

Biological filtration for membrane pre-treatment and other applications: towards the development of a practically-oriented performance parameter

P. M. Huck and M. M. Sozański

ABSTRACT

Biological processes have a long history of use in drinking water treatment. In recent decades, rapid biological filtration has achieved increasing acceptance. In terms of organic carbon, the classical application of biofiltration has been for the removal of easily biodegradable material, along with the related objectives of reduction in disinfectant demand and byproduct formation. Additional objectives receiving emphasis more recently include the removal of trace contaminants and the use of biofiltration as a membrane pre-treatment to reduce fouling. Although various models for biofiltration have been developed, these are generally complicated and research-oriented, and no modeling approach or performance parameter has found general acceptance in practice.

The paper summarizes previous modeling approaches and illustrates the usefulness of a previously developed index of contact time, X^* , referred to as dimensionless contact time. In addition to empty bed contact time, X^* contains other factors which are important for biofiltration performance: the reactor surface area available for biofilm development and substrate biodegradability and diffusivity. This paper shows how X^* can be simplified and also illustrates its application to the removal of trace contaminants (e.g. pharmaceuticals, endocrine disrupting substances or odorous compounds such as geosmin). A major section of the paper analyzes the requirements for biofiltration when used as a membrane pre-treatment to reduce fouling, and illustrates the applicability of X^* for this use of biofiltration. Thus X^* is presented as an overall unifying parameter. The last section of the paper introduces a new practically-oriented parameter, the Biofiltration Factor (BF). The use of this parameter, which is directly related to X^* , is proposed for biofiltration design and operation, once some additional work has been undertaken.

Key words | biofiltration, dissolved bacterial product material, drinking water, geosmin and MIB, membrane fouling, trace contaminants

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INTRODUCTION

Biological processes have been used for a long time in drinking water treatment, in the form of processes such as slow sand filtration and bank filtration. In recent decades, biological rapid filtration processes have come into use. Seminal work in this regard was the development of the Mülheim process (Sontheimer *et al.* 1978). Other early

applications of biological filtration are described by Rice & Robson (1982), and a recent comprehensive review is provided by Prévost *et al.* (2005). Although a filter will operate biologically whenever there is no disinfectant residual throughout the depth of the filter, a major impetus to biological filtration was given by the increased use of

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ozonation, which increases the concentration of easily biodegradable carbon.

The first modern quantification of biodegradable carbon in drinking water treatment was provided by van der Kooij (e.g. van der Kooij *et al.* 1982), who presented a method for what he termed Assimilable Organic Carbon (AOC). Another commonly used method measures Biodegradable Dissolved Organic Carbon (BDOC) (e.g. Servais *et al.* 1987). A comparison of various measurement methods is provided by Huck (1990). In addition to reducing concentrations of easily biodegradable organic carbon implicated in distribution system bacterial regrowth, biological filtration has also been shown to reduce levels of chlorine demand (e.g. Prévost *et al.* 1991) and chlorination byproduct formation potential (e.g. Niquette *et al.* 1999). Although biological filtration in drinking water treatment can also be used for other applications such as nitrogen removal (e.g. Rittmann & Huck 1989) the most common use is for the removal of organic carbon.

The uses of biological filtration mentioned just above can be considered as the classical applications, contributing to improved water quality within the distribution system, the importance of which is discussed by Sozański & Olańczuk–Neyman (2002). In addition to these, newer applications include the potential removal of trace contaminants such as pharmaceuticals and endocrine disrupting compounds that are biodegradable, the removal of odorous substances such as geosmin, and the use of biological filtration as a membrane pre-treatment.

Over the years various modeling approaches have been applied to biological filtration (normally for its classical applications) and the major models are reviewed briefly in this paper. However, no overall design framework for biological filtration has yet emerged. In part this is because specific generally accepted quantitative goals and therefore design criteria for biofiltration have not been defined (a notable exception being the Netherlands). In addition, the models are generally complicated and require a number of parameters which are not easily measured in practice.

In summary, there is as yet no paradigm or approach for biological filtration comparable to those that exist for other processes: for example the CT concept for disinfection, the use of G and GT in flocculation, and the use of the L/d ratio in granular filtration design. The objective of this paper

is to develop a unifying practically-oriented conceptual approach to biofiltration based on modeling. This approach is based on a previously-developed index for contact time, dimensionless contact time or X^* , which incorporates actual contact time and other factors important for biofiltration performance. The approach is discussed in terms of the various applications of biofiltration in relation to organic carbon – the classical one of the removal of easily biodegradable material, the removal of trace contaminants and, in somewhat more detail, the relatively new application of biological filtration as a membrane pre-treatment. For the latter application the approach needs further development. To relate X^* directly to practice, the last section of the paper proposes a new design-oriented parameter, the Biofiltration Factor (BF).

In principle both X^* and the Biofiltration Factor can be further developed for other uses of biofiltration in drinking water treatment such as nitrogen removal and the removal of iron and manganese. Similarly, the X^* and BF parameters could potentially be developed for the analysis of other traditional biological processes in drinking water such as slow sand or bank filtration and underground passage.

BACKGROUND

Factors affecting the performance of biofiltration

Various authors (e.g. Urfer *et al.* 1997) have reviewed the factors affecting the removal of biodegradable organic matter (BOM) in rapid biological filtration. The factors which are important are:

- Contact time
- Surface area
- Media
- Temperature
- Nature and concentration of the influent BOM, and
- Biofilm disruption (due to backwashing).

For a given bed depth, the surface area is inversely related to media diameter. The concentration of easily biodegradable organic matter entering a biofilter will be increased if there is upstream ozonation (e.g. van der Kooij *et al.* 1982).

From a practical point of view, the level of control which can be exerted over the various factors is important.

Table 1 (Huck *et al.* 2000) provides a qualitative assessment of both the degree of control and effect of the major parameters. For example the type of media can be specified and it has a moderate influence on process performance. On the other hand temperature has a major impact on performance and essentially cannot be controlled. The high effect shown for chlorination is because of its impact on biomass. (This figure was developed fairly early in the study and in fact the effect of chlorine at full scale was found to be less than expected.) The impact of the backwashing method may be lower than shown, except when biomass levels in the filter are low. The impact of salinity on the removal of micropollutants by biological carbon filters has been investigated by Sozański (1995).

Although the extent of biodegradation is related to the amount of biomass, the latter cannot be independently controlled because it is related to biodegradation through the yield coefficient. Thus the factors listed above which control the extent of biodegradation also essentially control the amount of biomass. Although various methods of measuring biomass have been used in biofiltration investigations, a number of investigators have estimated biomass by measuring phospholipid, based on a method by Findlay *et al.* (1989). In addition, Kijowska *et al.* (2001) have reported on the use of a metabolic activity test to investigate biodegradation in biological filters. Although other methods have been used, a detailed review of biomass measurement methods is outside the scope of this article.

In European practice a number of applications of biofiltration have been second-stage filters (typically with

GAC as media). A number of North American installations have implemented biofiltration in first-stage biofilters (i.e. where particle removal is also an objective). The approaches developed in this paper are applicable to both first and second-stage biofiltration.

Results from an American Water Works Association Research Foundation (AwwaRF) study (Huck *et al.* 2000) provide a useful context for a practically based approach to quantify biofiltration performance. In contrast to many investigations which are conducted at pilot scale, a substantial portion of this AwwaRF study was conducted on demonstration scale and full-scale biological filters. The objective was to optimize the performance of biological filters which were also intended for particle removal. Data were available from three locations and included a wide range of temperature conditions.

Figure 1 summarizes results from this study for the removal of BDOC above 10°C. There are two important messages in this figure:

- Removals of BDOC do increase with increasing empty bed contact time, but not proportionally.
- The removal of BDOC in these mature filters was essentially the same for both anthracite/sand and GAC/sand media. This was despite the fact that the biomass (as measured by the phospholipid method) was higher on the GAC media. GAC was shown to be better for BDOC removal at temperatures below 5°C and there were indications that it might also be better in the range from 5 to 10°C. GAC also did show an advantage above 10°C for the removal of a specific easily biodegradable compound (oxalate).

Table 1 | Level of control over parameters which influence (rapid) biological filtration

Parameter	Degree of Control (✓)		Effect (◆)	
	None	Low	Moderate	High
Media type			◆	✓
Chlorination			✓	◆
Filtration rate (EBCT)			✓◆	
Backwashing method			◆	✓
BOM loading		✓	◆	
Temperature	✓			◆
Time since startup	✓			◆

Source: Huck *et al.* (2000). Reprinted with permission.

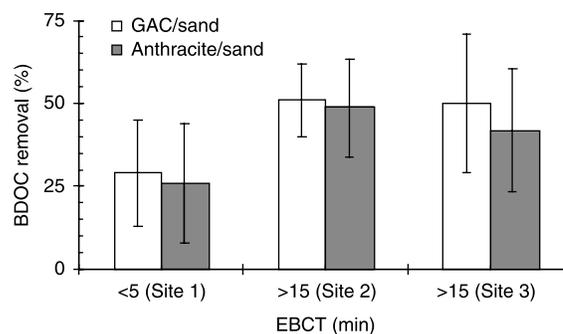


Figure 1 | Impact of EBCT on removal of BDOC (above 10°C). Source: Huck *et al.* (2000). Reprinted with permission.

Additional results from the study showed that the use of air scour in backwashing (under collapse pulsing conditions – Amirtharajah *et al.* 1991) did not affect BOM removals (oxalate, AOC and DOC), compared with water-only backwashing. These results were obtained using a GAC filter at temperatures above 10°C without chlorine in the backwash water. These data supported results of modeling conducted as part of the study (Hozalski & Bouwer 2001a,b, discussed later in this paper) that indicated backwashing would have to achieve biomass removals greater than 60% to have an effect on performance.

Additional investigations showed that the BOM removal capability of the full-scale filters was essentially unaffected by free chlorine in the backwash water. This was true whether air scour was present or absent. In contrast, the performance of bench scale filters backwashed using air scour was substantially impaired by free chlorine, although not by combined chlorine. From a practical perspective, it is encouraging that full-scale filters appear to be more robust. The reasons for the differences between bench- and full-scale performance could not be determined within the study. However, it is possible that the older, full-scale biofilms had become more resilient.

Winter data from Site 2 (low temperature, not shown) where lower biomass levels were observed suggested that the biomass level as measured by the phospholipid method only becomes important for performance if it drops below a certain level. The lack of a direct relationship between phospholipid biomass and removals was also noted by Urfer *et al.* (1997) in analyzing data published by Wang *et al.* (1995). Earlier, Hubele (1985) had concluded that the total biomass in a biofilter was only linked in a small way to the degradation of easily biodegradable TOC. The lack of a direct relationship between biomass and removals by biofiltration was also shown by Elhadi (2004) for geosmin and MIB.

In summary, important points for biofiltration modeling from this demonstration and full-scale study are:

- Removals increase with increasing EBCT, but in a less than proportional way,
- Backwashing likely does not have a measurable effect on BOM removal, and
- Biomass (as measured by the phospholipid method) is not directly quantitatively related to BOM removal.

Review of modeling approaches

Before considering the various available models, it is useful to place them in a contextual framework. Three important reasons for which modeling may be undertaken are:

- Design
- Operation
- Research

Table 2 shows some key attributes of models, in relation to these three goals. The number of diamonds indicates qualitatively the importance of each attribute, depending on the intended use of the model. For example, a model to be used for operation should be relatively easily understood by operations staff and based on easily measured parameters, whereas a very important aspect of a model used for research is its comprehensiveness. Models to be used for design would generally fall between these two categories in terms of their desired attributes. Most of the available models have been developed by researchers and their complexity, along with the general lack of specific quantitative goals for biofiltration, has hindered the use of modeling in practice. It is important to note that several of the models were developed in terms of a single rate-limiting substrate, whereas drinking water organic matter is complex, typically consisting of components with different rates of biodegradation.

Table 2 | Some key modeling attributes

Objective	Characteristic			
	Easily understood	Easily measured parameters	Comprehensiveness	Addresses multiple biofiltration objectives
Design	◆◆	◆◆	◆◆	◆◆(?)
Operation	◆◆◆	◆◆◆◆	◆	◆◆(?)
Research	◆	◆	◆◆◆◆	◆◆(?)

In their critical review of biological filtration, *Urfer et al. (1997)* summarized the practical usefulness of the most widely known models available at that time. Those authors note that although the models discussed provide important process insights for drinking water biofiltration, they are relatively complex and cannot be directly used by practitioners because they require parameters which cannot be readily obtained.

Table 3 is an updated version of a summary table provided by *Urfer et al. (1997)* showing the processes considered by each of the models, the major assumptions of each, the number of model parameters and the calculated outputs. The most important practical output, the substrate concentration, is an output of all of the models. The table has been updated by adding the models developed by *Uhl (2000, 2001)* and *Hozalski & Bouwer (2001a,b)*. These newer models are also complex. Thus all of the models in Table 3 fall into the category of ‘research models’ in Table 2.

In terms of a simpler approach, a first order empirical model was presented by *Huck & Anderson (1992)* and *Huck et al. (1994)* for the prediction of approximate performance. Those authors showed that the amount of BOM removed in a given biofilter was directly proportional to the influent concentration. Therefore removals can be approximated as a first-order process and a biofilter at apparent steady-state will essentially achieve a constant percentage removal, for a given EBCT and temperature. In addition to applying to the removal of AOC and BDOC, the relationship was shown to hold for THMFP and chlorine demand. *Gagnon et al. (1997)* demonstrated that it applied to the removal of carboxylic acids.

With the exception of the work by *Hozalski & Bouwer (2001a,b)* all of the models discussed above address steady-state or pseudo steady-state conditions. These are the most important in practice because, except for a few special circumstances such as process startup, biofilters can be generally considered to be operating at pseudo steady-state. *Hozalski and Bouwer’s* non-steady-state model was developed to capture the impacts on performance of the startup phase of a biological filter and recurring events such as backwashing. Their modeling provided good qualitative and quantitative agreement with full-scale data (*Hozalski & Bouwer 2001b*). As mentioned previously, their simulations demonstrated that backwashing

would have to remove a relatively high percentage of the attached biomass (60% or greater) to have an impact on performance. Thus as indicated by the results of the full/demonstration scale study (*Huck et al. 2000*) summarized earlier, for practically-oriented modeling a steady-state approach is normally adequate.

Although the recent book on biological filtration by *Prévost et al. (2005)* devotes several pages to a review of modeling, no overall modeling approach is proposed. Quantitative design-related criteria are not presented, except for the provision of some distribution system threshold BOM values. Also recently, *Westerhoff et al. (2005)* conducted a detailed investigation of ozone-biofiltration for geosmin and MIB removal. They conducted some first-order modeling of removal rates but did not provide an overall modeling approach. They did provide design criteria for new full-scale biofilters for geosmin/MIB removal. The process-related design recommendations were 48 inches of GAC and an EBCT of 7.5 minutes. It is thus evident (even in very recent publications) that an overall quantitative yet practically-based approach to biofiltration design has not been developed.

Urfer et al. (1997) describe the introduction of the steady-state biofilm model by *Rittmann & McCarty (1980a)* (Table 3) as being “seminal” work. The model considers microbial (Monod) kinetics and mass transport (Fick’s second law). The approach assumes that all nutrients are in excess except the rate-limiting substrate, S. An important contribution from this work was the introduction of the concept of S_{\min} , which is defined as the bulk substrate concentration below which a steady-state biofilm cannot exist. *Urfer et al. (1997)* note that several studies (*Rittmann & McCarty 1980b*; *Namkung & Rittmann 1987*; *Zhang 1996*; *Zhang & Huck 1996*) have shown that the principles of the steady-state biofilm model are applicable to the low substrate (oligotrophic) conditions usually encountered in drinking water biofiltration.

The basic equations for the model, which are written in dimensionless form, are shown below:

$$S_b^* = S_s^* + \int L^* / D^* \quad (1)$$

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

Table 3 | Summary of selected mechanistic models applicable to drinking water biofiltration

Reference	Processes considered	Major assumptions	Number of Parameters			Calculated output
			Measured or calculated	Assumed	Total	
Rittmann & McCarty (1980a)	Mass transport of substrate to biofilm, simultaneous diffusion and bioreaction of substrate in biofilm, definition of S_{min}	Substrate must be growth-limiting, predation not directly considered	5	3	8	Biofilm thickness, substrate flux into biofilm, bulk substrate concentration
Wanner & Gujer (1986)	General model with steady-state or dynamic cases possible, mass transport of substrate to biofilm, diffusion and bioreaction of substrate in biofilm, biomass exchange with bulk liquid	Biofilm biomass treated as continuum, one spatial dimension considered	Dependent on model formulation			Biofilm thickness, concentration of suspended microbes, bulk substrate concentrations
Billen <i>et al.</i> (1992)	Uptake of rapidly and slowly biodegradable DOC, temperature dependence of bacterial activity, physicochemical adsorption of bacteria, biological attachment of bacteria (biosynthesis of EPS), bacterial mortality and grazing by protozoa	No resistance to mass transfer of BDOC	10	2	12	Attached and suspended biomass, rapidly and slowly biodegradable DOC concentration
Wang & Summers (1995)	Mass transport of substrate to biofilm with substrate utilization on biofilm surface, definition of rapidly, slowly, and nonbiodegradable DOC fractions, biomass dependence on filter depth	Thin biofilm, expression of biomass dependence on filter depth must be known	7	3	10	Concentration of rapidly and slowly biodegradable DOC with filter depth
Zhang & Huck (1996)	Application of Rittmann & McCarty (1980a) approach to plug-flow bioreactors with AOC as substrate, definition of dimensionless EBCT	AOC is growth-limiting substrate and sole carbon and energy source, predation not directly considered	5	4	9	Substrate flux into biofilm, bulk effluent substrate concentration
Uhl (2001)	Substrate utilization, bacterial growth, maintenance and surface catalysis, suspended bacteria, grazing, non-steady state formulation also possible	No substrate concentration too low for degradation, growth of suspended bacteria neglected, mass transfer not limiting	12	3	15	Substrate concentration, removal of suspended bacteria and concentration of attached biomass as a function of filter depth

Table 3 | (continued)

Reference	Processes considered	Major assumptions	Number of Parameters		
			Measured or calculated	Assumed	Total
Hozalski & Bouwer (2002a)	As a function of time: transport of substrate to, and utilization by, biofilm, deposition of suspended bacteria, biofilm growth, decay and detachment, loss of biomass due to backwashing, mixtures of BOM	Biofilm homogeneous within a given filter segment, single substrate both diffusion and reaction limiting, no substrate competition in mixtures	17	4	21
					Biofilm thickness and substrate concentration as a function of time and filter depth, effect of mixture composition on effluent BOM concentration (Hozalski & Bouwer 2002b)

Source: Adapted from Urfer et al. (1997).

$$L_f^* = \frac{1 + S_{min}^* J^*}{S_{min}^*} = J^* k Y / b \tag{3}$$

where

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

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

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

where S_b is the bulk liquid concentration of the substrate ($M_s L^{-3}$); D is the free liquid diffusivity ($L^2 T^{-1}$); D_f ($L^2 T^{-1}$) is the diffusivity in the biofilm; K_s is the half velocity constant in the Monod expression ($M_s L^{-3}$); 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 ($M_x L^{-3}$); L_f is the biofilm thickness (L) and L denotes the thickness of the effective diffusion layer (L).

Because a direct analytical solution for this model is not possible, Rittmann and co-workers have developed pseudo-analytical solutions (Sáez & Rittmann 1988, 1992).

Zhang (1996) and Zhang & Huck (1996) obtained an analytical solution for this model for a plug flow reactor (i.e. a filter) after demonstrating that axial dispersion could be ignored. They then obtained the following equation for the depth of filter (X) required to achieve a specific concentration of substrate (at the biofilm outer surface):

$$X = \frac{v \tau}{\alpha D_f} \int_{S_{sx}^*}^{S_{so}^*} \frac{dS_s^*}{J^*} + \frac{v L}{\alpha D} \ln \left| \frac{J_o^*}{J_x^*} \right| \tag{4}$$

where v is the superficial flow velocity or hydraulic loading rate ($L T^{-1}$); α denotes the specific surface area which is the biofilm surface area in each unit volume of the bioreactor (L^{-1}); x is the longitudinal distance along the column (L); J_o is the flux of substrate into the biofilm at the inlet end of the column ($M_s L^{-2} T^{-1}$); J_x is the flux at the outlet end of the column ($M_s L^{-2} T^{-1}$); $S_{so}^* = S_{so} / K_s$ with S_{so} as the substrate concentration on the biofilm surface at the inlet end, ($M_s L^{-3}$); and $S_{sx}^* = S_{sx} / K_s$ with S_{sx} as the substrate concentration on the biofilm surface at depth X , ($M_s L^{-3}$).

DEVELOPMENT OF A UNIFYING PARAMETER – DIMENSIONLESS CONTACT TIME (X^*)

The most important contribution by Zhang (1996) and Zhang & Huck (1996) was the further transformation of Equation (4) to introduce the concept of an index for contact time, X^* . This parameter, referred to as dimensionless empty bed contact time (EBCT), is given by Equation (5):

$$X^* = \theta \frac{\alpha D_f}{\tau} = \int_{S_{sx}^*}^{S_{so}^*} \frac{dS_s^*}{J} + \frac{L^*}{D^*} \ln \frac{J_o}{J_X} \quad (5)$$

where $\theta = X/v$, i.e. the empty bed contact time.

Although Equation (5) provides the definition for X^* , the factors which it includes are shown more clearly in Equation (6), where τ is replaced by its constituent parameters.

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

It is thus evident that X^* includes, in addition to the actual contact time, the reactor specific surface area (surface area per unit filter volume), BOM diffusivity in the biofilm, biofilm density and biodegradation kinetic parameters.

Making use of pilot scale data from an earlier study (Huck *et al.* 1991), Zhang (1996) and Zhang & Huck (1996) modeled the removal of AOC and provided extensive interpretations related to the X^* parameter. They demonstrated a linear relationship between substrate removal and influent substrate concentration. The slope of this relationship (essentially the percentage removal) increases with X^* , as shown in Figure 2. (The values for several other dimensionless parameters are shown in the figure, as is the relative importance of the external mass transfer resistance (L^*/D^*)). Although the curves show some small influence of an increase in the external mass transfer resistance, the major message of the figure is that the slope (essentially percentage removal) initially increases essentially linearly with increasing X^* and then tapers off to approach a maximum value. Huck (1999) used this information to calculate approximate percentage removals, which are shown in Figure 3. It is thus evident that increasing the value of X^* , for example by increasing the residence time in a biofilter, will provide improvement, as would be expected, but with diminishing returns. This is consistent with, for example, the results for BDOC shown in Figure 1.

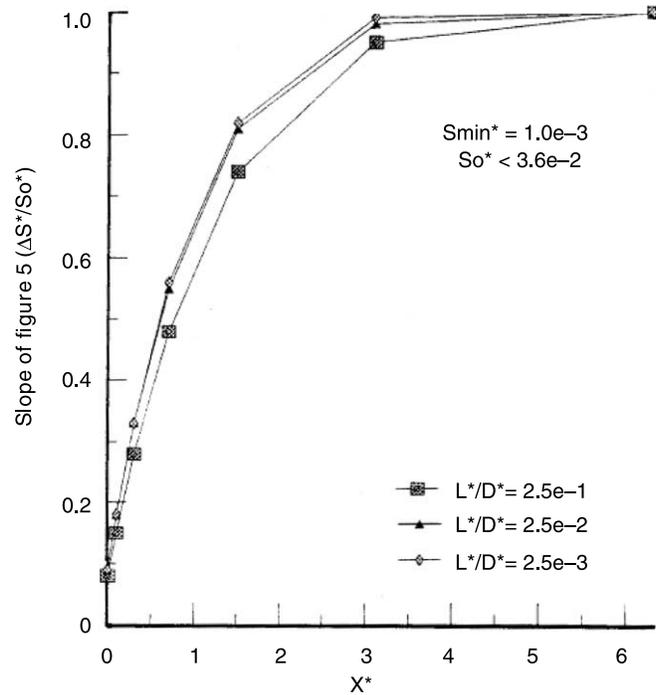


Figure 2 | Impact of X^* on change in substrate concentration. Source: Zhang & Huck (1996). Reprinted with permission.

An important aspect of X^* is that even though the physical parameters of a reactor (media and flow rate) may remain constant (and therefore contact time remains constant), X^* may vary, and therefore reactor efficiency may vary. This is illustrated conceptually in Figure 4(a), where the removal profile on the right represents a coarser media, which of course has a lower value of α , leading to a reduction in X^* , other factors such as flowrate remaining the same. A conceptually useful way to think of this is shown in Figure 4(b). If, in general, we consider that reactor effectiveness increases with reactor size, the effect of, for example, using coarser media is to “shrink” the filter with respect to its effectiveness as a bioreactor.

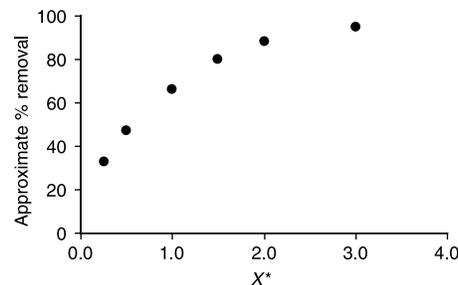


Figure 3 | Impact of X^* on percentage removal of substrate.

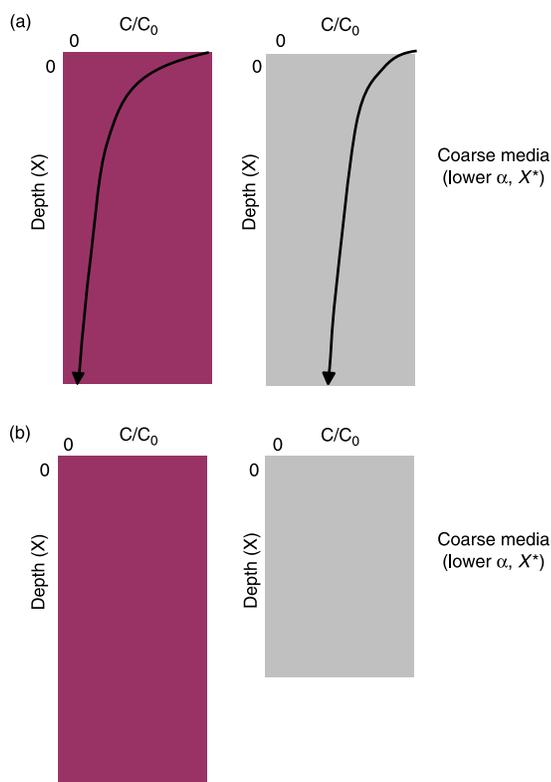


Figure 4 | (a) Conceptual illustration of impact of X^* on substrate removal in biofilter. (b) Illustration of concept that reduced X^* 'shrinks' biofilter (i.e. makes it less effective).

Based on their extensive analysis of the pilot scale AOC data from Huck *et al.* (1991), Zhang & Huck (1996) provided several important conclusions or inferences. These include the unimportance of external mass transport resistances and the fact that first order kinetics would be appropriate. Both of these conclusions have been subsequently confirmed in other experimental investigations, as will be discussed later.

Huck (1999) used the X^* concept to compare the expected bioreactor effectiveness of different typical filter designs. The hypothetical sand, dual-media and deep-bed monomedia filters considered all had an L/d ratio of 1,100, and hydraulic loadings of 5, 10 and 15 m/h respectively. The EBCT of the deep-bed monomedia filter was one third less than that of the sand filter, however the product of α and EBCT (and therefore X^*) decreased by a factor of three. In the same article he also used the X^* concept to interpret results from different filter configurations that had been evaluated by Ødegaard (1996) in an investigation of ozone/biofiltration for the removal of humic substances.

The next section discusses further development of the X^* parameter, including its usefulness in additional applications of biofiltration, and therefore its appropriateness as a unifying parameter. However, in order to make X^* useful in practice, it needs to be directly linked to filter design and operating conditions. This is done later in this paper through the introduction of a new directly-related parameter, the Biofiltration Factor (BF).

FURTHER DEVELOPMENT OF THE X^* PARAMETER

Work completed since the original development of X^* indicates that it can be simplified, which would assist in its acceptance by practitioners. In addition, the X^* concept can be extended, at least conceptually, to other applications of biological filtration (besides the classical ones of reduction in bacterial regrowth potential, chlorine demand and chlorination byproduct formation potential) such as trace contaminant removal and membrane pre-treatment.

Simplification of X^*

Two opportunities for simplification of drinking water biofiltration modeling, and of the X^* parameter in particular, are the ability, in most situations, to use first-order rather than Monod kinetics and to ignore mass transfer limitations. As indicated previously, the fact that external mass transfer limitations could be ignored and first-order kinetics could be used was identified by Zhang & Huck (1996). The appropriateness of first-order kinetics and the non-importance of both external and internal mass transfer resistances have been confirmed by further experimental work.

Booth *et al.* (2004) assessed the importance of both internal and external mass transfer resistances in an investigation of the removal of easily biodegradable compounds in pilot scale biofilters. Those authors calculated both an external mass transfer modulus, as suggested by Karel *et al.* (1985) and an internal mass transfer modulus, as suggested by Weisz (1973). In both cases, if the modulus is less than unity, biodegradation kinetics control the rate of substrate removal. For their system, Booth *et al.* (2004) showed that both moduli were much less than unity,

indicating that mass transfer limitations could be neglected. The authors noted that the biomass values measured in their research (using the phospholipid method of Findlay *et al.* (1989) with minor modifications) were in the general range of those measured in full scale biofilters, as reported by Huck *et al.* (2000). They thus concluded that it would be reasonable to neglect, at least as a first approximation, mass transfer limitations for easily biodegradable substances in drinking water biofilters.

The conclusion drawn by Booth *et al.* (2004) is supported by the work of several others. Wang (1995) showed that the rate of removal of BDOC in biological filtration was controlled by reaction rate, and not mass transfer. Urfer-Frund (1998) demonstrated that both external and internal mass transfer were of only minor importance for the removal of easily biodegradable compounds in his pilot scale biofilters. Gagnon & Huck (2001) showed that neither external nor internal mass transfer limited removals of easily biodegradable compounds in model drinking water distribution systems. Further, Elhadi (2004) demonstrated that mass transfer was not limiting for the removal of the odorous compounds geosmin and MIB in pilot scale biofilters.

The appropriateness of first order kinetics to represent the removal of easily biodegradable carbon in drinking water biofilters has been demonstrated in several instances. As described previously Huck *et al.* (1994) demonstrated this empirically, and Zhang & Huck (1996) also showed that a linear relationship between amount removed and influent concentration (i.e. first-order behaviour) could be derived from the steady-state biofilm model. Although they regarded their estimates of kinetic parameters for AOC as being tentative, Zhang & Huck (1996) also determined that the K_s value for AOC was relatively high (in the mg/L range) indicating that removals would be first order at concentrations normally experienced in drinking water. (An exception to this might be heavily polluted raw waters with very high concentrations (several mg/L) of easily biodegradable carbon.).

Trace contaminants are removed in biological processes through what is referred to as secondary utilization, where the biomass is sustained by biodegradation of the primary substrate. For the removal of geosmin and MIB at the ng/L level by biofiltration, Elhadi (2004) demonstrated that first order kinetics were reasonable. Ho *et al.* (2007) also

determined the biodegradation of these two compounds to be a pseudo-first-order reaction.

In terms of the simplification of X^* , if mass transfer is not limiting and kinetics can be considered to be first-order, Equation (6) can be simplified. Both the parameters D_f and K_s then become unimportant and can be grouped into a constant term, leading to Equation (7) below:

$$X^* = \theta\alpha K(kX_f)^{1/2} \quad (7)$$

where K is a constant.

The constant term is required to preserve the non-dimensionality of X^* and to maintain the relationship as an equation as opposed to a proportionality.

Since X^* was developed from a solution to the steady-state biofilm model, the elimination of two important aspects of that model (the substrate diffusivity and a parameter in the Monod relationship) make a strict theoretical interpretation of the simplified X^* more difficult. However, from a practical point of view the simplified X^* parameter can be very useful and provides important insights. Certainly θ and α can be relatively easily determined, and kX_f can be experimentally measured. K can be approximately determined from the relationship between percentage removal and X^* previously presented (Figure 3).

Equation (7) clearly shows that for a given substrate at a given temperature (i.e. given k), performance is essentially directly proportional to both EBCT and media surface area. Thus this expression allows comparison among different process conditions. Equation (7) also shows that the biomass term is less important. This is supported by results reported in the literature, as reviewed earlier in this paper.

The practically-oriented Biofiltration Factor introduced later in this paper does not require the simplified X^* . However the Biofiltration Factor may be more accessible to practitioners with the simplified X^* as an intermediate step.

Secondary utilization (trace contaminant removal)

Since biofiltration can provide removals of trace contaminants, it would be useful (for design purposes) to extend the X^* concept to secondary utilization. Biofiltration can be successful for the removal of odorous compounds (e.g. Elhadi *et al.* 2006; Metz *et al.* 2006). Figure 5 is an example of such removals, and incidentally also shows

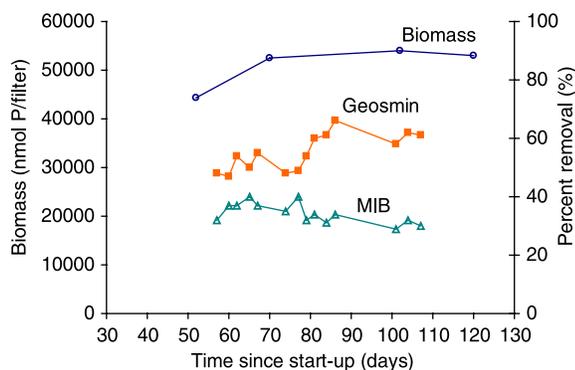


Figure 5 | Biomass and geosmin/MIB removals as a function of time in a biofilter (exhausted GAC, 20°C). Source: Elhadi (2004). Reprinted with permission.

that removals are not quantitatively related to biomass (as measured by the phospholipid method). In addition, recent work has shown removals of some pharmaceuticals at the ng/L level by biological filters (Guay 2007) and by biologically active roughing filters (Guay *et al.* 2007).

Using geosmin as an example, mass transfer is not limiting and first order kinetics can be considered reasonable, as was noted previously. The biomass in a biofilter used for the removal of trace contaminants is determined by primary utilization, so the parameter X^* remains important. (From a practical perspective it is reasonable to consider the entire biomass as a surrogate for organisms capable of degrading geosmin, although recent work (Hoefel *et al.* 2006) has shown that a consortium of three specific bacteria is necessary for geosmin biodegradation.) Therefore, the only additional parameter that is required is the first-order rate constant for geosmin biodegradation. Although it is not necessary to use the simplified expression for X^* to apply it to trace contaminant removal, it is conceptually simpler to do so.

Figure 6, based on Figure 3, shows conceptually the percentage removal of a trace contaminant as a function of X^* for two cases: in one case the first order rate constant for secondary utilization (k_2) is greater than that for the removal of easily biodegradable carbon (k) and so at a given X^* the percentage removal of the secondary contaminant is greater. The opposite is true when $k_2 < k$. Thus as a first approximation, removals of a trace contaminant (probably following acclimation) could be estimated for a given biofilter by laboratory determination of the biodegradation rate constant in comparison to that for the removal of easily biodegradable carbon.

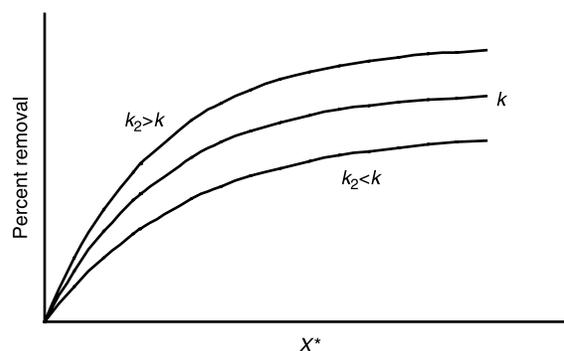


Figure 6 | Conceptual illustration of the impact of the biodegradation rate constant for a trace contaminant on its removal in relation to that of the primary substrate.

It should be noted that, based on Equations (6) and (7), the relative X^* values for a given removal of a secondary contaminant and the primary substrate will be related to $(k_2/k)^{1/2}$. This should be experimentally verified.

Biofiltration as membrane pre-treatment

Recent work has shown biofiltration to have promise as a membrane pre-treatment to reduce fouling. Because this is a relatively new application for biofiltration, the background is presented in more detail. The extension of X^* to this application requires considerable new development, which is also presented in detail.

In this section we first review the types of organic matter most responsible for organic and bio-fouling, two important types of membrane fouling. We then relate this information to biofiltration. In considering a biofilter as a way of reducing such fouling, it is important to consider the fact that biofilters not only remove organic matter but also produce both soluble and particulate organic material.

Organic matter and fouling

A number of studies (e.g. Yuan & Zydny 2000; Mosqueda-Jimenez *et al.* 2004) have determined that organic material with high molecular weight and high hydrophobic content contributes to the bulk of organic membrane fouling. However Carroll *et al.* (2000) suggested that a lower molecular weight, hydrophilic fraction is responsible. Investigating natural waters, Howe & Clark (2002) determined that very small colloids 3–20 nm in diameter (primarily organic material) seemed to be most important. Amy and coworkers

(e.g. Lee *et al.* 2004) have demonstrated that the NOM fractions contributing most to biofouling are colloidal biopolymers (polysaccharides, protein and amino sugars).

For charged ultrafiltration membranes, Cho *et al.* (2000) showed that the NOM foulants were the larger sized neutral and/or basic components. Investigating fouling during ultrafiltration of solutions of NOM fractions isolated from surface waters, Makdissy *et al.* (2004) found that the most significant flux decline (i.e. fouling) was due to the organic colloid fraction, a hydrophilic fraction consisting mostly of bacterial cell wall residues.

Jarusutthirak *et al.* (2002) investigated the fouling effects of wastewater effluent organic matter (EfOM) isolates on nanofiltration and ultrafiltration membranes. The colloidal EfOM fraction primarily consisted of polysaccharides, proteins and/or aminosugars and produced a high level of fouling. In particular, polysaccharides and/or aminosugars within the colloid fraction were found to be important in fouling. Haberkamp *et al.* (2007) also demonstrated the importance of EPS (primarily polysaccharides) for fouling of ultrafiltration membranes.

Flemming & Wingender (2001) discuss the role of extracellular polymeric substances (EPS) in biofouling and note that calcium precipitation can occur preferentially in the EPS matrix. Flemming *et al.* (1997) found that biofilm on an RO membrane contained more EPS under conditions of nutrient depletion. Also for RO membranes, Gabelich *et al.* (2004) attributed higher levels of fouling at pilot scale (compared to full scale) to production of low levels of natural polymers by biological communities upstream of the pilot scale membrane.

It is thus evident that specific fractions of organic matter are important for fouling of various types of membranes. Of particular importance is organic colloidal material which can include biopolymers and bacterial cell wall residues.

Biofouling

The traditional understanding of biofouling is that microorganisms in the feedwater become trapped on the membrane surface and proliferate because of the available nutrients (Uhl *et al.* 2003). The limiting nutrient in drinking water is usually considered to be biodegradable carbon.

Biofouling has been investigated by a number of researchers (e.g. Vrouwenvelder *et al.* 2000, 2007) and a comprehensive review of this work is outside the scope of the present article. Of significance for the present article is the fact that, for example, Flemming (2002) states that, for biofouling, nutrients are the most important fouling factors. Flemming *et al.* (1997) proposed a biofilter as a membrane pre-treatment for the removal of easily biodegradable material, but provided no specifics.

Uhl *et al.* (2003) have presented calculations to show that microbial cell material transported to the membrane surface (as opposed to growth of microorganisms on the membrane) can be sufficient to account for observed accumulations of biological foulants in some situations.

Thus although the literature indicates that reducing the concentration of both bacterial nutrients (with organic carbon usually being the limiting one) and also microbial product material will be important for reducing organic- and biofouling, the use of biofiltration to do this has received virtually no systematic investigation. Individual studies that have been conducted are reviewed briefly in a later section.

Microbial product material

Since it is evident that microbial product material can play a significant role in fouling, the paragraphs below briefly review the different types of soluble and particulate material that may be produced.

Laspidou & Rittmann (2002a) identify three groups of microbial products (EPS, soluble microbial products (SMP) and inert biomass) and provide a critical review of the relationships among them. SMP are defined as “soluble cellular components that are released during cell lysis, diffuse through the cell membrane, are lost during synthesis, or are excreted for some purpose”. Those authors report that these substances have moderate formula weights and are biodegradable. They note that although early biofilm research often assumed that polysaccharides were the most abundant component of EPS, in fact proteins and nucleic acids also appear in significant amounts. The relationship of the rate of EPS production to the biomass growth rate (or rate of substrate consumption) is not straightforward and various datasets appear to contradict each other (Laspidou & Rittmann 2002a).

SMP consists of two components. The first, defined as utilization-associated products (UAP), are produced as a direct result of substrate utilization. The second component is biomass-associated products (BAP), which are formed from biomass, presumably as part of the decay process. Both of these components are relatively biodegradable, however they have distinct degradation kinetics (Lapidou & Rittmann 2002a).

Although SMP are completely soluble, EPS are mostly, although not completely, associated with the solid phase and are therefore mostly insoluble (Lapidou & Rittmann 2002a). Those authors note that some researchers divide EPS into 'bound' and 'soluble'.

In a companion paper Lapidou & Rittmann (2002b) report on non-steady-state modeling of EPS, SMP and active and inert biomass. They report good agreement between their modeling results and experimental data obtained by others for a pure culture (*Pseudomonas atlantica*) degrading glucose. To conduct the modeling they used kinetic parameters for UAP and BAP obtained from other work. Of significance for the present paper is the fact that these kinetic parameters reflect much lower biodegradation rates for UAP and BAP than for glucose. The maximum specific utilization rate for UAP is more than an order of magnitude lower than that for glucose, and the rate for BAP is in turn more than an order of magnitude lower than that for UAP. Although the authors note that the set of parameters used would not necessarily apply to all systems, and although easily biodegradable BOM in drinking water would not necessarily be degraded as quickly as glucose, it is reasonable to expect that biodegradation rates for microbial products in drinking water biofilters would be substantially lower than for easily degradable BOM.

Thus microbial products (either entering the filter or generated within it) can be biodegraded and/or removed by physical filtration within the filter. The bed depths or contact times required are currently unknown but are likely to be greater than for the removal of easily biodegradable organic matter.

Effectiveness of biofiltration as a membrane pre-treatment

Before attempting to quantitatively assess the requirements for biofiltration as a membrane pre-treatment in subsequent

sections, this section briefly reviews investigations where biofiltration has been used for this application.

Work related to bank filtration and aquifer recharge has demonstrated that the organic components of interest as membrane foulants have the potential to be removed by biofiltration processes. Dillon *et al.* (2005) reported that organic matter degradation at aquifer storage and recovery sites occurred most rapidly for larger molecular weight materials, including polysaccharides. For wastewater effluent organic matter Sattler *et al.* (2005) reported that the polysaccharide components were readily removed in the upper vadose (unsaturated) zone. The removals of both polysaccharides and protein-like organic matter were attributable to biodegradation. Grünheid & Jekel (2005) also reported rapid removal of polysaccharides. Although these results suggest that biofiltration could be promising for the removal of specific organic components leading to membrane fouling, the detention times in rapid filters are much less than in bank filtration and aquifer recharge, and therefore X^* will be much lower.

Results from investigations of rapid biofiltration have, however, also demonstrated its promise as a membrane pre-treatment. Using a synthetic water containing easily biodegradable substances and humic acid, Basu & Huck (2004) found that placing a biofilter ahead of a low-pressure membrane reduced the rate of fouling. Results from subsequent work using the same experimental setup with some modifications are shown in Figure 7 (Mosqueda-

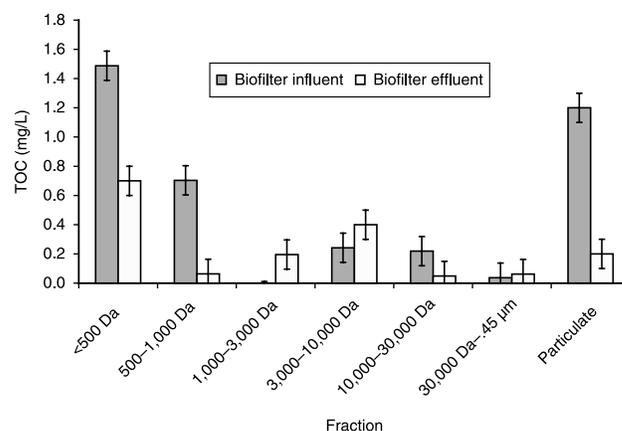


Figure 7 | Removal of various molecular weight fractions of organic carbon by a biofilter used as membrane pre-treatment. Source: Mosqueda-Jiménez & Huck (2006a). Reprinted with permission.

Jiménez & Huck 2006a). Here it is evident that the biofilter (placed ahead of the membrane) was capable of removing both low molecular weight (easily biodegradable) organic matter and particulate material retained on a $0.45\ \mu\text{m}$ filter (colloidal humic material). The membrane with the biofiltration pre-treatment showed a lower rate of flux decline than a second membrane operated in parallel without pre-treatment (Mosqueda-Jiménez & Huck 2006a). The reduction in fouling for both UF and NF membranes as a result of biofiltration pre-treatment is illustrated in Figure 8. In examining low flux NF installations for NOM removal from surface water, Uhl *et al.* (2007) found that biological activated carbon filtration at an EBCT of 34 minutes could provide significant mitigation of fouling. Dual-media filtration at an EBCT of 5.6 minutes had a limited effect. However, Xiong (2005) found that the effluent from a full-scale biofilter did not produce a lower rate of fouling with low-pressure membranes than did the filter influent. In very recent work using size exclusion chromatography with continuous organic carbon detection (LC-OCD) analysis, Guay (2007) obtained results demonstrating that biofiltration is capable of reducing the size of the chromatogram peak attributed to polysaccharides and proteins, the major organic fouling components.

Thus while biofiltration shows considerable promise as a membrane pre-treatment, to be effective it needs to achieve a net reduction in both the dissolved and

particulate components most important for fouling, including the easily biodegradable organic matter that would facilitate actual bacterial growth on the membrane.

As indicated by Laspidou & Rittmann (2002a,b), the relationships among the original substrate and dissolved and particulate bacterial material within the filter will be complex. We now demonstrate the general conceptual relevance of X^* with respect to dissolved and particulate bacterial material exiting a biofilter.

Net production of biomass

The net production of biomass by a biofilter is determined by the following factors:

- The amount of biomass entering the filter
- The amount of biomass generated within the filter
- The biomass removed or consumed within the filter

The biomass concentration entering the filter will be a function of raw water levels and/or upstream processes. The generation of biomass within the filter will be largely controlled by the consumption (removal) of easily-degradable BOM. Since the latter is related to X^* , biomass generation will also be related to X^* via a yield coefficient. The operating conditions of the filter may be important in determining the yield coefficient. For example Uhl (2000) determined a much lower yield coefficient (based on biodegraded TOC) than is typically used (e.g. Laspidou &

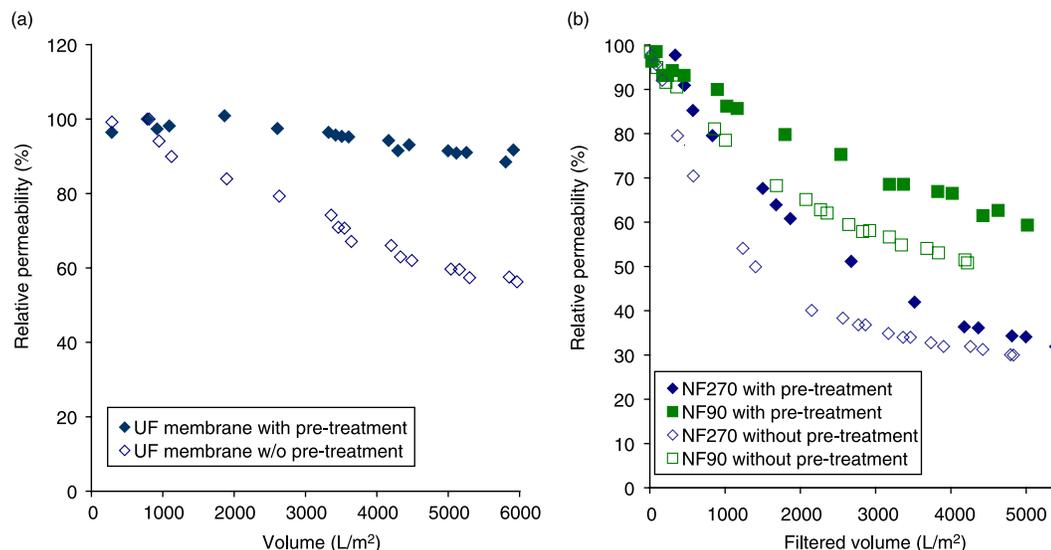


Figure 8 | Reduction in membrane fouling due to biofiltration pre-treatment. Source: Mosqueda-Jiménez & Huck (2006b). Reprinted with permission.

Rittmann 2002b). This may have been because he was investigating second-stage biological filters, which would have lower influent concentrations of easily biodegradable carbon than many first-stage filters.

Biomass produced within the filter is of course subject to decay and detachment. Detachment is a complex phenomenon (Stewart 1993) and is related both to the amount of biomass present and the growth rate.

Filtration mechanisms can act to remove biomass in the filter influent. Hubele (1985) noted that filtration mechanisms can remove, lower in the filter, detached biomass from higher in the filter (facilitating the removal of more slowly biodegradable DOC lower in the filter), and that therefore the filtration characteristics of biofilter media are important. Filtration mechanisms will apply to both active and inert biomass as well as to insoluble or solid-associated EPS. Uhl (2000, 2001) included the attachment/detachment of bacteria in his modeling of biofilter performance.

For the minimization of membrane fouling, it is important to minimize the level of biomass exiting the filter. The generation of biomass will largely occur in the initial part of the filter where most of the easily biodegradable material is removed. The concentration and type of easily biodegradable material will be a function of the incoming water quality and may not be appreciably affected by any upstream processes, with the exception of ozonation.

From a design and operating point of view, the major opportunities for minimizing the amount of biomass exiting the filter lie in maximizing biomass removal within the filter and/or in minimizing its generation. In general, this would mean increasing the contact time to facilitate filtration (and at least some consumption) of both incoming biomass and biomass generated within the filter.

With respect to quantifying filtration, the factors affecting its performance are well known (e.g. MWH 2005). These include flow rate and media depth and diameter as well as characteristics of the media and particles affecting attachment. Characteristics affecting attachment and the conditions affecting detachment can be represented in filtration modeling through coefficients. Usually in rapid filtration upstream coagulation plays a crucial role and has a substantial impact on the coefficients. In the case of biofiltration as a membrane-pre-treatment, it is also important to consider the case where such filters may be

operated without coagulation. This is possible (and desirable from a “green” perspective) since the membrane is capable of providing particle removal. The filter is therefore functioning as a bioreactor and to reduce turbidity peaks in the case of varying influent water quality. The filtration performance in terms of particle removal therefore does not need to be as high as if the membrane were not also present.

In terms of the major controllable physical factors affecting filtration performance (media depth (L) and diameter (d)), the ratio L/d is often used to guide design (e.g. Kawamura 1999). For a given flowrate, L directly determines θ , the empty bed contact time. Thus L/d is directly proportional to θ/d . Since the specific surface area of the media, α , is inversely proportional to d , θ/d is directly proportional to the product $\theta\alpha$, i.e. the physical elements of X^* (Equations (6) & (7)).

The L/d ratio is sometimes used as part of the exponent in a first order relationship to approximately describe particulate removal by filtration. Such a relationship implies diminishing returns with increases in filter depth, a behavior which parallels diminishing returns in terms of biofiltration performance for increasing values of X^* .

It is thus evident that the parameter X^* has relevance in terms of the net production of biomass by biofilters. Elements of this parameter are important both for the generation and the removal of biomass within a filter. Further work is required to quantitatively apply or adapt the X^* parameter to the net production of biomass by biological filters. The modeling work of Uhl (2000, 2001) and Hozalski & Bouwer (2001a,b) which includes bacterial attachment and detachment may provide useful insights in this regard.

Net generation of dissolved microbial product material

Although the considerations regarding the net production of dissolved microbial product material (SMP and dissolved EPS) in filters bear some similarity to those for the net generation of biomass, there is an important difference in that these soluble materials will not be subject to removal by physical filtration.

As is the case for biomass, generation of dissolved microbial product material within the filter is controlled largely but not entirely by the removal of easily-degradable BOM. The relationship is more complicated than for biomass

(Lapidou & Rittmann 2002a), but should generally be related to X^* . The biofilter influent will likely contain some of this material. Whether or not this is a significant amount in relation to that generated in the filter will be site-specific.

The removal of dissolved microbial product material within a biofilter is by biodegradation. Although the X^* value determined for the primary limiting substrate entering the filter cannot be directly applied to this removal, in principle, a second X^* value could be calculated for the second part of a biofilter (i.e. the part beginning where the original easily biodegradable substrate has been largely removed). The kinetics of biodegradation of dissolved microbial product material will undoubtedly be slower than for easily biodegradable material (Lapidou & Rittmann 2002b). Slowly biodegradable material in the filter influent will also be removed in this region of the filter. As noted earlier, Hubele (1985) discussed the importance of filtration mechanisms for retaining biomass lower in the filter for the removal of more slowly biodegradable DOC. Based on their simulations, Hozalski & Bouwer (2001b) note that the removal of slowly biodegradable material is significantly enhanced by the presence of easily biodegradable material, primarily because of the ability to maintain greater biomass levels in the filter. Biomass transport into the lower part of the filter will require modification to the X^* concept. Uhl (2001) has noted that biodegradation can occur at concentrations below the S_{\min} value, and has attributed this to biomass transport into the region of the filter where this biodegradation is occurring. First-order kinetics may not be applicable for dissolved microbial product material (and for influent slowly-biodegradable material) and diffusion limitations may not be able to be neglected. Whether or not it might be possible to use, as an approximation, the beginning of the filter bed as the starting point in terms of time or bed depth for the second X^* , would depend on how low the degradation rates were.

Minimization of the concentration of dissolved microbial product material in a biofilter effluent will likely involve increasing filter contact time above that which might be used for easily removable BOM. As mentioned previously, Uhl *et al.* (2007) reported much better performance in reducing fouling for BAC pre-treatment compared to dual-media pre-treatment (which would also have been biologically active). Although other factors may have played

a role, the much longer contact time provided in the BAC filter was likely an important contributor to this result. In general, whether the contact time required for the effective removal of dissolved microbial product material (or specific components that are most important for fouling) or for the minimization of particulate biomass exiting the filter will govern, will likely be site-specific. In terms of overall process train optimization it will also depend on the extent to which it is desired to minimize bio- and organic fouling through pre-treatment. These issues represent fruitful ground for well-conducted experimental investigations and further theoretical development.

DEFINITION OF A NEW PERFORMANCE PARAMETER – THE BIOFILTRATION FACTOR (BF)

This paper has demonstrated that the concept of an index for contact time, the dimensionless contact time or X^* , is relevant for carbon removal for various objectives in drinking water biofiltration. These objectives include the original classic one of the removal of easily biodegradable organic matter, the removal of trace contaminants, and the emerging and potentially important application of biofiltration as a membrane pre-treatment. However, for this concept to be useful in practice, it needs to be concretely related to process conditions. We therefore propose a new parameter, the Biofiltration Factor (BF), which is a simple linear transformation of X^* , but with a numerical reference point value tied to practice.

Examination of Figure 3 shows that approximately 50% AOC removal would be obtained at an X^* value of 0.5. For convenience, we therefore assign a BF value of 0.5 to a filter capable of achieving 50% removal of AOC at approximately 20°C. This is a relatively easy definition to remember. The BF value required for other treatment objectives (or at other temperatures) can then be directly related to this.

It is likely that a BF of 0.5 would correspond to empty bed contact times in the range of 3 to 5 minutes, for the usual filtration media diameters. More accurate quantification of this from the literature is outside the scope of the present article and is a topic for further work.

The Biofiltration Factor is thus in a general sense analogous to the parameters G or GT for coagulation, L/d for filtration or CT for disinfection. That is to say, that

although these parameters do not capture all the subtleties of the relevant processes, they are extremely useful as good engineering approximations for design. With the development of the appropriate database, it is expected that the Biofiltration Factor can also be used as a design and operating parameter. Although the BF is a parameter rather than a model, it is consistent with the desirable attributes proposed for design and operational models in Table 2.

Figure 9 shows conceptually the relationship of process performance to the Biofiltration Factor, in a very general way. Figure 9(a) illustrates the impact of temperature on AOC removal and, for a given removal, the relative BF values required for AOC and a trace contaminant with slower biodegradation kinetics. Figure 9(b) shows conceptually the reduction in membrane fouling as a function of the Biofiltration Factor. The relative location of the lines for fouling reduction by the removal of dissolved and particulate microbial product material is as yet unknown and may be site-specific.

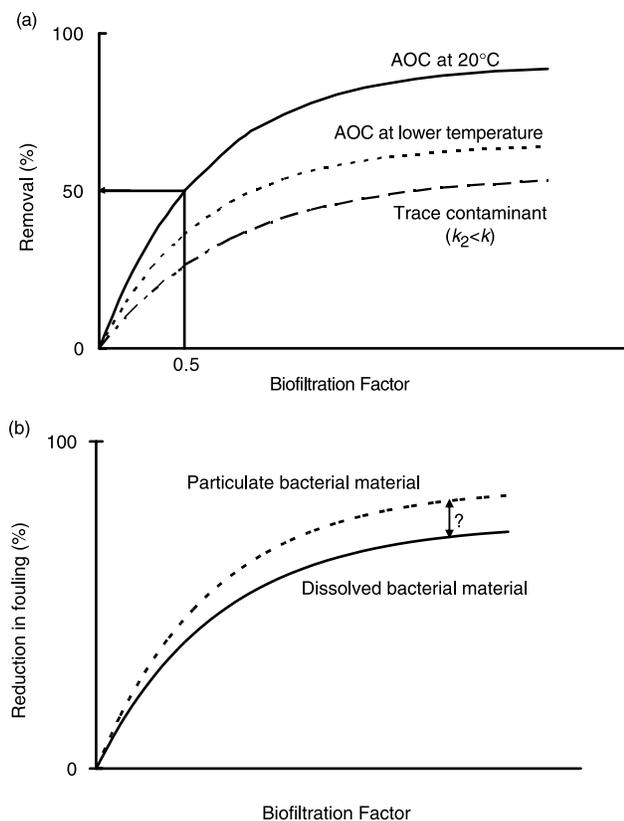


Figure 9 | Conceptual relationship of BF to performance (a) Impact of temperature and kinetics; (b) membrane fouling vs Biofiltration Factor.

Figure 10 shows conceptually the expected general sequence of Biofiltration Factors anticipated for various biofiltration applications. The exact value required, such as for reduction in membrane fouling in a particular situation will depend on site-specific water quality and operational conditions. Huck (1999) estimated X^* values of more than 100 for the slow sand filters investigated by Ødegaard (1996), and noted that they were considerably over-designed with respect to biodegradable TOC removal. This means that processes such as bank filtration are also likely considerably over-designed in this regard.

Considerable work remains to determine quantitative values of BF required for specific applications and as a function of temperature, so that a series of nomographs can be developed. Substantial progress in this regard can likely be made with data from the literature for classical applications of biofiltration and for the removal of at least some trace contaminants including geosmin and MIB, although additional experimental work will undoubtedly be required. The nature of these additional investigations can be better determined once information from the existing literature on biofiltration has been thoroughly analyzed for possible contributions to the quantification of X^* and BF. For membrane pre-treatment, additional theoretical work is required as previously indicated and experimental work will also be necessary. Initially, the experiments will likely require use of a control filter for which AOC removals are measured, to provide a reference point for quantification of BF values for various applications and situations. Although it is likely that most of the work would be conducted at pilot scale, ultimately confirmation at full or demonstration scale should be undertaken.

SUMMARY

This paper presents the development of a unifying practically-oriented conceptual approach and newly defined performance parameter for biological filtration in drinking water treatment for various applications. These applications include the classical objectives of removing easily biodegradable carbon which leads to bacterial regrowth in distribution systems, and the usually related objectives of reducing chlorine demand and concentrations of chlorination byproduct precursors in systems that use chlorine,

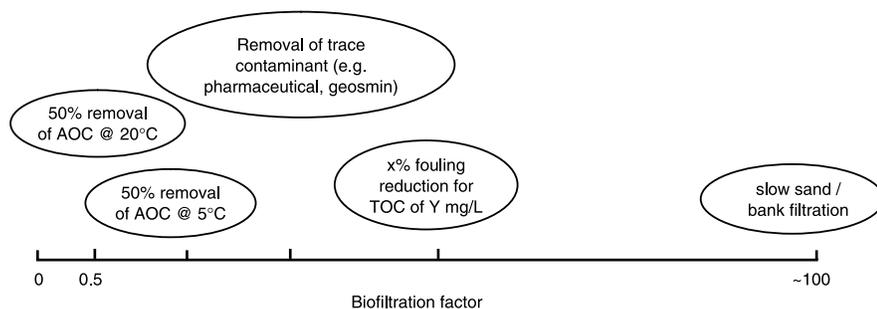


Figure 10 | Expected general sequence of BF required for various treatment goals (horizontal axis is not necessarily linear).

by reducing the concentration of biodegradable TOC. The additional applications addressed in the paper are the use of biological filtration for the removal of specific trace organic contaminants and as a pre-treatment for membrane filtration.

The major factors affecting biofiltration performance have been examined in a number of investigations. These factors include contact time, surface area of the media for biofilm growth, type of media, temperature, the nature and concentration of the influent BOM, and the potential disruption of the biofilm due to backwashing. With the exception of temperature and to some extent and nature of the influent BOM (depending on the presence or absence of upstream ozonation) these factors are subject to control through either design or operation. Of importance with respect to the development of a practically oriented modeling approach, full scale investigations have shown that BOM removals increase with increasing contact time (but in a less-than-proportional way), that backwashing likely does not have a measurable effect on BOM removals and that biomass (as measured by the frequently-used phospholipid method) is not quantitatively related to BOM removal.

Over the years a number of mathematical models have been developed for drinking water biofiltration, focusing largely on the classical and related process objectives mentioned above. Although some of these models have found localized use in practice, no generally accepted modeling approach has emerged. The major models, reviewed briefly in this paper, are generally complex, incorporating a number of parameters. Thus often their use has been restricted to the research investigation in which they have been developed. A practical modeling approach or conceptual framework that could be used in

the design and operation of full-scale biological filters should be relatively simple and incorporate the key controllable process parameters.

Arguably the seminal model for biofilm processes is the steady-state biofilm model developed by Rittmann & McCarty (1980a,b). Although not originally developed for drinking water biofilters, it has been used in this application. From the perspective of a practitioner this model, which is presented in dimensionless form, is complex. However an important development for the understanding of drinking water biofiltration, the concept of dimensionless contact time, X^* (Zhang 1996; Zhang & Huck 1996), is based on this model. The parameter X^* is effectively an index for contact time and incorporates, in addition to contact time, other factors: the reactor specific surface area, substrate diffusivity and biodegradation kinetic parameters, and the biofilm density. The effect of temperature is incorporated through the substrate-related parameters. Biofiltration performance (i.e. removal) increases essentially linearly with X^* for values of X^* up to about 0.5 and thereafter increases in a less than proportional way.

Although EBCT is often used as a parameter for biological filtration, and in fact may be the single design criterion used, the use of X^* is preferable as it allows for the meaningful comparison of biofilters incorporating different media sizes as well as having different contact times. It also provides insights into the removal, by the same biofilter, of substrates with different biodegradabilities or which are differently defined (e.g. AOC vs. BDOC).

In this paper we demonstrate that the X^* parameter is also relevant for additional biofiltration process objectives. However, we first show that X^* can be simplified, because in a number of cases mass transfer limitations can be

neglected and first order biodegradation kinetics can be used. The first of the additional applications demonstrated is the use of X^* to describe the removal of trace contaminants through secondary utilization. Such contaminants include the odorous compounds geosmin and MIB and biodegradable pharmaceutically active and endocrine disrupting compounds.

Recent investigations have shown that biological filtration can be effective as a membrane pre-treatment. It can act to reduce concentrations of both particulate matter and dissolved biodegradable material that have been shown to lead to membrane fouling. A major section of this paper is devoted to this use of biofiltration and to the extension of X^* to this application.

The application of biofiltration as a membrane pre-treatment places several important additional requirements on the biofiltration process. In addition to reducing concentrations of easily biodegradable carbon which could lead to bacterial growth on the membrane, the biofilter should minimize the concentrations of both particulate and dissolved microbial product material in the filter effluent.

Particulate biomass can be present in the biofilter influent and also be generated in the biofilter itself. Thus physical filtration mechanisms are important with regard to the level of particulate biomass exiting the filter. Important physical factors influencing filtration are bed depth and media diameter. Since filter specific surface area (α) is inversely proportional to media diameter, the two physical factors contained within X^* (EBCT and α) are important for physical biomass removal by the filter. This physical filtration can be expected to show the usually observed pattern of decreasing returns with increasing filter depth, which parallels the relationship between biodegradation and X^* .

Some dissolved microbial product material may enter the biofilter, and such material will certainly be generated within the filter. The generation will be related primarily to the removal of easily biodegradable material, and thus to X^* . The removal of this material by biodegradation within the filter will also be related to X^* , however it will likely be necessary to define a second X^* for the filter relating to this material and also slowly biodegradable carbon entering the filter. Because biomass transported from the upper part of the filter may be important for this biodegradation, some conceptual modification to the X^* parameter will likely be required for this

application. This area is a fruitful one for further experimental work and theoretical development.

In order to be useful for biofilter design and operation (regardless of the application), the index for contact time represented by X^* must be transformed into a parameter directly related to practice. We therefore introduce a new parameter, the Biofiltration Factor (BF), which is directly linearly related to X^* . For convenience, this parameter is assigned a value of 0.5 for a filter capable of achieving 50% AOC removal at approximately 20°C. Biofiltration Factors for the removal of trace contaminants and for the use of biofiltration as a membrane pre-treatment will undoubtedly be larger than this value. Although the initial development of a series of nomographs for the Biofiltration Factor for various applications can be undertaken based on data in the literature, additional experimental work will be required to complete this.

Although this paper focuses on rapid biofiltration applications for organic carbon removal, it should be possible to extend the X^* concept to other drinking water applications of biofiltration (e.g. nitrogen and iron/manganese removal) and to processes with much slower filtration rates such as bank filtration/ground passage and slow sand filtration. Since X^* is based on a solution to the steady-state biofilm model in which axial dispersion is neglected, it will be important to determine whether axial dispersion can also be neglected for processes with much lower filtration rates.

Through the analysis presented in this paper several areas for further work have been identified. These include:

- Experimental assessment of the predicted relationship $(k_2/k)^{1/2}$ for the relative removal of a trace contaminant with first-order rate constant k_2 and AOC with a rate constant of k ,
- Further development of analytical methods for quantitation of the various organic fractions important for membrane fouling, and determination of biodegradation rate constants for these fractions,
- Additional investigations into the production of dissolved microbial product material by biofilm organisms and the production and detachment of biomass from biofilms under conditions relevant for drinking water biofiltration,
- Further development to apply X^* to the microbial product materials mentioned in the point just above, including

the role of biomass transported to the lower regions of a filter,

- With respect to membrane fouling, determination of which is governing in terms of the necessary X^* : particulate or dissolved microbial product material (or specific components thereof),
- Quantification, based on the literature and further experimental studies, of Biofiltration Factors for various biofiltration applications, leading to the development of monographs for design and operational use, and
- Extension of the X^* parameter and the Biofiltration Factor to other drinking water applications of biofiltration (e.g. removal of nitrogen and Fe/Mn), and evaluation of the applicability of these two parameters to biological processes other than rapid filtration (e.g. slow sand and bank filtration, and underground passage).

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