

# Water Quality Modelling for Drinking Water Distribution Systems

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**Abstract:** A dynamic water quality model for drinking water distribution systems has been developed in this study, to include processes that occur in the bulk water, as well as those occurring in the biofilm of a distribution system. The model has been validated against water quality data obtained from extensive experimental studies conducted with biofilm reactors. Protein and carbohydrate densities in the biofilm represent biofilm biomass. This model is able to predict the disinfectant decay due to organic matter in the bulk water, as well as that due to biofilm. It simultaneously predicts the growth of biofilm in terms of carbohydrate and protein densities. While this model is complex enough to describe the water quality changes in a distribution system, it is also simple enough to be incorporated into a hydraulic model in order to describe the interaction between disinfectant and microbiological quality throughout a drinking water distribution system.

**Keywords:** *Biofilm, Bulk water, Drinking water quality, Modelling*

## 1. INTRODUCTION

Drinking water starts its journey within catchments, and is subsequently purified at treatment plants and delivered through distribution systems. The water quality generally deteriorates while it passes through the catchment. However, the subsequent treatment processes produce water of high quality. But, within the distribution system the quality of the water generally deteriorates again.

A water quality model for a distribution system should be coupled with a hydraulic model to adequately describe the microbiology of the system. This is where the problem lies. Although there are excellent hydraulic models available today, they do not have accurate water quality modules embedded in them. This may be due to the large computing time required to run a water quality module along with a hydraulic model. For example the EPANET developed by the United States Environmental Protection Agency could be used to model the water quality parameters such as temperature, pH and disinfectant residual throughout a distribution system (Rossman, 1994). However, it is unable to predict the bacterial regrowth and corresponding depletion of assimilable organic carbon (AOC) in the bulk water. Previous researchers using EPANET to predict transport of bacterial cells in a distribution system found that further development of EPANET is needed to incorporate the complex processes occurring within a distribution system.

Currently available water quality models (steady state model, SANCHO and BAM) are too complex to be coupled with hydraulic models (Gagnon et al., 1997). Identifying important processes and parameters through sensitivity analysis could reduce the complexity of a water quality model. Also, experiments are needed to quantify processes and parameters involved in a drinking water distribution system. This paper describes a basic water quality model that has been constructed through analyses of results obtained from extensive but carefully designed experiments. Such a model is sufficiently simple to be incorporated into a hydraulic model without a major increase in computing time.

## 2. EXPERIMENTAL

Two sets of experiments using a newly developed type of biofilm reactor were conducted in this study. One was to study nutrient limitation in tap water and the other concerned the effect of disinfectants (chlorine and chloramine) on drinking water. The biofilm reactors consisted of a stationary outer cylinder and a rotating inner cylinder, which provided both mixing and shear forces. Biofilms were formed on the surface of polycarbonate slide sections (17.5 mm x 67 mm) placed in the inner cylinder, when the reactors were fed with test waters, at a flow rate of 3 mL min<sup>-1</sup>. Reactors were run on each water type for specified days and biofilm and water phase samples were taken at regular intervals.

Aqueous phase samples from the inlet and outlet of each reactor were analysed for total organic carbon (TOC), bacterial cell counts, and total chlorine or chloramine concentration. Biofilms were sampled by removing slides from each reactor during sampling time. The slides were used to determine both total protein and total carbohydrate, and the total organic carbon.

Eight biofilm reactors were used to study the nutrient limitation in normal tap water under four different conditions in duplicate. Two reactors were fed with tap water as a control. Other pairs of reactors were fed with tap water that was supplemented with 100, 200 or 400 µg acetate carbon per litre. The tap water was chloraminated, and during the study over 120 days the average inlet chloramine concentration was 0.71 mg L<sup>-1</sup> as total chlorine.

In the second set of experiments, four biofilm reactors were used to study the effect of chlorine and chloramine at two different conditions. Tap water from a domestic line at the service reservoir situated at Emu Plains (New South Wales, Australia) was used in this study. The tap water was chlorinated, and during the study over 90 days the average inlet chlorine concentration was 0.58 mg L<sup>-1</sup> as total chlorine. Two reactors were fed with the tap water (chlorinated) and another two reactors were fed with water from the same tap, except that it was dosed with ammonium chloride to produce chloramine. On day 45, one of the chlorinated and chloraminated reactors were supplemented with 400 µg acetate carbon per litre, after which all the reactors were run for another 45 days.

### 3. MODELLING

Modelling bacterial growth and disinfectant decay in drinking water distribution systems is a challenging task due to the number of influential processes operating. Incorporating all the processes in a model would result in excellent predictions of bacterial growth and disinfectant decay, at the expense of prolonged computing time. On the other hand, a model including only a few processes may not be able to represent the system adequately. Thus, it is necessary to incorporate at least the fundamental processes that govern the water quality in the distribution system. In this paper, an attempt has been made to incorporate such processes and/or the processes that can be measured easily. Such a model can be incorporated into a hydraulic model to represent a distribution system, both in terms of its hydraulics and microbial water quality. Figure 1 shows the processes that are incorporated in the model presented in this paper.

### 3.1. Bacterial Attachment

Bacterial attachment onto surfaces and detachment (sloughing) of biofilm from surfaces are important processes governing microbial quality of drinking water supplied through distribution systems. Bacteria are transported to a surface where they may become attached. The mechanisms involved in bacterial transport are sedimentation, chemotaxis, Brownian motion and fluid dynamic forces (Savage and Fletcher, 1985). Cell surface hydrophobicity also contributes to bacterial transport. Once the bacteria are transported near to surfaces they attach to the surfaces either reversibly or irreversibly. Long range forces such as van der Waals and electric double-layer forces are responsible for reversible attachment (or adhesion). Irreversible adhesion is caused due to polymer bridging and short-range forces such as Born repulsion, hydration, chemical bonds, dipole interaction and hydrophobic bonding.

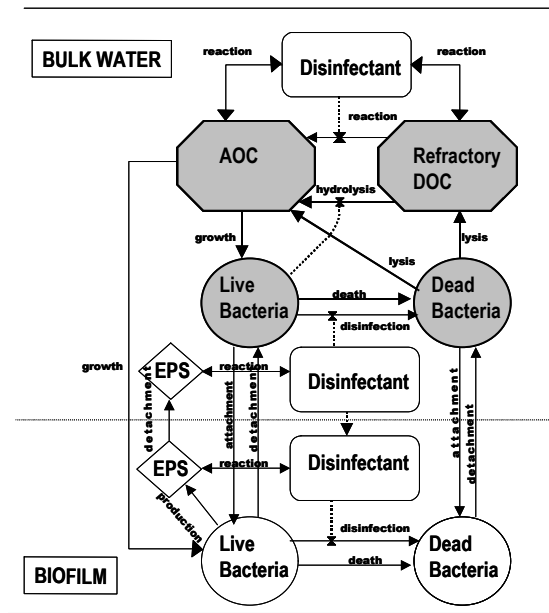
At present, the bacterial attachment onto, and detachment from, surfaces is modelled using Langmuir adsorption theory. In this approach, the adsorption of bacteria onto a surface is considered to depend on the concentration of bacteria in the bulk water and the availability of space on the surface. Thus the rate of attachment of live and dead bacterial cells can be given by:

$$(dX_L/dt)_{\text{attachment}} = k_{aX} \cdot X_L \cdot (1 - X_{Lb}/X_{\text{sat}}) \quad (1.1)$$

$$(dX_D/dt)_{\text{attachment}} = k_{aX} \cdot X_D \cdot (1 - X_{Db}/X_{\text{sat}}) \quad (1.2)$$

where,  $X_L$  and  $X_D$  are volumetric concentrations of live and dead bacterial cells in the bulk water, respectively;  $X_{Lb}$  and  $X_{Db}$  are surface concentration of live and dead bacterial cells in the biofilm, respectively;  $k_{aX}$  is the bacterial attachment coefficient, and  $X_{\text{sat}}$  is the saturation coefficient of cells in the biofilm.

The bacterial attachment coefficient ( $k_{aX}$ ) is assumed to be same for both live and dead bacterial cells and a value 0.00082 day<sup>-1</sup> was used in this study. The saturation coefficient ( $X_{\text{sat}}$ ) determines the maximum bacterial surface density in the biofilm. Thus, the value of  $X_{\text{sat}}$  is quantified from experimental results obtained for the maximum bacterial number in the biofilm and a value of  $4 \times 10^{10}$  cells/m<sup>2</sup> was used in this study.



**Figure 1** Interaction among different elements through various processes in a drinking water distribution system

### 3.2. Bacterial Detachment

Several models have been proposed in the literature for the detachment of biomass (Peyton and Characklis, 1993). However, in this study, only the bacterial detachment was considered. It has been reported that the bacterial coverage onto the inner surface of distribution pipes is only around 8-10%. Consequently, a complex model of detachment needs justification. In this study, live and dead bacterial cell detachment from biofilm to bulk water is assumed to be proportional to the surface concentration of bacterial cells. Since constant shear stress was applied on the biofilm in this study, the effect of shear stress was assumed to have no significant influence on detachment (Peyton and Characklis, 1993). Thus, the rate of detachment of live and dead bacterial cells can be given by:

$$(dX_{Lb}/dt)_{\text{detachment}} = k_{dX} \cdot X_{Lb} \quad (2.1)$$

$$(dX_{Db}/dt)_{\text{detachment}} = k_{dX} \cdot X_{Db} \quad (2.2)$$

where  $k_{dX}$  is the detachment coefficient, which was assumed to be the same for both live and dead cells. In this study, a value of  $0.00097 \text{ day}^{-1}$  has been assigned to  $k_{dX}$ .

### 3.3. Bacterial Growth

Bacterial growth in the bulk water and in the biofilm is modelled with modified Monod type growth to incorporate the influence of the disinfectant. Rate of growth of bacterial cells in bulk water and biofilm can be expressed as:

$$(dX_L/dt)_{\text{growth}} = \mu_{\text{max}} \cdot \text{minimum} [S/(S+K_S)] \cdot X_L \cdot 1/(1+k_{gCl} \cdot Cl) \quad (3.1)$$

$$(dX_{Lb}/dt)_{\text{growth}} = \mu_{\text{max-bio}} \cdot \text{minimum} [S/(S+K_S)] \cdot X_{Lb} \cdot 1/(1+k_{gCl} \cdot Cl) \quad (3.2)$$

where  $S$  denotes the substrates such as AOC, nitrogen and phosphorus,  $K_S$  is the half-saturation constant of those substrates,  $\mu_{\text{max}}$  and  $\mu_{\text{max-bio}}$  are maximum growth rates of bacteria in bulk water and biofilm, respectively,  $Cl$  is the concentration of disinfectant (either chlorine or chloramine) and  $k_{gCl}$  is the kinetic constant related to the inhibition of bacterial growth due to disinfection.

The maximum growth rate for bulk bacteria was obtained from a previous study (Jegatheesan et al., submitted) for the waters used in this study. A value of  $0.528 \text{ d}^{-1}$  was used for  $\mu_{\text{max}}$  for bulk water bacteria. Since the growth rate in the biofilm is much smaller than in bulk water, a value of  $0.0528 \text{ d}^{-1}$  was assigned to  $\mu_{\text{maxb}}$  (for biofilm bacteria) which agrees well with previously obtained values of  $\mu_{\text{maxb}}$  (van der Wende and Characklis, 1990).  $K_{\text{AOC}}$  was taken as  $0.1 \text{ mgL}^{-1}$ . The classical stoichiometric ratio of 100:10:1 was assumed for the utilisation of AOC, nitrogen and phosphorus. The bacterial cell yield,  $Y$  is defined as:

$$Y = \frac{\text{Growth of bacterial mass (mg/L)}}{\text{Amount of substrate utilized (mg C/L)}} \quad (3.3)$$

A yield of 0.2 was used in this study, based on the experimental results from previous work (Jegatheesan et al., submitted). Further,  $k_{gCl}$  was taken as  $1.5 \text{ m}^3 \text{ g}^{-1}$ .

### 3.4. Bacterial Mortality

Generally, bacterial mortality is considered to be proportional to the concentration of live cells (Bois et al., 1997). However, mortality of bacteria also seems to be linked to low or exhausted nutrients. To represent that situation, first order die-off rate has been combined with nutrient status. Thus, the rate of mortality of live cells in bulk water and in the biofilm can be expressed as follows:

$$(dX_L/dt)_{\text{mortality}} = \{k_m + k_{mCl} \cdot [Cl]\} \cdot X_L \cdot \exp[-k_{m\text{AOC}} \cdot \text{AOC}/(K_{m\text{AOC}} + \text{AOC})] \quad (4.1)$$

$$(dX_{Lb}/dt)_{\text{mortality}} = \{k_m + k_{mCl} \cdot [Cl]\} \cdot X_{Lb} \cdot \exp[-k_{m\text{AOC}} \cdot \text{AOC}/(K_{m\text{AOC}} + \text{AOC})] \quad (4.2)$$

where  $k_m$ ,  $k_{mCl}$ ,  $K_{m\text{AOC}}$  and  $k_{m\text{AOC}}$  are kinetic constants characterising the bacterial die-off. A value of  $0.04 \text{ /day}$  was assigned to  $k_m$  which was obtained from our previous work (Jegatheesan et al., submitted). The factor  $k_{mCl}$  is generally four

times less than  $k_m$  (Bois et al., 1991). Values for  $K_{mAOC}$  ( $0.5 \text{ mgL}^{-1}$ ) and  $k_{mAOC}$  ( $1.0$ ) were also obtained from the same work.

### 3.5. Bacterial Lysis

The lysis of dead bacterial cells was assumed to be proportional to the increase in the number of dead bacterial cells from  $t = 0$ . The rate of lysis of dead cells in the bulk water and in the biofilm can be expressed as:

$$(dX_D/dt)_{\text{lysis}} = k_l \cdot (X_D - X_{D0}) \quad (5.1)$$

$$(dX_{Db}/dt)_{\text{lysis}} = k_l \cdot (X_{Db} - X_{Dbo}) \quad (5.2)$$

Where,  $k_l$  is the kinetic constant characterising lysis and  $X_{D0}$  and  $X_{Dbo}$  are the concentration of dead bacterial cells in the bulk water and in the biofilm at  $t = 0$ . From our previous study (Jegatheesan et al., submitted),  $k_l$  was found to be around  $0.01 \text{ d}^{-1}$ . The increment of nutrients due to lysis was assumed to be proportional to the number of dead cells that were lysing. The proportionality constant can be different for different nutrients. However, the contribution of lysed cells to nutrient source is ignored in this study as it was small compared to the nutrients in the feed water.

### 3.6. Production of Carbohydrate and Protein

Generally, production of extra-cellular polymeric substances (EPS) by bacteria is assumed to depend on two kinetic parameters (Characklis and Marshall, 1990). The first is a coefficient associated with bacterial growth and while the second coefficient is non-growth associated. In this study, biofilm carbohydrate and protein are considered to represent EPS and their production in the biofilm is assumed to be a function of bacterial cell concentration in the biofilm and the AOC concentration in the bulk water. The production of carbohydrate and protein can be inhibited by the presence of disinfectant as well. In addition, there can be detachment of carbohydrate and protein from the biofilm to the bulk water. Thus, the production rates of carbohydrate and protein are expressed as:

$$dX_c/dt = k_{1c} \cdot X_{Lb} \cdot \text{AOC}/(k_{2c} + k_{3c} \cdot \text{Cl}) - k_{dc} \cdot X_c \quad (6.1)$$

$$dX_p/dt = k_{1p} \cdot X_{Lb} \cdot \text{AOC}/(k_{2p} + k_{3p} \cdot \text{Cl}) - k_{dp} \cdot X_p \quad (6.2)$$

where the subscripts c and p denote carbohydrate and protein respectively,  $k_1$ ,  $k_2$  and  $k_3$  are kinetic constants,  $k_{dc}$  and  $k_{dp}$  are detachment coefficients and  $X_c$  and  $X_p$  denote the concentration of carbohydrate and protein, respectively. The yield of carbohydrate ( $Y_c$ ) and protein ( $Y_p$ ) are defined similar to the cell yield given by equation (3.3).

### 3.7. Hydrolysis of Refractory Carbon

The refractory fraction of BDOC ( $C_{\text{refract}}$ ) will be hydrolysed by bacteria to form AOC, so the rate of hydrolysis is assumed to be proportional to  $X_L$ . The hydrolysis process can then be expressed as follows:

$$(d\text{AOC}/dt)_{\text{hydrolysis}} = k_h \cdot X_L \quad (7)$$

where  $k_h$  is the proportionality constant. From a previous work, a value of  $1 \times 10^{-12} \text{ g.cell}^{-1} \cdot \text{d}^{-1}$  (Jegatheesan et al., submitted).

### 3.8. Disinfectant Decay

Thorough understanding of disinfectant decay in bulk water has resulted from associated work (Kastl et al., 1999). In this study, first-order disinfectant decay in the bulk water was assumed, as the fast reacting organic carbon would have reacted with the disinfectant before the water entered the distribution system. Combined first-order disinfectant decay due to carbohydrate and protein was considered to occur in the biofilm. An additional first order decay component was included to describe the residual surface effect on disinfectant decay. Thus the disinfectant decay is expressed as:

$$d\text{Cl}/dt = -[k_{\text{Cl}} + (A/V)(k_{\text{Clp}} \cdot X_p + k_{\text{Clc}} \cdot X_c)] \cdot \text{Cl} - k_{\text{ClS}} \cdot \text{Cl} \quad (8)$$

Where,  $k_{\text{Cl}}$ ,  $k_{\text{ClS}}$ ,  $k_{\text{Clp}}$  and  $k_{\text{Clc}}$  are decay constants,  $A$  is the surface area of the biofilm and  $V$  is the corresponding bulk water volume,  $X_c$  and  $X_p$  denote the concentrations of carbohydrate and protein, respectively.

### 3.9. General Mass Balance

The mass balance for live and dead bacteria in the bulk water as well as in the biofilm are given by the following expressions:

In bulk water:

$$dX_L/dt = (dX_L/dt)_{\text{growth}} - (dX_L/dt)_{\text{attachment}} + (dX_{Lb}/dt)_{\text{detachment}} - (dX_L/dt)_{\text{mortality}} \quad (9.1)$$

$$dX_D/dt = (dX_L/dt)_{\text{mortality}} - (dX_D/dt)_{\text{attachment}} + (dX_{Db}/dt)_{\text{detachment}} - (dX_D/dt)_{\text{lysis}} \quad (9.2)$$

In biofilm:

$$dX_{Lb}/dt = (dX_{Lb}/dt)_{\text{growth}} + (dX_L/dt)_{\text{attachment}} - (dX_{Lb}/dt)_{\text{detachment}} - (dX_{Lb}/dt)_{\text{mortality}} \quad (10.1)$$

$$dX_{Db}/dt = (dX_{Lb}/dt)_{\text{mortality}} + (dX_p/dt)_{\text{attachment}} - (dX_{Db}/dt)_{\text{detachment}} - (dX_{Db}/dt)_{\text{lysis}} \quad (10.2)$$

The mass balance for AOC can be given by the following expression:

$$d(\text{AOC})/dt = (d\text{AOC}/dt)_{\text{hydrolysis}} - Y \cdot (dX_L/dt)_{\text{growth}} - Y \cdot (dX_{Lb}/dt)_{\text{growth}} \quad (11)$$

### 3.10. Model Construction

The model was constructed within the AQUASIM software package by introducing all the processes described above. AQUASIM contains a dynamic equation solver, which can perform parameter estimation to find the best fit of the model output to the experimental data (Reichert, 1998). In this study, the model output from the AQUASIM software was fitted to experimental measurements of concentration of chlorine or chloramine, and surface density of carbohydrate and protein in the biofilm. The weighted error between experimental data and model ( $\chi^2$ ) can be used as a criterion of goodness of fit of a model to the experimental data. The weighted error,  $\chi^2$  is defined as:

$$c^2(p) = \sum_{i=1}^n \left[ \frac{f_{meas,i} - f_i(p)}{s_{meas,i}} \right]^2 \quad (12)$$

where  $f_{meas,i}$  is the  $i$ th measured value,  $f_i(p)$  is the calculated value from the model, using parameter values  $p$  and  $s_{meas,i}$  is the estimated standard deviation of  $f_{meas,i}$ . The kinetic parameters that were adjusted during the fitting are listed in Table 1.

## 4. RESULTS AND DISCUSSION

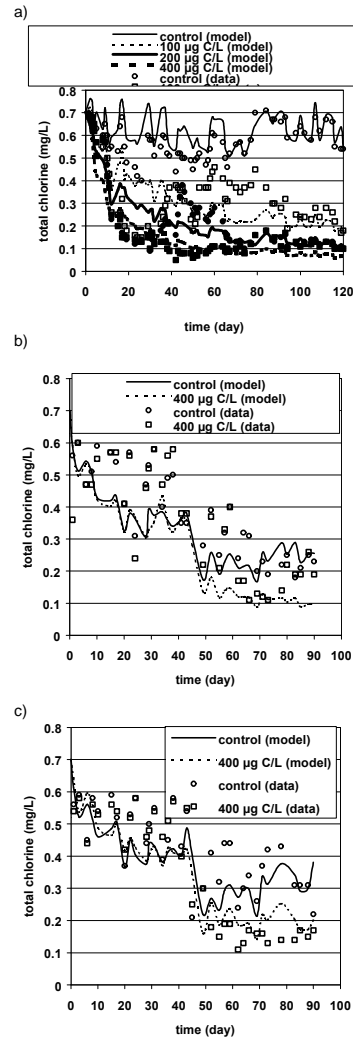
### 4.1. Disinfectant Decay

Figures 2(a) to 2(c) show the disinfectant decay in biofilm reactors for different experimental conditions. The decay of chloramine increases as the concentration of acetate carbon increases. The tap water used in the nutrient limitation study had an AOC level of about  $100 \mu\text{g L}^{-1}$ . Thus, if the supplemented acetate is considered to be AOC,  $200 \mu\text{g AOC L}^{-1}$  was found to produce significant decay of chloramine (see fig.2(a)). Further increase in AOC up to  $500 \mu\text{g L}^{-1}$  increased the decay of chloramine marginally. This trend was observed in the experiments conducted with Emu Plains water as well, which generally had an AOC level of  $250 \mu\text{g L}^{-1}$  (see fig.2(b) and 2(c)).

### 4.2. Production of Protein in the biofilm

Production of protein also increases as the concentration of acetate carbon increases. In the nutrient limitation study, a protein production level of  $4.5 \mu\text{g cm}^{-2}$  was observed in the biofilm after 120 days, when  $400 \mu\text{g acetate carbon.L}^{-1}$  was supplemented. But this trend was not repeated with the Emu Plains water. In the chloraminated case, a protein density of  $3.47 \mu\text{g cm}^{-2}$  in the biofilm was achieved when  $400 \mu\text{g acetate carbon.L}^{-1}$  was supplemented for 45 days. The chlorinated Emu Plains water had a protein surface density of  $7.26 \mu\text{g cm}^{-2}$  in the biofilm when  $400 \mu\text{g acetate carbon.L}^{-1}$  was supplemented for 45 days. This may be due to

the difference in the quantities of carbon sources in those waters. The process proposed in this study for the production of protein could model these observations within the experimental error.



**Figure 2** Disinfectant decay in

- nutrients limitation experiments
- Emu Plains – Chloraminated
- Emu plains - chlorinated

From Table (1), it can be seen that the synthesis of protein ( $Y_p$ ) and detachment of protein from the biofilm to bulkwater ( $k_{dp}$ ) are higher for the experiments conducted with Emu Plains water compared to that of the laboratory tap water.

### 4.3. Production of Carbohydrate in the biofilm

When the production of carbohydrate in biofilm for different experimental conditions was analysed, the production of carbohydrate increased as the concentration of acetate carbon

increased. From Table (1), it can be seen that the synthesis of carbohydrate ( $Y_c$ ) is similar in all experiments. However, the detachment of carbohydrate from the biofilm to bulkwater ( $k_{dc}$ ) is higher for the experiments conducted with Emu Plains water compared to that of the laboratory tap water. Also, the chlorinated Emu Plains water showed more detachment compared to that of chloraminated Emu Plains water.

**Table 1** Kinetic coefficients derived from the model

Process	Kinetic Coeff.	Nutrient limitation Experiments	Emu Plains Chloramine	Emu Plains Chlorine
Disinfectant	$k_{Cl}$	0.143	0.153	0.120
Decay	$k_{Clp}$	0.711	0.740	0.340
	$k_{Clc}$	25.85	53.39	48.22
	$k_{ClS}$	0.079	0.246	0.290
	$k_{1c}$	$6.48 \times 10^{-7}$	$6.30 \times 10^{-7}$	$9.75 \times 10^{-7}$
Production of Carbohydrate	$k_{2c}$	2.95	27.30	11.26
	$k_{3c}$	62.48	0.055	0.749
	$k_{dc}$	0.31	1.43	10.00
	$Y_c$	0.008	0.002	0.001
	$k_{1p}$	$2.11 \times 10^{-7}$	$5.81 \times 10^{-8}$	$8.74 \times 10^{-7}$
Production of Protein	$k_{2p}$	47.39	8.54	67.92
	$k_{3p}$	55.83	41.50	20.87
	$k_{dp}$	1.64	9.92	9.94
	$Y_p$	0.9	3.6	4.0

$k_{Cl}$  ( $\text{day}^{-1}$ );  $k_{Clp}$  ( $\text{m}^3 \text{g}^{-1} \text{day}^{-1}$ );  $k_{Clc}$  ( $\text{m}^3 \text{g}^{-1} \text{day}^{-1}$ );  $k_{ClS}$  ( $\text{mgL}^{-1} \text{day}^{-1}$ );  $k_{1c}$  ( $\text{m}^3 \text{day}^{-1}$ );  $k_{2c}$ -non dimensional;  $k_{3c}$  ( $\text{m}^3 \text{g}^{-1}$ );  $k_{dc}$  ( $\text{gm}^{-2} \text{d}^{-1}$ );  $Y_c$  (gcarbohydrate. gAOC $^{-1}$ );  $k_{1p}$  ( $\text{m}^3 \text{day}^{-1}$ );  $k_{2p}$ -non dimensional;  $k_{3p}$  ( $\text{m}^3 \text{g}^{-1}$ );  $k_{dp}$  ( $\text{gm}^{-2} \text{d}^{-1}$ );  $Y_p$  (gprotein. gAOC $^{-1}$ )

## 5. CONCLUSIONS

A water quality model to run along with a hydraulic model for drinking water distribution system has been developed in this study. Extensive experimental studies conducted with biofilm reactors to examine effects of nutrient levels and disinfectant type were used in the development of the model. The model is able to predict the disinfectant decay due to organic matter in bulk water, as well as biofilm, and the growth of biofilm in terms of carbohydrate and protein. The accurate prediction of disinfectant decay in the distribution system is important to maintain the water quality in the distribution system. While this model is complex enough to describe the microbiological water quality changes in a distribution system, it is also simple enough to be incorporated into a hydraulic network model of the system to test alternative management strategies such as limiting the input of nutrients from treated water, optimising the disinfectant regime for control of biofilm activities, adopting regimes for rapid water movement through the system and optimising the type and timing of mains cleaning, to ultimately

improve drinking water quality. Incorporation of this model into a network model (EPANET) is underway through a project funded by Cooperative Research Centre for Water Quality and Treatment (CRCWQT) and Sydney Water Corporation.

## 6. ACKNOWLEDGMENTS

The authors acknowledge the partial funding provided by the CRCWQT for this work under the Program 4 entitled "Maintaining Water Quality in Distribution Systems".

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