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A UNIFIED APPROACH TO MODELLING URBAN STORMWATER TREATMENT

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ABSTRACT

The mechanisms promoted in the removal of stormwater pollutants encompass physical, chemical and biological processes. Owing to the intermittent nature of stormwater inflow, physical processes associated with detention for sedimentation and filtration (either through vegetated systems or through an infiltration medium) are the principal mechanisms by which stormwater contaminants are first intercepted. Subsequent chemical and biological processes can influence the transformation of these contaminants. In this paper, it is asserted by the authors that the various stormwater treatment measures by which contaminants are first intercepted and detained can be described using a unified model. Grass swales, wetlands, ponds and infiltration systems are considered to be a single continuum of treatment based around flow attenuation and detention, and particle sedimentation and filtration. Hydraulic loading, vegetation density and areal coverage, hydraulic efficiency and the characteristics of the target pollutants (eg. particle size distribution and contaminant speciation) largely influence their differences in performance. In this context, infiltration systems are simply vertical filtration systems compared to the horizontal filtration systems of grass swales and wetlands, reliant on enhanced sedimentation and surface adhesion (promoted by biofilm growth) for removal of fine particles.

The validity of this unified conceptual approach to simulating the operation of stormwater treatment measures is demonstrated by empirical analysis of observed water quality (predominantly TSS) improvements in swales, wetlands, ponds and infiltration basins and also by fitting observed water quality data from these treatment systems to a unified stormwater treatment model (USTM) developed by the authors. The USTM provides an efficient mechanism by which urban catchment and waterway managers can predict and assess the performance of stormwater treatment measures.

KEYWORDS

Stormwater, pollutants, treatment, infiltration, wetlands, swales

1. INTRODUCTION

Increasingly over recent years, initiatives to protect the aquatic environment of urban areas have been a focus of many federal, state and local government organisations and community groups. Many of these initiatives have successfully reduced point sources such as sewage discharge and industrial effluent. Urban stormwater and its role in conveying pollutants to our urban waterways is now widely recognised as the next major issue to tackle. However, the sources of urban pollutants are diffuse and inherently more difficult to manage. The nature of pollutants emanating from different landuses is different and, as a consequence, the appropriate treatment techniques for improving the resulting stormwater quality will vary, and may involve several treatment measures. These treatment measures are often used in series or in parallel in an integrated treatment sequence to improve the overall performance of the treatment system, leading to a sustainable strategy which can overcome site factors that limit the effectiveness of any single measure.

In order to prioritise the implementation of stormwater treatment measures, urban waterway managers need to be able to predict and assess their performance, both singly and in combination. This paper presents a unified approach to predicting the performance of a range of stormwater treatment measures, gives examples of its application, and outlines future development to refine the approach.

2. STORMWATER TREATMENT PROCESSES

The mechanisms promoted in the removal of stormwater pollutants encompass physical, chemical and biological processes. Owing to the intermittent nature of stormwater inflow, physical processes associated with detention for sedimentation and filtration (either through vegetated systems or through an infiltration medium) are the principal mechanisms by which stormwater contaminants are first intercepted. Subsequent chemical and biological processes can influence the transformation of these contaminants.

In this paper, it is asserted by the authors that the various stormwater treatment measures by which contaminants are first intercepted and detained can be described using a unified model. Grass swales, wetlands, ponds and infiltration systems are considered to be a single continuum of treatment based around flow attenuation and detention, and particle sedimentation and filtration. Grass swales are simply ephemeral vegetated systems operating at a higher hydraulic loading than constructed wetlands. Constructed wetlands are simply shallow densely vegetated systems compared to ponds which are characterised by deeper open water and fringing vegetation. Hydraulic loading, vegetation density and areal coverage, hydraulic efficiency and the characteristics of the target pollutants (eg. particle size distribution and contaminant speciation) largely influence their differences in performance. In this context, infiltration systems are simply vertical filtration systems compared to the horizontal filtration systems of grass swales and wetlands, reliant on enhanced sedimentation and surface adhesion (promoted by biofilm growth) for removal of fine particles.

The validity of this unified conceptual approach to simulating the operation of stormwater treatment measures is demonstrated by empirical analysis of observed water quality (predominantly TSS) improvements in swales, wetlands, ponds and infiltration basins and also by fitting observed water quality data from these treatment systems to a unified stormwater treatment model (USTM) developed by the authors. The USTM provides an efficient mechanism by which urban catchment and waterway managers can predict and assess the performance of stormwater treatment measures.

3. MODELLING POLLUTANT REMOVAL

3.1 THE 1ST ORDER KINETIC MODEL

A simple model commonly adopted in describing the pollutant removal process is a two-parameter first order decay function, which expresses the rate (k) at which pollutant concentration moves towards an equilibrium or background concentration (C*), with distance along the treatment measure, as a linear function of the concentration. The model, known as the "k-C* model", assumes steady and plug flow conditions and is typically expressed as follows:-

$$q\frac{dC}{dx} = -k(C - C^*) \qquad (1)$$

where
- q = hydraulic loading rate (m/y), defined as the ratio of the inflow and the surface area of the system
- x = fraction of distance from inlet to outlet
- C = concentration of the water quality parameter
- C* = background concentration of the water quality parameter
- k = areal decay rate constant (m/y)

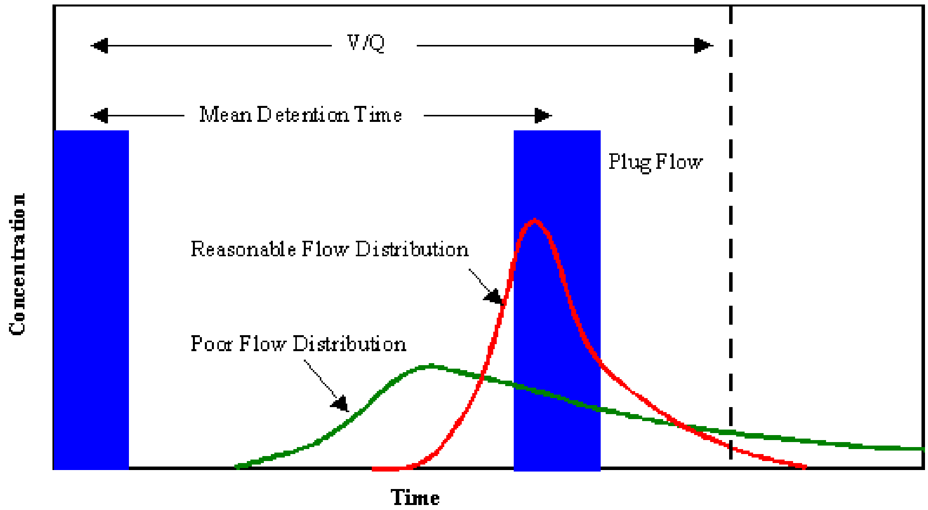
The parameters k and C* are "lumped" parameters representing the combined effects of a number of pollutant removal mechanisms. A high value of k results in a rapid approach to equilibrium, and hence a higher treatment capacity (provided that the background concentration (C*) is less than the inflow concentration). Wong and Geiger (1997) discussed possible impacts of intermittent loading conditions in stormwater wetlands on these parameters compared with typical parameter values applicable to wastewater wetland systems with less variable flow.

3.2 THE CONTINUOUSLY STIRRED TANK REACTOR MODEL

Kadlec and Knight (1996) describe a distribution function of hydraulic residence time, referred to as the Retention Time Distribution (RTD) function, to reflect the degree to which the hydraulic residence time varies. Under plug flow conditions, the concentration-time distribution is simply a spike with a very small standard deviation about the mean residence time as shown in Figure 1. This suggests that all individual parcels of tracer entering the wetland experience a similar period of detention. For fully mixed flow conditions, the concentration-time distribution takes the form of an exponential function, where the effect of flow dilution in steady flow conditions progressively reduces the tracer concentration at the outflow.

Plug or continuously stirred flow conditions never occur in natural systems and the concentration-time distribution of natural wetland systems lies somewhere in between the distributions of plug flow and fully mixed flow conditions. According to Kadlec and Knight (1996), flow hydrodynamics within a wetland system may be modelled as a combination of plug flow (ie. a time delay before tracer outflow is observed) and a number of

Figure 1. Illustration of Tracer Concentration-Time Distribution



continuously stirred tanks reactors (CSTRs). A single CSTR will result in a pollutant hydraulic residence time distribution represented by an exponential function. As the number of CSTRs in series approaches infinity, the residence time distribution approaches that of plug flow. The higher the number of CSTRs, therefore, the higher the hydraulic efficiency. The concentration-time distribution takes the form of a positively skewed distribution function with the tail of the distribution extending as flow conditions for the entire detention system approach fully mixed conditions. The extent to which flow conditions depart from an idealised plug flow condition is reflected in the spread of the distribution function. Generally, an outflow concentration distribution with a large standard deviation suggest the presence of short-circuit flow paths and flow re-circulating zones. In some cases, the combined effect of short-circuit flow paths and re-circulating zones can result in the outflow concentration-time distribution exhibiting multiple peaks, or in other cases in a flat extended peak.

The hydraulic efficiency of ponds and wetlands needs to reflect two basic features in the hydrodynamic performance of a stormwater detention system. The first is the ability to distribute the inflow evenly across the detention system and the second is the amount of mixing or re-circulation, ie. deviations from plug flow. Persson et al. (1999) developed a quantitative measure of the wetland hydrodynamic behaviour to allow a consistent basis for evaluating the *hydraulic efficiency* of wetlands. The measure, *Hydraulic efficiency* (λ), is expressed as follows:-

$$\lambda = \left(\frac{t_{50}}{t_n}\right)^2 \left(\frac{t_{50}}{t_{50} - t_p}\right) \quad \text{or} \quad \lambda = e^2 N \qquad (2)$$

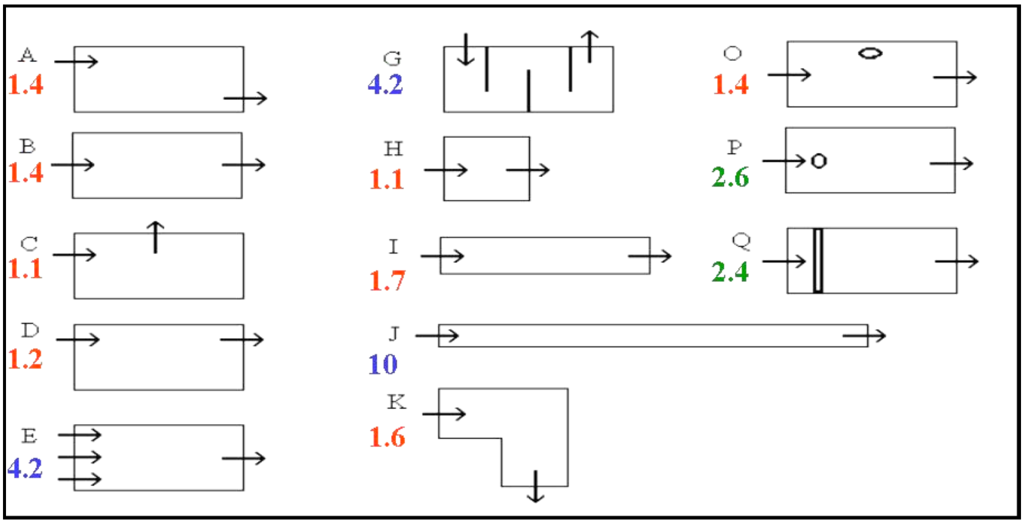
where t_{50} is the time of the 50th percentile of the hydraulic residence time distribution, t_n is the nominal detention period computed as the ratio of the detention volume and the discharge (V/Q), t_p is the time of the peak outflow concentration, and e is the effective volume ratio.

The number of continuously stirred tanks (N) can be approximately related to the hydraulic efficiency of the treatment facility as follows:-

$$\lambda \approx 1 - \frac{1}{N_{CSTR}} \qquad (3)$$

With this measure of *hydraulic efficiency*, it is possible to examine the relative effects of modifications to the shape, inlet and outlet locations, bathymetry and vegetation types, layout and density on the hydrodynamic behaviour of these detention systems, and the appropriate number of continuously stirred tank reactors selected for modelling. This is illustrated in Figure 2, adapted from the results of Persson et al. (1999).

Figure 2. Hydraulic Efficiencies of Ponds and Wetlands, showing the appropriate number of CSTRs (adapted from Persson et al., 1999)



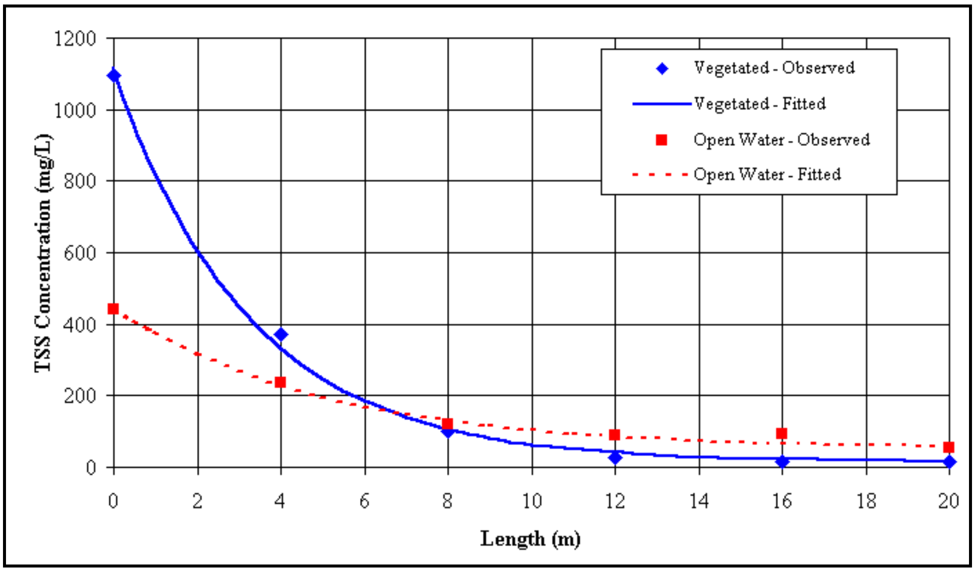
4. APPLICABILITY OF THE 1ST ORDER MODEL

4.1 PONDS AND WETLANDS

Wong et al. (2000) describe field measurements carried out in two parallel channels established in the Hallam Valley stormwater treatment wetland in Melbourne, Australia. Each channel was 3m wide, 20m long, and 250mm deep. One was densely vegetated with *Eleocharis acuta* (Slender spikerush), while the other was open water with all vegetation removed. Under steady flow conditions a high concentration of graded sediment was introduced via a mixing box to the upstream end of the channels.

The resulting TSS concentrations along the two channels are shown in Figure 3, together with eyefit curves of the k-C* form. The fit is very good in each case. Compared with the open water channel, concentrations in the vegetated channel fall more rapidly (i.e. higher k) to a lower background level (i.e. lower C*). The vegetated channel represents a well designed stormwater treatment wetland. The open channel is more like a pond, although shallower than is usually the case. In each case the first order kinetic model appears to be highly appropriate.

Figure 3. TSS Concentrations at Hallam Valley Wetland (after Wong et al., 2000)



4.2 GRASS SWALES

4.2.1 NARROW SWALES

Application of the k-C* model to vegetated swales followed a review of both the approaches used to model swale behaviour, and actual data from experiments testing the performance of swales in field or laboratory conditions.

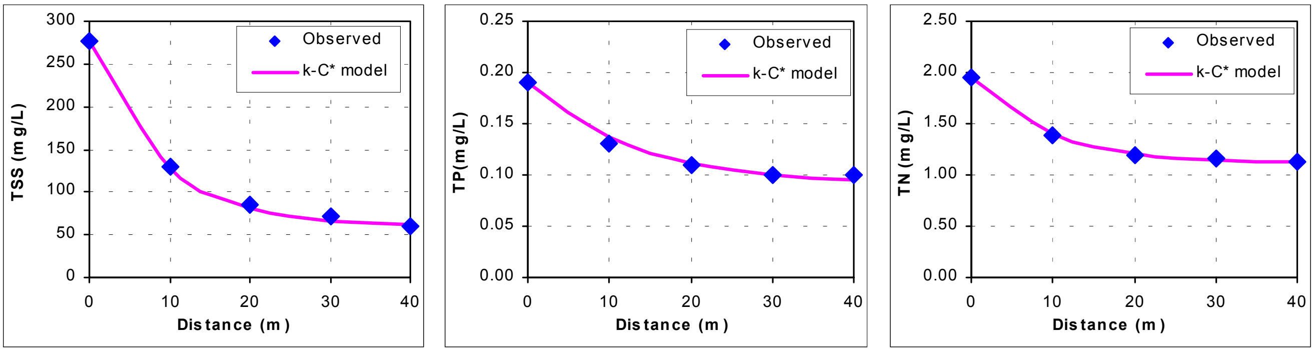
Several approaches have been taken to modelling swale and buffer strip performance (e.g. Barling and Moore, 1993; Dillaha and Inamdar, 1996; Flanagan et al., 1989; Gold and Kellog, 1996; Knisel, 1980), although many have been in non-urban situations. More importantly, many of these approaches require input of detailed site and process variables, which are often not available to urban waterway managers. An appropriate modelling approach must balance the need to understand the processes occurring in swales, with the information available to provide input to the model. Performance data from previous studies were therefore reviewed, to test the applicability of the k-C* model.

There have been a number of studies of the pollutant removal performance of grass swales within an urban environment (e.g. Barrett et al., 1998; Kercher, 1983; Walsh et al., 1997; Yousef et al., 1987). Whilst most provide a useful summary of the *overall performance* of swales, very few have been able to provide the experimental control or quantification of key variables (e.g. pollutant characteristics, hydraulic load, swale dimensions), necessary to develop reliable models from the results.

Researchers at the University of Texas (Barrett et al., 1998; Walsh et al., 1997) undertook both field and laboratory experiments on the performance of grass swales, and the latter provided the necessary data to fit and calibrate the k-C* model. The experiment was undertaken in a 40 x 0.75 m constructed swale, at an average slope of 0.44%, with soil and grass overlying a layer of gravel. A constant-head tank discharged to an initial mixing basin, where known concentrations of pollutants were added. Water quality monitoring was undertaken using dedicated sampling tubes within the swale, and from the downstream discharge weir.

A k-C* model was applied to the results of these experiments. Whilst the results vary between experimental runs, the overall fit between the observed data and the k-C* prediction is encouraging. Three of the best examples (for TSS, TP and TN) are shown in Figure 4. Field experiments are now being undertaken in Australia to further test the application of the k-C* model, and to calibrate the model parameters to local conditions.

Figure 4. Example of k-C* model application to swale performance data from Walsh *et al.* (1997).



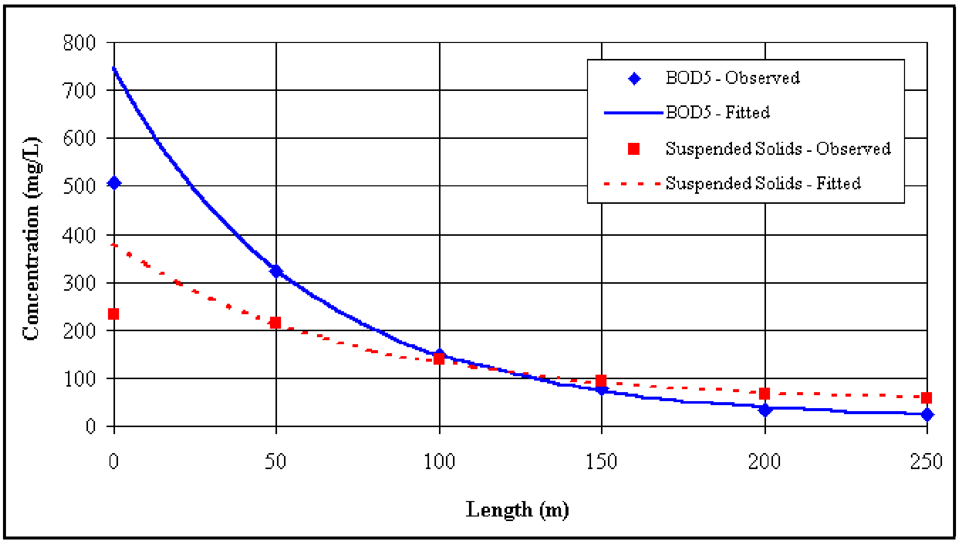
Inadequate data have so far been found to test the applicability of the k-C* model to buffer strips. Whilst many studies of buffer strip behaviour have been undertaken, none of those reviewed to date have provided data sufficient to test this approach. Further work in this area will be undertaken in the next few years.

4.2.1 BROAD SWALES

The Western Treatment Plant at Werribee treats part of Melbourne's sewage by a combination of primary settlement, land filtration, grass filtration, and lagooning. In the grass filtration process, settled sewage flows through irrigation bays planted with appropriate grass species. Bays are typically 10m wide and 300m long, with slopes of 0.1 to 0.4%. They may be viewed as either broad swales or shallow wetlands.

Scott & Fulton (1978) describe a measurement program which took water quality samples from the inlet and at 50 metre intervals in four parallel bays over one winter irrigation season. Measured concentrations of TSS and BOD_5 at each distance, averaged over the four bays, are shown in Figure 5, together with eyefitted curves of the k-C* form. In each case the treatment over the first 50m is less than suggested by the first order decay curve, but for subsequent samples the fit is very good. The initial discrepancy is probably due to turbulence near the inlets, but may also be associated with anaerobic conditions observed near the start of the bays. Scott & Fulton (1978) present results for 19 water quality parameters, and the great majority exhibit behaviour of the form shown.

Figure 5. TSS and BOD_5 Concentrations in Grass Filtration Bays (after Scott & Fulton (1978))

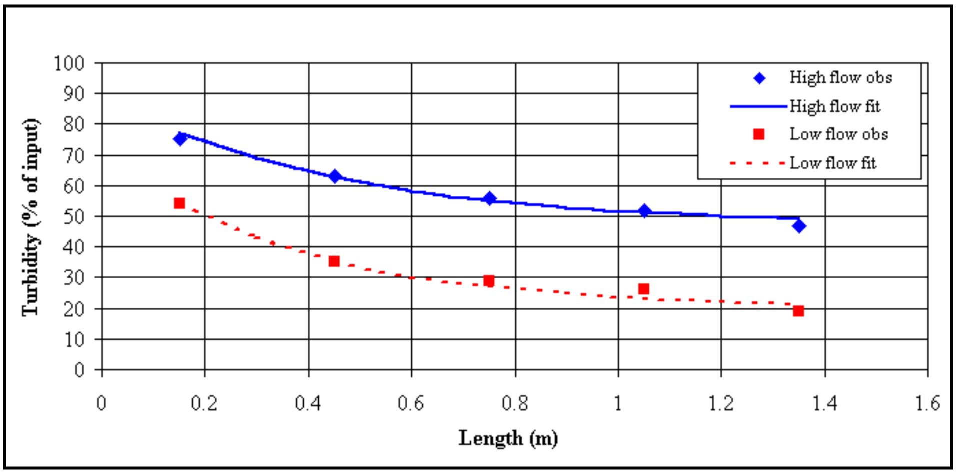


4.3 GRAVEL FILTERS

Sivakumar (1980) describes a program of laboratory measurements of turbidity in a horizontal flow gravel filter. The filter comprises a rectangular box 1.8m long, 400mm wide, and 500mm deep with an overflow set at 450mm depth. The box is filled with gravel ranging from 2 to 12mm in diameter. Tests were carried out for several flow rates, and for both high and low input turbidity. All results are presented in terms of percent removal.

Sivakumar (1980) fitted turbidity *removal* as a power function of flow rate, input turbidity, depth of measurement, and length of filter. But if the results are expressed as output percent (i.e. turbidity not removed), which is more analogous to output concentration, the data can again be closely fitted by a curve of the k-C* form as shown in Figure 6.

Figure 6. Turbidity in a Gravel Filter (after Sivakumar (1980))



A review of the technical literature on sand and gravel filter performance shows that media particle size, and hence surface area, is a highly significant explanatory variable for performance. The larger the surface area the better the performance, and thus the higher value of the parameter k in the 1st order kinetic model. There is obvious analogy here with the effect of vegetation density in a wetland.

5. DISCUSSION

The goodness of fit of the first order kinetic model in these very different situations is striking, particularly for the gravel filter, which on first sight appears to have little in common with the others. Nevertheless, observation shows that there is an underlying unity of behaviour, which suggests in turn an underlying unity of process. At a theoretical level, the nature and extent of this unity requires further investigation. At a practical level, the observed unity of behaviour can be used to develop a model which can be fitted to the various treatment facilities by changing the input conditions – hydraulic loading, background concentrations, and the like – rather than by changing the model structure.

This unified approach provides some real advantages. With only two parameters, it provides a well-defined focus for future research activities. Thus, future research will be aimed at improving our understanding of the variability of k and C*, and how these interact with characteristics of both the catchment (e.g. geology, particle size and settling velocity distributions) and the particular stormwater treatment measure (e.g. hydraulic efficiency and hydraulic loading). Perhaps more importantly, this approach minimises the number of parameters that urban waterway managers will need to calibrate for use in their own catchments.

Utilisation of the USTM approach is based on the premise that the processes by which stormwater pollutants are first intercepted and treated are largely physical. Future research will need to investigate the role of biological processes, in the subsequent transformation and removal of pollutants, particularly those in the soluble form. Similarly, much of the research into the behaviour of pollutants within stormwater treatment facilities has been conducted in event conditions. It is likely that the relative contribution of physical, chemical and biological processes will be different between the event and inter-event period, and refinement of the USTM to reflect these differences is required.

This Unified Stormwater Treatment Model has been developed as part of a broader project, aimed at developing a model for urban stormwater improvement conceptualisation. This broader model will incorporate not only performance of treatment measures, but information on their lifecycle costing. It will also provide for the prediction of ecosystem responses to given stormwater treatment strategies, which is currently an important gap in our understanding.

6. CONCLUSIONS

It is proposed that grass swales, wetlands, ponds, and infiltration systems all form a continuum of treatment based on flow attenuation and detention, and on particle sedimentation and filtration. It is further proposed that the short term water quality treatment behaviour of all these measures can be modelled using a first order kinetic model (or k-C* model). A wide range of experimental data provides strong support for the proposition. Differences in performance between the various treatment measures are accommodated, not by change to the model structure, but by the use of appropriate treatment facility and pollutant characteristics. Treatment facility characteristics include hydraulic loading, hydraulic efficiency, vegetation density and areal coverage, and filter medium surface area. Pollutant characteristics include particle size distribution and contaminant speciation.

The Unified Stormwater Treatment Model provides urban waterway managers with an efficient means of predicting and assessing the performance of stormwater treatment measures, and provides researchers with a focus for continued improvement in our understanding of stormwater treatment mechanisms.

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