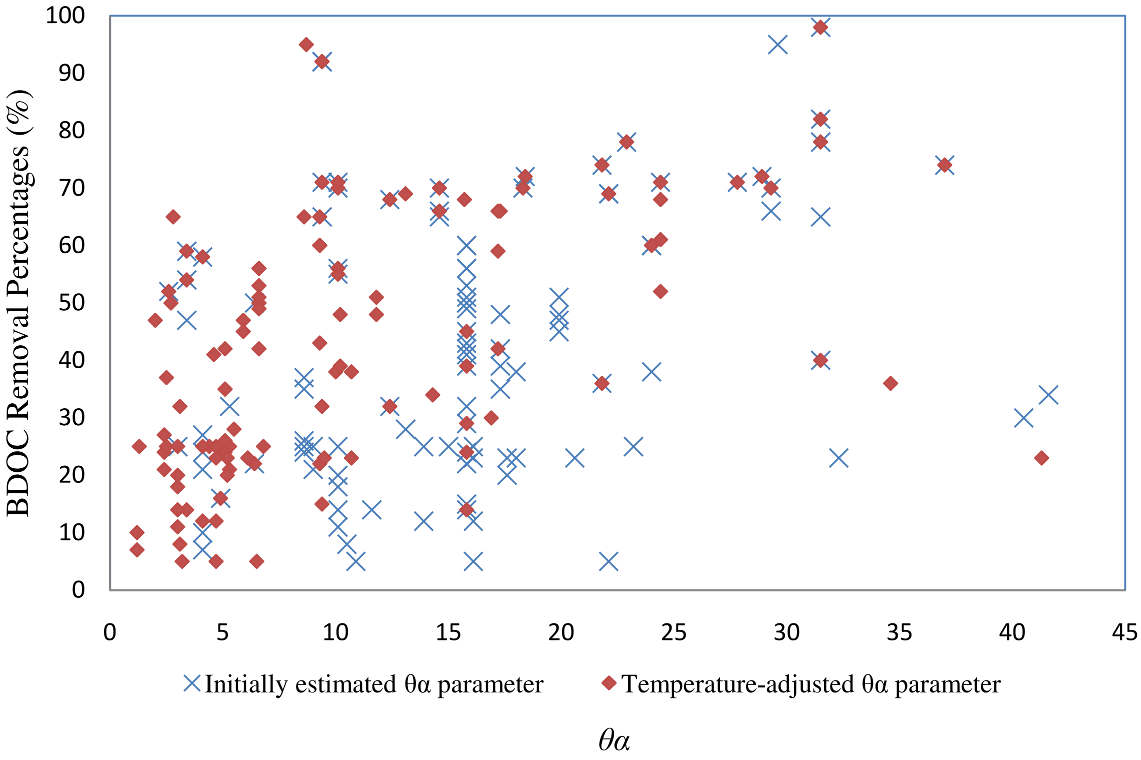

Figure 4-5 Comparison of temperature-adjusted $\theta\alpha$ values and initially estimated $\theta\alpha$ values

Figures 4-6, 4-7, 4-8 and 4-9 compare distributions of temperature-adjustment $\theta\alpha$ parameter values under different temperature ranges with the corresponding temperature-adjusted BDOC removal-$\theta\alpha$ relationships.

Figure 4-6 particularly presents comparison between the established BDOC removal-$\theta\alpha$ relationship at around 20°C and distribution of $\theta\alpha$ parameter values for temperature-favored biofiltration processes (in this study, processes with raw water temperature above 15°C). As shown in Appendix B-7, the temperature-adjustment coefficients for temperature-favored biofiltration processes were set as 1. $\theta\alpha$ parameter values associated with temperature-favored biofiltration processes were not temperature-adjusted because it is assumed that temperature above 15°C but under 20°C would just have minor impairment of performances of biofilters, and the stimulatory effect of temperatures over 20°C on biofiltration is not a concern in practice. The size of the analyzed data pool is larger significantly by including BDOC removal data associated with this temperature range. In other words, $\theta\alpha$ parameter values in Figure 4-6 are $\theta\alpha$ parameter values without adjustment.

It appears that in Figure 4-6, with some exceptions where associated BDOC removal efficiencies were much lower than expected, distribution of initially estimated $\theta\alpha$ parameter values falls into a reasonable range of the BDOC removal-$\theta\alpha$ relationship, based on the BDOC-X* relationship established in Section 4.3. It confirms again that the approaches and the chosen parameters in section 4.3 are reasonable, and the established BDOC removal-X* relationship is a useful tool to evaluate and predict performances of biofilters.

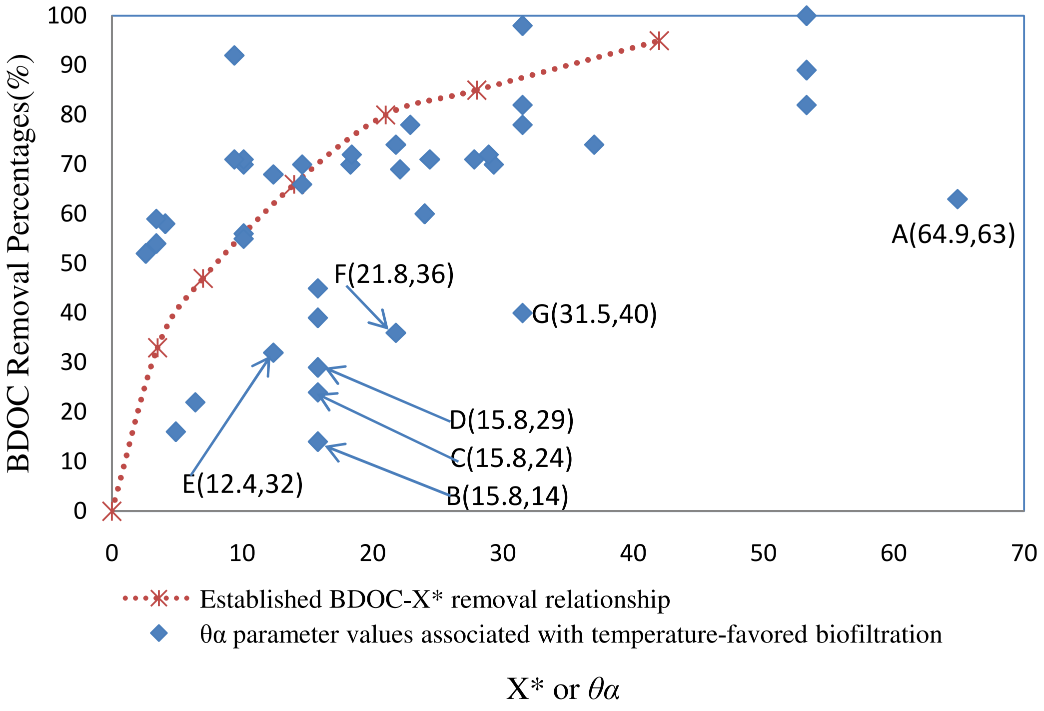

Figure 4-6 Distribution of $\theta\alpha$ associated with temperature-favored (i.e., temperature ≥15°C) biofiltration

“Substantially diverging” points in Figure 4-6 are associated with removal efficiencies much lower than expected. Table 4-10 summarizes detailed information of these points.

Table 4-10 “Substantially diverging” points in Figure 4-6

(Initially estimated $\theta\alpha$ values were much less than expected $\theta\alpha$ parameter values for temperature-favored biofiltration)

Source	$\theta\alpha$ Values	Expected $\theta\alpha$ Parameter	BDOC Removal (%)

Nishijima and Speitel, 2004 (Point A)	64.9	12.0	63
Servais et al., 1991 (Point B, C, D)	15.8	1.0	14
	15.8	2.0	24
	15.8	3.0	29
Wang and Summers, 1996 (Point E, F, G)	12.4	4.0	32
	21.8	7.0	36
	31.5	10.0	40

For the BDOC removal study in Nishijima and Speitel (2004), fulvic acids extracted from soils were specially treated and used as filter influent; biodegradable fractions in the extracted fulvic acids were removed before experiments by aeration for two weeks, thus there was no BDOC in the influent before ozonation process. Low biodegradability of significantly modified influent water and more limited range of substrate are likely to be responsible for the low BDOC removal.

For the full-scale study in Servais et al. (1991), three removal cases listed in Table 4-10 were associated with much higher BDOC and DOC levels in the influent than the average loading, whereas the treatment plant might use a uniform ozonation dosage, rather than carbon-normalized ozone dosage according to water quality, or the treatment plant might not have adequate time to respond to changes of influent water quality by adjusting applied ozone dosages to reach a preset organic carbon-normalized dosage. BDOC biodegradability of these three removal cases would then be smaller because of inadequate oxidation and cause low removal percentages. Another possible reason is that the microbial community may not have had the ability to adapt to water quality transients without acclimation.

The influent of the biofilter was a specially treated NOM stock solution (average molecular weight of 1500) in Wang and Summers (1996). Low biodegradability of the significantly modified influent water and a more limited range of substrate are also likely to be responsible for the low BDOC removal. It is an interesting finding that in that study AOC removal was quite efficient (59% AOC removal at EBCT of 0.4 minutes). This observation is also consistent with the conclusion in other studies, such as Charnock and Kionno (2000), that it is likely that the AOC and BDOC analyses target different fractions of BOM. Prévost et al. (1992) also observed different removal trends between AOC and BDOC.

Figures 4-7 and 4-8 and to some extent Figure 4-9 show that, based on visual examination as discussed in relation to Figure 3-5, plotting of temperature-adjusted $\theta\alpha$ parameter values based on actual process configuration all reasonably match the BDOC removal- $\theta\alpha$ relationships under temperature ranges of 10-15°C, 5-10°C and <5°C respectively, especially for the low ends where temperature-adjusted $\theta\alpha$ parameter values are less than 20. As a matter of fact, most practically applied temperature-adjusted $\theta\alpha$ parameter values are less than 20.

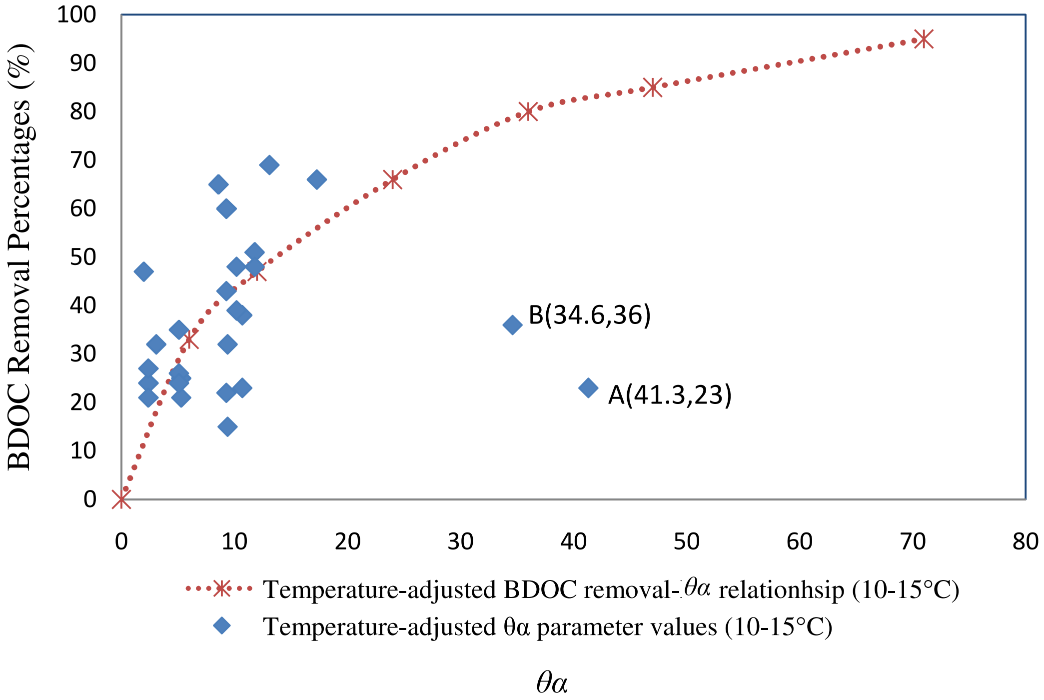

Figure 4-7 Temperature-adjusted BDOC-$\theta\alpha$ relationship and distribution of temperature-adjusted $\theta\alpha$ parameter (10-15°C)

Two “substantially diverging” points are manifest in Figure 4-7. Point A is associated (Heinicke et al., 2006), with a situation in which biofilters received raw water without pre-treatment and therefore the influent water had a low biodegradability; point B is associated with a water temperature of just 10°C (Melin et al., 2002). According to the approach of determining the representative temperature range in Chapter 3, if the provided temperature was just 10°C, the upper division (10-15°C) was chosen as the representative temperature range. Therefore, the initially estimated $\theta\alpha$ parameter was adjusted by 1.69 (for range of 10-15°C), rather than by 2.40

(for range of 5-10°C). If it were in the lower division (10-15°C), point B would not be a "substantially diverging" point in Figure 4-7.

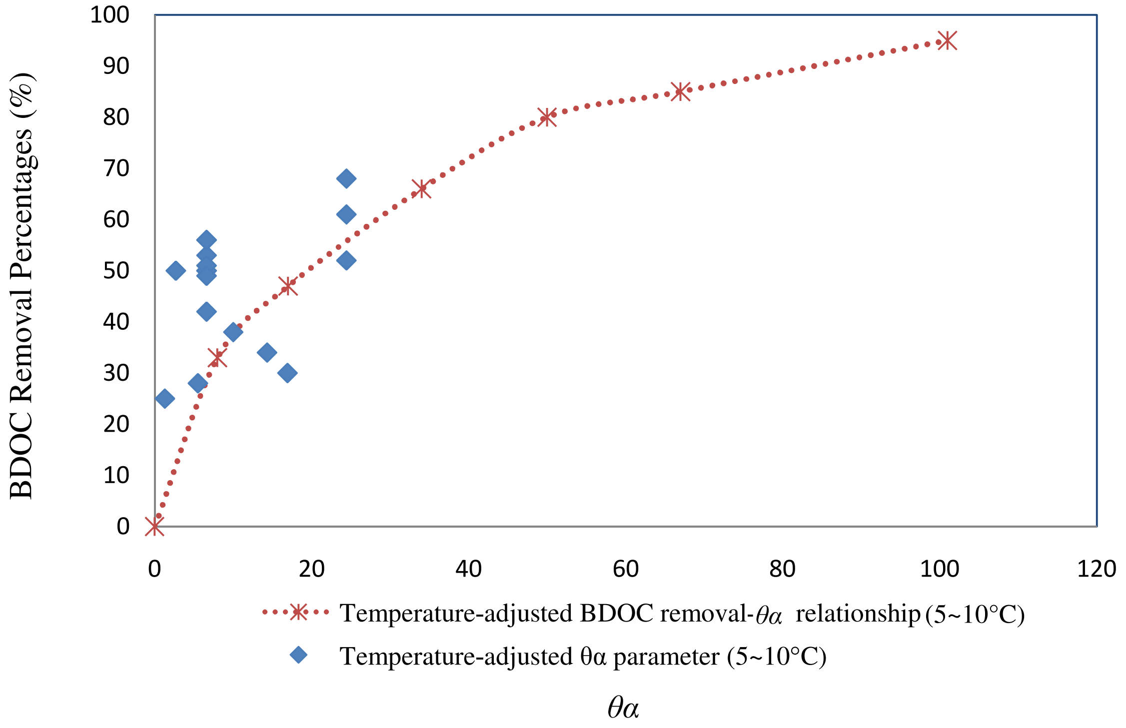

Figure 4-8 Temperature-adjusted BDOC-$\theta\alpha$ relationship and distribution of temperature-adjusted $\theta\alpha$ (5~10°C)

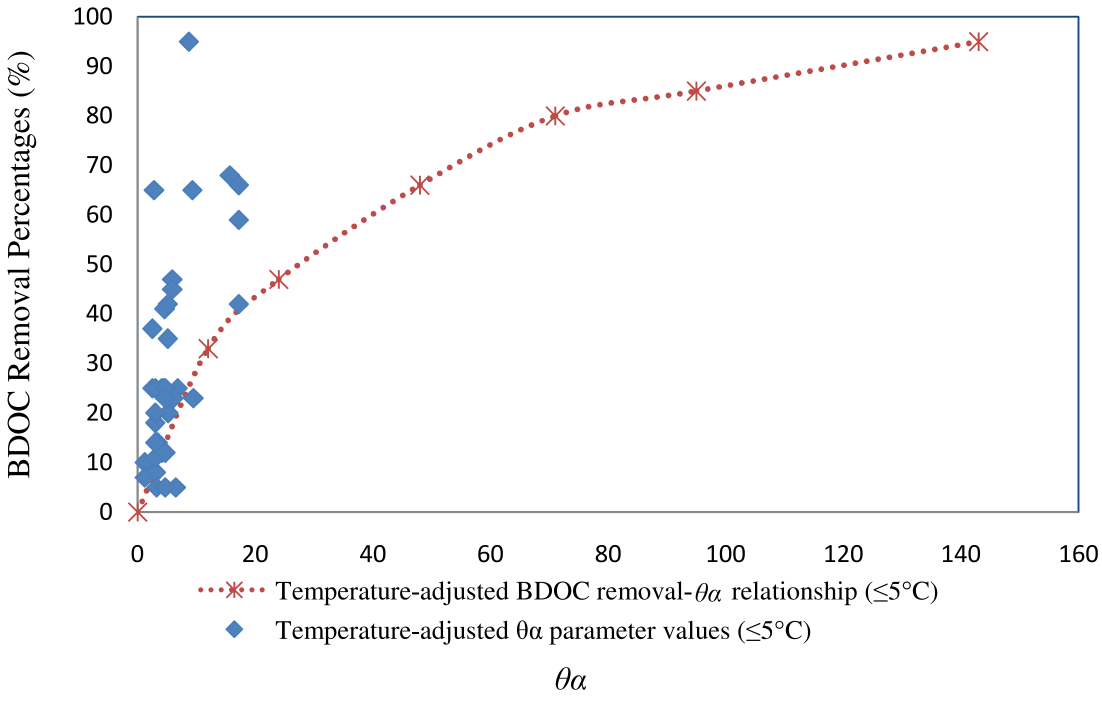

Figure 4-9 Temperature-adjusted BDOC-*θα* relationship and distribution of temperature-adjusted *θα* (≤5°C)

In Figure 4-9, there are quite a few "substantially diverging" points s with the temperature-adjusted *θα* parameter values much smaller than the necessarily expected *θα* values at this temperature range. In other words, for these "substantially diverging" points, the associated biofiltration processes performed much better than expected in terms of BDOC removal efficiencies.

From Figure 4-9, the *θα* value needs to be greater than around 30 to achieve 50% BDOC removal, for which a very long contact time is required. However, there are a number of reports (associated with the "substantially diverging" points) in which fair to good BDOC removals were still achieved under 5°C and with moderate EBCTs. Melin et al. (2002) observed there was no correlation between water temperatures and BDOC removals and significant BDOC removals were still able to be obtained if water temperature was only 3-4°C. Laurent et al. (1999) attributed the underestimation of BDOC removals at low temperature to the ability of biomass to acclimate to a cold environment, less predation mortality and even relative composition changes of organic compounds in winter. The modified model considering these factors in Laurent et al.

(1999) then predicted BDOC removals satisfactorily, based on visual observation as defined in relation to Figure 3-5.

4.6 Consideration of Effect of Ozonation on Biofiltration Processes and $\theta\alpha$ Values

4.6.1 Collecting Ozonation-Related Information and Calculation of Ozone Dosages

As summarized in the literature review, pre-ozonation is also one of the most important factors affecting biofiltration performance of for BDOC removal. Table 4-11 records ozonation information for the collected cases of BDOC removal by biofiltration. A detailed description of extracting information from the literature, calculation methods and calculation results discussion was given in Section 3.6.1, Chapter 3. Only cases with adequate ozone dosing information were chosen for the following discussion.

Table 4-11 TOC and pre-ozonation information for the collected BDOC removal cases

Source	TOC(mg/L)**	Ozone Dosage(mg/L)**	Ozone Dosage (mgO_3/mg TOC)** ***
Fonseca and Summers, 2003			0 mgO_3/mg DOC$^{(1)(2)}$ 1.30 mgO_3/mg DOC$^{(1)(3)(4)}$
Griffini, et al., 1999			0.70-1.60 mgO_3/mg DOC$^{(1)}$
Heinicke et al., 2006	4.7	0$^{(5)}$	0
Melin et al., 2002			1.50
Moll et al., 1999			1.30
Niquette et al., 1998		NA$^{(6)}$	NA
Nishijima and Speitel, 2004			NA$^{(7)}$
Persson et al., 2006		0$^{(5)}$	0
Prévost et al., 1992		NA	NA
Sang et al., 2003		0$^{(5)}$	0
Servais et al., 1991	NA	1.00-2.00	NA

Servais et al., 1994		NA	NA
Thiel, et al., 2006			0(8) (with other pre-oxidation)
Wang et al., 2000	11	1.5(9)	
Wang and Summers, 1996			0.35(1) (mg O_3/mg DOC)
Zappia et al., 2007			0(8) (with other pre-oxidation)

Note: * Average is used for calculation;
** Refer to 3.6.1 for collecting TOC and ozonation information;
*** Refer to 3.6.1 for calculation of ozone dosages;

1. Expressed as mg O_3/mg DOC;
2. No ozonation, modified BDOC measurement method, data excluded in calculations;
3. First-stage ozonation, modified BDOC measurement method, data excluded in calculations;
4. Transfer efficiency between 70% to 90%, average used for calculation;
5. Receiving raw water, nutrient addition, aeration,; data excluded in calculations;
6. Ozone residual concentration is available, dosage information is not available;
7. Special treatment for influent water;
8. Other irregular pre-treatment processes, data excluded in calculations;
9. Water reclamation context.

4.6.2 Effect of Ozonation on Biofiltration and $\theta\alpha$ Values

Appendix B-9 lists calculation results of carbon-normalized ozone dosages for the chosen BDOC removal cases by biofiltration. Five rows in the top of Appendix B-9 are associated with biofilters without pre-ozonation processes. Only immediate pre-ozonation data was included. Because a modified BDOC measurement method was used and the removal results were irregularly high in Fonseca and Summers (2003), data associated with this study were excluded. Data were also excluded if there were other pre-oxidation or irregular pre-treatment units/operations before biofiltration, such as peroxide addition, chlorination and chloramination, even if normalized ozone dosages were given or could be calculated. Only immediate pre-ozonation data were included. Figure 4-10 presents plotting of ozone dosages vs. BDOC removal percentages. Comparing Figure 3-14 in Chapter 3 for AOC removal analysis, the data pool for Figure 4-10 is much smaller.

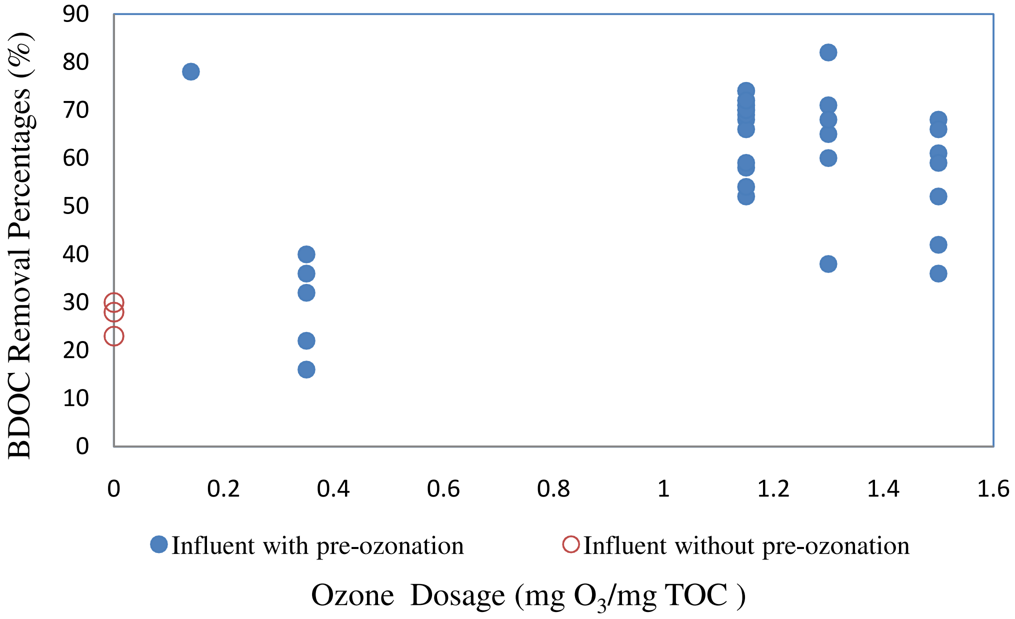

Figure 4-10 Ozone dosages and BDOC removal percentages

For biofiltration processes with pre-ozonation, the average ozone dosage is above 1.10 O_3/mg TOC, which may suggest that a higher ozone dosage is needed to achieve satisfactory BDOC removal than AOC removal. An ANOVA test performed by the author of this thesis indicated there was a significant difference in average BDOC removal efficiencies between ozonated and non-ozonated influent for biofilters, with the average BDOC percentage removals of 60% and 27% respectively ($p<0.05$). This may also suggest ozonation not only increases the absolute amount of BDOC for the following biofilter but also increase the biodegradation kinetics of BDOC given that the contact time in a filter is much shorter than that for the BDOC test) by changing the relative composition of various BOM components.

For reference, Figure 4-11 also presents comparison of BDOC removal percentages vs. the corresponding temperature-adjusted $\theta\alpha$ parameter for biofilters with and without pre-ozonation for the data pool in this section.

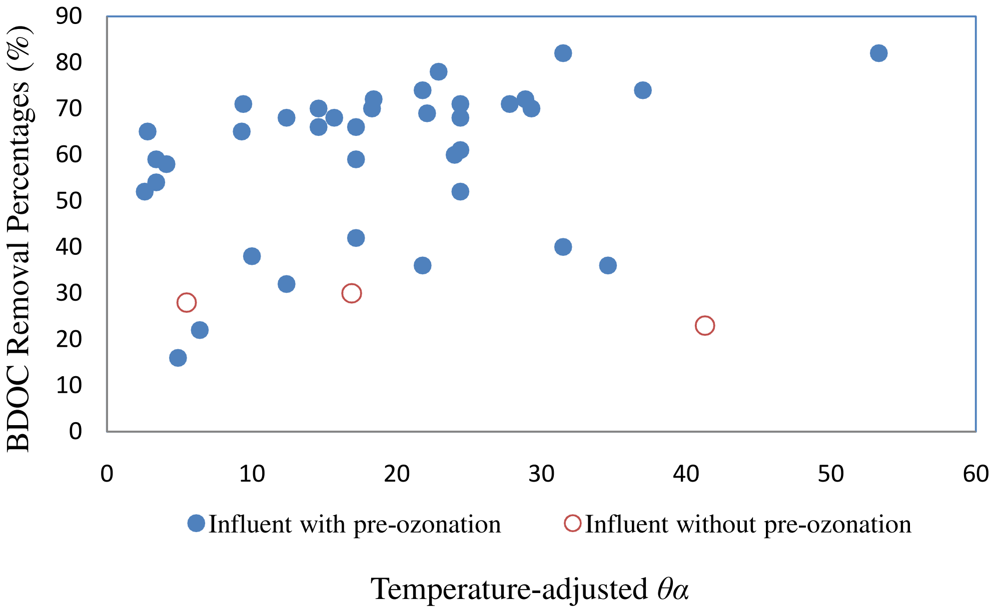

Figure 4-11 Temperature-adjusted $\theta\alpha$ values for biofilters with and without pre-ozonation

4.7 Consideration of Effect of BOM Measurement on Biofilter Performance

BDOC measurement requires a long incubation time for tested water samples (five days to three weeks, depending on the method). Several researchers therefore reported attempts to simplify the procedure and shorten the incubation time (Wang and Summers, 1994; Carlson et al., 1996; Carlson and Amy, 1999).

Figure 4-12 sketches an approximate hypothetical BDOC profile with incubation time. It shows with increasing incubation time, BDOC measurement results increases rapidly with a short time and then barely increases after further incubation time. The rapidly degradable fraction of BDOC ($BDOC_{rapid}$) can likely be substantially removed in a biofilter and the slowly degradable fraction of BDOC ($BDOC_{slow}$) is likely largely unaffected by a biofilter. Ozone dose was found to increase $BDOC_{rapid}$ within a certain range, while $BDOC_{slow}$ was insensitive to ozone dose (Wang and Summers, 1994; Carlson et al., 1996; Carlson and Amy, 1999).

Carlson and Amy (1999) concluded that BDOC measurement from biofilters with 60 minutes contact time ($BDOC_{filter}$) could be used as a surrogate for $BDOC_{slow}$ since DOC levels were barely reduced further after 60 minutes.

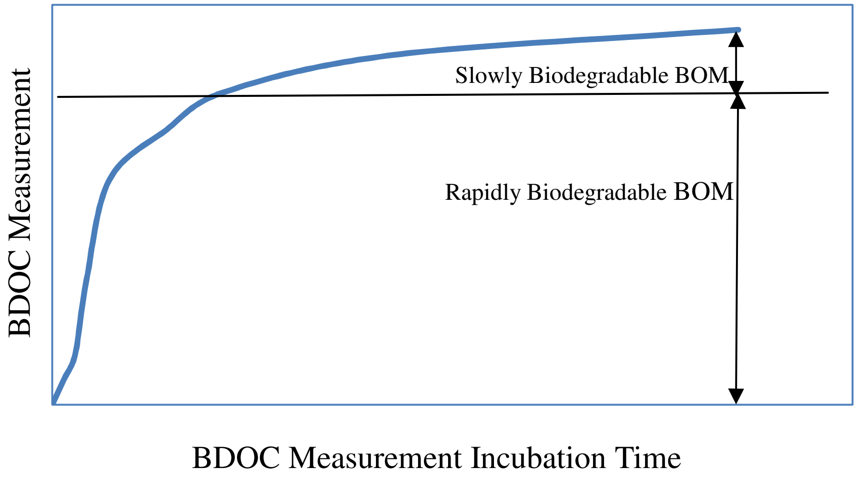

Figure 4-12 Relationship between BDOC measurement and incubation time

The discussion above may indicate that the incubation time for the standard BDOC measurement is longer than necessary in terms of describing removal of BDOC by biofiltration processes. This may be true also for AOC measurements and may also explain some unusual removal data encountered in this study.

4.8 Conclusions

X* analysis was performed for BDOC removal by biofiltration in this chapter. Wide ranges of media sizes, EBCTs, temperatures and a high diversity of pre-treatment and operation conditions for the collected cases were able to well represent biofiltration practices for BDOC removal by biofiltration.

No significant co-relationship between EBCT and BDOC removal percentage were found, indicating that EBCT alone is not able to guide biofiltration design and operation. BDOC removal efficiencies were found lower than those of AOC.

Using the model-based AOC removal-X* relationship and comparing kinetics parameter between AOC and BDOC removal, a standard BDOC removal-X* relationship was established. The results confirm that more than 10 times X* or EBCT is required to achieve a similar removal percentage as that of AOC.

Expected X* values were obtained. $\theta\alpha$ parameter values were estimated by comparison of ratios of $\theta\alpha$ products based on properly chosen calculation bases. The finding can be also considered as a "survey" of results of θ, α, $\theta\alpha$ and $\theta\alpha$ parameter value among various studies and can narrow down the scope of BDOC removal biofiltration process configuration and treatability investigations.

The raw water temperature range for each collected case was determined with the assistance of information on temperatures, temperature ranges and conducted season and location. The distribution of $\theta\alpha$ parameter values for temperature-favored biofiltration processes matched the established BDOC-$\theta\alpha$ relationship reasonably, based on visual comparison, and as defined in relation to Figure 3-5, i.e. with no systematic departure of the data set from the expected relationship. Really high BDOC percentage removals were not observed for the collected cases.

Using the average value of the temperature activity coefficients in Elhadi (2004), temperature-adjustment coefficients for $\theta\alpha$ parameter values were determined under different temperature ranges for all types of biofilters. Temperature-adjusted BDOC removal-$\theta\alpha$ relationships were developed and temperature-adjusted $\theta\alpha$ parameter values were also estimated. Temperature adjusted $\theta\alpha$ values were distributed in a relatively narrow range. Comparisons were conducted between temperature adjusted AOC removal- $\theta\alpha$ relationships and temperature- adjusted $\theta\alpha$ parameter values and reasonable matches based on visual comparison were found. "Substantially diverging" points far from the expected relationship were analyzed for possible reasons.

For BDOC removal by biofiltration at low temperature (i.e. under 5°C), there are quite a few "substantially diverging" points with temperature-adjusted $\theta\alpha$ parameter values much smaller than necessarily expected $\theta\alpha$ values at this temperature range. In other words, for these "substantially diverging" points, the associated biofiltration processes performed much better than expected in terms of BDOC removal efficiencies. Underestimation of the ability of biomass acclimation to cold environment may be the main reason.

TOC-normalized ozone dosages were calculated for part of the collected BDOC removal cases. The average ozone dosage was 1.10 O_3/mg TOC for biofiltration processes with pre-ozonation and falls into the suggested range reasonably. A higher ozone dosage is needed to achieve satisfactory BDOC removal than AOC removal. Although no relationship was found between ozone dosages and BDOC removal percentages, an ANOVA test indicated there was significant difference of average BDOC removal efficiencies between ozonated and non-ozonated influent for biofilters. This may also suggest ozonation not only increase the absolute amount of BDOC for the following biofilter but also increase the biodegradation kinetics of BDOC after ozonation by changing the relative composition of various BOM components.

BDOC measurement and fractionation were discussed, indicating that the incubation time for the standard BDOC measurement may be longer than necessary in terms of describing removal of BDOC by biofiltration processes. This may also explain some unusual removal data encountered in this study.

In general, examining X* (or partially estimated X*, which is $\theta\alpha$ parameter) provided more useful information in terms of evaluation and prediction of BDOC removal by biofiltration than simply considering EBCT, which confirms that X* is a better parameter for biofiltration design and operation.

Chapter 5

X* Analysis for Other Biofiltration Applications and Using the X* Concept to Evaluate Biofiltration Performance

In addition to having the ability of reducing levels of biodegradable organic carbon (e.g. AOC and BDOC) and controlling bacterial regrowth potential in distribution systems, biological filtration processes have been used in drinking water treatment for additional objectives in recent years, such as removing some disinfectant by-products (and precursors for others), trace containments and being used as a pre-treatment process for membrane processes. As discussed in Huck and Sozański (2008), the X* concept can be extended to other applications as well as to AOC and BDOC reductions in Chapter 3 and 4.

Using trace contaminants removal (secondary utilization) as an example, this chapter considers X* (or at least X* components) for the performance of the other applications and also explores the promise and appropriateness of the X* concept as a unifying parameter for biofiltration design and operation. Analysis results and discussions are presented here. Calculation results are presented in Appendix C.

5.1 Attempting to Use the X* Concept to Analyze the Removal of Odorous Compounds by Biofiltration Processes

Occurrences of trace contaminants (such as taste/odorous compounds and pharmaceuticals/ personal care products) are often at very low concentrations (e.g., at ng/L levels) in raw water, therefore biodegradation of these compounds in biofilters was hypothesized as occurring through secondary utilization (Huck et al., 1995). Two most problematic odorous compounds, MIB and geosmin, are chosen in this chapter to illustrate the extension of X* concept to secondary utilization. Studies have shown biofilters were able to remove MIB and geosmin from drinking water satisfactorily through biodegradation (Yagi et al., 1988, Terauchi, et al., 1995 and Sinha et al., 2003).

5.1.1 Applicability of Simplified X* Expression

Recalling X* expression and X* simplified expression:

$$X^* = \theta \alpha {D_f}^{1/2} (kX_f/K_s)^{1/2} \qquad (5-1)$$

$$X^* = \theta \alpha K \left(kX_f\right)^{\frac{1}{2}} \qquad (5-2)$$

(As indicated previously, the simplified X* expression (Equation (5-2)) is obtained by assuming first order reaction and that mass transfer is not limiting for Equation (5-1)).

Although first-order reaction kinetics have been well documented for removal of organic carbon parameters by biofiltration (Huck et al., 1994), studies regarding the biodegradation rates of MIB and geosmin are limited and inconsistent. Rittmann et al. (1995), Ho et al. (2007) and Westerhoff et al. (2005) proposed a second-order, pseudo-first-order and pseudo-zero-order reaction respectively. Elhadi (2004), who had a somewhat limited data set from the perspective of establishing reaction order, found the experimental data fit a second-order, first-order and zero-order reaction all satisfactorily and for in-depth modeling, selected the zero-order reaction kinetics for its simplicity in the study.

As Huck and Sozański (2008) suggested, although it is not necessary to use the simplified expression for X* to apply it to secondary utilization, it is conceptually simpler to do so. However, it should be kept in mind that adopting first-order reaction kinetics and then applying the simplified X* expression for MIB and geosmin degradation in biofilters may receive comment, although it can be considered a reasonable approach.

5.1.2 Collecting Data on MIB and Geosmin Removal by Biofiltration

The collected cases for MIB and geosmin removal by biofiltration are summarized in Table 5-1; information includes source and study scale, media type, summary of pre-treatment and operation conditions and range of percentage removal. Appendix C-1 lists detailed reference information.

Table 5-1 Summary of the collected cases for MIB and geosmin removal by biofiltration processes

Source/Scale$^{(1)}$	**Media Type**	**Pre-treatment and Operation Conditions**	**Taste and Odor Compounds Removal (%)**
Elhadi et al., 2004 (B)	GAC/Sand	No ozonation; Typical ozonation by-products amended; MIB and Geosmin spiked; Temperature 12-16°C.	MIB 50 Geosmin 82
Elhadi et al., 2006 (B)	GAC/Sand; Anthracite/Sand	No ozonation, Typical ozonation by-products amended; MIB and Geosmin spiked to high/low level; Temperature 8 and 20°C.	MIB 1-36 Geosmin 4-60
McDowall, et al., 2007a (B)	Sand	MIB and Geosmin spiked; Room temperature.	MIB 12-95 Geosmin 36-95
McDowall, et al., 2007b (B)	Sand	MIB and Geosmin spiked; Room temperature.	MIB around 95 Geosmin around 95
McDowall, et al., 2009 (B)	Sand	Seeding filter with specific Geosmin bacteria; Room temperature.	Geosmin 40-75
Persson et al., 2007 (B)	GAC; Expanded Clay	MIB and Geosmin spiked; Temperature 6-12°C and 15°C	MIB 37-97 Geosmin 45-97

Note: (1) B-Bench scale

Media types include granular activated carbon, sand, anthracite, expanded clay (EC), as well as dual-media; a consortium of geosmin-degrading bacteria was seeded into the sand filter columns in one study; MIB and geosmin were spiked in prepared influent or raw water; temperature ranged from 8°C to 20°C; recorded EBCT values range from 3.5 minutes to 30 minutes. MIB and geosmin percentage removal ranged from 1% and 4% respectively, to 97%.

Filter media sizes for the collected biofiltration cases are summarized and calculated in Appendix C-2. Calculation results of actively specific surface area of filter media particles are presented in Appendix C-3. Appendix C-4-1 and C-4-2 list empty bed contact time values, percentage removals, calculated EBCT-normalized percentage removals and $\theta\alpha$ parameter values for MIB and geosmin respectively. The ways used to obtain these values are also indicated. The calculation methods can be found in Chapter 3.

Figure 5-1 and 5-2 present the relationship between percentage removals and EBCTs for MIB and geosmin respectively.

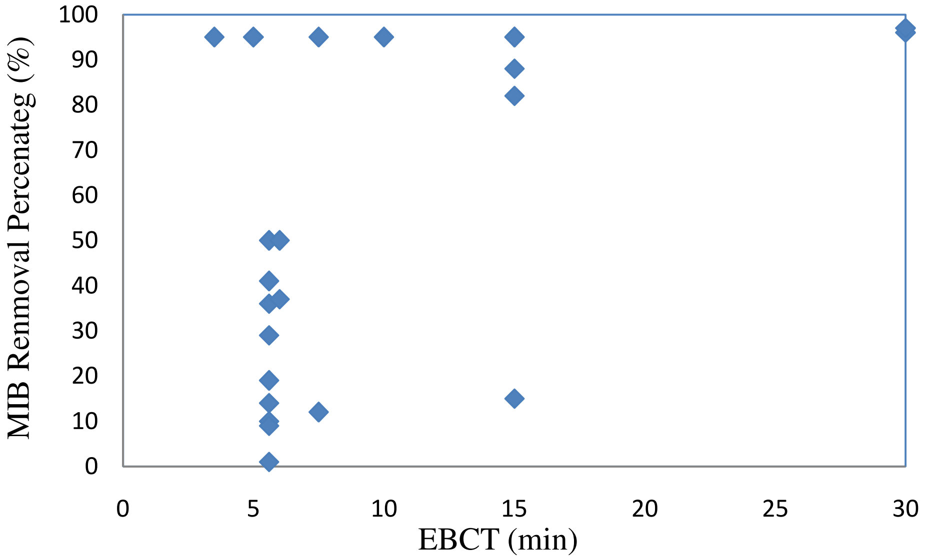

Figure 5-1 Distribution of EBCTs over MIB removal percentages

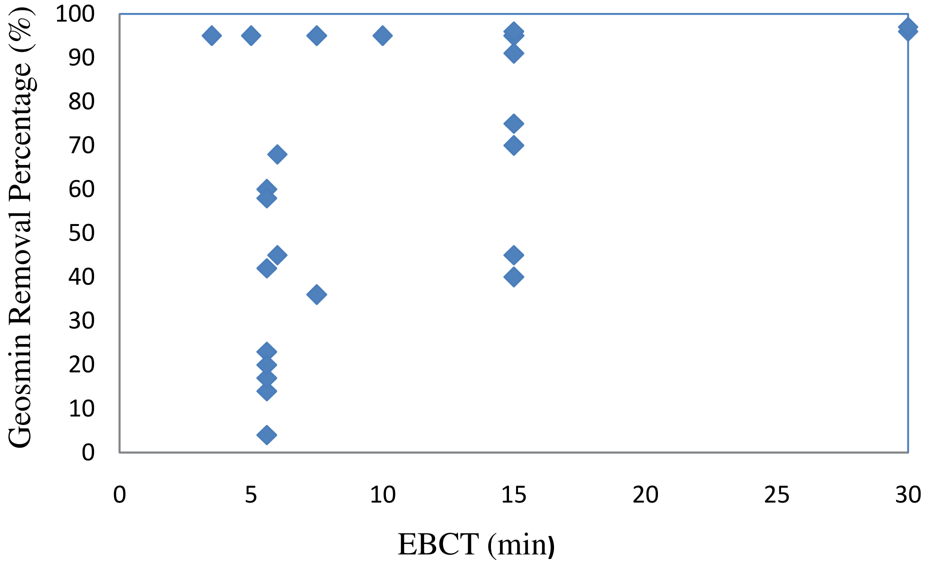

Figure 5-2 Distribution of EBCTs over geosmin removal percentages

Figures 5-1 and 5-2 demonstrate that the biofiltration process is able to reduce odorous compounds, however no significant co-relationship between EBCT and removal percentage could be found. This observation confirms the consideration that EBCT is not adequate to guide biofiltration design and operation.

5.1.3 Attempt to Use the X* Concept for MIB and Geosmin Removal by Biofiltration

Although one investigation (Eaton and Sandusky, 2009) has found that a consortium of several specific bacteria strains is responsible for MIB and geosmin biodegradation, Huck and Sozański (2008) suggested it was reasonable to consider the entire biomass (X_f) as a surrogate for the degrading organisms.

The reported first-order reaction rate k for MIB and geosmin varies greatly from one study to another. For example, Lauderdale et al. (2004) determined a k value of 0.034 h^{-1} to 0.045 h^{-1}, while Ho et al. (2007) reported a k value of 0.064 d^{-1} for MIB; for geosmin, Hoefel et al. (2009) determined a k value of 0.010 h^{-1} to 0.029 h^{-1}, while Ho et al. (2007) reported a k value of 0.075 d^{-1} for MIB. The great differences may be because of different bacteria strains seeded and different test columns.

Based on the discussion above, it may not be realistic to estimate X* values or values of the $\theta\alpha$ product for collected cases of MIB and geosmin removal by biofiltration or establish an X*-removal percentage relationship for them. However, it may be useful to plot removal percentage for MIB and geosmin removal vs. $(\theta\alpha)'$ (the product of θ and α) (Figures 5-3 and 5-4).

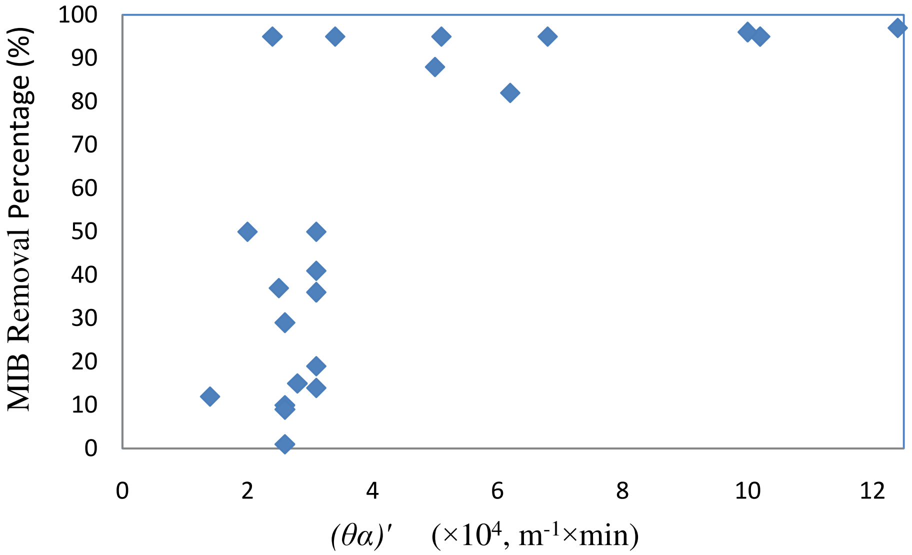

Figure 5-3 Relationship between $(\theta\alpha)'$ and removal percentage for MIB

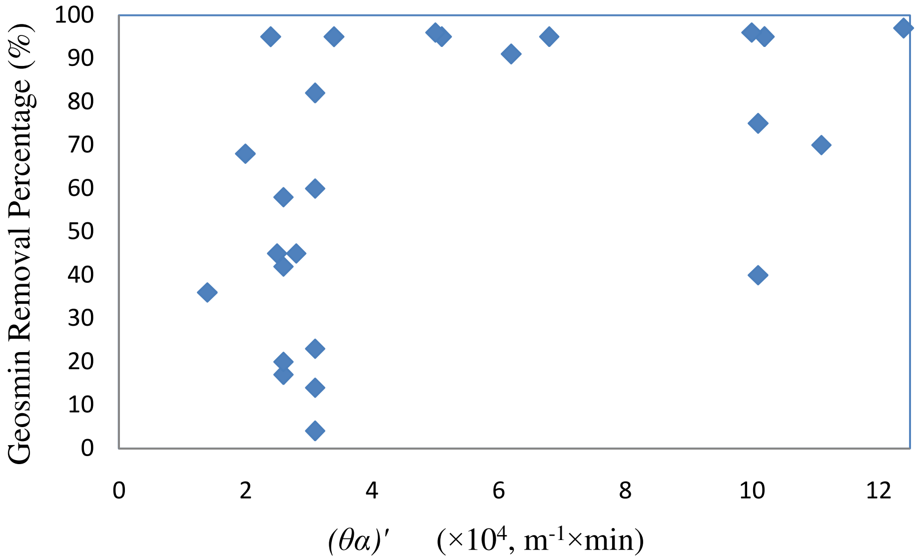

Figure 5-4 Relationship between $(\theta\alpha)'$ and removal percentage for geosmin

Visually, the product $(\theta\alpha)'$ has more of a relationship with percentage removal on Figures 5-3 and 5-4 than θ (EBCT) alone has on Figure 5-1 and 5-2. However, for $(\theta\alpha)'$ values less than about 3, removals seem unrelated to the product$(\theta\alpha)'$. This also confirms that other factors, such as temperature, affect MIB and geosmin removal by biofiltration. Products $(\theta\alpha)'$ were not corrected for temperature in this chapter because of unavailability of comparable biodegradation kinetic parameters between AOC/BDOC and MIB/geosmin from the collected literature. Ozonation can also affect odorous compounds removal in two ways: (1) Direct oxidation; (2) Producing more easily biodegradable compounds by oxidizing NOM so that more biomass can be sustained in the biofilters.

5.2 Using the X* Parameter to Evaluate Biofiltration Performance

Chapter 3 and 4 analyzed AOC and BDOC removal by biofiltration using the concept of X* and Section 5.1 in this chapter attempted, in a limited way, to extend the X* concept to secondary utilization. The results showed that the parameter X*, which incorporates physical contact time, media surface area, kinetics and other factors affecting the biofiltration process, is a better indicator than contact time alone for biofilter performance and a useful tool to compare treatment efficiency among different studies and different removal targets.

As Huck and Sozański (2008) discussed, if the X* concept can be applied for other biofiltration objectives, such as all removing biodegradable trace contaminants, reducing chlorine demand, disinfection by-products and reducing organic fouling for membrane processes, it would assist in its acceptance by consultants and water works as a "universal" design and operational parameter for biofiltration. Applying the simplified X* expression only needs the kinetics ratio between AOC and the removal target, since the original X*-percentage removal relationship was developed for AOC.

The X*-percentage removal relationship can be also used to avoid "over-design" or "under-operated", which were discussed in chapter 2 and 3 since "over-design" or "under-operated" phenomenon can be found in a significant number of the collected cases.

Over-design or under-operated refers to biofiltration process with an unnecessary configuration for a certain removal target (e.g. AOC percentage removal of 80%), such as a longer EBCT than necessary.

A utilization factor η therefore is proposed to guide biofilter design and operation.

$$\eta = \frac{\frac{Applied\ X^*}{Expected\ X^*}}{Safety\ Factor} \approx \frac{\frac{Applied\ \theta\alpha\ Parameter}{Expected\ X^*}}{Safety\ Factor} \qquad (5-3)$$

where the safety factor is suggested in this study to be assigned a value of 1.2-1.5 to assist in coping with loading shocks and changes of the ambient environment. The applied $\theta\alpha$ parameter should be temperature-adjusted. It should also be noted that experience may suggest a different value for this safety factor.

If the utilization factor η is less than 1, the biofilters is not able to accomplish the assigned removal task, if the safety factor is not considered. If the utilization factor η is greater than 1, the process can be considered as "over-designed" or "under-operated". For example in LeChevallier et al. (1992a, b), when EBCT increased from 5 to 20 minutes, the AOC removal percentage increased from 83% to only 89%. The corresponding expected X* increased from 1.8 to 2.4, while the applied $\theta\alpha$ parameter increased from 1.8 to 7.1 (Section 3.4.1 and Appendix A-6). Therefore, the utilization factor for EBCT of 20 minutes is calculated as: $\eta=\frac{7.1}{2.4}/1.2=2.5$ (1.2 is assigned for the safety factor in this calculation). The calculation result shows that the biofilters with 20 minutes EBCT in this study were significantly under-utilized for the task of removing around 90% AOC, which also suggests the contact time could be reduced to 20/2.5=8 minutes by increasing flow rate (producing more water) or shortening the bed depth to 1/2.5=40% of the current depth, or the current filter media could be replaced with coarser filter media, if it is of economical benefit.

Equation (5-3) also can be used to initially determine process configuration at pre-design stage or treatability study stage.

5.3 Conclusions

MIB and geosmin were chosen as examples in an attempt to apply the X* concept to secondary utilization in biofilters in this chapter. Applicability of using the simplified X* expression was also discussed.

No significant co-relationships between EBCTs and removal percentages were found for the collected cases of MIB and geosmin removal by biofiltration processes. It may not be realistic to obtain the estimated X* values or values of the $\theta\alpha$ parameter from the data available. Factors other than contact time and media size affect treatment efficiency. However, plotting the $\theta\alpha$ product vs. removal percentage for the collected MIB and geosmin removal cases shows more positive co-relationships than the EBCT-removal percentage visually.

The appropriateness of extending the X* concept to other biofiltration applications was discussed. A utilization factor η was developed to guide biofilter design and operation and to assess “over-design” and “under-operated”.

Chapter 6

Conclusions and Recommendations

By collecting, screening and investigating literature of AOC, BDOC and odorous compounds removal by biofiltration process, this study applied the X* concept on these collected investigations and attempted to extend X* concept to other biofiltration applications and establish a framework to assess process performance among different target parameters, different filters and different investigations. This work involved considerable effort to find, extract, and in some cases estimate information from the literature, and is the first such effort to the author's knowledge. Quantitative comparison of the literature results to the expected percentage removal *vs.* X* relationship would have been difficult statistically. Therefore, in this exploratory research, comparisons were made visually, looking for general agreement. In cases where general agreement was found it would be appropriate, in subsequent research, to attempt quantification, provided the necessary statistical conditions could be fulfilled.

6.1 Conclusions

The wide ranges of particle sizes, EBCTs, temperatures and high diversity of pre-treatment and operation conditions for the collected cases were considered to be able to well represent biofiltration practices for studied removal targets.

This section lists the most significant overall conclusions identified from this research:

- No significant co-relationship between EBCTs and removal percentages were found, indicating that EBCT alone is not able to guide biofiltration design and operation.
- Based on kinetics parameter comparison, BDOC removal-X* relationship was established.
- A new parameter, $\theta\alpha$, was developed in this thesis to refer to partially estimated X*

values only considering EBCT and particle size. $\theta\alpha$ parameter values were estimated by comparison of ratios of $\theta\alpha$ products ($(\theta\alpha)'$) based on the properly chosen calculation bases. Distribution of the $\theta\alpha$ values for temperature-favored (i.e., temperature ≥15°C) AOC and BDOC removal biofiltration processes matched the established removal-X* relationship reasonably, based on visual inspection.

- With the assistance of supporting information, raw water temperature ranges for AOC and BDOC removal biofiltration processes were determined; by adopting available temperature activity coefficients, temperature-adjustment coefficients for $\theta\alpha$ values were determined for the different temperature ranges. Temperature-adjusted AOC and BDOC removal- $\theta\alpha$ relationships were developed and temperature-adjusted $\theta\alpha$ parameter values for AOC and BDOC removal were also estimated. Comparisons were conducted between temperature-adjusted removal-$\theta\alpha$ relationships and temperature-adjusted $\theta\alpha$ values, showing fair matches for most of the temperature ranges, i.e. general agreement based on visual inspection.
- No relationships were found between ozone dosages and AOC/BDOC removal percentages and the ANOVA test indicated there was significant difference of removal efficiencies between ozonated and non-ozonated influents for biofilters, suggesting ozonation may not only increase the amount of BOM for following biofilter and increase the biodegradability of bulk water; it may also increase the biodegradability of AOC and BDOC themselves.
- BOM Measurement incubation time may be longer than necessary. Different BOM measurement methods may obtain different calculation results of removal efficiencies.
- It may not be realistic to obtain the estimated $\theta\alpha$ values for MIB and geosmin removal by biofiltration. However, plotting $\theta\alpha$ product vs. removal percentage for the collected MIB and geosmin removal cases shows more positive co-relationships than EBCT-removal percentage relationships visually.
- A new utilization factor η was proposed to guide biofilter design and operation and to assess “over-design” and “under-operated”. Biofilter over-design or under-operated is common for the collected cases.
- The X* concept can be extended to other biofiltration applications.

In general, examining X* (or partially estimated X*, which is $\theta\alpha$) provided more useful information in terms of evaluation and prediction of biodegradable organic compounds removal by biofiltration, which confirms that X* is a better parameter for biofiltration design and operated.

6.2 Recommendations

The X* concept was firstly derived from modeling work by Zhang and Huck (1996a), which was based on the pseudo-analytical solution of the biofilm model originally developed by Rittmann and McCarty (1980a). Since 1980, especially in recent ten years, more in-depth modeling work has been done to modify the original biofilm model. Taking into account these updated modeling work and recent understanding of biofilm and biofiltration would be beneficial for the X* concept framework and expression.

Biodegradation kinetics parameter (maximum utilization rate or first-order reaction rate) is a key to develop removal-X* (removal-$\theta\alpha$) relationship and determine X* values (or $\theta\alpha$ values) for removal targets other than AOC by comparing AOC removal kinetic and other removal targets removal kinetics. Currently, kinetic information in this regard is very limited and inconsistent, thus become a main obstacle to extend X* concept to other biofiltration applications. Obtaining kinetic parameter values for various removal targets is highly recommended.

The effect of biofilm density on X* and biofiltration performance was not studied in this study and the effect of ozonation was not studied thoroughly. Understanding of these two factors on X* needs to be studied.

Over-design or under-operated cases was a headache for the author during the study, because it confounded the X* values without apparent outstanding signal. Practically, unnecessary conservative over-design or under-operated may be of a waste of the utilities.

For cases where the literature results fit the expected relationship reasonably, based on visual examination, efforts could be made to provide quantitation of the fit using an error-in-variables approach after obtaining an equation for the expected relationship, provided reasonable variance estimates could be obtained for the $\theta\alpha$ parameter and X* values estimated from the various literature studies.

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Appendix A-2 Filter media particle sizes for AOC removal cases

Source	Media Type**	Average Size d(mm)	Effective Size d_{10}(mm)	Uniformity Coefficient (UC)
Chien et al., 2007*, 2008	GAC		0.55	1.90
Heinicke et al., 2006	GAC		0.63	1.70
Huck et al.,1991	GAC		0.55-0.75	<1.90
Lai et al., 2002*	GAC		0.95	2.30
LeChevallier et al., 1992a,b	GAC		0.80-0.90	<1.90
Krasner et al., 1993	GAC		1.00-1.10	<1.70
Persson et al. 2006	GAC		0.80-1.00	<2.10
Shu et al., 2008	GAC	1.00		
Vahala et al., 1998a,b	GAC		0.55-0.75	1.90
van der Aa et al., 2003	GAC	1.50-1.70		
van der Hoek et al., 2000	GAC		0.69	1.20
Chien et al., 2008	Anthracite		1.00-1.20	1.77
Hijnen and van der Kooij, 1992	Sand	0.42-0.60		
Moll et al., 1999	Sand		0.40	1.40-1.70***
Wang, et al., 1995	Sand		0.44	1.41
Wang and Summers et al., 1996	Sand		0.44	1.41****

Persson et al., 2006	EC (expanded clay)	Bio-ceramic	0.90-1.00	<1.50
Persson et al., 2006	EC (expanded clay)		2.45-2.75	<1.50
Hu et al., 1999	Bio-ceramic	2.00—5.00		
Hu et al., 2005	Zeolite	0.50-2.50		
Ahmad et al.,1998	Dual-Anthracite(60cm) Dual-Sand(30cm)		Anthracite 0.95 Sand 0.44	Anthracite 1.5 Sand 1.44
Daniel and Teefy, 1995	Dual-GAC(30inches) Dual-Sand(11inches)		GAC 1.10 Sand 1.10	GAC 1.40 Sand 1.40
Daniel and Teefy, 1995	Dual-Anthracite(30inches) Dual-Sand(11inches)		Anthracite 1.10 Sand 1.10	Anthracite 1.40 Sand 1.40
LeChevallier et al., 1992a,b	Mixed- Anthracite (18 inches) Mixed- Sand (9 inches) Mixed- Garnet (3inches)		Anthracite 1.10 Sand 0.43 Garnet 0.24	Anthracite <1.80 Sand <2.00 Garnet <3.00
LeChevallier et al., 1992a,b	Dual- GAC (20inches) Dual- Sand (10inches)		GAC 0.85 Sand 0.43	GAC<1.90 Sand <2.00
Najm et al., 2005	Dual- GAC (27inches) Dual- Sand (12inches)		GAC0.95 Sand 0.45	GAC <2.40 Sand 1.44
Najm et al., 2005	Dual- GAC (27inches) Dual- Sand (12inches)		GAC0.74 Sand 0.45	GAC 1.65 Sand 1.44
Najm et al., 2005	Dual- GAC (27inches) Dual- Sand (12inches)		GAC0.96 Sand 0.45	GAC 1.77 Sand 1.44
Wang, et al., 1995	Dual- Anthracite (20inches) Dual- Sand (10inches)		Anthracite 1.02 Sand 0.44	Anthracite 1.32 Sand 1.41
Wang, et al., 1995	Dual- GAC (20inches) Dual- Sand (10inches)		GAC 0.64 Sand 0.44	GAC 1.74 Sand 1.41
Wang, et al., 1995	Dual- GAC (20inches) Dual- Sand (10inches)		GAC 0.68 Sand 0.44	GAC 1.74 Sand 1.41
Wang, et al., 1995	Dual- GAC (20inches) Dual- Sand (10inches)		GAC 1.52 Sand 0.44	GAC 1.4 Sand 1.41
Wert et al., 2008*	Dual- Sand (25cm) Dual- Anthracite(53cm)		Sand 0.50 Anthracite 1.20	Sand <1.5 Anthracite <1.7
Zappia et al., 2007	Dual-sand (0.30m) Dual-Anthracite (1.75m)		Sand 0.65 Anthracite 1.10	Sand 1.40 Anthracite 1.60

Zappia et al., 2007	Dual-Sand (0.30m) Dual-Anthracite (0.65m)		Sand 0.65 Anthracite 1.10	Sand 1.40 Anthracite 1.60
Zappia et al., 2007	Dual-sand (0.30m) Dual-GAC (1.75m)		Sand 0.65 GAC 1.30	Sand 1.40 GAC 1.40
Zappia et al., 2007	Dual-sand (0.30m) Dual-GAC (1.75m)		Sand 0.65 GAC 0.70	Sand 1.40 GAC 1.70
Zhang et al., 2004	Dual-GAC (0.50m) Dual-sand (0.40m)	GAC 0.90-1.25mm Sand 0.45-0.61mm		

Note: Refer to section 3.1.1 for collecting and determining representative sizes of filter media particles;

* Contacted authors for information;

** For dual-media and multi-media filter, whichever is listed in upper place of a cell in any Appendix table is the toper layer in the corresponding biofilter, whichever is listed in bottom place of a cell is the bottom layer in the biofilter;

*** From Moll D.M., 1998, Civil Engineering PhD Dissertation, University of Cincinnati, Cincinnati, OH;

**** From Wang J.Z., 1995, Civil Engineering PhD Dissertation, University of Cincinnati, Cincinnati, OH.

Appendix A-3 Collection and calculation results of EBCTs, AOC removal percentages and EBCT-normalized percentage removals

Source	Media Type	EBCT[(1)] (min)	AOC Removal (%)[(2)]	EBCT-Normalized Percentage Removal (min^{-1}) [(8)]
Chien et al., 2007	GAC	6.6[(3)]	60[(3)]	9.1
Chien et al., 2008	GAC	6.6[(3)]	86[(3)]	13.0
Heinicke et al., 2006	GAC	34[(4)]	19[(3)]	0.6
Huck et al.,1991	GAC	36[(3)]	33[(4)]	0.9
Huck et al.,1991	GAC	19[(3)]	52[(4)]	2.7
Huck et al.,1991	GAC	36[(3)]	50[(4)]	1.4
Huck et al.,1991	GAC	19[(3)]	88[(4)]	4.6
Huck et al.,1991	GAC	19[(3)]	68[(4)]	3.6
Huck et al.,1991	GAC	36[(3)]	82[(4)]	2.3
Huck et al.,1991	GAC	19[(3)]	86[(4)]	4.5
Huck et al.,1991	GAC	36[(3)]	68[(4)]	1.9
Huck et al.,1991	GAC	19[(3)]	69[(4)]	3.6
Huck et al.,1991	GAC	36[(3)]	77[(4)]	2.1
Huck et al.,1991	GAC	19[(3)]	83[(4)]	4.4
Lai et al., 2002	GAC	7.1[(3)]	88[(5)]	12.4
Lai et al., 2002	GAC	12.5[(3)]	76[(5)]	6.1
LeChevallier et al., 1992a, b	GAC	14[(3)]	89[(3)]	6.4
Krasner et al., 1993	GAC	1.4[(3)]	72[(3)]	51.4
Krasner et al., 1993	GAC	4.2[(3)]	74[(3)]	17.6
Persson et al. 2006	GAC	31[(3)]	22[(3)]	0.7
Shu et al., 2008	GAC	8.6[(3)]	35[(4)]	4.1
Vahala et al., 1998a	GAC	13[(3)]	53[(3)]	4.1
Vahala et al., 1998a	GAC	13[(3)]	69[(3)]	5.3

Vahala et al., 1998a	GAC	13(3)	73(3)	5.6
Vahala et al., 1998a	GAC	13(3)	76(3)	5.8
Vahala et al., 1998a	GAC	13(3)	59(3)	4.5
Vahala et al., 1998a	GAC	13(3)	69(3)	5.3
Vahala et al., 1998a	GAC	13(3)	75(3)	5.8
Vahala et al., 1998a	GAC	13(3)	79(3)	6.1
Vahala et al., 1998a	GAC	13(3)	75(3)	5.8
Vahala et al., 1998a	GAC	13(3)	27(3)	2.1
Vahala et al., 1998a	GAC	13(3)	51(3)	3.9
Vahala et al., 1998a	GAC	13(3)	64(3)	4.9
Vahala et al., 1998a	GAC	13(3)	49(3)	3.8
Vahala et al., 1998b	GAC	15(3)	73(3)	4.9
Vahala et al., 1998b	GAC	15(3)	76(3)	5.1
van der Aa et al., 2003	GAC	7(3)	57(4)	8.1
van der Aa et al., 2003	GAC	23(3)	64(4)	2.8
van der Aa et al., 2003	GAC	40(3)	71(4)	1.8
van der Hoek et al., 2000	GAC	8(3)	64(5)	8
van der Hoek et al., 2000	GAC	16(3)	75(5)	4.7
van der Hoek et al., 2000	GAC	32(3)	76(5)	2.4
van der Hoek et al., 2000	GAC	40(3)	80(5)	2.0
Chien et al., 2008	Anthracite	6.6(3)	17(3)	2.6
Hijnen and van der Kooij, 1992	Sand	2.4(4)	72(3)	30.0
Hijnen and van der Kooij, 1992	Sand	6(4)	72(3)	12.0
Hijnen and van der Kooij, 1992	Sand	12(4)	72(3)	6.0
Hijnen and van der Kooij, 1992	Sand	60(4)	76(3)	1.3
Moll et al., 1999	Sand	7.0(3)	43(3)	6.1

Moll et al., 1999	Sand	7.0(3)	55(3)	7.8
Moll et al., 1999	Sand	7.0(3)	57(3)	8.1
Wang, et al., 1995	Sand	9.2(3)	43(3)	4.7
Wang and Summers, 1996	Sand	0.4(6)	59(5)	147.5
Wang and Summers, 1996	Sand	0.8(6)	61(5)	76.3
Wang and Summers, 1996	Sand	1.6(6)	56(5)	35.0
Wang and Summers, 1996	Sand	2.2(6)	65(5)	29.5
Wang and Summers, 1996	Sand	3.6(6)	68(5)	18.9
Wang and Summers, 1996	Sand	3.8(6)	72(5)	18.9
Wang and Summers, 1996	Sand	5.6(6)	64(5)	11.4
Wang and Summers, 1996	Sand	6.8(6)	76(5)	11.2
Persson et al., 2006	EC (expanded clay)	31(3)	35(3)	1.1
Persson et al., 2006	EC (expanded clay)	31(3)	41(3)	1.3
Hu et al., 1999	Bio-ceramic	40(4)	44(3)	1.1
Hu et al., 2005	Zeolite	7.5(3)	4(5)	0.5
Hu et al., 2005	Zeolite	30(3)	45(5)	1.5
Hu et al., 2005	Zeolite	60(3)	53(5)	0.9
Ahmad et al.,1998	Dual-Anthracite/Sand	5.4(4)	79(4) (7)	14.6
Ahmad et al.,1998	Dual-Anthracite/Sand	5.4(4)	78(4) (7)	14.4
Ahmad et al.,1998	Dual-Anthracite/Sand	5.4(4)	55(5)	10.2
Ahmad et al.,1998	Dual-Anthracite/Sand	5.4(4)	21(5)	3.9
Ahmad et al.,1998	Dual-Anthracite/Sand	5.4(4)	80(5)	14.8
Daniel and Teefy, 1995	Dual-GAC/Sand	6.4(3)	70(3)	10.9
Daniel and Teefy, 1995	Dual-Anthracite/Sand	6.4(3)	70(3)	10.9
LeChevallier et al., 1992a,b	Mixed-Anthracite/Sand/Garnet	5(3)	75(3)	15.0
LeChevallier et al., 1992a,b	Dual-GAC/Sand	5(3)	86(3)	17.2

LeChevallier et al., 1992a,b	Dual-GAC/Sand	7(3)	85(3)	12.1
LeChevallier et al., 1992a,b	Dual-GAC/Sand	10(3)	91(3)	9.1
LeChevallier et al., 1992a,b	Dual-GAC/Sand	10(3)	85(3)	8.5
LeChevallier et al., 1992a,b	Dual-GAC/Sand	10(3)	75(3)	7.5
LeChevallier et al., 1992a,b	Dual-GAC/Sand	10(3)	79(3)	7.9
LeChevallier et al., 1992a,b	Dual-GAC/Sand	10(3)	84(3)	8.4
LeChevallier et al., 1992a,b	Dual-GAC/Sand	7(3)	82(3)	11.7
LeChevallier et al., 1992a,b	Dual-GAC/Sand	7(3)	84(3)	12.0
LeChevallier et al., 1992a,b	Dual-GAC/Sand	5(3)	83(3)	16.6
LeChevallier et al., 1992a,b	Dual-GAC/Sand	10(3)	84(3)	8.4
LeChevallier et al., 1992a,b	Dual-GAC/Sand	15(3)	84(3)	5.6
LeChevallier et al., 1992a,b	Dual-GAC/Sand	20(3)	89(3)	4.5
Najm et al., 2005	Dual- GAC/Sand	10(3)	>75(3)	7.5
Najm et al., 2005	Dual- GAC/Sand	55(3)	>75(3)	1.4
Wang, et al., 1995	Dual-Anthracite/Sand	9.2(3)	39(3)	4.2
Wang, et al., 1995	Dual-Anthracite/Sand	9.2(3)	37(3)	4.0
Wang, et al., 1995	Dual-Anthracite/Sand	9.2(3)	48(3)	5.2
Wang, et al., 1995	Dual---GAC/Sand	9.2(3)	51(3)	5.5
Wang, et al., 1995	Dual-GAC/Sand	9.2(3)	47(3)	5.1
Wang, et al., 1995	Dual-GAC/Sand	9.2(3)	42(3)	4.6
Wert et al., 2008	Dual-Sand/ Anthracite	3.2(3)	>60(3)	18.8
Wert et al., 2008	Dual-Sand/ Anthracite	9.7(3)	>60(3)	6.2
Wert et al., 2008	Dual-Sand/ Anthracite	25(3)	>60(3)	2.4
Zappia et al., 2007	Dual-Sand/Anthracite	8(3)	76(4)	9.5
Zappia et al., 2007	Dual-Sand/Anthracite	8(3)	79(4)	9.9

Zappia et al., 2007	Dual-Sand/Anthracite	8(3)	42(4)	5.3
Zappia et al., 2007	Dual-Sand/Anthracite	8(3)	57(4)	7.1
Zappia et al., 2007	Dual-Sand/GAC	8(3)	68(4)	8.5
Zappia et al., 2007	Dual-Sand/GAC	8(3)	76(4)	9.5
Zhang et al., 2004	Dual-GAC/Sand	6.8(3)	84(4)	12.4

Note: 1. Refer to section 3.1.2 for collecting and determining EBCTs;
2. Refer to section 3.1.3 for determining AOC removal percentages;
3. Data provided in the literature;
4. Calculated using provided supporting data;
4.1. EBCT calculation is as Equation (3-5):

$$EBCT = \frac{h}{HLR} = \frac{h}{Q/A} \qquad (3-5)$$

4.2. AOC removal percentage calculation is as Equation (3-7):

$$AOC\ Removal\ Percentage = \frac{AOC_{Filter\ Influent} - AOC_{Filter\ Effluent}}{AOC_{Filter\ Influent}} \qquad (3-7)$$

where AOC value refers to summary of AOC-P17 and AOC-NOX, if it was expressed separately;

5. Measured from figures in the literature and calculated using Equation (3-5) and (3-7);
6. Measured from figures in the literature;
7. Average values;
8. EBCT-normalized percentage removal calculation is as Equation (3-8):

$$EBCT\ Normalized\ Percentage\ Removal = \frac{AOC\ Removal\ Percentage}{EBCT} \qquad (3-8)$$

Appendix A-4 Actively specific surface area of filter media particles

Source	Media Type	Representative Size(mm)	Specific Surface Area for Ideal Spherical Shape (m^{-1})	Actively Specific Surface Area (m^{-1})
Chien et al., 2007, 2008	GAC	0.80	7500	5820
Heinicke et al., 2006	GAC	0.85	7060	5480
Huck et al.,1991	GAC	0.94	6380	4980
Lai et al., 2002	GAC	1.57	3820	2970
LeChevallier et al., 1992a,b	GAC	1.23	4880	3790
Krasner et al., 1993	GAC	1.42	4230	3300
Persson et al. 2006	GAC	1.40	4290	3590
Shu et al., 2008	GAC	1.00	6000	4660
Vahala et al., 1998a,b	GAC	0.94	6380	4960
van der Aa et al., 2003	GAC	1.60	3750	2910
van der Hoek et al., 2000	GAC	0.76	7890	6150
Chien et al., 2008	Anthracite	1.49	4030	3130
Hijnen and van der Kooij	Sand	0.51	11760	9170
Moll et al., 1999	Sand	051	11760	9170
Wang, et al., 1995	Sand	0.53	11320	8790
Wang and Summers, 1996	Sand	0.53	11320	8790

Persson et al., 2006	EC (expanded clay)	1.19	5040	3470
Persson et al., 2006	EC (expanded clay)	3.25	1850	1120
Hu et al., 1999	Bio-ceramic	3.50	1710	1330
Hu et al., 2005	Zeolite	3.50	1714	1340
Ahmad et al.,1998	Dual-Anthracite(60cm) Dual-Sand(30cm)	Anthracite 1.19 Sand 0.54	Anthracite 5040 Sand 11110	Anthracite 3930 Sand 8670
Daniel and Teefy, 1995	Dual-GAC(30inches) Dual-Sand(11inches)	GAC 1.32 Sand 1.32	GAC 4550 Sand 4550	GAC 3530 Sand 3530
Daniel and Teefy, 1995	Dual-Anthracite(30inches) Dual-Sand(11inches)	Anthracite 1.32 Sand 1.32	Anthracite 4550 Sand 4550	Anthracite 3530 Sand 3530
LeChevallier et al.,1992a,b	Mixed-Anthracite(18 inches) Mixed-Sand (9 inches) Mixed-Garnet (3inches)	Anthracite 1.54 Sand 0.65 Garnet 0.48	Anthracite 3900 Sand 9230 Garnet 12500	Anthracite 3040 Sand 7170 Garnet 9750
LeChevallier et al.,1992a,b	Dual-GAC (20inches) Dual-Sand (10inches)	GAC 1.23 Sand 0.65	GAC 4880 Sand 9230	GAC 3790 Sand 7170
Najm et al., 2005	Dual-GAC (27inches) Dual-Sand (12inches)	GAC 1.62 Sand 0.55	GAC 3700 Sand 10910	GAC 2880 Sand 8470
Najm et al., 2005	Dual-GAC (27inches) Dual---Sand (12inches)	GAC 0.98 Sand 0.55	GAC 6120 Sand 10910	GAC 4750 Sand 8470
Najm et al., 2005	Dual-GAC (27inches) Dual-Sand (12inches)	GAC 1.33 Sand 0.55	GAC 4510 Sand 10910	GAC 3500 Sand 8470
Wang, et al., 1995	Dual-Anthracite (20inches) Dual-Sand (10inches)	Anthracite 1.18 Sand 0.53	Anthracite 5080 Sand 11320	Anthracite 3920 Sand 8830
Wang, et al., 1995	Dual-GAC (20inches) Dual-Sand (10inches)	GAC 0.88 Sand 0.53	GAC 6820 Sand 11320	GAC 5320 Sand 8830
Wang, et al., 1995	Dual-GAC (20inches) Dual-Sand (10inches)	GAC 0.82 Sand 0.53	GAC 7320 Sand 11320	GAC 5710 Sand 8830
Wang, et al., 1995	Dual-GAC (20inches) Dual-Sand (10inches)	GAC 1.82 Sand 0.53	GAC 3300 Sand 11320	GAC 2570 Sand 8830
Wert et al., 2008	Dual-Sand (25cm) Dual-Anthracite(53cm)	Sand 0.63 Anthracite 1.62	Sand 9520 Anthracite 3700	Sand 7390 Anthracite 2880
Zappia et al., 2007	Dual-Sand (0.3m) Dual-Anthracite (1.75m)	Sand 0.78 Anthracite 1.43	Sand 7690 Anthracite 4200	Sand 5970 Anthracite 3260

Zappia et al., 2007	Dual-Sand (0.3m) Dual-Anthracite (0.65m)	Sand 5970 Anthracite 3260	Sand 7690 Anthracite 4200	Sand 5970 Anthracite 3260
Zappia et al., 2007	Dual-Sand (0.3m) Dual-GAC (1.75m)	Sand 5970 GAC 2990	Sand 7690 GAC 3850	Sand 5970 GAC 2990
Zappia et al., 2007	Dual-Sand (0.3m) Dual-GAC (1.75m)	Sand 5970 GAC 5000	Sand 7690 GAC 6320	Sand 5970 GAC 5000
Zhang et al., 2004	Dual-GAC (0.50m) Dual-Sand (0.40m)	GAC 1.08 Sand 0.53	GAC 5560 Sand 11320	GAC 4340 Sand 8830

Note: 1. Refer to section 3.1.1 for collection and determination of representative sizes of filter media particles;

2. Refer to section 3.2 for calculation of actively specific surface area of filter media;

3. Actively specific surface area of filter media was calculated by Equation (3-9):

$$\alpha = \frac{6 \times \beta \times \varphi \times (1 - \epsilon)}{\bar{d}} \qquad (3-9)$$

Appendix A-5 Initial estimation of $\theta\alpha$ parameter values

Source	Media Type	α (m^{-1})	EBCT (min)	α × EBCT (×10^{4}, m^{-1}×min)	Initially Estimated $\theta\alpha$ Parameter Values$^{(1)}$
Chien et al.,2007	GAC	5820	6.6	3.8	2.7
Chien et al., 2008	GAC	5820	6.6	3.8	2.7
Heinicke et al., 2006	GAC	5480	34	18.6	13.4
Huck et al.,1991	GAC	4980	36	17.9	14.0
Huck et al.,1991	GAC	4980	19	9.5	7.4
Lai et al., 2002	GAC	2970	7.1	2.1	1.5
Lai et al., 2002	GAC	2970	12.5	3.7	2.7
LeChevallier et al., 1992a, b	GAC	3790	14	5.3	3.8
Krasner et al., 1993	GAC	3300	1.4	0.5	0.4
Krasner et al., 1993	GAC	3300	4.2	1.4	1.1
Persson et al. 2006	GAC	3590	31	11.3	8.1
Shu et al., 2008	GAC	4660	8.6	4	2.9
Vahala et al., 1998a	GAC	4960	13	6.4	4.6
Vahala et al., 1998b	GAC	4960	15	7.4	5.3
van der Aa et al., 2003	GAC	2910	7	2.0	1.4
van der Aa et al., 2003	GAC	2910	23	6.7	4.8

van der Aa et al., 2003	GAC	2910	40	11.6	8.4
van der Hoek et al., 2000	GAC	6150	8	4.9	3.5
van der Hoek et al., 2000	GAC	6150	16	9.8	6.9
van der Hoek et al., 2000	GAC	6150	32	19.6	13.9
van der Hoek et al., 2000	GAC	6150	40	24.5	17.4
Chien et al., 2008	Anthracite	3130	6.6	2.1	1.5
Hijnen and van der Kooij, 1992	Sand	9170	2.4	2.2	1.6
Hijnen and van der Kooij, 1992	Sand	9170	6	5.5	4.0
Hijnen and van der Kooij, 1992	Sand	9170	12	11	8.0
Hijnen and van der Kooij, 1992	Sand	9170	60	55	40.0
Moll et al., 1999	Sand	9170	7.0	6.4	5.0
Moll et al., 1999	Sand	9170	7.0	6.4	5.0
Moll et al., 1999	Sand	9170	7.0	6.4	5.0
Wang, et al., 1995	Sand	8830	9.2	8.1	5.8
Wang and Summers, 1996	Sand	8830	0.4	0.4	0.3
Wang and Summers, 1996	Sand	8830	0.8	0.7	0.5
Wang and Summers, 1996	Sand	8830	1.6	1.4	1.0
Wang and Summers, 1996	Sand	8830	2.2	1.9	1.4

Wang and Summers, 1996	Sand	8830	3.6	3.2	2.3
Wang and Summers, 1996	Sand	8830	3.8	3.4	2.4
Wang and Summers, 1996	Sand	8830	5.6	4.9	3.5
Wang and Summers, 1996	Sand	8830	6.8	6.0	4.3
Persson et al., 2006	EC (expanded clay)	3590	31	10.7	7.7
Persson et al., 2006	EC (expanded clay)	1120	31	3.4	2.5
Hu et al., 1999	Bio-ceramic	1340	40	5.3	3.8
Hu et al., 2005	Zeolite	3110	7.5	2.3	1.7
Hu et al., 2005	Zeolite	3110	30	9.3	6.7
Hu et al., 2005	Zeolite	3110	60	18.6	13.4
Ahmad et al.,1998	Dual-Anthracite(60cm) Dual-Sand(30cm)	Anthracite 3930 Sand 8670	5.4	3.8	2.8
Daniel and Teefy, 1995	Dual-GAC(30inches) Dual-Sand(11inches)	GAC 3530 Sand 3530	6.4	2.3*	1.7
Daniel and Teefy, 1995	Dual-Anthracite(30inches) Dual-Sand(11inches)	Anthracite 3530 Sand 3530	6.4	2.3*	1.7
LeChevallier et al., 1992a,b	Mixed-Anthracite(18 inches) Mixed-Sand (9 inches) Mixed-Garnet (3inches)	Anthracite 3040 Sand 7170 Garnet 9750	5	2.5*	1.8
LeChevallier et al., 1992a,b	Dual----GAC (20inches) Dual----Sand (10inches)	GAC 3790 Sand 7170	5	2.5*	1.8
LeChevallier et al., 1992a,b	Dual----GAC (20inches) Dual----Sand (10inches)	GAC 3790 Sand 7170	7	3.5*	2.5
LeChevallier et al., 1992a,b	Dual----GAC (20inches) Dual----Sand (10inches)	GAC 3790 Sand 7170	10	4.9*	3.5
LeChevallier et al., 1992a,b	Dual----GAC (20inches) Dual----Sand (10inches)	GAC 3790 Sand 7170	5	2.5*	1.8

LeChevallier et al., 1992a,b	Dual----GAC (20inches) Dual----Sand (10inches)	GAC 3790 Sand 7170	10	4.9*	3.5
LeChevallier et al., 1992a,b	Dual----GAC (20inches) Dual----Sand (10inches)	GAC 3790 Sand 7170	15	7.4*	5.3
LeChevallier et al., 1992a,b	Dual----GAC (20inches) Dual----Sand (10inches)	GAC 3790 Sand 7170	20	9.8*	7.1
LeChevallier et al., 1992a,b	Dual----GAC (20inches) Dual----Sand (10inches)	GAC 3790 Sand 7170	10	4.9*	3.5
Najm et al., 2005	Dual--- GAC (27inches) Dual---Sand (12inches)	GAC 2880 Sand 8470	10	4.6*	3.3
Najm et al., 2005	Dual--- GAC (27inches) Dual---Sand (12inches)	GAC 2880 Sand 8470	55	25.3*	18.2
Wang, et al., 1995	Dual---Anthracite(20inches) Dual---Sand (10inches)	Anthracite 3920 Sand 8830	9.2	5.1*	3.7
Wang, et al., 1995	Dual---GAC (20inches) Dual---Sand (10inches)	GAC 5320 Sand 8830	9.2	6.0*	4.3
Wang, et al., 1995	Dual---GAC (20inches) Dual---Sand (10inches)	GAC 5710 Sand 8830	9.2	6.2*	4.5
Wang, et al., 1995	Dual---GAC (20inches) Dual---Sand (10inches)	GAC 2570 Sand 8830	9.2	2.3*	1.7
Wert et al., 2008	Dual- Sand (25cm) Dual- Anthracite(53cm)	Sand 7390 Anthracite 2880	3.2	1.4*	1.0
Wert et al., 2008	Dual- Sand (25cm) Dual- Anthracite(53cm)	Sand 7390 Anthracite 2880	9.7	4.2*	3.0
Wert et al., 2008	Dual- Sand (25cm) Dual- Anthracite(53cm)	Sand 7390 Anthracite 2880	25	10.8*	7.8
Zappia et al., 2007	Dual-sand (0.3m) Dual-Anthracite (1.75m)	Sand 5970 Anthracite 3260	8	2.9*	2.1
Zappia et al., 2007	Dual-sand (0.3m) Dual-Anthracite (0.65m)	Sand 5970 Anthracite 3260	8	3.3*	2.4
Zappia et al., 2007	Dual-sand (0.3m) Dual-GAC (1.75m)	Sand 5970 GAC 2990	8	2.7*	1.9
Zappia et al., 2007	Dual-sand (0.3m) Dual-GAC (1.75m)	Sand 5970 GAC 5000	8	4.1*	3.0
Zhang et al., 2004	Dual-GAC (0.50m) Dual-sand (0.40m)	GAC 8830 Sand 4340	6.75	4.6*	3.3

Note: 1. Refer to section 3.4 for initial estimation of $\theta\alpha$ parameter values;

2. Estimated $\theta\alpha$ parameter value was calculated by Equation (3-12):

$$\theta\alpha\ parameter\ value = (\theta\alpha)'\frac{1.8}{2.5} \tag{3-12}$$

3. (*): dual-media filters or multi-media,

Calculation of EBCT for each layer is as Equation (3-6):

$$EBCT_i = \frac{h_i}{h} \times EBCT_{Overall} \tag{3-6}$$

$\theta\alpha$ calculation is as Equation (3-11):

$$(\theta\alpha)'_{overall} = \sum(\theta\alpha)'_{each\ layer} \tag{3-11}$$

4. Underlined are the calculation bases.

Appendix A-6 Initially estimated $\theta\alpha$ parameter values, expected $\theta\alpha$ parameter values, physical EBCTs and AOC removal percentages

Source	Media Type	Initially Estimated $\theta\alpha$ parameter values	Expected $\theta\alpha$ Parameter Values(1)	EBCT(min)	AOC Removal (%)
Chien et al.,2007	GAC	2.7	0.8	6.6	60
Chien et al., 2008	GAC	2.7	2.1	6.6	86
Heinicke et al., 2006	GAC	13.4	0.1	34	19
Huck et al.,1991	GAC	14.0	0.3	36	33
Huck et al.,1991	GAC	7.4	0.5	19	52
Huck et al.,1991	GAC	14.0	0.5	36	50
Huck et al.,1991	GAC	7.4	2.3	19	88
Huck et al.,1991	GAC	7.4	1.1	19	68
Huck et al.,1991	GAC	14.0	1.6	36	82
Huck et al.,1991	GAC	7.4	2.1	19	86
Huck et al.,1991	GAC	14.0	1.1	36	68
Huck et al.,1991	GAC	7.4	1.1	19	69
Huck et al.,1991	GAC	14.0	1.4	36	77
Huck et al.,1991	GAC	7.4	1.8	19	83
Lai et al., 2002	GAC	1.5	2.3	7.1	88

Lai et al., 2002	GAC	2.7	1.3	12.5	76
LeChevallier et al., 1992a, b	GAC	3.8	2.4	14	89
Krasner et al., 1993	GAC	0.4	1.2	1.4	72
Krasner et al., 1993	GAC	1.1	1.3	4.2	74
Persson et al. 2006	GAC	8.1	0.1	31	22
Shu et al., 2008	GAC	2.9	0.3	8.6	35
Vahala et al., 1998a	GAC	4.6	0.6	13	53
Vahala et al., 1998a	GAC	4.6	1.1	13	69
Vahala et al., 1998a	GAC	4.6	1.2	13	73
Vahala et al., 1998a	GAC	4.6	1.3	13	76
Vahala et al., 1998a	GAC	4.6	0.7	13	59
Vahala et al., 1998a	GAC	4.6	1.1	13	69
Vahala et al., 1998a	GAC	4.6	1.3	13	75
Vahala et al., 1998a	GAC	4.6	1.5	13	79
Vahala et al., 1998a	GAC	4.6	1.3	13	75
Vahala et al., 1998a	GAC	4.6	0.2	13	27
Vahala et al., 1998a	GAC	4.6	0.5	13	51
Vahala et al., 1998a	GAC	4.6	0.9	13	64
Vahala et al., 1998a	GAC	4.6	0.5	13	49

Vahala et al., 1998b	GAC	5.3	1.2	15	73
Vahala et al., 1998b	GAC	5.3	1.5	15	76
van der Aa et al., 2003	GAC	1.4	0.8	7	57
van der Aa et al., 2003	GAC	4.8	1.0	23	64
van der Aa et al., 2003	GAC	8.4	1.2	40	71
van der Hoek et al., 2000	GAC	3.5	0.9	8	64
van der Hoek et al., 2000	GAC	6.9	1.3	16	75
van der Hoek et al., 2000	GAC	13.9	1.3	32	76
van der Hoek et al., 2000	GAC	17.4	1.5	40	80
Chien et al., 2008	Anthracite	1.5	0.1	6.6	17
Hijnen and van der Kooij, 1992	Sand	1.6	1.2	2.4	72
Hijnen and van der Kooij, 1992	Sand	4.0	1.2	6	72
Hijnen and van der Kooij, 1992	Sand	8.0	1.2	12	72
Hijnen and van der Kooij, 1992	Sand	40.0	1.3	60	76
Moll et al., 1999	Sand	5.0	0.4	7.0	43
Moll et al., 1999	Sand	5.0	0.7	7.0	55
Moll et al., 1999	Sand	5.0	0.8	7.0	57
Wang, et al., 1995	Sand	5.8	0.4	9.2	43

Wang and Summers, 1996	Sand	0.3	0.8	0.4	59
Wang and Summers, 1996	Sand	0.5	0.9	0.8	61
Wang and Summers, 1996	Sand	1.0	0.7	1.6	56
Wang and Summers, 1996	Sand	1.4	1.0	2.2	65
Wang and Summers, 1996	Sand	2.3	1.1	3.6	68
Wang and Summers, 1996	Sand	2.4	1.2	3.8	72
Wang and Summers, 1996	Sand	3.5	1.0	5.6	64
Wang and Summers, 1996	Sand	4.3	1.4	6.8	76
Persson et al., 2006	EC (expanded clay)	7.7	0.3	31	35
Persson et al., 2006	EC (expanded clay)	2.5	0.4	31	41
Hu et al., 1999	Bio-ceramic	3.8	0.4	40	44
Hu et al., 2005	Zeolite	1.7	0.1	7.5	4
Hu et al., 2005	Zeolite	6.7	0.4	30	45
Hu et al., 2005	Zeolite	13.4	0.5	60	53
Ahmad et al.,1998	Dual-Anthracite/Sand	2.8	1.5	5.4	79
Ahmad et al.,1998	Dual-Anthracite/Sand	2.8	1.4	5.4	78
Ahmad et al.,1998	Dual-Anthracite/Sand	2.8	0.7	5.4	55
Ahmad et al.,1998	Dual-Anthracite/Sand	2.8	0.1	5.4	21
Ahmad et al.,1998	Dual-Anthracite/Sand	2.8	1.5	5.4	80

Daniel and Teefy, 1995	Dual-GAC/Sand	1.7	1.1	6.4	70
Daniel and Teefy, 1995	Dual-Anthracite/Sand	1.7	1.1	6.4	70
LeChevallier et al., 1992a,b	Mixed-Anthracite/Sand/Garnet	1.8	1.3	5	75
LeChevallier et al., 1992a,b	Dual-GAC/Sand	1.8	2.1	5	86
LeChevallier et al., 1992a,b	Dual-GAC/Sand	2.4	2.0	7	85
LeChevallier et al., 1992a,b	Dual-GAC/Sand	3.5	1.7	10	91
LeChevallier et al., 1992a	Dual-GAC/Sand	3.5	1.7	10	85
LeChevallier et al., 1992a,b	Dual-GAC/Sand	3.5	1.3	10	75
LeChevallier et al., 1992a,b	Dual-GAC/Sand	3.5	1.5	10	79
LeChevallier et al., 1992a,b	Dual-GAC/Sand	3.5	2.0	10	84
LeChevallier et al., 1992a,b	Dual-GAC/Sand	2.4	1.7	7	82
LeChevallier et al., 1992a,b	Dual-GAC/Sand	2.4	1.9	7	84
LeChevallier et al., 1992a,b	Dual-GAC/Sand	1.8	1.8	5	83
LeChevallier et al., 1992a,b	Dual-GAC/Sand	3.5	1.9	10	84
LeChevallier et al., 1992a,b	Dual-GAC/Sand	5.3	1.9	15	84
LeChevallier et al., 1992a,b	Dual-GAC/Sand	7.1	2.4	20	89
Najm et al., 2005	Dual- GAC/Sand	3.3	1.3	10	75
Najm et al., 2005	Dual- GAC/Sand	18.2	1.3	55	75

Wang, et al., 1995	Dual-Anthracite/Sand	3.7	0.3	9.2	39
Wang, et al., 1995	Dual-Anthracite/Sand	3.7	0.3	9.2	37
Wang, et al., 1995	Dual-Anthracite/Sand	3.7	0.5	9.2	48
Wang, et al., 1995	Dual---GAC/Sand	4.3	0.5	9.2	51
Wang, et al., 1995	Dual-GAC/Sand	4.5	0.5	9.2	47
Wang, et al., 1995	Dual-GAC/Sand	1.7	0.4	9.2	42
Wert et al., 2008	Dual-Sand/ Anthracite	1.0	0.7	3.2	60
Wert et al., 2008	Dual-Sand/ Anthracite	3.0	0.7	9.7	60
Wert et al., 2008	Dual-Sand/ Anthracite	7.8	0.7	25	60
Zappia et al., 2007	Dual-Sand/Anthracite	2.1	1.3	8	76
Zappia et al., 2007	Dual-Sand/Anthracite	2.4	1.5	8	79
Zappia et al., 2007	Dual-Sand/Anthracite	2.4	0.4	8	42
Zappia et al., 2007	Dual-Sand/Anthracite	2.4	0.7	8	57
Zappia et al., 2007	Dual-Sand/GAC	1.9	1.1	8	68
Zappia et al., 2007	Dual-Sand/GAC	3.0	1.3	8	76
Zhang et al., 2004	Dual-GAC/Sand	3.3	2.0	6.8	84

Note: 1. Refer to section 3.3, Figure 3-3 and Table 3-5 for determining expected X* values.

Appendix A-7 Representative temperature range and temperature adjustment coefficients

Source	Media Type	Initially Estimated $\theta\alpha$ Parameter Values	Temperature, Season, Location	Representative Temperature Range[12]	Temperature Adjustment Coefficient [13]
Chien et al.,2007	GAC	2.7	25-30°C	≥15°C	1
Chien et al., 2008	GAC	2.7	25-30°C	≥15°C	1
Heinicke et al., 2006	GAC	13.4	Sweden [1]	10-15°C	1.29
Huck et al.,1991	GAC	14.0	Summer (4-25°C) [9] Alberta, Canada	10-15°C	1.29
Huck et al.,1991	GAC	7.4	Summer (4-25°C) [9] Alberta, Canada	10-15°C	1.29
Huck et al.,1991	GAC	14.0	Summer (4-25°C) [9] Alberta, Canada	10-15°C	1.29
Huck et al.,1991	GAC	7.4	Summer (4-25°C) [9] Alberta, Canada	10-15°C	1.29
Huck et al.,1991	GAC	7.4	Fall-Winter(7-14°C) [9] Alberta, Canada	10-15°C	1.29
Huck et al.,1991	GAC	14.0	Fall-Winter(7-14°C) [9] Alberta, Canada	10-15°C	1.29
Huck et al.,1991	GAC	7.4	Fall-Winter(7-14°C) [9] Alberta, Canada	10-15°C	1.29
Huck et al.,1991	GAC	14.0	Spring(6-9°C) Alberta, Canada	5-10°C	1.53
Huck et al.,1991	GAC	7.4	Spring(6-9°C) Alberta, Canada	5-10°C	1.53
Huck et al.,1991	GAC	14.0	Spring(6-9°C) Alberta, Canada	5-10°C	1.53
Huck et al.,1991	GAC	7.4	Spring(6-9°C) Alberta, Canada	5-10°C	1.53
Lai et al., 2002	GAC	1.5	19.8-25.4°C	≥15°C	1

Lai et al., 2002	GAC	2.7	19.8-25.4°C	≥15°C	1
LeChevallier et al., 1992a, b	GAC	3.8	10.5°C	10-15°C	1.29
Krasner et al., 1993	GAC	0.4	California(2), May-September, 15-25°C	≥15°C	1
Krasner et al., 1993	GAC	1.1	California(2), May-September, 15-25°C	≥15°C	1
Persson et al. 2006	GAC	8.1	Sweden(1) 1.5-20.2°C, median 7.2°C	5-10°C	1.53
Shu et al., 2008	GAC	2.9	Southern China(3)	10-15°C	1.29
Vahala et al., 1998a	GAC	4.6	7.3°C	5-10°C	1.53
Vahala et al., 1998a	GAC	4.6	12.1°C	10-15°C	1.29
Vahala et al., 1998a	GAC	4.6	13.6°C	10-15°C	1.29
Vahala et al., 1998a	GAC	4.6	11.6°C	10-15°C	1.29
Vahala et al., 1998a	GAC	4.6	5.8°C	5-10°C	1.53
Vahala et al., 1998a	GAC	4.6	4.1°C	<5°C	1.81
Vahala et al., 1998a	GAC	4.6	4.0°C	<5°C	1.81
Vahala et al., 1998a	GAC	4.6	5.1°C	5-10°C	1.53
Vahala et al., 1998a	GAC	4.6	10.0°C	10-15°C	1.29
Vahala et al., 1998a	GAC	4.6	9.9°C	5-10°C	1.53
Vahala et al., 1998a	GAC	4.6	9.8°C	5-10°C	1.53
Vahala et al., 1998a	GAC	4.6	13.6°C	10-15°C	1.29
Vahala et al., 1998a	GAC	4.6	11.8°C	10-15°C	1.29

Vahala et al., 1998b	GAC	5.3	8.2-9.6°C	5-10°C	1.53
Vahala et al., 1998b	GAC	5.3	8.2-9.6°C	5-10°C	1.53
van der Aa et al., 2003	GAC	1.4	Netherlands(4)	5-10°C	1.53
van der Aa et al., 2003	GAC	4.8	Netherlands(4)	5-10°C	1.53
van der Aa et al., 2003	GAC	8.4	Netherlands(4)	5-10°C	1.53
van der Hoek et al., 2000	GAC	3.5	Netherlands(4)	5-10°C	1.53
van der Hoek et al., 2000	GAC	6.9	Netherlands(4)	5-10°C	1.53
van der Hoek et al., 2000	GAC	13.9	Netherlands(4)	5-10°C	1.53
van der Hoek et al., 2000	GAC	17.4	Netherlands(4)	5-10°C	1.53
Chien et al., 2008	Anthracite	1.5	25-30°C	≥15°C	1
Hijnen and van der Kooij, 1992	Sand	1.6	Netherlands 10-18°C	10-15°C	2.11
Hijnen and van der Kooij, 1992	Sand	4.0	Netherlands 10-18°C	10-15°C	2.11
Hijnen and van der Kooij, 1992	Sand	8.0	Netherlands 10-18°C	10-15°C	2.11
Hijnen and van der Kooij, 1992	Sand	40.0	Netherlands 10-18°C	10-15°C	2.11
Moll et al., 1999	Sand	5.0	5°C	5-10°C	3.46
Moll et al., 1999	Sand	5.0	20°C	≥15°C	1
Moll et al., 1999	Sand	5.0	35°C	≥15°C	1
Wang, et al., 1995	Sand	5.8	Ohio, USA 13-27°C	≥15°C	1

Wang and Summers, 1996	Sand	0.3	Ohio, USA(10)	≥15°C	1
Wang and Summers, 1996	Sand	0.5	Ohio, USA(10)	≥15°C	1
Wang and Summers, 1996	Sand	1.0	Ohio, USA(10)	≥15°C	1
Wang and Summers, 1996	Sand	1.4	Ohio, USA(10)	≥15°C	1
Wang and Summers, 1996	Sand	2.3	Ohio, USA(10)	≥15°C	1
Wang and Summers, 1996	Sand	2.4	Ohio, USA(10)	≥15°C	1
Wang and Summers, 1996	Sand	3.5	Ohio, USA(10)	≥15°C	1
Wang and Summers, 1996	Sand	4.3	Ohio, USA(10)	≥15°C	1
Persson et al., 2006	EC (expanded clay)	7.7	Sweden(1) 1.5-20.2°C, median 7.2°C	5-10°C	3.46
Persson et al., 2006	EC (expanded clay)	2.5	Sweden(1) 1.5-20.2°C, median 7.2°C	5-10°C	3.46
Hu et al., 1999	Bio-ceramic	3.8	Northern China(5) Spring	5-10°C	1.53
Hu et al., 2005	Zeolite	1.7	25-30°C	≥15°C	1
Hu et al., 2005	Zeolite	6.7	25-30°C	≥15°C	1
Hu et al., 2005	Zeolite	13.4	25-30°C	≥15°C	1
Ahmad et al.,1998	Dual-Anthracite/Sand	2.8	Alberta, Canada(11)	10-15°C	2.11
Ahmad et al.,1998	Dual-Anthracite/Sand	2.8	Alberta, Canada(11)	10-15°C	2.11
Ahmad et al.,1998	Dual-Anthracite/Sand	2.8	Alberta, Canada(11)	10-15°C	2.11
Ahmad et al.,1998	Dual-Anthracite/Sand	2.8	Alberta, Canada(11)	10-15°C	2.11

Ahmad et al.,1998	Dual-Anthracite/Sand	2.8	Alberta, Canada(11)	10-15°C	2.11
Daniel and Teefy, 1995	Dual-GAC/Sand	1.7	California, USA(2) Summer	≥15°C	1
Daniel and Teefy, 1995	Dual-Anthracite/Sand	1.7	California, USA(2) Summer	≥15°C	1
LeChevallier et al., 1992a,b	Mixed-Anthracite/Sand/Garnet	1.8	3.5°C	<5°C	5.70
LeChevallier et al., 1992a,b	Dual-GAC/Sand	1.8	3.5°C	<5°C	1.81
LeChevallier et al., 1992a,b	Dual-GAC/Sand	2.4	10.5°C	10-15°C	1.29
LeChevallier et al., 1992a,b	Dual-GAC/Sand	3.5	4.7-12.7°C	5-10°C	1.53
LeChevallier et al., 1992a,b	Dual-GAC/Sand	3.5	4.7-12.7°C	5-10°C	1.53
LeChevallier et al., 1992a,b	Dual-GAC/Sand	3.5	23.7°C	≥15°C	1
LeChevallier et al., 1992a,b	Dual-GAC/Sand	3.5	23.7°C	≥15°C	1
LeChevallier et al., 1992a,b	Dual-GAC/Sand	3.5	23.7°C	≥15°C	1
LeChevallier et al., 1992a,b	Dual-GAC/Sand	2.4	19.4°C	≥15°C	1
LeChevallier et al., 1992a,b	Dual-GAC/Sand	2.4	19.4°C	≥15°C	1
LeChevallier et al., 1992a,b	Dual-GAC/Sand	1.8	New Jersey, USA(6)	≥15°C	1
LeChevallier et al., 1992a,b	Dual-GAC/Sand	3.5	New Jersey, USA(6)	≥15°C	1
LeChevallier et al., 1992a,b	Dual-GAC/Sand	5.3	New Jersey, USA(6)	≥15°C	1
LeChevallier et al., 1992a,b	Dual-GAC/Sand	7.1	New Jersey, USA(6)	≥15°C	1
Najm et al., 2005	Dual- GAC/Sand	3.3	North Carolina, USA(7) Summer, highest of 28°C	≥15°C	1

Najm et al., 2005	Dual- GAC/Sand	18.2	North Carolina, USA(7) Winter, lowest of 11°C	10-15°C	1.29
Wang, et al., 1995	Dual-Anthracite/Sand	3.7	Ohio, USA 13-27°C	≥15°C	1
Wang, et al., 1995	Dual-Anthracite/Sand	3.7	Ohio, USA 13-27°C	≥15°C	1
Wang, et al., 1995	Dual-Anthracite/Sand	3.7	Ohio, USA 13-27°C	≥15°C	1
Wang, et al., 1995	Dual-GAC/Sand	4.3	Ohio, USA 13-27°C	≥15°C	1
Wang, et al., 1995	Dual-GAC/Sand	4.5	Ohio, USA 13-27°C	≥15°C	1
Wang, et al., 1995	Dual-GAC/Sand	1.7	Ohio, USA 13-27°C	≥15°C	1
Wert et al., 2008	Dual-Sand/ Anthracite	1.0	Nevada, USA Around 15-19°C	≥15°C	1
Wert et al., 2008	Dual-Sand/ Anthracite	3.0	Nevada, USA Around 15-19°C	≥15°C	1
Wert et al., 2008	Dual-Sand/ Anthracite	7.8	Nevada, USA Around 15-19°C	≥15°C	1
Zappia et al., 2007	Dual-Sand/Anthracite	2.1	Australia(8), March	5-10°C	3.46
Zappia et al., 2007	Dual-Sand/Anthracite	2.4	Australia(8), March	5-10°C	3.46
Zappia et al., 2007	Dual-Sand/Anthracite	2.4	Australia(8), March	5-10°C	3.46
Zappia et al., 2007	Dual-Sand/Anthracite	2.4	Australia(8), March	5-10°C	3.46
Zappia et al., 2007	Dual-Sand/GAC	1.9	Australia(8), March	5-10°C	3.46
Zappia et al., 2007	Dual-Sand/GAC	3.0	Australia(8), March	5-10°C	3.46
Zhang et al., 2004	Dual-GAC/Sand	3.3	8°C	5-10°C	3.46

Note: 1. http://en.wikipedia.org/wiki/Sweden (2009-08-10);
2. http://en.wikipedia.org/wiki/Climate_of_California (2009-08-10);
3. http://en.wikipedia.org/wiki/Changsha (2009-08-10);
4. http://en.wikipedia.org/wiki/Netherlands (2009-08-10);
5. http://en.wikipedia.org/wiki/Beijing (2009-08-10);
6. Estimation base, assuming no effect of temperature, refer to discussion in section 3.6.5;
7. http://en.wikipedia.org/wiki/North_Carolina (2009-08-10);
8. http://en.wikipedia.org/wiki/Climate_of_Australia (2009-08-10);
9. Temperature of most operation days stayed in the range of 10-15°C;
10. Determination of temperature range followed Wang et al. (1995);
11. Determination of temperature range followed Huck et al. (1991);
12. Refer to section 3.6.5 for determination approaches of representative temperature ranges;
13. Refer to Table 3-9 for temperature adjustment coefficients.

Appendix A-8 Temperature-adjusted $\theta\alpha$ parameter values

Source	Media Type	Initially Estimated $\theta\alpha$ Parameter Values	Temperature-adjusted $\theta\alpha$ Parameter Values	EBCT (min)	AOC Removal (%)
Chien et al.,2007	GAC	2.7	2.7	6.6	60
Chien et al., 2008	GAC	2.7	2.7	6.6	86
Heinicke et al., 2006	GAC	13.4	10.4	34	19
Huck et al.,1991	GAC	14.0	10.9	36	33
Huck et al.,1991	GAC	7.4	5.7	19	52
Huck et al.,1991	GAC	14.0	10.9	36	50
Huck et al.,1991	GAC	7.4	5.7	19	88
Huck et al.,1991	GAC	7.4	5.7	19	68
Huck et al.,1991	GAC	14.0	10.9	36	82
Huck et al.,1991	GAC	7.4	5.7	19	86
Huck et al.,1991	GAC	14.0	9.2	36	68
Huck et al.,1991	GAC	7.4	4.8	19	69
Huck et al.,1991	GAC	14.0	9.2	36	77
Huck et al.,1991	GAC	7.4	4.8	19	83
Lai et al., 2002	GAC	1.5	1.5	7.1	88
Lai et al., 2002	GAC	2.7	2.7	12.5	76

LeChevallier et al., 1992a, b	GAC	3.8	2.9	14	89
Krasner et al., 1993	GAC	0.4	0.4	1.4	72
Krasner et al., 1993	GAC	1.1	1.1	4.2	74
Persson et al. 2006	GAC	8.1	5.3	31	22
Shu et al., 2008	GAC	2.9	2.2	8.6	35
Vahala et al., 1998a	GAC	4.6	3.0	13	53
Vahala et al., 1998a	GAC	4.6	3.6	13	69
Vahala et al., 1998a	GAC	4.6	3.6	13	73
Vahala et al., 1998a	GAC	4.6	3.6	13	76
Vahala et al., 1998a	GAC	4.6	3.0	13	59
Vahala et al., 1998a	GAC	4.6	2.5	13	69
Vahala et al., 1998a	GAC	4.6	2.5	13	75
Vahala et al., 1998a	GAC	4.6	3.0	13	79
Vahala et al., 1998a	GAC	4.6	3.6	13	75
Vahala et al., 1998a	GAC	4.6	3.0	13	27
Vahala et al., 1998a	GAC	4.6	3.0	13	51
Vahala et al., 1998a	GAC	4.6	3.6	13	64
Vahala et al., 1998a	GAC	4.6	3.6	13	49
Vahala et al., 1998b	GAC	5.3	3.5	15	73

Vahala et al., 1998b	GAC	5.3	3.5	15	76
van der Aa et al., 2003	GAC	1.4	0.9	7	57
van der Aa et al., 2003	GAC	4.8	3.1	23	64
van der Aa et al., 2003	GAC	8.4	5.5	40	71
van der Hoek et al., 2000	GAC	3.5	2.3	8	64
van der Hoek et al., 2000	GAC	6.9	4.5	16	75
van der Hoek et al., 2000	GAC	13.9	9.1	32	76
van der Hoek et al., 2000	GAC	17.4	11.3	40	80
Chien et al., 2008	Anthracite	1.5	1.5	6.6	17
Hijnen and van der Kooij, 1992	Sand	1.6	0.8	2.4	72
Hijnen and van der Kooij, 1992	Sand	4.0	1.9	6	72
Hijnen and van der Kooij, 1992	Sand	8.0	3.8	12	72
Hijnen and van der Kooij, 1992	Sand	40.0	19.0	60	76
Moll et al., 1999	Sand	5.0	1.4	7.0	43
Moll et al., 1999	Sand	5.0	5.0	7.0	55
Moll et al., 1999	Sand	5.0	5.0	7.0	57
Wang, et al., 1995	Sand	5.8	5.8	9.2	43
Wang and Summers, 1996	Sand	0.3	0.3	0.4	59

Wang and Summers, 1996	Sand	0.5	0.5	0.8	61
Wang and Summers, 1996	Sand	1.0	1.0	1.6	56
Wang and Summers, 1996	Sand	1.4	1.4	2.2	65
Wang and Summers, 1996	Sand	2.3	2.3	3.6	68
Wang and Summers, 1996	Sand	2.4	2.4	3.8	72
Wang and Summers, 1996	Sand	3.5	3.5	5.6	64
Wang and Summers, 1996	Sand	4.3	4.3	6.8	76
Persson et al., 2006	EC (expanded clay)	7.7	2.2	31	35
Persson et al., 2006	EC (expanded clay)	2.5	0.7	31	41
Hu et al., 1999	Bio-ceramic	3.8	2.5	40	44
Hu et al., 2005	Zeolite	1.7	1.7	7.5	4
Hu et al., 2005	Zeolite	6.7	6.7	30	45
Hu et al., 2005	Zeolite	13.4	13.4	60	53
Ahmad et al.,1998	Dual-Anthracite/Sand	2.8	1.3	5.4	79
Ahmad et al.,1998	Dual-Anthracite/Sand	2.8	1.3	5.4	78
Ahmad et al.,1998	Dual-Anthracite/Sand	2.8	1.3	5.4	55
Ahmad et al.,1998	Dual-Anthracite/Sand	2.8	1.3	5.4	21
Ahmad et al.,1998	Dual-Anthracite/Sand	2.8	1.3	5.4	80
Daniel and Teefy, 1995	Dual-GAC/Sand	1.7	1.7	6.4	70

Daniel and Teefy, 1995	Dual-Anthracite/Sand	1.7	1.7	6.4	70
LeChevallier et al., 1992a,b	Mixed-Anthracite/Sand/Garnet	1.8	0.3	5	75
LeChevallier et al., 1992a,b	Dual-GAC/Sand	1.8	1.0	5	86
LeChevallier et al., 1992a,b	Dual-GAC/Sand	2.4	1.9	7	85
LeChevallier et al., 1992a,b	Dual-GAC/Sand	3.5	2.3	10	91
LeChevallier et al., 1992a,b	Dual-GAC/Sand	3.5	2.3	10	85
LeChevallier et al., 1992a,b	Dual-GAC/Sand	3.5	3.5	10	75
LeChevallier et al., 1992a,b	Dual-GAC/Sand	3.5	3.5	10	79
LeChevallier et al., 1992a,b	Dual-GAC/Sand	3.5	3.5	10	84
LeChevallier et al., 1992a,b	Dual-GAC/Sand	2.4	2.4	7	82
LeChevallier et al., 1992a,b	Dual-GAC/Sand	2.4	2.4	7	84
LeChevallier et al., 1992a,b	Dual-GAC/Sand	1.8	1.8	5	83
LeChevallier et al., 1992a,b	Dual-GAC/Sand	3.5	3.5	10	84
LeChevallier et al., 1992a,b	Dual-GAC/Sand	5.3	5.3	15	84
LeChevallier et al., 1992a,b	Dual-GAC/Sand	7.1	7.1	20	89
Najm et al., 2005	Dual- GAC/Sand	3.3	3.3	10	75
Najm et al., 2005	Dual- GAC/Sand	18.2	14.1	55	75
Wang, et al., 1995	Dual-Anthracite/Sand	3.7	3.7	9.2	39

Wang, et al., 1995	Dual-Anthracite/Sand	3.7	3.7	9.2	37
Wang, et al., 1995	Dual-Anthracite/Sand	3.7	3.7	9.2	48
Wang, et al., 1995	Dual-GAC/Sand	4.3	4.3	9.2	51
Wang, et al., 1995	Dual-GAC/Sand	4.5	4.5	9.2	47
Wang, et al., 1995	Dual-GAC/Sand	1.7	1.7	9.2	42
Wert et al., 2008	Dual-Sand/ Anthracite	1.0	1.0	3.2	60
Wert et al., 2008	Dual-Sand/ Anthracite	3.0	3.0	9.7	60
Wert et al., 2008	Dual-Sand/ Anthracite	7.8	7.8	25	60
Zappia et al., 2007	Dual-Sand/Anthracite	2.1	0.6	8	76
Zappia et al., 2007	Dual-Sand/Anthracite	2.4	0.7	8	79
Zappia et al., 2007	Dual-Sand/Anthracite	2.4	0.7	8	42
Zappia et al., 2007	Dual-Sand/Anthracite	2.4	0.7	8	42
Zappia et al., 2007	Dual-Sand/GAC	1.9	0.5	8	68
Zappia et al., 2007	Dual-Sand/GAC	3.0	0.9	8	76
Zhang et al., 2004	Dual-GAC/Sand	3.3	2.2	6.8	84

Note: Refer to 3.5.5 for determining temperature-adjustment $\theta\alpha$ parameter values.

Appendix A-9 Ozonation dosages and AOC removal efficiencies

Source	EBCT (min)	Temperature-Adjusted $\theta\alpha$ Parameter Values	Ozone Dosage (mgO_3/mg TOC)	AOC Removal (%)
Heinicke et al., 2006	34	11.4	0	19
Hu et al., 1999	40	2.5	0	44
Persson et al., 2006	31	2.2	0	35
Persson et al., 2006	31	0.7	0	41
Persson et al., 2006	31	5.2	0	22
Ahmad et al.,1998	5.4	2.2	1.00	79
Ahmad et al.,1998	5.4	2.2	1.00	78
Ahmad et al.,1998	5.4	2.2	1.00	55
Ahmad et al.,1998	5.4	2.2	1.00	21
Ahmad et al.,1998	5.4	2.2	1.00	80
Daniel and Teefy, 1995	6.4	1.7	0.60	70
Daniel and Teefy, 1995	6.4	1.7	0.60	70
Huck et al.,1991	36	11.9	0.50	33
Huck et al.,1991	19	6.3	1.00	52
Huck et al.,1991	36	11.9	0.50	50
Huck et al.,1991	19	6.3	1.00	88
Huck et al.,1991	19	6.3	0.50	68
Huck et al.,1991	36	11.9	1.00	82
Huck et al.,1991	19	6.3	1.00	86
Huck et al.,1991	36	11.9	0.50	68
Huck et al.,1991	19	6.3	1.00	69
Huck et al.,1991	36	11.9	0.50	77
Huck et al.,1991	19	6.3	1.00	83

Krasner et al., 1993	1.4	0.4	0.60	72
Krasner et al., 1993	4.2	1.1	0.60	74
Lai et al., 2002	7.1	1.5	0.43	88
Lai et al., 2002	12.5	2.7	0.60	76
LeChevallier et al., 1992a, b	14	3.2	0.75	89
LeChevallier et al., 1992a,b	5	0.4	0.75	75
LeChevallier et al., 1992a,b	5	1.2	0.75	86
LeChevallier et al., 1992a,b	7	2.0	0.75	85
LeChevallier et al., 1992a,b	10	3.0	0.75	91
LeChevallier et al., 1992a,b	10	3.5	0.75	75
LeChevallier et al., 1992a,b	7	2.4	0.75	82
LeChevallier et al., 1992a,b	5	1.8	0.75	83
LeChevallier et al., 1992a,b	10	3.5	0.75	84
LeChevallier et al., 1992a,b	15	5.3	0.75	84
LeChevallier et al., 1992a,b	20	7.1	0.75	89
Moll et al., 1999	7.0	1.4	1.30	43
Moll et al., 1999	7.0	5	1.30	55
Moll et al., 1999	7.0	5	1.30	57
Najm et al., 2005	10	3.3	0.78	75
Najm et al., 2005	55	15.4	0.78	75
Shu et al., 2008	8.6	2.5	0.39	35
Vahala et al., 1998a	13	3.0	0.52	53
Vahala et al., 1998a	13	3.9	0.52	69
Vahala et al., 1998a	13	3.9	0.46	73
Vahala et al., 1998a	13	3.9	0.52	76
Vahala et al., 1998a	13	3.0	0.39	59

Vahala et al., 1998a	13	3.0	0.39	69
Vahala et al., 1998a	13	3.0	0.32	75
Vahala et al., 1998a	13	3.0	0.28	79
Vahala et al., 1998a	13	3.9	0.31	75
Vahala et al., 1998a	13	3.0	0.39	27
Vahala et al., 1998a	13	3.0	0.22	51
Vahala et al., 1998a	13	3.9	1.04	64
Vahala et al., 1998a	13	3.9	1.23	49
van der Hoek et al., 2000	8	2.3	0.43	64
van der Hoek et al., 2000	16	4.5	0.43	75
van der Hoek et al., 2000	32	9.0	0.43	76
van der Hoek et al., 2000	40	11.2	0.43	80
Wang, et al., 1995	9.2	4.5	0.88	43
Wang, et al., 1995	9.2	2.9	0.88	39
Wang, et al., 1995	9.2	2.9	0.88	37
Wang, et al., 1995	9.2	2.9	0.88	48
Wang, et al., 1995	9.2	3.6	0.88	51
Wang, et al., 1995	9.2	3.8	0.88	47
Wang, et al., 1995	9.2	1.4	0.88	42
Wang and Summers, 1996	0.4	0.2	0.35	59
Wang and Summers, 1996	0.8	0.4	0.35	61
Wang and Summers, 1996	1.6	0.8	0.35	56
Wang and Summers, 1996	2.2	1.1	0.35	65
Wang and Summers, 1996	3.6	1.8	0.35	68
Wang and Summers, 1996	3.8	1.9	0.35	72
Wang and Summers, 1996	5.6	2.7	0.35	64

Wang and Summers, 1996	6.8	3.4	0.35	76
Wert et al., 2008	3.2	0.8	0.57	60
Wert et al., 2008	9.7	2.3	0.57	60
Wert et al., 2008	25	6.1	0.57	60

Note: 1. Refer to 3.7 for calculation of normalized ozone dosages and related discussions;
2. First five rows are associated with biofiltration processes without pre-ozonation.

Appendix B-1 Reference for collected BDOC removal cases

Fonseca A.C. and Summers R.S. 2003. Evaluation of different ozonation strategies and of temperature effects on biological filter performance. Proceedings AWWA water quality and technology conference, Philadelphia, PA

Griffini O., Bao M.L., Barbieri K., Burrini D., Santianni D. and Pantani F. 1999. Formation and removal of biodegradable ozonation by-products during ozonation-biofiltration treatment: pilot-scale evaluation. *Ozone Science and Engineering*, 21, 1: 79-98

Heinicke G., Persson F., Uhl W., Hermansson M. and Hedberg T. 2006. The effect of biological pre-filtration on the performance of conventional surface water treatment. *Journal of Water Supply: Research and Technology-AQUA*, 55, 2: 109-119

Melin E.S., Eikebrokk B., Brugger M. and Ødegaard H. 2002. Treatment of humic surface water at cold temperatures by ozonation and biofiltration. *Water Science and Technology: Water Supply*, 2: 451-457

Moll D.M., Summers R.S., Fonseca A.C. and Matheis W. 1999. Impact of temperature on drinking water biofilters performance and microbial community structure. *Environmental Science and Technology*, 33: 2377-2382

Niquette P., Prévost M., Maclean R.G., Thibault D., Coallier J., Desjardins R. and Lafrance P. 1998. Backwashing first-stage sand-BAC filters. *Journal of American Water Works Association*, 90, 1: 86-97

Nishijima W. and Speitel G.E. 2004. Fate of biodegradable dissolved organic carbon produced by ozonation on biological activated carbon. *Chemosphere*, 56: 113-119

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Servais P.G., Billen C. and Bouillot P. 1994. Biological colonization of granular activated carbon filters in drinking water treatment. *Journal of Environmental Engineering*, 120, 4: 888-898

Thiel P., Zappia L., Warton B., Nolan P., Scott D., Alessandrino M., Franzmann P., Hiller B., Heitz A. and Masters D. 2006. Activated carbon vs. anthracite as primary dual media filters-A pilot plant study. Proceedings 69th Annual water industry engineers and operators conference, Bendigo, Australia

Wang X.C., Wang S. and Jin P.K. 2000. Tertiary treatment of domestic wastewater by low-concentration-ozonation and biofiltration for water reclamation. Proceedings International conference on advances in ozone science and engineering, Hongkong

Wang J.Z. and Summers R.S. 1996. Biodegradation behavior of ozonated natural organic matter in sand filters. *Revue des sciences de l'eau*, 9, 1: 3-16

Zappia L.R., Warton B., Alessandrino M., Scott D., Wylie J.T., Heitz A., Hiller B., Masters D., Nolan P., Thiel P., Kagi R.I., Joll C.A. and Franzmann P.D. 2007. Pilot scale testing of biofilter post-treatment of MIEX® treated water. *Journal of Water Supply: Research and Technology-AQUA*, 56, 4: 217-232

Appendix B-2 Filter media particle sizes for BDOC removal cases

Source	**Media Type***	**Average Size d(mm)**	**Effective Size d_{10}(mm)**	**Uniformity Coefficient**
Griffini et al., 1999	GAC		0.88	1.17
Griffini et al., 1999	GAC		0.90-1.10	1.88
Griffini et al., 1999	GAC		1.07	1.25
Griffini et al., 1999	GAC		0.90-1.10	1.42
Heinicke et al., 2006	GAC		0.63	1.70
Niquette et al., 1998	GAC		1.00	1.70
Persson et al. 2006	GAC		0.80-1.00	<2.10
Prevost et al., 1992	GAC		0.64	<1.60
Servais et al., 1991	GAC		0.80	1.80
Servais et al., 1994	GAC		0.80	1.80
Wang et al., 2002	GAC		1.00	1.14*
Fonseca and Summers, 2003	Sand		0.44	1.4-1.7
Moll et al., 1999	Sand		0.40	1.40-1.70***
Wang and Summers, 1996	Sand		0.44	1.40-1.70
Griffini et al., 1999	Biolite	1.20		
Sang et al., 2003	Bio-ceramic	2.00-5.00		

Persson et al., 2006	EC (expanded clay)		0.90-1.00	<1.50
Persson et al., 2006	EC (expanded clay)		2.45-2.75	<1.50
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)		GAC 1.00 Sand 0.50	GAC 1.70 Sand 1.50
Nishijima and Speitel, 2004	Dual-GAC (1/3) Dual-Sand (2/3)	Sand 1.20-2.00	GAC 0.64	GAC <1.60
Melin et al., 2002	Mixed-GAC (0.8m) Mixed-phonolith (1.1m) Mixed-calcium carbonate (0.4m)	GAC 0.50-2.50 Phonolith 0.30-0.80 Calcium carbonate 1.00-3.00		
Thiel, et al., 2006	Dual- Anthracite (1.75m) Dual-Sand (0.65m)	Anthracite 1.10 Sand 0.65		
Thiel, et al., 2006	Dual- GAC (1.75m) Dual- Sand (0.65m)	GAC 1.30 Sand 0.65		
Thiel, et al., 2006	Dual-GAC (1.75m) Dual-Sand (0.65m)	GAC 0.70 Sand 0.65		
Wang, et al., 1995	Dual- Anthracite (20inches) Dual- Sand (10inches)		Anthracite 1.02 Sand 0.44	Anthracite 1.32 Sand 1.41
Zappia et al., 2007	Dual-sand (0.30m) Dual-Anthracite (1.75m)		Sand 0.65 Anthracite 1.10	Sand 1.40, Anthracite 1.60
Zappia et al., 2007	Dual-sand (0.30m) Dual-Anthracite (0.65m)		Sand 0.65 Anthracite 1.10	Sand 1.40 Anthracite 1.60
Zappia et al., 2007	Dual-sand (0.30m) Dual-GAC (1.75m)		Sand 0.65 GAC 1.30	Sand 1.40 GAC 1.40
Zappia et al., 2007	Dual-sand (0.30m) Dual-GAC (1.75m)		Sand 0.65 GAC 0.70	Sand 1.40 GAC 1.70

Note: 1.Refer to section 3.1.1 for collecting and determining representative sizes of filter media particles;

* For dual-media and multi-media filter, whichever is listed in upper place of a cell in any Appendix table is the toper layer in the corresponding biofilter, whichever is listed in bottom place of a cell is the bottom layer in the biofilters.

Appendix B-3 Actively specific surface area of filter media particles

Source	Media Type	Representative Size (mm)	Specific Surface Area for Ideal Spherical Shape (m^{-1})	Actively Specific Surface Area (m^{-1})
Griffini et al., 1999	GAC	0.95	6320	4930
Griffini et al., 1999	GAC	1.44	4170	3250
Griffini et al., 1999	GAC	1.20	5000	3900
Griffini et al., 1999	GAC	1.21	4960	3870
Heinicke et al., 2006	GAC	0.85	7060	5480
Niquette et al., 1998	GAC	1.35	4440	3460
Persson et al. 2006	GAC	1.40	4290	3590
Prevost et al., 1992	GAC	0.83	7230	5640
Servais et al., 1991	GAC	1.12	5360	4180
Servais et al., 1994	GAC	1.12	5360	4180
Wang et al., 2002	GAC	1.07	5610	4380
Fonseca and Summers, 2003	Sand	0.56	10710	8350
Moll et al., 1999	Sand	051	11760	9170
Wang and Summers, 1996	Sand	0.56	10710	8350
Sang et al., 2003	Bio-ceramic	3.50	1710	1340
Griffini et al., 1999	Biolite	1.20	5000	3900

Persson et al., 2006	EC (expanded clay)	1.19	5040	3470
Persson et al., 2006	EC (expanded clay)	3.25	1850	1120
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 1.35 Sand 0.63	GAC 4440 Sand 9520	GAC 3460 Sand 7430
Nishijima and Speitel, 2004	Dual-GAC (1/3) Dual-Sand (2/3)	GAC 0.83 Sand 1.60	GAC 7230 Sand 3750	GAC 5640 Sand 2930
Melin et al., 2002	Mixed-GAC (0.8m) Mixed-Phonolith (1.1m) Mixed-Calcium carbonate (0.4m)	GAC 1.50 Phonolith 0.55 Calcium carbonate 2.00	GAC 4000 Phonolith 10910 Calcium carbonate 3000	GAC 3120 Phonolith 8510 Calcium carbonate 2340
Thiel, et al., 2006	Dual-Anthracite(1.75m) Dual-Sand(0.65m)	Anthracite 1.10 Sand 0.65	Anthracite 5450 Sand 9230	Anthracite 4250 Sand 7200
Thiel, et al., 2006	Dual-GAC(1.75m) Dual-Sand(0.65m)	GAC 1.30 Sand 0.65	GAC 4620 Sand 9230	GAC 3600 Sand 7200
Thiel, et al., 2006	Dual-GAC(1.75m) Dual-Sand(0.65m)	GAC 0.70 Sand 0.65	GAC 8570 Sand 9230	GAC 6680 Sand 7200
Wang, et al., 1995	Dual-Anthracite (20inches) Dual-Sand (10inches)	Anthracite 1.18 Sand 0.53	Anthracite 5080 Sand 11320	Anthracite 3920 Sand 8830
Zappia et al., 2007	Dual-Sand (0.3m) Dual-Anthracite (1.75m)	Sand 0.78 Anthracite 1.43	Sand 7690 Anthracite 4200	Sand 5970 Anthracite 3260
Zappia et al., 2007	Dual-Sand (0.3m) Dual-Anthracite (0.65m)	Sand 5970 Anthracite 3260	Sand 7690 Anthracite 4200	Sand 5970 Anthracite 3260
Zappia et al., 2007	Dual-Sand (0.3m) Dual-GAC (1.75m)	Sand 5970 GAC 2990	Sand 7690 GAC 3850	Sand 5970 GAC 2990
Zappia et al., 2007	Dual-Sand (0.3m) Dual-GAC (1.75m)	Sand 5970 GAC 5000	Sand 7690 GAC 6320	Sand 5970 GAC 5000

Note: 1. Refer to section 3.1.1 for collection and determination of representative sizes of filter media particles;

2. Refer to section 3.2 for calculation of actively specific surface area of filter media;

3. Actively specific surface area of filter media was calculated by Equation (3-9):

$$\alpha = \frac{6 \times \beta \times \varphi \times (1 - \epsilon)}{\bar{d}} \qquad (3-9)$$

Appendix B-4 Collection and calculation results of EBCTs, BDOC removal percentages and EBCT-normalized percentage removals

Source	Media Type	EBCT(1) (min)	BDOC Removal (%)(2)	EBCT-Normalized Percentage Removal (min^{-1})
Griffini et al., 1999	GAC	2.3(3)	58(3)	25
Griffini et al., 1999	GAC	10(3)	70(3)	7.0
Griffini et al., 1999	GAC	15(3)	71(3)	4.7
Griffini et al., 1999	GAC	20(3)	74(3)	3.7
Griffini et al., 1999	GAC	2.3(3)	52(3)	22.6
Griffini et al., 1999	GAC	10(3)	68(3)	6.8
Griffini et al., 1999	GAC	15(3)	72(3)	4.8
Griffini et al., 1999	GAC	20(3)	71(3)	3.6
Griffini et al., 1999	GAC	2.3(3)	54(3)	23.5
Griffini et al., 1999	GAC	10(3)	66(3)	6.6
Griffini et al., 1999	GAC	15(3)	69(3)	4.6
Griffini et al., 1999	GAC	20(3)	70(3)	3.5
Griffini et al., 1999	GAC	2.3(3)	59(3)	25.7
Griffini et al., 1999	GAC	10(3)	70(3)	7.0
Griffini et al., 1999	GAC	15(4)	74(3)	4.9
Griffini et al., 1999	GAC	20(4)	72(3)	3.6
Heinicke et al., 2006	GAC	34(3)	23(3)	0.7
Niquette et al., 1998	GAC	4(6)	32(6)	8.0
Niquette et al., 1998	GAC	7(6)	25(6)	3.6
Niquette et al., 1998	GAC	7(6)	21(6)	3.0
Niquette et al., 1998	GAC	12(6)	32(6)	2.7
Niquette et al., 1998	GAC	12(6)	15(6)	1.3
Niquette et al., 1998	GAC	14(6)	38(6)	2.7

Niquette et al., 1998	GAC	14(6)	23(6)	1.6
Heinicke et al., 2006	GAC	34(4)	23(3)	0.7
Persson et al., 2006	GAC	31(3)	34(3)	1.1
Prevost et al., 1992	GAC	5(3)	8(6)	1.6
Prevost et al., 1992	GAC	7.5(3)	41(6)	5.5
Prevost et al., 1992	GAC	14(3)	95(6)	6.8
Servais et al., 1991	GAC	2(3)	25(4)	12.5
Servais et al., 1991	GAC	4(3)	50(4)	12.5
Servais et al., 1991	GAC	10(3)	53(4)	5.3
Servais et al., 1991	GAC	10(3)	39(4)	3.9
Servais et al., 1991	GAC	10(3)	14(4)	1.4
Servais et al., 1991	GAC	10(3)	22(4)	2.2
Servais et al., 1991	GAC	10(3)	60(4)	6.0
Servais et al., 1991	GAC	10(3)	60(4)	6.0
Servais et al., 1991	GAC	10(3)	50(4)	5.0
Servais et al., 1991	GAC	10(3)	42(4)	4.2
Servais et al., 1991	GAC	10(3)	50(4)	5.0
Servais et al., 1991	GAC	10(3)	24(4)	2.4
Servais et al., 1991	GAC	10(3)	45(4)	4.5
Servais et al., 1991	GAC	10(3)	29(4)	2.9
Servais et al., 1994	GAC	10(3)	43(4)	4.3
Servais et al., 1994	GAC	10(3)	56(4)	5.6
Servais et al., 1994	GAC	10(3)	51(4)	5.1
Servais et al., 1994	GAC	10(3)	49(4)	4.9
Servais et al., 1994	GAC	10(3)	56(4)	5.6
Wang et al., 2002	GAC	14(4)	78(5)	5.6

Fonseca and Summers, 2003	Sand	3(3)	92(3)	31
Fonseca and Summers, 2003	Sand	10(3)	98(3)	9.8
Fonseca and Summers, 2003	Sand	17(3)	100(3)	5.9
Fonseca and Summers, 2003	Sand	3(3)	92(3)	31
Fonseca and Summers, 2003	Sand	10(3)	78(3)	7.8
Fonseca and Summers, 2003	Sand	17(3)	89(3)	5.2
Fonseca and Summers, 2003	Sand	3(3)	71(3)	24
Fonseca and Summers, 2003	Sand	10(3)	82(3)	8.2
Fonseca and Summers, 2003	Sand	17(3)	82(3)	4.8
Fonseca and Summers, 2003	Sand	3(3)	65(3)	22
Fonseca and Summers, 2003	Sand	10(3)	65(3)	6.5
Fonseca and Summers, 2003	Sand	17(3)	68(3)	4.0
Moll et al., 1999	Sand	7.0(3)	38(3)	5.4
Moll et al., 1999	Sand	7.0(3)	60(3)	8.6
Moll et al., 1999	Sand	7.0(3)	60(4)	8.6
Wang and Summers, 1996	Sand	1.5(3)	16(4)	11
Wang and Summers, 1996	Sand	2(3)	22(4)	11
Wang and Summers, 1996	Sand	4(3)	32(4)	8.0
Wang and Summers, 1996	Sand	7(3)	36(4)	5.1
Wang and Summers, 1996	Sand	10(3)	40(4)	4.0
Sang et al., 2003	Bio-ceramic	20(4)	56(5)	2.8
Sang et al., 2003	Bio-ceramic	20(4)	55(4)	2.8
Sang et al., 2003	Bio-ceramic	20(4)	70(4)	3.5
Sang et al., 2003	Bio-ceramic	20(4)	71(4)	3.6
Griffini et al., 1999	Biolite	2.3(3)	47(3)	20
Griffini et al., 1999	Biolite	10(3)	65(3)	6.5

Griffini et al., 1999	Biolite	15(3)	69(3)	4.6
Griffini et al., 1999	Biolite	20(3)	66(3)	3.3
Persson et al., 2006	EC (expanded clay)	31(3)	30(4)	1.0
Persson et al., 2006	EC (expanded clay)	31(3)	28(4)	0.9
Niquette et al., 1998	Dual-GAC/sand	3(6)	24(5)	8.0
Niquette et al., 1998	Dual-GAC/sand	6(6)	26(5)	4.3
Niquette et al., 1998	Dual-GAC/sand	12(6)	42(5)	3.5
Niquette et al., 1998	Dual-GAC/sand	14(6)	47(5)	3.4
Niquette et al., 1998	Dual-GAC/sand	3(6)	10(5)	3.3
Niquette et al., 1998	Dual-GAC/sand	6(6)	37(5)	6.2
Niquette et al., 1998	Dual-GAC/sand	3(6)	7(5)	2.3
Niquette et al., 1998	Dual-GAC/sand	6(6)	25(5)	4.2
Niquette et al., 1998	Dual-GAC/sand	12(6)	35(5)	2.9
Niquette et al., 1998	Dual-GAC/sand	14(6)	45(5)	3.2
Niquette et al., 1998	Dual-GAC/sand	3(6)	21(5)	7.0
Niquette et al., 1998	Dual-GAC/sand	6(6)	35(5)	5.8
Niquette et al., 1998	Dual-GAC/sand	12(6)	39(5)	3.3
Niquette et al., 1998	Dual-GAC/sand	14(6)	51(5)	3.6
Niquette et al., 1998	Dual-GAC/sand	3(6)	27(5)	9
Niquette et al., 1998	Dual-GAC/sand	6(6)	24(5)	4
Niquette et al., 1998	Dual-GAC/sand	12(6)	48(5)	4
Niquette et al., 1998	Dual-GAC/sand	14(6)	48(5)	3.4
Nishijima and Speitel, 2004	Dual-GAC/sand	45(6)	63(5)	1.4
Melin et al., 2002	Mixed	26-30(3) (7)	36(3)	1.3
Melin et al., 2002	Mixed	26-30(3) (7)	68(3)	2.4

Melin et al., 2002	Mixed	26-30(3) (7)	61(3)	2.2
Melin et al., 2002	Mixed	26-30(3) (7)	66(3)	2.4
Melin et al., 2002	Mixed	26-30(3) (7)	42(3)	1.5
Melin et al., 2002	Mixed	26-30(3) (7)	59(3)	2.1
Melin et al., 2002	Mixed	26-30(3) (7)	52(3)	1.9
Thiel, et al., 2006	Dual-Anthracite/Sand	8(3)	25(5)	3.1
Thiel, et al., 2006	Dual-GAC/Sand	8(3)	25(5)	3.1
Thiel, et al., 2006	Dual-GAC/Sand	8(3)	12(5)	1.5
Thiel, et al., 2006	Dual-GAC/Sand	8(3)	20(5)	2.5
Thiel, et al., 2006	Dual-GAC/Sand	8(3)	23(5)	2.9
Zappia et al., 2007	Dual-Sand/Anthracite	8(3)	5(4)	0.6
Zappia et al., 2007	Dual-Sand/Anthracite	16(3)	5(4)	0.3
Zappia et al., 2007	Dual-Sand/Anthracite	16(3)	14(4)	1.0
Zappia et al., 2007	Dual-Sand/Anthracite	8(3)	25(4)	3.1
Zappia et al., 2007	Dual-Sand/GAC	8(3)	18(4)	2.3
Zappia et al., 2007	Dual-Sand/GAC	16(3)	23(4)	1.4
Zappia et al., 2007	Dual-Sand/GAC	8(3)	25(4)	3.1
Zappia et al., 2007	Dual-Sand/GAC	8(3)	11(4)	1.4
Zappia et al., 2007	Dual-Sand/GAC	8(3)	20(4)	2.5
Zappia et al., 2007	Dual-Sand/GAC	8(3)	14(4)	1.8
Zappia et al., 2007	Dual-Sand/GAC	8(3)	5(4)	0.6
Zappia et al., 2007	Dual-Sand/GAC	16(3)	23(4)	1.4
Zappia et al., 2007	Dual-Sand/GAC	8(3)	25(4)	3.1
Zappia et al., 2007	Dual-Sand/GAC	8(3)	23(4)	2.9
Zappia et al., 2007	Dual-Sand/GAC	8(3)	12(4)	1.5

Note: 1. Refer to section 3.1.2 for collecting and determining EBCTs;

2. Refer to section 3.1.3 for determining BDOC removal percentages;

3. Data provided in the literature;

4. Calculated using provided supporting data;

4.1. EBCT calculation is as Equation (3-5):

$$EBCT = \frac{h}{HLR} = \frac{h}{Q/A} \qquad (3-5)$$

4.2. BDOC removal percentage calculation is as below:

$$BDOC\ Removal\ Percentage = \frac{BDOC_{Filter\ Influent} - BDOC_{Filter\ Effluent}}{BDOC_{Filter\ Influent}} \qquad (B-1)$$

5. Measured from figures in the literature and calculated using Equation (3-4) and equation above;

6. Measured from figures in the literature;

7. Average values;

8. EBCT-normalized percentage removal calculation is below:

$$EBCT\ Normalized\ Percentage\ Removal = \frac{BDOC\ Removal\ Percentage}{EBCT} \qquad (B-2)$$

Appendix B-5 Initial estimation of $\theta\alpha$ parameter values

Source	Media Type	α (m^{-1})	EBCT (min)	α × EBCT (×10^{4}, m^{-1}×min)	Initially Estimated $\theta\alpha$ Parameter Values$^{(1)}$
Griffini et al., 1999	GAC	4930	2.3	1.1	4.1
Griffini et al., 1999	GAC	4930	10	4.9	18.3
Griffini et al., 1999	GAC	4930	15	7.4	27.8
Griffini et al., 1999	GAC	4930	20	9.9	37.
Griffini et al., 1999	GAC	3250	2.3	0.7	2.6
Griffini et al., 1999	GAC	3250	10	3.3	12.4
Griffini et al., 1999	GAC	3250	15	4.9	18.4
Griffini et al., 1999	GAC	3250	20	6.5	24.4
Griffini et al., 1999	GAC	3900	2.3	0.9	3.4
Griffini et al., 1999	GAC	3900	10	3.9	14.6
Griffini et al., 1999	GAC	3900	15	5.9	22.1
Griffini et al., 1999	GAC	3900	20	7.8	29.3
Griffini et al., 1999	GAC	3870	2.3	0.9	3.4
Griffini et al., 1999	GAC	3870	10	3.9	14.6
Griffini et al., 1999	GAC	3870	15	5.8	21.8
Griffini et al., 1999	GAC	3870	20	7.7	28.9

Heinicke et al., 2006	GAC	5480	34	18.6	69.8
Niquette et al., 1998	GAC	3460	4	1.4	5.3
Niquette et al., 1998	GAC	3460	7	2.4	9.0
Niquette et al., 1998	GAC	3460	7	2.4	9.0
Niquette et al., 1998	GAC	3460	12	4.2	15.8
Niquette et al., 1998	GAC	3460	12	4.2	15.8
Niquette et al., 1998	GAC	3460	14	4.8	18.0
Niquette et al., 1998	GAC	3460	14	4.8	18.0
Persson et al., 2006	GAC	3590	31	11.1	41.6
Prevost et al., 1992	GAC	5640	5	2.8	10.5
Prevost et al., 1992	GAC	5640	7.5	4.2	15.8
Prevost et al., 1992	GAC	5640	14	7.9	29.6
Servais et al., 1991	GAC	4180	2	0.8	3.0
Servais et al., 1991	GAC	4180	4	1.7	6.4
Servais et al., 1991	GAC	4180	10	4.2	15.8
Servais et al., 1991	GAC	4180	10	4.2	15.8
Servais et al., 1991	GAC	4180	10	4.2	15.8
Servais et al., 1991	GAC	4180	10	4.2	15.8
Servais et al., 1991	GAC	4180	10	4.2	15.8

Servais et al., 1991	GAC	4180	10	4.2	15.8
Servais et al., 1991	GAC	4180	10	4.2	15.8
Servais et al., 1991	GAC	4180	10	4.2	15.8
Servais et al., 1991	GAC	4180	10	4.2	15.8
Servais et al., 1991	GAC	4180	10	4.2	15.8
Servais et al., 1991	GAC	4180	10	4.2	15.8
Servais et al., 1991	GAC	4180	10	4.2	15.8
Servais et al., 1994	GAC	4180	10	4.2	15.8
Servais et al., 1994	GAC	4180	10	4.2	15.8
Servais et al., 1994	GAC	4180	10	4.2	15.8
Servais et al., 1994	GAC	4180	10	4.2	15.8
Servais et al., 1994	GAC	4180	10	4.2	15.8
Wang et al., 2002	GAC	4380	14	6.1	22.9
Fonseca and Summers, 2003	Sand	8350	3	2.5	9.4
Fonseca and Summers, 2003	Sand	8350	10	8.4	31.5
Fonseca and Summers, 2003	Sand	8350	17	14.2	53.3
Fonseca and Summers, 2003	Sand	8350	3	2.5	9.4
Fonseca and Summers, 2003	Sand	8350	10	8.4	31.5

Fonseca and Summers, 2003	Sand	8350	17	14.2	53.3
Fonseca and Summers, 2003	Sand	8350	3	2.5	9.4
Fonseca and Summers, 2003	Sand	8350	10	8.4	31.5
Fonseca and Summers, 2003	Sand	8350	17	14.2	53.3
Fonseca and Summers, 2003	Sand	8350	3	2.5	9.4
Fonseca and Summers, 2003	Sand	8350	10	8.4	31.5
Fonseca and Summers, 2003	Sand	8350	17	14.2	53.3
Moll et al., 1999	Sand	9170	7.0	6.4	24
Moll et al., 1999	Sand	9170	7.0	6.4	24
Moll et al., 1999	Sand	9170	7.0	6.4	24
Wang and Summers, 1996	Sand	8350	1.5	1.3	4.9
Wang and Summers, 1996	Sand	8350	2	1.7	6.4
Wang and Summers, 1996	Sand	8350	4	3.3	12.4
Wang and Summers, 1996	Sand	8350	7	5.8	21.8
Wang and Summers, 1996	Sand	8350	10	8.4	31.5
Sang et al., 2003	Bio-ceramic	1340	20	2.7	10.1
Sang et al., 2003	Bio-ceramic	1340	20	2.7	10.1
Sang et al., 2003	Bio-ceramic	1340	20	2.7	10.1

Sang et al., 2003	Bio-ceramic	1340	20	2.7	10.1
Griffini et al., 1999	Biolite	3900	2.3	0.9	3.4
Griffini et al., 1999	Biolite	3900	10	3.9	14.6
Griffini et al., 1999	Biolite	3900	15	5.9	22.1
Griffini et al., 1999	Biolite	3900	20	7.8	29.3
Persson et al., 2006	EC (expanded clay)	3470	31	10.8	40.5
Persson et al., 2006	EC (expanded clay)	1120	31	3.5	13.1
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 3460 Sand 7430	3	1.1*	4.1
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 3460 Sand 7430	6	2.3*	8.6
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 3460 Sand 7430	12	4.6*	17.3
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 3460 Sand 7430	14	5.3*	19.9
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 3460 Sand 7430	3	1.1*	4.1
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 3460 Sand 7430	6	2.3*	8.6
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 3460 Sand 7430	3	1.1*	4.1
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 3460 Sand 7430	6	2.3*	8.6
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 3460 Sand 7430	12	4.6*	17.3
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 3460 Sand 7430	14	5.3*	19.9
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 3460 Sand 7430	3	1.1*	4.1

Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 3460 Sand 7430	6	2.3*	8.6
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 3460 Sand 7430	12	4.6*	17.3
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 3460 Sand 7430	14	5.3*	19.9
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 3460 Sand 7430	3	1.1*	4.1
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 3460 Sand 7430	6	2.3*	8.6
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 3460 Sand 7430	12	4.6*	17.3
Niquette et al., 1998	Dual-GAC (2.0m) Dual-Sand (0.2m)	GAC 3460 Sand 7430	14	5.3*	19.9
Nishijima and Speitel, 2004	Dual-GAC (1/3) Dual-Sand (2/3)	GAC 5640 Sand 2930	45	17.3*	64.9
Melin et al., 2002	Mixed-GAC (0.8m) Mixed-Phonolith (1.1m) Mixed-Calcium carbonate (0.4m)	GAC 3120 Phonolith 8510 Calcium carbonate 2340	28	15.6*	58.5
Melin et al., 2002	Mixed-GAC (0.8m) Mixed-Phonolith (1.1m) Mixed-Calcium carbonate (0.4m)	GAC 3120 Phonolith 8510 Calcium carbonate 2340	28	15.6*	58.5
Melin et al., 2002	Mixed-GAC (0.8m) Mixed-Phonolith (1.1m) Mixed-Calcium carbonate (0.4m)	GAC 3120 Phonolith 8510 Calcium carbonate 2340	28	15.6*	58.5
Melin et al., 2002	Mixed-GAC (0.8m) Mixed-Phonolith (1.1m) Mixed-Calcium carbonate (0.4m)	GAC 3120 Phonolith 8510 Calcium carbonate 2340	28	15.6*	58.5
Melin et al., 2002	Mixed-GAC (0.8m) Mixed-Phonolith (1.1m) Mixed-Calcium carbonate (0.4m)	GAC 3120 Phonolith 8510 Calcium carbonate 2340	28	15.6*	58.5
Melin et al., 2002	Mixed-GAC (0.8m) Mixed-Phonolith (1.1m) Mixed-Calcium carbonate (0.4m)	GAC 3120 Phonolith 8510 Calcium carbonate 2340	28	15.6*	58.5
Melin et al., 2002	Mixed-GAC (0.8m) Mixed-Phonolith (1.1m) Mixed-Calcium carbonate (0.4m)	GAC 3120 Phonolith 8510 Calcium carbonate 2340	28	15.6*	58.5

Thiel, et al., 2006	Dual-Anthracite(1.75m) Dual-Sand(0.65m)	Anthracite 4250 Sand 7200	8	4.0*	15
Thiel, et al., 2006	Dual-GAC(1.75m) Dual-Sand(0.65m)	GAC 3600 Sand 7200	8	3.7*	13.9
Thiel, et al., 2006	Dual-GAC(1.75m) Dual-Sand(0.65m)	GAC 3600 Sand 7200	8	3.7*	13.9
Thiel, et al., 2006	Dual-GAC(1.75m) Dual-Sand(0.65m)	GAC 6680 Sand 7200	8	4.7*	17.6
Thiel, et al., 2006	Dual-GAC(1.75m) Dual-Sand(0.65m)	GAC 6680 Sand 7200	8	4.7*	17.6
Zappia et al., 2007	Dual-Sand(0.3m) Dual- Anthracite(1.75m)	Sand 5970 Anthracite 3260	8	2.9*	10.9
Zappia et al., 2007	Dual-Sand(0.3m) Dual- Anthracite(1.75m)	Sand 5970 Anthracite 3260	16	5.9*	22.1
Zappia et al., 2007	Dual-Sand(0.3m) Dual- Anthracite(0.65m)	Sand 5970 Anthracite 3260	16	3.1*	11.6
Zappia et al., 2007	Dual-Sand(0.3m) Dual- Anthracite(0.65m)	Sand 5970 Anthracite 3260	8	6.2*	23.2
Zappia et al., 2007	Dual-Sand (0.3m) Dual-GAC (1.75m)	Sand 5970 GAC 2990	8	2.7*	10.1
Zappia et al., 2007	Dual-Sand (0.3m) Dual-GAC (1.75m)	Sand 5970 GAC 2990	16	5.5*	20.6
Zappia et al., 2007	Dual-Sand (0.3m) Dual-GAC (1.75m)	Sand 5970 GAC 2990	8	2.7*	10.1
Zappia et al., 2007	Dual-Sand (0.3m) Dual-GAC (1.75m)	Sand 5970 GAC 2990	8	2.7*	10.1
Zappia et al., 2007	Dual-Sand (0.3m) Dual-GAC (1.75m)	Sand 5970 GAC 2990	8	2.7*	10.1
Zappia et al., 2007	Dual-Sand (0.3m) Dual-GAC (1.75m)	Sand 5970 GAC 2990	8	2.7*	10.1
Zappia et al., 2007	Dual-Sand (0.3m) Dual-GAC (1.75m)	Sand 5970 GAC 5000	8	4.3*	16.1
Zappia et al., 2007	Dual-Sand (0.3m) Dual-GAC (1.75m)	Sand 5970 GAC 5000	16	8.6*	32.3
Zappia et al., 2007	Dual-Sand (0.3m) Dual-GAC (1.75m)	Sand 5970 GAC 5000	8	4.3*	16.1

Zappia et al., 2007	Dual-Sand (0.3m) Dual-GAC (1.75m)	Sand 5970 GAC 5000	8	4.3*	16.1
Zappia et al., 2007	Dual-Sand (0.3m) Dual-GAC (1.75m)	Sand 5970 GAC 5000	8	4.3*	16.1

Note: 1. Refer to section 4.4 for initial estimation of $\theta\alpha$ parameter values;

2. Estimated $\theta\alpha$ parameter value was calculated by Equation (4-3):

$$\theta\alpha\ parameter\ value = (\theta\alpha)'\frac{15}{4.0} \qquad (4-3)$$

3. (*): dual-media filters or multi-media,

Calculation of EBCT for each layer is as Equation (3-6):

$$EBCT_i = \frac{h_i}{h} \times EBCT_{Overall} \qquad (3-6)$$

$\theta\alpha$ calculation is as Equation (3-11):

$$(\theta\alpha)'_{overall} = \sum(\theta\alpha)'_{each\ layer} \qquad (3-11)$$

4. Underlined are the calculation bases.

Appendix B-6 Initially $\theta\alpha$ parameter values, expected parameter values, physical EBCTs and BDOC removal percentages

Source	Media Type	Initially Estimated $\theta\alpha$ parameter Values	Expected $\theta\alpha$ Parameter Values[1]	EBCT (min)	BDOC Removal (%)
Griffini et al., 1999	GAC	4.1	12	2.3	58
Griffini et al., 1999	GAC	18.3	16	10	70
Griffini et al., 1999	GAC	27.8	17	15	71
Griffini et al., 1999	GAC	37.	18	20	74
Griffini et al., 1999	GAC	2.6	9	2.3	52
Griffini et al., 1999	GAC	12.4	15	10	68
Griffini et al., 1999	GAC	18.4	17	15	72
Griffini et al., 1999	GAC	24.4	17	20	71
Griffini et al., 1999	GAC	3.4	8	2.3	54
Griffini et al., 1999	GAC	14.6	14	10	66
Griffini et al., 1999	GAC	22.1	15	15	69
Griffini et al., 1999	GAC	29.3	16	20	70
Griffini et al., 1999	GAC	3.4	11	2.3	59
Griffini et al., 1999	GAC	14.6	16	10	70
Griffini et al., 1999	GAC	21.8	18	15	74

Griffini et al., 1999	GAC	28.9	17	20	72
Heinicke et al., 2006	GAC	69.8	3	34	23
Niquette et al., 1998	GAC	5.3	3.5	4	32
Niquette et al., 1998	GAC	9.0	3	7	25
Niquette et al., 1998	GAC	9.0	3	7	21
Niquette et al., 1998	GAC	15.8	4	12	32
Niquette et al., 1998	GAC	15.8	1	12	15
Niquette et al., 1998	GAC	18.0	4	14	38
Niquette et al., 1998	GAC	18.0	3	14	23
Persson et al., 2006	GAC	41.6	4	31	34
Prevost et al., 1992	GAC	10.5	1	5	8
Prevost et al., 1992	GAC	15.8	4	7.5	41
Prevost et al., 1992	GAC	29.6	42	14	95
Servais et al., 1991	GAC	3.0	3	2	25
Servais et al., 1991	GAC	6.4	8	4	50
Servais et al., 1991	GAC	15.8	9	10	53
Servais et al., 1991	GAC	15.8	4	10	39
Servais et al., 1991	GAC	15.8	1	10	14
Servais et al., 1991	GAC	15.8	2	10	22

Servais et al., 1991	GAC	15.8	11	10	60
Servais et al., 1991	GAC	15.8	11	10	60
Servais et al., 1991	GAC	15.8	9	10	50
Servais et al., 1991	GAC	15.8	8	10	42
Servais et al., 1991	GAC	15.8	9	10	50
Servais et al., 1991	GAC	15.8	2	10	24
Servais et al., 1991	GAC	15.8	6	10	45
Servais et al., 1991	GAC	15.8	3	10	29
Servais et al., 1994	GAC	15.8	5	10	43
Servais et al., 1994	GAC	15.8	10	10	56
Servais et al., 1994	GAC	15.8	9	10	51
Servais et al., 1994	GAC	15.8	8	10	49
Servais et al., 1994	GAC	15.8	10	10	56
Wang et al., 2002	GAC	22.9	20	14	78
Fonseca and Summers, 2003	Sand	9.4	39	3	92
Fonseca and Summers, 2003	Sand	31.5	43	10	98
Fonseca and Summers, 2003	Sand	53.3	43	17	100
Fonseca and Summers, 2003	Sand	9.4	39	3	92
Fonseca and Summers, 2003	Sand	31.5	20	10	78

Fonseca and Summers, 2003	Sand	53.3	34	17	89
Fonseca and Summers, 2003	Sand	9.4	17	3	71
Fonseca and Summers, 2003	Sand	31.5	23	10	82
Fonseca and Summers, 2003	Sand	53.3	23	17	82
Fonseca and Summers, 2003	Sand	9.4	13	3	65
Fonseca and Summers, 2003	Sand	31.5	13	10	65
Fonseca and Summers, 2003	Sand	53.3	15	17	68
Moll et al., 1999	Sand	24	4	7.0	38
Moll et al., 1999	Sand	24	11	7.0	60
Moll et al., 1999	Sand	24	11	7.0	60
Wang and Summers, 1996	Sand	4.9	2	1.5	16
Wang and Summers, 1996	Sand	6.4	2	2	22
Wang and Summers, 1996	Sand	12.4	4	4	32
Wang and Summers, 1996	Sand	21.8	4	7	36
Wang and Summers, 1996	Sand	31.5	5	10	40
Sang et al., 2003	Bio-ceramic	10.1	10	20	56
Sang et al., 2003	Bio-ceramic	10.1	10	20	55
Sang et al., 2003	Bio-ceramic	10.1	16	20	70

Sang et al., 2003	Bio-ceramic	10.1	17	20	71
Griffini et al., 1999	Biolite	3.4	7	2.3	47
Griffini et al., 1999	Biolite	14.6	13	10	65
Griffini et al., 1999	Biolite	22.1	16	15	69
Griffini et al., 1999	Biolite	29.3	14	20	66
Persson et al., 2006	EC (expanded clay)	40.5	3	31	30
Persson et al., 2006	EC (expanded clay)	13.1	3	31	28
Niquette et al., 1998	Dual-GAC/sand	4.1	2	3	24
Niquette et al., 1998	Dual-GAC/sand	8.6	2	6	26
Niquette et al., 1998	Dual-GAC/sand	17.3	5	12	42
Niquette et al., 1998	Dual-GAC/sand	19.9	7	14	47
Niquette et al., 1998	Dual-GAC/sand	4.1	1	3	10
Niquette et al., 1998	Dual-GAC/sand	8.6	4	6	37
Niquette et al., 1998	Dual-GAC/sand	4.1	1	3	7
Niquette et al., 1998	Dual-GAC/sand	8.6	2	6	25
Niquette et al., 1998	Dual-GAC/sand	17.3	4	12	35
Niquette et al., 1998	Dual-GAC/sand	19.9	6	14	45
Niquette et al., 1998	Dual-GAC/sand	4.1	2	3	21
Niquette et al., 1998	Dual-GAC/sand	8.6	4	6	35

Niquette et al., 1998	Dual-GAC/sand	17.3	5	12	39
Niquette et al., 1998	Dual-GAC/sand	19.9	8	14	51
Niquette et al., 1998	Dual-GAC/sand	4.1	3	3	27
Niquette et al., 1998	Dual-GAC/sand	8.6	2	6	24
Niquette et al., 1998	Dual-GAC/sand	17.3	7	12	48
Niquette et al., 1998	Dual-GAC/sand	19.9	7	14	48
Nishijima and Speitel, 2004	Dual-GAC/sand	64.9	12	45	63
Melin et al., 2002	Mixed	58.5	4	26-30	36
Melin et al., 2002	Mixed	58.5	15	26-30	68
Melin et al., 2002	Mixed	58.5	12	26-30	61
Melin et al., 2002	Mixed	58.5	13	26-30	66
Melin et al., 2002	Mixed	58.5	5	26-30	42
Melin et al., 2002	Mixed	58.5	11	26-30	59
Melin et al., 2002	Mixed	58.5	9	26-30	52
Thiel, et al., 2006	Dual-Anthracite/Sand	15	2	8	25
Thiel, et al., 2006	Dual-GAC/Sand	13.9	2	8	25
Thiel, et al., 2006	Dual-GAC/Sand	13.9	1	8	12
Thiel, et al., 2006	Dual-GAC/Sand	17.6	1	8	20

Thiel, et al., 2006	Dual-GAC/Sand	17.6	2	8	23
Zappia et al., 2007	Dual-Sand/Anthracite	10.9	0.5	8	5
Zappia et al., 2007	Dual-Sand/Anthracite	22.1	0.5	16	5
Zappia et al., 2007	Dual-Sand/Anthracite	11.6	1	16	14
Zappia et al., 2007	Dual-Sand/Anthracite	23.2	3	8	25
Zappia et al., 2007	Dual-Sand/GAC	10.1	2	8	18
Zappia et al., 2007	Dual-Sand/GAC	20.6	2	16	23
Zappia et al., 2007	Dual-Sand/GAC	10.1	2	8	25
Zappia et al., 2007	Dual-Sand/GAC	10.1	1	8	11
Zappia et al., 2007	Dual-Sand/GAC	10.1	2	8	20
Zappia et al., 2007	Dual-Sand/GAC	10.1	1	8	14
Zappia et al., 2007	Dual-Sand/GAC	16.1	0.5	8	5
Zappia et al., 2007	Dual-Sand/GAC	32.3	2	16	23
Zappia et al., 2007	Dual-Sand/GAC	16.1	3	8	25
Zappia et al., 2007	Dual-Sand/GAC	16.1	2	8	23
Zappia et al., 2007	Dual-Sand/GAC	16.1	1	8	12

Note: 1. Refer to section 4.3, Figure 4-2 and Table 4-4 for determining expected X* values.

Appendix B-7 Representative temperature range and temperature adjustment coefficients

Source	Media Type	Initially Estimated $\theta\alpha$ Parameter Values	Temperature, Season, Location	Representative Temperature Range(7)	Temperature Adjustment Coefficient (8)
Griffini et al., 1999	GAC	4.1	15-28°C	>15°C	1
Griffini et al., 1999	GAC	18.3	15-28°C	>15°C	1
Griffini et al., 1999	GAC	27.8	15-28°C	>15°C	1
Griffini et al., 1999	GAC	37.	15-28°C	>15°C	1
Griffini et al., 1999	GAC	2.6	15-28°C	>15°C	1
Griffini et al., 1999	GAC	12.4	15-28°C	>15°C	1
Griffini et al., 1999	GAC	18.4	15-28°C	>15°C	1
Griffini et al., 1999	GAC	24.4	15-28°C	>15°C	1
Griffini et al., 1999	GAC	3.4	15-28°C	>15°C	1
Griffini et al., 1999	GAC	14.6	15-28°C	>15°C	1
Griffini et al., 1999	GAC	22.1	15-28°C	>15°C	1
Griffini et al., 1999	GAC	29.3	15-28°C	>15°C	1
Griffini et al., 1999	GAC	3.4	15-28°C	>15°C	1
Griffini et al., 1999	GAC	14.6	15-28°C	>15°C	1
Griffini et al., 1999	GAC	21.8	15-28°C	>15°C	1
Griffini et al., 1999	GAC	28.9	15-28°C	>15°C	1

Heinicke et al., 2006	GAC	69.8	Sweden (1)	10-15°C	1.69
Niquette et al., 1998	GAC	5.3	12°C	10-15°C	1.69
Niquette et al., 1998	GAC	9.0	12°C	10-15°C	1.69
Niquette et al., 1998	GAC	9.0	12°C	10-15°C	1.69
Niquette et al., 1998	GAC	15.8	12°C	10-15°C	1.69
Niquette et al., 1998	GAC	15.8	12°C	10-15°C	1.69
Niquette et al., 1998	GAC	18.0	12°C	10-15°C	1.69
Niquette et al., 1998	GAC	18.0	12°C	10-15°C	1.69
Persson et al., 2006	GAC	41.6	Sweden(1) 1.5-20.2°C, median 7.2°C	5-10°C	2.40
Prevost et al., 1992	GAC	10.5	January, Quebec(2)	<5°C	3.40
Prevost et al., 1992	GAC	15.8	January, Quebec(2)	<5°C	3.40
Prevost et al., 1992	GAC	29.6	January, Quebec(2)	<5°C	3.40
Servais et al., 1991	GAC	3.0	March, France(3)	5-10°C	2.40
Servais et al., 1991	GAC	6.4	March, France(3)	5-10°C	2.40
Servais et al., 1991	GAC	15.8	March, France(3)	5-10°C	2.40
Servais et al., 1991	GAC	15.8	August, France(3)	>15°C	1
Servais et al., 1991	GAC	15.8	May, France(3)	>15°C	1
Servais et al., 1991	GAC	15.8	October, France(3)	10-15°C	1.69
Servais et al., 1991	GAC	15.8	October, France(3)	10-15°C	1.69

Servais et al., 1991	GAC	15.8	October, France(3)	10-15°C	1.69
Servais et al., 1991	GAC	15.8	March, France(3)	5-10°C	2.40
Servais et al., 1991	GAC	15.8	March, France(3)	5-10°C	2.40
Servais et al., 1991	GAC	15.8	March, France(3)	5-10°C	2.40
Servais et al., 1991	GAC	15.8	June, France(3)	>15°C	1
Servais et al., 1991	GAC	15.8	August, France(3)	>15°C	1
Servais et al., 1991	GAC	15.8	August, France(3)	>15°C	1
Servais et al., 1994	GAC	15.8	April, France(3)	10-15°C	1.69
Servais et al., 1994	GAC	15.8	November, France(3)	5-10°C	2.40
Servais et al., 1994	GAC	15.8	November, France(3)	5-10°C	2.40
Servais et al., 1994	GAC	15.8	November, France(3)	5-10°C	2.40
Servais et al., 1994	GAC	15.8	November, France(3)	5-10°C	2.40
Wang et al., 2002	GAC	22.9	May-October, Northern China(4)	>15°C	1
Fonseca and Summers, 2003	Sand	9.4	>12°C	>15°C	1
Fonseca and Summers, 2003	Sand	31.5	>12°C	>15°C	1
Fonseca and Summers, 2003	Sand	53.3	>12°C	>15°C	1
Fonseca and Summers, 2003	Sand	9.4	>12°C	>15°C	1
Fonseca and Summers, 2003	Sand	31.5	>12°C	>15°C	1

Fonseca and Summers, 2003	Sand	53.3	>12°C	>15°C	1
Fonseca and Summers, 2003	Sand	9.4	>12°C	>15°C	1
Fonseca and Summers, 2003	Sand	31.5	>12°C	>15°C	1
Fonseca and Summers, 2003	Sand	53.3	>12°C	>15°C	1
Fonseca and Summers, 2003	Sand	9.4	3°C	<5°C	3.40
Fonseca and Summers, 2003	Sand	31.5	3°C	<5°C	3.40
Fonseca and Summers, 2003	Sand	53.3	3°C	<5°C	3.40
Moll et al., 1999	Sand	24	5°C	5-10°C	2.40
Moll et al., 1999	Sand	24	20°C	>15°C	1
Moll et al., 1999	Sand	24	35°C	>15°C	1
Wang and Summers, 1996	Sand	4.9	Ohio, USA(5)*	>15°C	1
Wang and Summers, 1996	Sand	6.4	Ohio, USA(5)*	>15°C	1
Wang and Summers, 1996	Sand	12.4	Ohio, USA(5)*	>15°C	1
Wang and Summers, 1996	Sand	21.8	Ohio, USA(5)*	>15°C	1
Wang and Summers, 1996	Sand	31.5	Ohio, USA(5)*	>15°C	1
Sang et al., 2003	Bio-ceramic	10.1	10-23°C	>15°C	1
Sang et al., 2003	Bio-ceramic	10.1	10-23°C	>15°C	1
Sang et al., 2003	Bio-ceramic	10.1	10-23°C	>15°C	1

Sang et al., 2003	Bio-ceramic	10.1	10-23°C	>15°C	1
Griffini et al., 1999	Biolite	3.4	7-14°C	10-15°C	1.69
Griffini et al., 1999	Biolite	14.6	7-14°C	10-15°C	1.69
Griffini et al., 1999	Biolite	22.1	7-14°C	10-15°C	1.69
Griffini et al., 1999	Biolite	29.3	7-14°C	10-15°C	1.69
Persson et al., 2006	EC (expanded clay)	40.5	Sweden(1) 1.5-20.2°C, median 7.2°C	5-10°C	2.40
Persson et al., 2006	EC (expanded clay)	13.1	Sweden(1) 1.5-20.2°C, median 7.2°C	5-10°C	2.40
Niquette et al., 1998	Dual-GAC/sand	4.1	14°C	10-15°C	1.69
Niquette et al., 1998	Dual-GAC/sand	8.6	14°C	10-15°C	1.69
Niquette et al., 1998	Dual-GAC/sand	17.3	0.5°C	<5°C	3.40
Niquette et al., 1998	Dual-GAC/sand	19.9	0.5°C	<5°C	3.40
Niquette et al., 1998	Dual-GAC/sand	4.1	0.5°C	<5°C	3.40
Niquette et al., 1998	Dual-GAC/sand	8.6	0.5°C	<5°C	3.40
Niquette et al., 1998	Dual-GAC/sand	4.1	0.5°C	<5°C	3.40
Niquette et al., 1998	Dual-GAC/sand	8.6	0.5°C	<5°C	3.40
Niquette et al., 1998	Dual-GAC/sand	17.3	0.5°C	<5°C	3.40
Niquette et al., 1998	Dual-GAC/sand	19.9	0.5°C	<5°C	3.40
Niquette et al., 1998	Dual-GAC/sand	4.1	12°C	10-15°C	1.69

Niquette et al., 1998	Dual-GAC/sand	8.6	12°C	10-15°C	1.69
Niquette et al., 1998	Dual-GAC/sand	17.3	12°C	10-15°C	1.69
Niquette et al., 1998	Dual-GAC/sand	19.9	12°C	10-15°C	1.69
Niquette et al., 1998	Dual-GAC/sand	4.1	12°C	10-15°C	1.69
Niquette et al., 1998	Dual-GAC/sand	8.6	12°C	10-15°C	1.69
Niquette et al., 1998	Dual-GAC/sand	17.3	12°C	10-15°C	1.69
Niquette et al., 1998	Dual-GAC/sand	19.9	12°C	10-15°C	1.69
Nishijima and Speitel, 2004	Dual-GAC/sand	64.9	25°C	>15°C	1
Melin et al., 2002	Mixed	58.5	10.0°C	10-15°C	1.69
Melin et al., 2002	Mixed	58.5	8.5°C	5-10°C	2.40
Melin et al., 2002	Mixed	58.5	7.7°C	5-10°C	2.40
Melin et al., 2002	Mixed	58.5	2.8°C	<5°C	3.40
Melin et al., 2002	Mixed	58.5	3.3°C	<5°C	3.40
Melin et al., 2002	Mixed	58.5	3.9°C	<5°C	3.40
Melin et al., 2002	Mixed	58.5	6.5°C	5-10°C	2.40
Thiel, et al., 2006	Dual-Anthracite/Sand	15	Groundwater Australia(6)	<5°C	3.40
Thiel, et al., 2006	Dual-GAC/Sand	13.9	Groundwater Australia(6)	<5°C	3.40
Thiel, et al., 2006	Dual-GAC/Sand	13.9	Groundwater Australia(6)	<5°C	3.40

Thiel, et al., 2006	Dual-GAC/Sand	17.6	Groundwater Australia(6)	<5°C	3.40
Thiel, et al., 2006	Dual-GAC/Sand	17.6	Groundwater Australia(6)	<5°C	3.40
Zappia et al., 2007	Dual-Sand/Anthracite	10.9	Groundwater March, Australia(6)	<5°C	3.40
Zappia et al., 2007	Dual-Sand/Anthracite	22.1	Groundwater March, Australia(6)	<5°C	3.40
Zappia et al., 2007	Dual-Sand/Anthracite	11.6	Groundwater March, Australia(6)	<5°C	3.40
Zappia et al., 2007	Dual-Sand/Anthracite	23.2	Groundwater March, Australia(6)	<5°C	3.40
Zappia et al., 2007	Dual-Sand/GAC	10.1	Groundwater March, Australia(6)	<5°C	3.40
Zappia et al., 2007	Dual-Sand/GAC	20.6	Groundwater March, Australia(6)	<5°C	3.40
Zappia et al., 2007	Dual-Sand/GAC	10.1	Groundwater March, Australia(6)	<5°C	3.40
Zappia et al., 2007	Dual-Sand/GAC	10.1	Groundwater March, Australia(6)	<5°C	3.40
Zappia et al., 2007	Dual-Sand/GAC	10.1	Groundwater March, Australia(6)	<5°C	3.40
Zappia et al., 2007	Dual-Sand/GAC	10.1	Groundwater March, Australia(6)	<5°C	3.40
Zappia et al., 2007	Dual-Sand/GAC	16.1	Groundwater March, Australia(6)	<5°C	3.40
Zappia et al., 2007	Dual-Sand/GAC	32.3	Groundwater March, Australia(6)	<5°C	3.40
Zappia et al., 2007	Dual-Sand/GAC	16.1	Groundwater March, Australia(6)	<5°C	3.40
Zappia et al., 2007	Dual-Sand/GAC	16.1	Groundwater March, Australia(6)	<5°C	3.40
Zappia et al., 2007	Dual-Sand/GAC	16.1	Groundwater March, Australia(6)	<5°C	3.40

Note: 1. http://en.wikipedia.org/wiki/Sweden (2009-08-10);
2. http://en.wikipedia.org/wiki/Geography_of_Quebec (2009-08-10);
3. http://en.wikipedia.org/wiki/Paris (2009-08-10);
4. http://en.wikipedia.org/wiki/Beijing (2009-08-10);
5. http://en.wikipedia.org/wiki/Ohio (2009-08-10);
6. http://en.wikipedia.org/wiki/Climate_of_Australia (2009-08-10);
7. Refer to section 3.6.5 for determination approaches of representative temperature ranges;
8. Refer to Table 4-8 for determination of temperature adjustment coefficients.

Appendix B-8 Temperature-adjusted $\theta\alpha$ parameter values

Source	Media Type	Initially Estimated $\theta\alpha$ Parameter Values	Temperature-adjusted $\theta\alpha$ Parameter Values	EBCT (min)	BDOC Removal (%)
Griffini et al., 1999	GAC	4.1	4.1	2.3	58
Griffini et al., 1999	GAC	18.3	18.3	10	70
Griffini et al., 1999	GAC	27.8	27.8	15	71
Griffini et al., 1999	GAC	37.0	37.0	20	74
Griffini et al., 1999	GAC	2.6	2.6	2.3	52
Griffini et al., 1999	GAC	12.4	12.4	10	68
Griffini et al., 1999	GAC	18.4	18.4	15	72
Griffini et al., 1999	GAC	24.4	24.4	20	71
Griffini et al., 1999	GAC	3.4	3.4	2.3	54
Griffini et al., 1999	GAC	14.6	14.6	10	66
Griffini et al., 1999	GAC	22.1	22.1	15	69
Griffini et al., 1999	GAC	29.3	29.3	20	70
Griffini et al., 1999	GAC	3.4	3.4	2.3	59
Griffini et al., 1999	GAC	14.6	14.6	10	70
Griffini et al., 1999	GAC	21.8	21.8	15	74
Griffini et al., 1999	GAC	28.9	28.9	20	72

Heinicke et al., 2006	GAC	69.8	41.3	34	23
Niquette et al., 1998	GAC	5.3	3.1	4	32
Niquette et al., 1998	GAC	9.0	5.3	7	25
Niquette et al., 1998	GAC	9.0	5.3	7	21
Niquette et al., 1998	GAC	15.8	9.4	12	32
Niquette et al., 1998	GAC	15.8	9.4	12	15
Niquette et al., 1998	GAC	18.0	10.7	14	38
Niquette et al., 1998	GAC	18.0	10.7	14	23
Persson et al., 2006	GAC	41.6	14.3	31	34
Prevost et al., 1992	GAC	10.5	3.1	5	8
Prevost et al., 1992	GAC	15.8	4.6	7.5	41
Prevost et al., 1992	GAC	29.6	8.7	14	95
Servais et al., 1991	GAC	3.0	1.3	2	25
Servais et al., 1991	GAC	6.4	2.7	4	50
Servais et al., 1991	GAC	15.8	6.6	10	53
Servais et al., 1991	GAC	15.8	15.8	10	39
Servais et al., 1991	GAC	15.8	15.8	10	14
Servais et al., 1991	GAC	15.8	9.3	10	22
Servais et al., 1991	GAC	15.8	9.3	10	60

Servais et al., 1991	GAC	15.8	9.3	10	60
Servais et al., 1991	GAC	15.8	6.6	10	50
Servais et al., 1991	GAC	15.8	6.6	10	42
Servais et al., 1991	GAC	15.8	6.6	10	50
Servais et al., 1991	GAC	15.8	15.8	10	24
Servais et al., 1991	GAC	15.8	15.8	10	45
Servais et al., 1991	GAC	15.8	15.8	10	29
Servais et al., 1994	GAC	15.8	9.3	10	43
Servais et al., 1994	GAC	15.8	6.6	10	56
Servais et al., 1994	GAC	15.8	6.6	10	51
Servais et al., 1994	GAC	15.8	6.6	10	49
Servais et al., 1994	GAC	15.8	6.6	10	56
Wang et al., 2000	GAC	22.9	22.9	14	78
Fonseca and Summers, 2003	Sand	9.4	9.4	3	92
Fonseca and Summers, 2003	Sand	31.5	31.5	10	98
Fonseca and Summers, 2003	Sand	53.3	53.3	17	100
Fonseca and Summers, 2003	Sand	9.4	9.4	3	92
Fonseca and Summers, 2003	Sand	31.5	31.5	10	78

Fonseca and Summers, 2003	Sand	53.3	53.3	17	89
Fonseca and Summers, 2003	Sand	9.4	9.4	3	71
Fonseca and Summers, 2003	Sand	31.5	31.5	10	82
Fonseca and Summers, 2003	Sand	53.3	53.3	17	82
Fonseca and Summers, 2003	Sand	9.4	2.8	3	65
Fonseca and Summers, 2003	Sand	31.5	9.3	10	65
Fonseca and Summers, 2003	Sand	53.3	15.7	17	68
Moll et al., 1999	Sand	24	10	7.0	38
Moll et al., 1999	Sand	24	24	7.0	60
Moll et al., 1999	Sand	24	24	7.0	60
Wang and Summers, 1996	Sand	4.9	4.9	1.5	16
Wang and Summers, 1996	Sand	6.4	6.4	2	22
Wang and Summers, 1996	Sand	12.4	12.4	4	32
Wang and Summers, 1996	Sand	21.8	21.8	7	36
Wang and Summers, 1996	Sand	31.5	31.5	10	40
Sang et al., 2003	Bio-ceramic	10.1	10.1	20	56
Sang et al., 2003	Bio-ceramic	10.1	10.1	20	55
Sang et al., 2003	Bio-ceramic	10.1	10.1	20	70

Sang et al., 2003	Bio-ceramic	10.1	10.1	20	71
Griffini et al., 1999	Biolite	3.4	2.0	2.3	47
Griffini et al., 1999	Biolite	14.6	8.6	10	65
Griffini et al., 1999	Biolite	22.1	13.1	15	69
Griffini et al., 1999	Biolite	29.3	17.3	20	66
Persson et al., 2006	EC (expanded clay)	40.5	16.9	31	30
Persson et al., 2006	EC (expanded clay)	13.1	5.5	31	28
Niquette et al., 1998	Dual-GAC/sand	4.1	2.4	3	24
Niquette et al., 1998	Dual-GAC/sand	8.6	5.1	6	26
Niquette et al., 1998	Dual-GAC/sand	17.3	5.1	12	42
Niquette et al., 1998	Dual-GAC/sand	19.9	5.9	14	47
Niquette et al., 1998	Dual-GAC/sand	4.1	1.2	3	10
Niquette et al., 1998	Dual-GAC/sand	8.6	2.5	6	37
Niquette et al., 1998	Dual-GAC/sand	4.1	1.2	3	7
Niquette et al., 1998	Dual-GAC/sand	8.6	2.5	6	25
Niquette et al., 1998	Dual-GAC/sand	17.3	5.1	12	35
Niquette et al., 1998	Dual-GAC/sand	19.9	5.9	14	45
Niquette et al., 1998	Dual-GAC/sand	4.1	2.4	3	21

Niquette et al., 1998	Dual-GAC/sand	8.6	5.1	6	35
Niquette et al., 1998	Dual-GAC/sand	17.3	10.2	12	39
Niquette et al., 1998	Dual-GAC/sand	19.9	11.8	14	51
Niquette et al., 1998	Dual-GAC/sand	4.1	2.4	3	27
Niquette et al., 1998	Dual-GAC/sand	8.6	5.1	6	24
Niquette et al., 1998	Dual-GAC/sand	17.3	10.2	12	48
Niquette et al., 1998	Dual-GAC/sand	19.9	11.8	14	48
Nishijima and Speitel, 2004	Dual-GAC/sand	64.9	64.9	45	63
Melin et al., 2002	Mixed	58.5	34.6	26-30	36
Melin et al., 2002	Mixed	58.5	24.4	26-30	68
Melin et al., 2002	Mixed	58.5	24.4	26-30	61
Melin et al., 2002	Mixed	58.5	17.2	26-30	66
Melin et al., 2002	Mixed	58.5	17.2	26-30	42
Melin et al., 2002	Mixed	58.5	17.2	26-30	59
Melin et al., 2002	Mixed	58.5	24.4	26-30	52
Thiel, et al., 2006	Dual-Anthracite/Sand	15	4.4	8	25
Thiel, et al., 2006	Dual-GAC/Sand	13.9	4.1	8	25
Thiel, et al., 2006	Dual-GAC/Sand	13.9	4.1	8	12

Thiel, et al., 2006	Dual-GAC/Sand	17.6	5.2	8	20
Thiel, et al., 2006	Dual-GAC/Sand	17.6	5.2	8	23
Zappia et al., 2007	Dual-Sand/Anthracite	10.9	3.2	8	5
Zappia et al., 2007	Dual-Sand/Anthracite	22.1	6.5	16	5
Zappia et al., 2007	Dual-Sand/Anthracite	11.6	3.4	16	14
Zappia et al., 2007	Dual-Sand/Anthracite	23.2	6.8	8	25
Zappia et al., 2007	Dual-Sand/GAC	10.1	3.0	8	18
Zappia et al., 2007	Dual-Sand/GAC	20.6	6.1	16	23
Zappia et al., 2007	Dual-Sand/GAC	10.1	3.0	8	25
Zappia et al., 2007	Dual-Sand/GAC	10.1	3.0	8	11
Zappia et al., 2007	Dual-Sand/GAC	10.1	3.0	8	20
Zappia et al., 2007	Dual-Sand/GAC	10.1	3.0	8	14
Zappia et al., 2007	Dual-Sand/GAC	16.1	4.7	8	5
Zappia et al., 2007	Dual-Sand/GAC	32.3	9.5	16	23
Zappia et al., 2007	Dual-Sand/GAC	16.1	4.7	8	25
Zappia et al., 2007	Dual-Sand/GAC	16.1	4.7	8	23
Zappia et al., 2007	Dual-Sand/GAC	16.1	4.7	8	12

Note: Refer to 4.5.4 for determining temperature-adjustment $\theta\alpha$ parameter values.

Appendix B-9 Ozonation dosages and BDOC removal efficiencies

Source	EBCT (min)	Temperature-Adjusted $\theta\alpha$ Parameter Values	Ozone Dosage (mgO_3/mg TOC)	BDOC Removal (%)
Heinicke et al., 2006	34	41.3	0	23
Persson et al., 2006	31	16.9	0	30
Persson et al., 2006	31	5.5	0	28
Fonseca and Summers, 2003	3	9.4	1.30	71
Fonseca and Summers, 2003	10	31.5	1.30	82
Fonseca and Summers, 2003	17	53.3	1.30	82
Fonseca and Summers, 2003	3	2.8	1.30	65
Fonseca and Summers, 2003	10	9.3	1.30	65
Fonseca and Summers, 2003	17	15.7	1.30	68
Griffini et al., 1999	2.3	4.1	1.15	58
Griffini et al., 1999	10	18.3	1.15	70
Griffini et al., 1999	15	27.8	1.15	71
Griffini et al., 1999	20	37.0	1.15	74
Griffini et al., 1999	2.3	2.6	1.15	52
Griffini et al., 1999	10	12.4	1.15	68
Griffini et al., 1999	15	18.4	1.15	72
Griffini et al., 1999	20	24.4	1.15	71
Griffini et al., 1999	2.3	3.4	1.15	54
Griffini et al., 1999	10	14.6	1.15	66
Griffini et al., 1999	15	22.1	1.15	69
Griffini et al., 1999	20	29.3	1.15	70
Griffini et al., 1999	2.3	3.4	1.15	59
Griffini et al., 1999	10	14.6	1.15	70

Griffini et al., 1999	15	21.8	1.15	74
Griffini et al., 1999	20	28.9	1.15	72
Melin et al., 2002	28	34.6	1.50	36
Melin et al., 2002	28	24.4	1.50	68
Melin et al., 2002	28	24.4	1.50	61
Melin et al., 2002	28	17.2	1.50	66
Melin et al., 2002	28	17.2	1.50	42
Melin et al., 2002	28	17.2	1.50	59
Melin et al., 2002	28	24.4	1.50	52
Moll et al., 1999	7.0	10	1.30	38
Moll et al., 1999	7.0	24	1.30	60
Moll et al., 1999	7.0	24	1.30	60
Wang et al., 2000	14	22.9	0.14	78
Wang and Summers, 1996	1.5	4.9	0.35	16
Wang and Summers, 1996	2	6.4	0.35	22
Wang and Summers, 1996	4	12.4	0.35	32
Wang and Summers, 1996	7	21.8	0.35	36
Wang and Summers, 1996	10	31.5	0.35	40

Note: 1. Refer to 3.7 for calculation of normalized ozone dosages and related discussions;
2. First three rows are associated with biofiltration processes without pre-ozonation.

Appendix C-1 Reference for collected MIB and geosmin removal cases

Elhadi S.L.N., Huck P.M. and Slawson R.M. 2004. Removal of geosmin and 2-methylisoborneol by biological filtration. *Water Science and Technology*, 49, 9: 273-280

Elhadi S.L.N., Huck P.M. and Slawson R.M. 2006. Factors Affecting the Removal of Geosmin and MIB in Drinking Water Biofilters. *Journal of American Water Works Association*, 98, 8: 108-119

McDowall B., Ho L., Saint C.P. and Newcombe G. 2007a. Removal of geosmin and 2-methylisoborneol through biologically active sand filters. *International Journal of Environment and Waste Management*, 1, 4: 311-320

McDowall B., Ho L., Saint C.P. and Newcombe G. 2007b. Biological removal of MIB and geosmin through rapid gravity filters. *Water: Journal of the Australian Water Association*, 34, 7: 48-54

McDowall B., Hoefel D., Newcombe G., Saint C.P. and Ho L. 2009. Enhancing the biofiltration of geosmin by seeding sand filter columns with a consortium of geosmin-degrading bacteria. *Water Research*, 43: 433-440

Persson F., Heinicke G., Hedberg T., Hermansson M. and Uhl W. 2007. Removal of geosmin and MIB by biofiltration - An investigation discriminating between adsorption and biodegradation. *Environmental Technology*, 28, 1: 95-104

Appendix C-2 Filter media particle sizes for MIB and geosmin removal cases

Source	Media type	Average size d(mm)	Effective Size d_{10}(mm)	Uniformity Coefficient (UC)
Elhadi et al., 2004	Dual-GAC(1) Dual-Sand		GAC 0.72 Sand 0.50	GAC 1.8(2) Sand 1.5(3)
Elhadi, et al., 2006	Dual-GAC Dual-Sand		GAC 0.72 Sand 0.50	GAC 1.8(2) Sand 1.5(3)
Elhadi, et al., 2006	Dual-Anthracite Dual-Sand		Anthracite 1.10 Sand 0.50	Anthracite 1.5(3) Sand 1.5(3)
McDowall, et al., 2007a	Sand		0.55	1.5(3)
McDowall, et al., 2007a	Sand		2.00	1.5(3)
McDowall, et al., 2007b	Sand		0.55	1.5(3)
McDowall, et al., 2009	Sand		0.60	1.3(4)
McDowall, et al., 2009	Sand		0.55	1.3(5)
Persson et al., 2007	GAC		0.80-1.00	<2.10
Persson et al., 2007	EC		0.85-0.95	1.5

Note: Refer to section 3.1.1 for collecting and determining representative sizes of filter media particles;

(1) For dual-media and multi-media filter, whichever is listed in upper place of a cell in any Appendix table is the toper layer in the corresponding biofilter, whichever is listed in bottom place of a cell is the bottom layer in the biofilter;

(2) P-830, *PICA USA Inc.*, Columbus, OH. Data from:
Buchanan W., Roddick F. and Porter N. 2008. Removal of VUV pre-treated natural organic matter by biologically activated carbon columns. *Water Research*, 42: 3335-3342;

(3) Assuming UC=1.5;

(4) http://www.riversands.com.au/divisions_waterfiltration.php#filter_sand (2009-10-06);

(5) Assuming UC is the same as for (4).

Appendix C-3 Actively specific surface area of filter media particles

Source	Media Type	Representative Size(mm)	Speicific Surface Area for Ideal Spherical Shape (m^{-1})	Actively Specific Surface Area (m^{-1})$^{(1)}$
Elhadi et al., 2004	Dual-GAC (50 cm) Dual-Sand (25cm)	GAC 1.00 Sand 0.63	GAC 6000 Sand 9520	GAC 4680 Sand 7430
Elhadi et al., 2006	Dual-GAC (50 cm) Dual-Sand (25cm)	GAC 1.00 Sand 0.63	GAC 6000 Sand 9520	GAC 4680 Sand 7430
Elhadi, et al., 2006	Dual-Anthracite (50 cm) Dual-Sand (25 cm)	Anthracite 1.38 Sand 0.63	Anthracite 4350 Sand 9520	Anthracite 3390 Sand 7430
McDowall, et al., 2007a	Sand	0.69	8700	6790
McDowall, et al., 2007a	Sand	2.50	2400	1870
McDowall, et al., 2007b	Sand	0.69	8700	6790
McDowall, et al., 2009	Sand	0.69	8700	6790
McDowall, et al., 2009	Sand	0.63	9520	7430
Persson et al., 2007	GAC	1.40	4290	3350
Persson et al., 2007	Sand	1.13	5310	4140

Note: 1. Refer to section 3.1.1 for collection and determination of representative sizes of filter media particles;
2. Refer to section 3.2 for calculation of actively specific surface area of filter media;
3. Actively specific surface area of filter media was calculated by Equation (3-9):

$$\alpha = \frac{6 \times \beta \times \varphi \times (1 - \epsilon)}{\bar{d}} \qquad (3-9)$$

Appendix C-4-1 Collection and calculation results of EBCTs, percentages removal, EBCT-normalized percentage removals, actively specific surface area of filter media particles and $\theta\alpha$ product values for MIB removal

Source	Media Type	EBCT(1) (min)	MIB Removal (%)(2)	EBCT-Normalized Removal (min^{-1}) (5)	α (m^{-1})	α × EBCT ($\times 10^4$, $m^{-1} \times min$)
Elhadi et al., 2004	Dual-GAC/Sand	5.6(3)	50(4)	8.9	GAC 4680 Sand 7430	3.1(6)
Elhadi et al., 2006	Dual-GAC/Sand	5.6(3)	41(4)	7.3	GAC 4680 Sand 7430	3.1(6)
Elhadi et al., 2006	Dual-Anthracite/Sand	5.6(3)	29(4)	5.2	Anthracite 3390 Sand 7430	2.6(6)
Elhadi et al., 2006	Dual-GAC/Sand	5.6(3)	19(3, 4)	3.4	GAC 4680 Sand 7430	3.1(6)
Elhadi et al., 2006	Dual-Anthracite/Sand	5.6(3)	10(3, 4)	1.8	Anthracite 3390 Sand 7430	2.6(6)
Elhadi et al., 2006	Dual-GAC/Sand	5.6(3)	36(3, 4)	6.4	GAC 4680 Sand 7430	3.1(6)
Elhadi et al., 2006	Dual-Anthracite/Sand	5.6(3)	9(3, 4)	1.6	Anthracite 3390 Sand 7430	2.6(6)
Elhadi et al., 2006	Dual-GAC/Sand	5.6(3)	14(3, 4)	2.5	GAC 4680 Sand 7430	3.1(6)
Elhadi et al., 2006	Dual-Anthracite/Sand	5.6(3)	1(3, 4)	0.6	Anthracite 3390 Sand 7430	2.6(6)
McDowall, et al., 2007a	Sand	7.5(3)	95(4)	12.7	6790	5.1
McDowall, et al., 2007a	Sand	15(3)	95(4)	6.3	6790	10.2
McDowall, et al., 2007a	Sand	7.5(3)	12(4)	1.6	1870	1.4
McDowall, et al., 2007a	Sand	15(3)	15(4)	1.0	1870	2.8
McDowall, et al., 2007b	Sand	3.5(3)	95(3, 4)	27.1	6790	2.4

McDowall, et al., 2007b	Sand	5(3)	95(3, 4)	19.0	6790	3.4
McDowall, et al., 2007b	Sand	10(3)	95(3, 4)	9.5	6790	6.8
McDowall, et al., 2007b	Sand	15(3)	95(3, 4)	6.3	6790	10.2
Persson et al., 2007	GAC	6(3)	50(4)	8.3	3350	2.0
Persson et al., 2007	GAC	15(3)	88(4)	5.9	3350	5.0
Persson et al., 2007	GAC	30(3)	96(4)	3.2	3350	10.0
Persson et al., 2007	EC	6(3)	37(4)	6.2	4140	2.5
Persson et al., 2007	EC	15(3)	82(4)	5.5	4140	6.2
Persson et al., 2007	EC	30(3)	97(4)	3.2	4140	12.4

Note: 1. Refer to section 3.1.2 for collecting and determining EBCTs;
2. Refer to section 3.1.3 for determining removal percentages;
3. Data provided in the literature;
4. Measured and estimated from figures in the literature;
5. EBCT-normalized percentage removal calculation is as below:

$$EBCT\ Normalized\ Percentage\ Removal\ = \frac{MIB\ Removal\ Percentage}{EBCT} \qquad (C-1)$$

6. Dual-media filters or multi-media,

Calculation of EBCT for each layer is as Equation (3-6):

$$EBCT_i = \frac{h_i}{h} \times EBCT_{Overall} \qquad (3-6)$$

$\theta\alpha$ calculation is as Equation (3-11):

$$(\theta\alpha)'_{overall} = \sum(\theta\alpha)'_{each\ layer} \qquad (3-11)$$

Appendix C-4-2 Collection and calculation results of EBCTs, percentages removal, EBCT-normalized percentage removals, actively specific surface area of filter media particles and $\theta\alpha$ product values for geosmin removal

Source	Media Type	EBCT(1) (min)	Geosmin Removal (%)(2)	EBCT-Normalized Percentage Removal (min^{-1}) (5)	α (m^{-1})	α × EBCT (×10^4, m^{-1}×min)
Elhadi et al., 2004	Dual-GAC/Sand	5.6(3)	82(4)	14.6	GAC 4680 Sand 7430	3.1(6)
Elhadi et al., 2006	Dual-GAC/Sand	5.6(3)	60(3,4)	10.7	GAC 4680 Sand 7430	3.1(6)
Elhadi et al., 2006	Dual-Anthracite/Sand	5.6(3)	42(3,4)	7.5	Anthracite 3390 Sand 7430	2.6(6)
Elhadi et al., 2006	Dual-GAC/Sand	5.6(3)	23(3,4)	4.1	GAC 4680 Sand 7430	3.1(6)
Elhadi et al., 2006	Dual-Anthracite/Sand	5.6(3)	58(3,4)	10.4	Anthracite 3390 Sand 7430	2.6(6)
Elhadi et al., 2006	Dual-GAC/Sand	5.6(3)	14(3,4)	2.5	GAC 4680 Sand 7430	3.1(6)
Elhadi et al., 2006	Dual-Anthracite/Sand	5.6(3)	17(3,4)	3.0	Anthracite 3390 Sand 7430	2.6(6)
Elhadi et al., 2006	Dual-GAC/Sand	5.6(3)	4(3,4)	0.7	GAC 4680 Sand 7430	3.1(6)
Elhadi et al., 2006	Dual-Anthracite/Sand	5.6(3)	20(3,4)	3.6	Anthracite 3390 Sand 7430	2.6(6)
McDowall, et al., 2007a	Sand	7.5(3)	95(4)	12.7	6790	5.1
McDowall, et al., 2007a	Sand	15(3)	95(4)	6.3	6790	10.2
McDowall, et al., 2007a	Sand	7.5(3)	36(4)	4.8	1870	1.4
McDowall, et al., 2007a	Sand	15(3)	45(4)	3.0	1870	2.8
McDowall, et al., 2007b	Sand	3.5	95(3, 4)	27.1	6790	2.4

McDowall, et al., 2007b	Sand	5	95(3, 4)	19.0	6790	3.4
McDowall, et al., 2007b	Sand	10	95(3, 4)	9.5	6790	6.8
McDowall, et al., 2007b	Sand	15	95(3, 4)	6.3	6790	10.2
McDowall, et al., 2009	Sand	15(3)	40(4)	2.7	6790	10.1
McDowall, et al., 2009	Sand	15(3)	70(4)	4.7	7430	11.1
McDowall, et al., 2009	Sand	15(3)	75(4)	5.0	6790	10.1
Persson et al., 2007	GAC	6(3)	68(4)	11.3	3350	2.0
Persson et al., 2007	GAC	15(3)	96(4)	6.4	3350	5.0
Persson et al., 2007	GAC	30(3)	96(4)	3.2	3350	10.0
Persson et al., 2007	EC	6(3)	45(4)	7.5	4140	2.5
Persson et al., 2007	EC	15(3)	91(4)	6.1	4140	6.2
Persson et al., 2007	EC	30(3)	97(4)	3.2	4140	12.4

Note: 1. Refer to section 3.1.2 for collecting and determining EBCTs;
2. Refer to section 3.1.3 for determining removal percentages;
3. Data provided in the literature;
4. Measured and estimated from figures in the literature;
5. EBCT-normalized percentage removal calculation is as Equation

$$EBCT\ Normalized\ Percentage\ Removal = \frac{Geosmin\ Removal\ Percentage}{EBCT} \qquad (C-2)$$

6. Dual-media filters or multi-media,
Calculation of EBCT for each layer is as Equation (3-6):

$$EBCT_i = \frac{h_i}{h} \times EBCT_{Overall} \qquad (3-6)$$

$\theta\alpha$ calculation is as Equation (3-11):

$$(\theta\alpha)'_{overall} = \sum(\theta\alpha)'_{each\ layer} \qquad (3-11)$$