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ADDENDUM

p iv line 5: delete "Associated"

p 1-1 line 18: delete "and heavy metals"

p 1-2 line 2: delete "practice" and read "urban planning and design"

p 1-2 line 15: change "perform" to "performs"

p 1-4 line 13: at the end of the sentence cite a reference "(Wong et al., 1999a)"

p 1-5 line 13: delete "surface" and read "bed"

p 1-8 line 11: delete "will be" and read "is"

p 1-9 line 11: delete "will also be" and read "is also"

p 2-6 line 8: after bracket add "and hydraulic properties such as residence time and flow velocity."

p 2-15 line 14: after "to settle" add "or never settle"

p 2-16: add a new subsection 2.2.3 after the subsection 2.2.2:

2.2.3 The biochemical processes that affect sediment behaviour in constructed stormwater wetlands

Prior to pursuing further the physical processes, let us consider the biochemical processes which affect pollutant behaviours and removal efficiency in the soil-water-plant wetlands. When stormwater enters a constructed wetland at a low flow velocity, it allows suspended solids to settle out via physical sedimentation of particles, filtration and coagulation of small, colloidal particles (Wong et al., 1998). Meanwhile, through biochemical reactions with the soil, water and air interfaces, plants can increase the assimilation of pollutants and their biomass production within wetlands (Strecker et al., 1992). The growth, dieback and decomposition of plant biomass create internal storage compartments that may be temporary or permanent in nature. Therefore, in addition to the influent solids, surface flow wetlands have internally generated loads (WERF, 2006).

The generation of sedimentary material in wetlands can happen in a number of ways as illustrated in Figure 2-1 (Kadlec and Wallace, 2009). The processes of plant death, litter fall and attrition in wetlands can result in new solids. This occurs for biota at a number of size scales, ranging from macrophytes down to bacteria. For example, algal debris can be a major producer of suspended solids, and adding pollen and seeds to the water can be the second possibility. Some parts of leaf and stem litter contribute to TSS either via direct
attrition or via microbial decomposition. Generally, the TSS produced within the wetland is organic in character (Kadlec and Knight, 1996) and high concentrations of incoming nutrients can cause more internal generation of TSS (Kadlec and Wallace, 2009).

In addition, some chemical reactions can produce particulate materials under certain circumstances. For example, many ionic species (e.g. metals) dissolve or precipitate in response to changes in the solution chemistry with the wetland environment (Strecker et al., 1992). Similarly, oxyhydroxides of iron and calcium carbonate are formed under aerobic conditions. These and other compounds may undergo dissolution and be recharged into the flow from the sediment bed due to the change of redox and pH in a wetland (Kadlec and Knight, 1996).

![Diagram](image)

**Figure AD - 1: Processes affecting particulate matter removal and generation in free surface flow wetlands** (Source: Kadlec and Wallace, 2009).

Kadlec and Knight (1996) recommended that random chemical and biological events have pronounced effects on effluent concentrations, which are strongly related to internal ecosystem processes. It is no doubt important to recognize the existence of these processes, because they contribute to a background level of TSS in a wetland. However, the solids produced by them are at this stage virtually impossible to predict and quantify.

Settling TSS in the influent, plus those generated within the wetland, can accrete as either movable sediments or consolidated immovable new soils. The build-up of new sediments typically occurs preferentially in the inlet section of the wetland. The removal of suspended moveable solids from the water to the wetland sediment bed is considered as the net result of a set of processes, including sedimentation, re-suspension and generation of transportable solids by wetland biota (Kadlec and Wallace, 2009).

p 2-18 line 11: change the title to “Processes of solid removal to be modelled”

p 2-21 line 19: delete “(US Environmental Protection Agency)”

p 2-21 line 23: delete “does not always be”, read “is not always”
p 2-21 line 28: after “commercial”, add “and public”

p 2-22 line 4: delete “hydrological” and read “transverse and vertical”

p 2-43 line 15: after “re-suspended particles” add “except for very fine ones in wetlands”

p 2-45 line 17: delete “develop”, and read “employ”

p 2-46 line 5: delete the last sentence

p 3-16 line 6: after “glass-fibre filter paper” add “(1.2 µm pore size)”

p 4-8 line 1: after “The tubes” add “at a diameter of 10 mm”

p 5-5 line 5: at the end of the paragraph, cite “(Fletcher et al. 2004)”

p 6-17 line 5: before “deposition” add “net”

p 6-17 line 12: before “Thus” add a sentence “In addition, inevitable internal biochemical process within field wetlands will also result in irreducible background TSS concentration as discussed in Section 2.2.3.”

p 6-18: line 18: delete “Re-suspension due to environmental factors” and read “The overall effects of wash-off, particle re-suspension due to environmental factors, and internal generation of new solids via biochemical processes in wetlands”

p 6-28 line 13: delete “about” and read “above”

p 7-17 line 14: after “due to” add “internal biochemical processes,”

p 10-8 line 11: after “and etc.” add two sentences “Additionally, due to internal biochemical processes in wetlands which generate and circulate solids, biochemical events can have obvious effects on effluent concentrations (Kadlec and Wallace, 2009). These effects also should be combined into the background concentration C* of a wetland.”

p 10-13 line 1: after “a water column” add “, together with internal TSS generation in wetlands.”

p R-2 line 7: “f’ for “of”


Modelling Sediment Behaviour in Constructed Stormwater Wetlands

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Summary

Constructed stormwater wetlands are widely used to improve stormwater quality by means of a range of complex physical, chemical and biological processes. Modelling the sedimentation of suspended solids in stormwater wetlands is essential, since a significant proportion of urban stormwater pollutants are transported in a particulate form. Many models have been developed for sediment removal, but their reliability, robustness and complexity limit practical applications. To overcome this, a new approach has been developed in this study for the long-term continuous prediction of the trapping efficiency of sediment particles, through comprehensive laboratory, field and modelling studies.

More than 80 steady-state experiments were undertaken in four mesocosm wetlands and one non-vegetated pond at the Monash Hydraulics Laboratory, aiming to explore the most important factors controlling sedimentation. These experiments were divided into two different groups: wet weather and dry weather. The laboratory simulations of real wetlands were achieved by similarities in three governing dimensionless numbers associated with sediment deposition, re-suspension and wash-off of fine particles: Particle Fall Number, $N_f$ (a time ratio of a particle travelling in horizontal and vertical directions), Particle Shear Velocity Reynolds Number, $Re^*$, and Turbulent Reynolds Number, $Re_T$. Experimental results covered a range of vegetation densities, hydraulic loading rates, input sediment concentrations and particle size distributions (PSD). TSS concentration and sediment PSD along the mesocosm flumes and the non-vegetated pond were measured. There was an exponential decrease of TSS concentrations along the flume with the decay coefficient being highly dependent on particle size. Both $Re^*$ and $Re_T$ did not significantly influence sediment trapping, and flow-induced particle re-suspension was not significant for typical flow velocities experienced by appropriately designed stormwater wetlands. Vegetation density was also found to be relatively unimportant, but particle size and flow characteristics were observed to influence significantly the trapping efficiency of particles.

Based on statistical analyses of the experimental results, a non-linear regression algorithm was developed for the prediction of the sediment trapping efficiency in constructed
stormwater wetlands. It is a function of $N_f$ for wet weather conditions, and of both $N_f$ and the time ratio $t^*$ (a ratio between the actual time since the last storm event and the mean detention time in the wetland) for dry weather conditions. The proposed model is physically-based, with the key coefficients being independent of flow rate and sediment characteristics.

The field data collected from a monitored wetland was used to test the findings of the laboratory study. The model was modified in terms of an apparent background concentration to account for the effects of wash-off and re-suspension of fine particles caused by environmental factors. Together with a common flow-hydrodynamic model (known as the CSTRs model), it can deal with variable flow conditions in real wetlands. The modelling results were compared with those generated from the widely-used, first-order kinetic decay ($k-C^*$) model. The model was better suited than the $k-C^*$ model to different flow conditions in constructed stormwater wetlands without complex calibration. However, further field studies are required for model verification and future refinements.
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\( M_{Rn} \) Reduction rate in pollutant in \( n \)-th tank (Figure 2-9)

\( m_1, m_2 \) Duplicate results in TSS measurement

\( m \) Mean of the duplicate results in TSS measurement

\( N \) Number of continuously stirred tanks

\( N_f \) Particle Fall Number

\( n \) Number of observations

\( n_t \) Turbulence and short-circuiting parameter

\( O_i \) Observed value

\( O_{mean} \) Mean observed value

\( P_i \) Predicted value

\( PSD_{wet,s} \) Percents of particle fraction \( s \) of total weight based on the wet weather inflow particle size distribution

\( PSD_{dry,s} \) Percents of particle fraction \( s \) of total weight based on the dry weather particle size distribution

\( P_m \) Relative error in TSS measurement (difference between duplicate results)

\( P_{mean} \) Mean predicted value

\( p \) Probability of obtaining values of the test statistic

\( \overline{Q} \) Mean flow rate in the treatment cell over a time step \( \Delta t \)

\( Q_{baseflow} \) Base flow in the wetland

\( Q_{ET} \) Evapotranspiration rate

\( Q_{gwin} \) Groundwater inflow rate

\( Q_{gwout} \) Groundwater outflow rate

\( Q_n \) Inflow rate of a treatment system

\( Q_{out} \) Outflow rate of the \( n \)-th tank (Figure 2-9)

\( Q_p \) Net precipitation rate

\( Q_r \) Tidal inflow (+) or outflow (-) rate

\( q \) Hydraulic loading rate

\( q_{mean} \) Mean hydraulic loading rate over a time step \( \Delta t \)

\( R \) Flow hydraulic radius

\( R^2 \) Coefficient of determination (cross-correlation coefficient)

\( R_b \) Roughness function in Equation 2.25

\( R_e \) Flow Reynolds Numbers

\( R_e^* \) Shear Velocity Reynolds Number

\( R_p \) Particle Reynolds Number

\( R_{eT} \) Turbulent Reynolds Number

\( RMSE \) Root Mean Square Error

\( R_s \) Spacing hydraulic radius

\( r_s \) Spearman correlation coefficients

\( S \) Water storage

\( S_{mean} \) Mean storage volume of the cell over a time step \( \Delta t \)

\( S_k \) Source or sink

\( S_{slope} \) Slope gradient

\( T_{rs} \) Trapping efficiency of particles of the size \( s \)

\( t \) Time or the actual time since the last storm event

\( t^* \) Normalized time ratio parameter (= \( t/t_d \))

\( t_d \) Mean detention time for treatment cell

\( t_{mean} \) Mean detention time over a time step \( \Delta t \)

\( t_m \) Mean detention time
$t_n$  Theoretical or nominal detention time
$t_p$  Actual detention time of the peak concentration through the facility
$t_s$  Settling time for particle $s$
$U^*$  Shear velocity
$U_{\text{cm}}$  Individual uncertainty
$U_{\text{dup}}$  Variations in measuring the same water sample
$U_{\text{filter}}$  Mass of the filter
$U_{\text{mass}}$  Mass of the filters weighed by analytical balance
$U_{\text{rm}}$  Variations in sample (sediment) recovery from the method used
$U_{\text{sample}}$  Mass of the filter and residue
$U_{\text{volume}}$  Volume of sample
$V$  Mean flow velocity
$V_D$  Velocity ratio
$V_{cr}$  Average critical flow velocity at incipient motion of sediments
$V_{cs}$  Particle critical settling velocity
$V_s$  Terminal settling velocity of particle $s$
$V_{\text{ss}}$  Modified settling velocity
$V_{xs}, V_{ys}, V_{zs}$  Flow velocities along $X, Y, Z$ directions
$V_{50}$  Median particle settling velocity
$W$  Width of stormwater treatment system
$W_{ss}$  Distance between two plant stems
$X_s$  Fraction of particles with settling velocity less than $V_{cs}$ (Equation 2.19)
x  Distance of the treatment system
$y$  Fractional distance from inlet to outlet (Equation 2.9)
z  A parameter in Rouse’s equation ($= V_s/k'U^*$)
$ΔS$  Change in storage of the CSTR cell over a time step $Δt$
$ΔX$  Length of a treatment cell in Figures 9-1 and 9-5
$ε$  Re-suspension rate
$λ$  Hydraulic efficiency factor
$μ$  Dynamic viscosity of water
$ν$  Kinematic viscosity of sediment and water mixture
$θ$  Dimensionless weighting factor in the Muskingum method
$ρ_s$  Density of the particles $s$
$ρ_w$  Density of water
$τ_b$  Bottom shear stress calculated by $τ_b = \rho_wU^*^2$
$τ_{cd}$  Critical shear stress for deposition
$τ_{cr}$  Critical shear stress for re-suspension
Chapter 1

Introduction

1.1 Background

Urbanization leads to significant changes in stormwater quantity and quality, with the most evident effect being the increase in the magnitude of stormwater discharge and pollutant concentration, and the consequential negative impact on receiving waters. For example, the increase in the quantity and rate of stormwater runoff (generated from the increased impervious areas and the reduction in catchment’s storage) results in extensive channel erosion and an increased frequency of flooding (Wong et al., 2000). Associated with the higher concentration of human activities in urban areas, stormwater may become highly contaminated. Pollutants typically include suspended solids, heavy metals, nutrients, organics, oxygen-demanding materials, pathogenic bacteria and viruses. Stormwater quality is as varied as the sources and characteristics of the pollutants it carries. The pollutants mainly come from transportation, industrial activities, decaying vegetation, soil erosion, animals, fertilizer and pesticide application, dryfall (atmospheric pollution) and general litter (Nix, 1994). However, the pollutants in stormwater may also originate from leaking sanitary sewers, landfills, poorly operating septic systems, etc. As a result, the receiving watercourses suffer from excessive inputs of nutrients and heavy metals, which may lead to nuisance phytoplankton growth in the upper zone of the water bodies and anoxia in the deeper zone. This results in a decrease in the ecosystem health and a consequent impact on fishery and recreation values (Novotny, 2003). The recent inquiry into Australia’s urban water management commissioned by the parliament of Australia (Commonwealth of Australia, 2002) concluded that stormwater control should become a high priority at both state and local levels, and that it represents one of the major threats to water quality.
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In Australia, stormwater management practices are commonly incorporated within the practice and more commonly referred to as Water Sensitive Urban Design (WSUD). WSUD is the concept of integrated urban planning, urban design, landscape architecture and stormwater management for providing more ecologically sustainable urban environments (Mouritz et al., 2006). Traditionally, urban stormwater management was focused on flood control, arising from storm events for the protection of urban areas (Roesner and Bledsoe, 2001). WSUD manages stormwater as a valuable resource, rather than as a burden (Wong and Breen, 2006). WSUD practices have resulted in innovations beyond the end-of-pipe stormwater treatment approaches, upon their adoption and enforcement in urban developments and retrofits (Mouritz et al., 2006).

To achieve WSUD objectives, the appropriate integration of Best Planning Practices (BPPs) and Best Management Practices (BMPs) is needed (Mouritz et al., 2006). The former is defined as the best planning approach for achieving water resource management objectives in an urban situation (e.g. land use controls and town planning requirements) and the latter refers to the structural and non-structural elements of the design that perform prevention, collection, treatment, conveyance, storage, and reuse functions of a water management scheme (Mouritz et al., 2006). Structural practices may involve using a combination of measures in series or simultaneously, as an integrated stormwater treatment train, incorporating a range of treatment measures such as buffer strips, grassed swales, bioretention systems, detention basins, infiltration basins, constructed wetlands and aquifer recharge. The key design principle requires an integrated combination of systems that match opportunities and constraints at a particular site.

The selection of correct stormwater treatment measures will vary from site to site, after estimating the specific site characteristics and land capability (especially targeted pollutants), and is determined by a number of technical disciplines. Wong (2000) proposed a method for selection of appropriate treatment measures based on the pollutant particle size grading and hydraulic loading rate. This is outlined in Figure 1.1, where the treatment train approach is designed to remove larger material first, before removing medium and then finer material. Following this concept, surface flow wetlands are typically used to remove pollutants ranging from coarse suspended solids to very fine particulates, as well as
dissolved materials. In fact, the use of constructed wetlands and ponds for the treatment of urban stormwater pollution is now a well-established technology, and they are commonly used in WSUD to detain water and reduce the level of pollutants, in order to protect the aquatic environment (Bavor et al., 2001; Wong, 2002). In addition to their primary role in stormwater quality treatment, constructed wetlands and ponds also provide a range of other important benefits to the community, including flood protection, fauna habitat, recreational opportunities, landscape amenity and the opportunity for stormwater harvesting (Dallmer, 2002).

![Figure 1-1: Operating hydraulic loading and target particle size of stormwater treatment measures (Source: Wong, 2000).](image)

### 1.2 Constructed stormwater wetlands

A wetland is a transitional area between terrestrial and aquatic ecosystems, which is either permanently or periodically inundated with shallow water, and supports the growth of aquatic macrophytes (Kadlec and Knight, 1996; Wong and Breen, 2006). Wetlands can be natural or artificial. Constructed wetlands are artificially created ecosystems to mimic the function of natural wetlands containing pond, marsh, and swamp features. As a general rule, a constructed wetland normally consists of an open-water inlet zone, an ephemeral area, vegetation zones and an outlet pool with discharge structures. A typical layout for a constructed wetland is presented in Figure 1.2. The inlet zone is commonly a permanent pool designed for the dissipation of energy, the reduction of flow velocity, the uniform distribution of inflow and the pre-treatment sedimentation of coarse to medium-sized...
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Sediments prior to stormwater discharge into the downstream system. As stormwater quality treatment facilities, wetlands are essentially detention systems with an extended detention depth, to allow the temporary storage (and treatment) of stormwater. Hence, the macrophyte (vegetated) zone contains a minimum of two ‘compartments’, i.e., a permanent pool and an extended detention storage. Areas between the permanent pool water level and extended detention water level are ephemeral, and are subjected to a regular wetting and drying cycle during storm events and between events. The macrophyte zone is relatively shallow (typically with a permanent water level of 20 - 30 cm with 30 - 75 cm of extended detention depth) to allow particle settling and adhesion to vegetation, an important factor in the effective removal of fine particulates and soluble pollutants (Breen et al., 2006). Thus, wetlands are predominantly vegetated systems with limited open-water zones in a marsh-like configuration. Ponds are mainly deep, open bodies, typically with fringing vegetation and submerged macrophytes with very limited water level fluctuations. The main distinction between them is often based on the density of the vegetation cover within the systems.

Figure 1-2: Typical layout of a constructed stormwater wetland.

The function of the riser outlet is to control the water level in the system and the discharge rate of the wetland. Typically, the riser outlet will be designed to maintain a relatively constant detention time (with 72 hours being typical), regardless of the water level (Wong and Breen, 2006). Moreover, stormwater can also bypass the whole system when inflow exceeds the predefined flow, or when the water level reaches a pre-determined level. For
example, a common design is to have the invert of the bypass channel set at the same level as
the obvert of the riser outlet, enabling inflows to bypass the system once the wetland is full,
and allowing the water already in the wetland to be properly treated according to the design
detention time.

Stormwater treatment processes in constructed stormwater wetlands and ponds involve four
primary mechanisms of pollutant removal: sedimentation, filtration, chemical adsorption
and biological uptake. The former two physical processes dominate during storm events and
the latter two chemical processes usually become more dominant during dry periods (in
between storm events). The process of sedimentation involves settling suspended solids and
attached pollutants from the water column down to the wetland bottom. Vegetation may
assist this in a process - termed as 'enhanced sedimentation' - whereby the vegetation surface
acts to intercept sediment in the water column, allowing it to adhere either to the vegetation
surface, or to drop to the wetland surface. This process will assist in the filtration of colloidal
substances from the stormwater (Lloyd et al., 1998). Most soluble pollutants in stormwater
are quickly adsorbed onto suspended particles, and then trapped via sedimentation and
filtration (Lawrence and Breen, 1998). This is particularly the case for many heavy metals
and for phosphorus, but less so for nitrogen (Taylor et al., 2005). Biological uptake
transforms dissolved pollutants into algal, plant and bacterial particulate matter by biofilms
attached to the surfaces of emergent aquatic microphyte and plant roots; these can then be
intercepted via sedimentation or other biological transformations (Kadlec and Knight 1996;
Breen et al., 2006). An efficient surface flow stormwater wetland aims to reduce the level of
total suspended solids (TSS), total phosphorus (TP), total nitrogen (TN) and heavy metals in
stormwater, both within and between individual storm events.

Given their cost and land requirements, it is highly desirable that the performance of
constructed wetlands be predicted reliably. There are clear benefits when appropriate models
are used to predict and assess the performance of the treatment systems. Reliable models of
wetland performance will give a number of important benefits:

- better understanding of the cost-effective stormwater management strategies by
  comparing alternatives;
• optimization of the design and operation methods (e.g. real-time control) to accomplish the best treatment and management of the system; and

• predicting impacts on receiving waters, by simulating the levels of key water quality parameters in stormwater discharges.

The reliability, robustness and complexity (or simplicity) of models are all crucial for their usefulness. It may also be desirable that model parameters be transferable between sites and operating conditions, and therefore should be constant. In particular, modelling sediment trapping efficiency is an important requirement to evaluate and predict the performance of constructed stormwater wetlands and ponds, because a range of key pollutants including phosphorus and heavy metals are largely transported in particulate forms (Pitt & Amy, 1973; Diessner, 1992; Kadlec and Knight, 1996; Sansalone et al., 1998; Wong et al., 1998; Braskerud, 2001; Deletic and Orr, 2003; WERF, 2005).

Many models have been developed for predicting the performance of constructed stormwater wetlands, ranging from simple regression models to extremely complicated deterministic models. Some of these are simple and often of low reliability, while others are very complex with many coefficients which are difficult to calibrate. Commonly used models are the USEPA stormwater management models SWMM and XP-SWMM, the environmental management support system model EMSS, and the event-based model SWITCH. However, as pointed out by Zoppou (2001), all these models suffer from several deficiencies, such as low reliability, extensive input data requirements and difficulties during calibration (e.g. some calibration coefficients are not constant even within the same catchments). Few models have an explicit sediment transport module and many are not designed as operational models (being used instead as research tools). It is important to ensure that future models are useable by stormwater managers, and thus simplicity and robustness become important criteria.

In Australia, the most popular model is the Model for Urban Stormwater Improvement Conceptualization (MUSIC) developed by the CRC for Catchment Hydrology (CRCCH, 2005). This model uses the first-order kinetic decay equation (known as the $k-C^*$ equation) to describe water quality behavior in various forms of stormwater treatment, combined with
the Continuously Stirred Tank Reactors (CSTRs) model to express flow hydrodynamic behavior (Kadlec and Knight, 1996; Wong et al., 2006). The combination of these two methods is used as a ‘unified model’ for the prediction of treatment of a wide range of WSUD systems, for three main pollutants (Total Suspended Solids, Total Phosphorus and Total Nitrogen). Whilst the simplicity of such a model is attractive, the relatively few datasets that have been available to calibrate it have demonstrated that the two parameters - $k$ and $C^*$ - can vary, even at the same site. Since only simple models can be calibrated for most operational systems (Kadlec, 2000), further work is needed to develop a model which maintains this attractive simplicity, but includes robust parameters that remain relatively constant between sites and operating conditions.

Furthermore, it is important to estimate long-term performances of treatment systems during both storm events and the long inter-event (dry) periods to which wetlands are subjected. Modelling flow (e.g., flow velocity and Reynolds Number) and sediment behaviours will differ in dry weather from that in wet weather. Unfortunately, this issue is ineffectively addressed in the research literature, and remains a key gap in the modelling of sediment transport within constructed wetlands. It is therefore necessary to develop a method that considers simultaneously both wet and dry weather conditions, in order to provide a robust prediction of the overall performance of constructed wetlands.

1.3 Aim and scope of the research

This study aims to examine sediment behaviour in constructed stormwater wetlands under both wet and dry weather conditions.

Since broad types of wetlands have been used in water quantity and quality management, this study only focuses on surface flow constructed wetlands (and ponds) for controlling urban stormwater quality. The investigations concentrate on two main physical processes in constructed urban stormwater wetlands and ponds during both wet and dry weather periods: deposition and re-suspension. Unquestionably, the removal of other pollutants associated with stormwater qualities such as nitrogen, phosphorus, heavy metals and pathogens is also very important. However, examinations of these pollutants and other physical, chemical, and
biological treatment processes lie outside the scope of this study.

Modelling sedimentation in stormwater treatment facilities requires the simulation of the two principal processes: sediment behaviour and flow hydrodynamic behaviour, both of which significantly affect the removal of sediment. As flow behaviours in constructed stormwater wetlands have been studied widely in the literature (Kadlec and Knight, 1996; Persson et al., 1999; Some, 1999; CRCCH, 2005), most of this project is focused on sediment behaviour.

1.4 Outline of the thesis

In this project, the investigation of sediment behaviour in constructed stormwater wetlands and ponds, under both wet and dry weather conditions, is undertaken through a series of laboratory experiments and monitoring field studies. A model called as “the $N_f$ model” will be proposed and its results compared with those generated using the previously-developed $k-C^*$ model. Rigorous sensitivity testing of the proposed model and $k-C^*$ model is conducted. The structure of the thesis is presented in Figure 1.3.

The importance of using constructed wetlands in improving stormwater quality has been identified in this chapter (Chapter 1).

Chapter 2 contains an overall review of current literature relevant to the study and discusses the inadequacies of previous modelling work in sediment behavior in constructed stormwater wetlands. The needs and directions for the current study are identified from this review. The last section of the chapter identifies specific aims and objectives of the work, as well as underlying hypotheses.

Chapter 3 depicts the laboratory investigation carried out in three mesocosm stormwater wetlands (vegetated with a well-established canopy at different densities) and one mesocosm open-water pond. After introducing the experimental hypotheses and describing the scaling approach used in the physical modelling of stormwater wetlands and ponds, the details for laboratory installation, procedures and programmes in both wet and dry weather conditions are presented. Results and discussion are also given.
Chapter 4 focuses on the laboratory study in an open-water flume and its main findings. The objective is to further identify and assess the most important processes that impacts on sediment behaviour in stormwater non-vegetated ponds, by achieving similarity for Particle Fall Number $N_f$ and Flow Reynolds Number $R_e$ in order to have the same real systems values.

In Chapter 5, the data collected from mesocosm stormwater wetlands experiments (Chapter 3) are used to test the $k-C^*$ model. In particular, a sensitivity analysis of the $k-C^*$ model is presented in comparison to laboratory data.

Chapter 6 elucidates the development of the $N_f$ model based on the laboratory investigation for wet and dry weather conditions, respectively. The detailed description on the integrated model will also be provided, including the governing equations, model structure, input data requirements and model parameters.

Chapter 7 addresses the field water monitoring project in the Ruffey’s Creek Wetland. The monitoring programme includes water quality sampling and flow management at the inlet and outlet of the wetland, together with the collection of particle size distribution data.

Chapter 8 considers the application of MUSIC to the Ruffey’s Creek monitoring data. The results of sensitivity testing of the $k-C^*$ model are also given.

Chapter 9 presents the testing of the $N_f$ model against field monitoring data. Hydrologic routing for flow through the monitoring wetland obtained by MUSIC is adopted for the $N_f$ model testing.

Chapter 10 discusses the strengths and weaknesses of the laboratory study and monitoring field data. A comparison between the $N_f$ model and the $k-C^*$ model in their modelling approaches is also made.

Chapter 11 draws the conclusions of the thesis and provides a summary of the outcomes of the research undertaken, including their implications for wetland design. Recommendations for further research are outlined.
Figure 1-3: Organization of research program.
Chapter 2

Literature Review

2.1 Introduction

Wetlands have been recognized as important features of landscape and natural resources throughout human history. In the last few decades, constructed wetlands have been used as more cost-effective facilities than advanced wastewater treatment systems to treat municipal and industrial wastewater (Hammer, 1989; Moshiri, 1993; Green and Upton 1994; Breaux et al., 1995; Mokry, 1995; Cooper et al., 1996; Kadlec & Knight, 1996; Wong and Geiger, 1997; Liebig et al., 1999; Ktinzer et al., 2004; Scholz, 2006). They are now widely utilized for control of stormwater to provide the services of water storage, peak-flow attenuation, sediment settling, nutrient cycling and metal sequestration (DeLaney, 1995; Kadlec & Knight, 1996; Reddy, et al., 1999; Somes et al., 1999; Frehmann et al., 2004). In recent years, considerable investigations to explore this relatively new technique for stormwater management have been undertaken around the world. The studies have been driven by increasing concerns regarding receiving water degradation in urbanized areas due to urban stormwater (Vassilios and Rizwan, 1997; Burton and Pitt 2002; Mungasavalli and Viraraghavan, 2006; Scholz, 2006).

The primary focus of this thesis is on modelling sediment behaviour in constructed stormwater wetlands. In this chapter, an overview is presented for the functions of constructed wetlands in urban stormwater quality improvement, as well as their modelling approaches. Flow behaviour, pollutant transport and removal are considered as the three key processes in controlling the performance of constructed wetlands (Wong et al., 2006). The merits and drawbacks of a range of modelling approaches are discussed. A brief summary of the limitation of current modelling approaches will be given, along with
potentials in overcoming them. The chapter will outline the key features and specific objectives of the project, based on the research needs identified from the review of the literature.

2.1.1 Wetland definition and treatment wetland classification

A wetland is a dynamic, transitional zone between terrestrial ecosystems and aquatic systems that are inherently different from each other and it is yet highly dependent on both (Mitsch & Gosselink, 2000). Water within a wetland can be still or flowing, fresh, brackish or saline. The United States Army Corps of Engineers (USACE) and the United States Environmental Protection Agency (USEPA) jointly defined wetlands in the following way (Environmental Laboratory, 1987):

"Those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas."

The technical meaning of the term “wetland” describes systems that are among the most variable and productive ecosystems (Wong and Breen, 2006). Climate and geomorphology define the degree to which natural wetlands can exist (Mitsch and Gosselink, 2000). The distinguishing features of wetlands include the following aspects (Hammer, 1992; Mitsch and Gosselink, 2000; USACE, 2000):

- the presence of shallow water or saturated conditions, either on the surface or within the root zone, is dependent on periodic inundation and drying;
- the substrate is predominantly unique wetland soils (hydric soil) that differ from upland soils; and
- the existence of vegetation adapted to the wet conditions (hydrophytes).

In response to their hydrology, soils, climate, inflow sources and regional plant communities, wetlands exist in a diversity of forms (Kadlec and Knight, 1996). There are many ways of classifying treatment wetland types. For example, treatment wetlands can be
classified by the general treated water sources such as municipal wastewater, mine
drainage, stormwater and non-point source, landfill leachate, agricultural wastewater and
an array of industrial wastewaters. However, the basic types of treatment wetlands used for
improving water qualities include natural wetlands and constructed wetlands (Kadlec and
Knight, 1996; Mitsch and Gosselink, 2000).

Kadlec and Knight (1996) categorized natural wetlands into four types, based on the
characteristics of water salinity (salt water vs. freshwater) and plant form (herbaceous vs.
woody): salt marsh, mangrove wetland, freshwater marsh, and freshwater swamp. Wong
and Breen (2006) presented a broader description of the supporting hydrology of seventeen
types of natural wetlands from coastal flats, inland flats, bogs, deep marsh, fen, shallow
marsh, salt marsh, seagrass beds, deep salt pans, deep open water, shallow open water, wet
heath, mangrove, scrub swamp, to forest swamp-wet, forest swamp-ephemeral and forest
swamp-dry. Classifications of wetlands include the inundation water type (freshwater,
saline or marine), frequency of drying, inundation depth, and the duration of regularity of
inundation. Understanding the classification of wetlands is important to field assessment so
as to ensure that potential threats to the wetland can be appropriately managed (Wong and
Breen, 2006).

As opposed to natural wetlands, constructed wetlands are engineered treatment systems for
one or more of four primary purposes: creation of habitat, water quality management, flood
or flow control, and aquaculture (Kadlec and Knight, 1996). These systems are constructed
to utilize natural processes involving wetland vegetation, soils and associated microbial for
enhancement of water quality, but under a more controlled condition (Vymazal, 2005) than
is offered by natural systems. Constructed wetland treatment systems generally fall into
two categories according to the basic flow configurations (Shutes et al., 2004):

1. surface flow constructed wetlands, and
2. subsurface flow constructed wetlands.

Free surface flow systems are designed to mimic natural wetlands with water flowing over
the soil surface at shallow depths. These systems are planted with emergent, submerged
and/or floating wetland macrophyte plants (Kadlec and Knight, 1996; Shutes et al., 2004). Such wetlands typically have hydrological regime similar to natural wetlands. Conversely, subsurface flow systems are designed to pass flow horizontally or vertically through a permeable medium planted with wetland plants, and to keep the water being treated below the surface (Trepel et al., 2000). Purification occurs during contact with the soil filter medium and plant roots. Sub-surface treatment systems more closely resemble wastewater treatment plants than wetlands. Free surface flow wetlands are commonly designed to maximize wetland habitat values and to offer other benefits such as recreation and landscape amenity, which tend to be less provided by subsurface flow wetlands (USEPA, 1993).

Constructed wetlands are typically designed to meet a range of urban design objectives, including (Livingston, 1989; Schueler, 1992; Dallmer, 2002; Breen et al., 2006):

- the enhancement of urban runoff water quality;
- the control of flooding;
- the control of erosion and sedimentation;
- the creation of aesthetic amenities;
- the water supply of irrigation; and,
- the ecological services.

### 2.1.2 Characteristics of constructed stormwater wetlands

The development of characteristics unique to constructed wetland ecosystems results from the interaction of three important components: (1) wetland hydrology, (2) wetland soil, and (3) wetland vegetation (Environmental Laboratory, 1987; Hammer, 1992). Each of these components is discussed below.

*Hydrology* is considered to be the driving force defining the structure and function of wetlands (Hammer, 1992; Mitsch and Gosselink, 2000). Wetland hydrologic conditions determine many abiotic factors, including water availability, nutrient availability, soil anaerobiosis, soil particle size and composition, and related conditions such as water level,
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Water velocity, flow patterns within the system, flood duration and frequency, water and soil chemistry. These, in turn, affect biotic components, especially plants in the wetlands.

The hydrologic pathways of a wetland include streamflow, surface runoff, groundwater, precipitation and evapotranspiration (Kadlec and Knight, 1996). Evapotranspiration is driven by solar radiation that occurs with seasonal cycles. The presence of groundwater is related to the bottom soil conditions. In practice, constructed wetlands typically are constructed with a natural or artificial impermeable soil stratum to limit seepage, i.e. groundwater discharge and recharge. The location and function of wetlands determines which of these sources and its amount of flow will be contributing to a wetland’s flow balance.

Constructed stormwater wetlands may be located either on-line within the major flow path accepting all flows from the catchments, or off-line on the side of the major flow path receiving parts of flows into the wetlands, or having bypass diversion around the wetlands to discharge safely extreme flows (Somes, 2000; Breen et al., 2006). The high flow variability is experienced by stormwater wetlands and the stochasticity of flows is an important driver for their functions (Kadlec and Knight, 1996) owing to the inherent high variability and stochasticity of storm rainfalls, patterns and durations. The dynamic overall water budget is the balance between the input and output of water within a constructed stormwater wetland as shown in Figure 2-1, which can be described in terms of volume per unit time (Kadlec and Knight, 1996; Mitsch and Gosselink, 2000; Scholz, 2006)

$$\frac{dS}{dt} = Q_{in} + Q_{gwin} + Q_{p} - Q_{out} - Q_{gwout} - Q_{ET} \pm Q_T$$  \hspace{1cm} (2.1)

where $S$ is water storage in wetland, $t$ is time, $Q_{in}$ is surface inflow including stream inflow, $Q_{gwin}$ is groundwater inflows, $Q_{p}$ is net precipitation, $Q_{out}$ is surface outflow including stream outflow, $Q_{gwout}$ is groundwater outflows, $Q_{ET}$ is evapotranspiration and $Q_T$ is tidal inflow (+) or outflow (-). Each term in the water budget may affect a given wetland, but seldom do all terms contribute significantly (Kadlec and Knight, 1996).

2-5
Mitsch and Gosselink (2000) proposed a conceptual model to describe the effects of hydrology on wetland function and the biotic feedbacks to wetland hydrology, as shown in Figure 2-2. The hydrology directly modifies and determines a wetland's physicochemical environment (chemical and physical properties), such as sediments and other pollutants transport, and pollutants load, thereby, leading to a direct impact on the biota in the wetland. On the other hand, the build-up of sediments can change the basin geometry, thus modifying wetlands hydrology such as hydrologic inflows or outflows (i.e. water budget, pathway A in Figure 2-2). Moreover, the biotic processes have impacts upon the hydrological conditions of the wetland through a variety of mechanisms (pathway B in Figure 2-2). For example, plants can cause alterations in their physical environment through sediment trapping, nutrient retention, water shading, and transpiration.

Hydrology is therefore regarded as the most important factor in the design of constructed stormwater wetlands for pollution control (Mitsch and Gosselink, 2000; Zhang and Mitsch, 2005; Scholz, 2006). Several parameters including hydraulic loading rate, detention time, hydroperiod, water depth, hydraulic regime and flooding frequency are used to describe the hydrologic conditions of constructed wetlands (Kadlec and Knight, 1996; Mitsch and Gosselink, 2000). In order to optimize treatment of the stormwater in the wetlands, Wong and Somes (1995) highlighted the following three principal components for designing
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wetland,

- hydrological effectiveness,
- hydraulic efficiency, and
- facilitation and optimisation of water quality treatment processes.

Figure 2-2: Conceptual diagram illustrating the effects of hydrology on wetland function and the biotic feedbacks that affect wetland hydrology. Pathways A and B are feedbacks to the hydrology and physicochemistry of the wetland (Source: Mitsch and Gosselink, 2000).

Figure 2-3 (Wong et al., 1998) illustrates the inter-relationship between these design components.

The selection of treatment system detention time is based on consideration of the target pollutant characteristics, and of the time necessary for effective treatment or removal of these constituents (Breen et al., 2006). The term "hydrological effectiveness" describes the interaction among the proportion of runoff to be treated, the detention time required for effective treatment, and the wetland volume subsequently required. For example, Figure
2-4 shows the proportion of mean annual flow which will be treated by a wetland within Sydney, for a range of design detention times. As the detention time gets longer (in order to promote effective removal of nitrogen, for example), the larger wetland is required in order to treat the same proportion of mean annual flow. Catchment-scale factors (climate in particular) are important in determining the required wetland volume, while local-scale factors are important in determining the effective use of this volume for runoff-treatment purposes.

![Diagram of wetland design elements](image)

**Figure 2-3: Interaction of wetland design elements** (Source: Wong *et al.*, 1998).

Wetland hydraulic efficiency describes the uniformity of detention time within the wetland, and thus reflects the influence of wetland bathymetry, hydraulic structure such as inlets, outlets and berms, and the type, extent and distribution of vegetation. For example, a wetland with stagnant areas and short-circuits will have low hydraulic efficiency, because some water will be held in the wetland much longer than the design detention time, whilst other water will be discharged via preferential flow paths, resulting in shorter than intended detention times.
The hydrologic regime describes the long-term spatial variation in water depths and the period of inundation of a wetland system for controlling wetland vegetation distribution. The subsequent treatment performance is a combined effect of the wetland's hydrological effectiveness, hydraulic efficiency and treatment efficiency via physicochemical and biological treatment processes that are induced by the interaction of plants, micro-organisms, soil and pollutants (Stottmeister et al., 2003).

**Wetland soils** are the foundation for supporting the growth of plant and microbial. Wetland soils have been classified as hydric, which are characterized by anaerobic conditions depending on the duration of saturation, flooding, or ponding (Environmental Laboratory, 1987; Kadlec and Knight, 1996). On the other hand, constructed wetland soils are different from those in natural wetlands, in that they are dynamic in character due to the accumulation of sediments above the wetland substrates. Antecedent soils are changed and replaced by new organic soils generated from deposited sediments, organic matter and mineral solids. Their accretion rate affects the potential removal of conservative elements such as phosphorus and heavy metals. In contrast, during infrequent storm events, soil particulates can be re-suspended (Kadlec and Knight, 1996) and potentially discharged from the wetland, changing the soil properties of the wetland.
**Wetland vegetation** is adapted to specific wetting and drying cycles, owing to the stochastic nature of stormwater. Most wetlands are structurally and functionally dominated by their populations of microbes and plant life (Kadlec and Knight, 1996). In particular, wetland plants can withstand the reducing conditions (absence of soil oxygen) caused by hydric soils, and can survive fluctuation in water levels. In turn, the fluctuating water levels are controlled by factors such as climate and the wetland’s physical characteristics, and vegetation diversity is thus affected by these factors (Koning, 2005). Microbial populations in constructed wetlands include the diverse flora of bacteria, fungi, and algae that are important for nutrient cycling and pollutant transformations. Macrophytic plants provide structures for microbes and create physical and biological conditions required for enhancing water quality in wetland treatment systems (Faulkner and Richardson, 1989; May et al., 1990; Rogers et al., 1991; Kadlec and Knight, 1996; Wong et al., 1998; Breen et al., 2006). Thus, soil, sediments, microbes and macrophytes may all play important and interacting roles in pollutant removal mechanisms (Kadlec, 1994).

### 2.2 Treatment in constructed stormwater wetlands

To understand how constructed stormwater wetlands operate, we must explore:

1. the key stormwater pollutants related to water quality; and
2. the key processes involved in stormwater treatment.

A brief review of the large body of work undertaken on stormwater pollutants is presented below, followed by a review of the studies on key processes that are vital to treatment in constructed wetlands.

#### 2.2.1 The key stormwater pollutants

Catchment urbanization leads to changes in wetlands hydrology and high pollution loads from stormwater runoff (Novotny, 2003; WERF, 2005; Wong and Breen, 2006). Urban stormwater pollutants range from gross solids through fine particulates to soluble
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contaminants. The following are recognized as the key stormwater pollutants by the USA EPA (1983): TSS (Total suspended solids), BOD (Biochemical oxygen demand), COD (Chemical oxygen demand), TP (Total phosphorus), SP (Soluble phosphorus), TKN (Total Kjeldahl nitrogen), NO$_2$ & NO$_3$ (Nitrite & nitrate), Cu (Total copper), Pb (Total lead), Zn (Total zinc) (Stahre and Urbonas, 1990).

Stormwater quality may vary widely in urban areas (WERF, 2005). Residential, commercial, and industrial development, construction and similar activities, transport and recreational activities all contribute to the pollution of urban stormwater. These sources and land uses may deliver a variety of pollutants to receiving waters (NCHRP, 2006). Vassilios et al. (1997) summarized the typical urban non-point source pollution process as shown in Figure 2.5. Although many different components can be found in urban runoff, neither the Nationwide Urban Runoff Program (NURP) introduced by the Environmental Protection Agency (1983) in the USA nor the review of worldwide data by Duncan (1997) found significant differences among the mean concentrations of pollutants in urban runoff. Similarly, geographic locations, runoff volumes, and other watershed factors were of little use in explaining overall site-to-site or event-to-event variability (Novotny, 2003; WERF, 2005).

Duncan (1997) analysed the results of many investigations reported in the literature. He conducted a statistical overview of urban stormwater quality data, and the interaction with land use and other catchment characteristics. The results from his work are summarized in Table 2-1 for different use subgroups.

In the recent past, controlling urban stormwater pollution has often been regarded as a matter of controlling sediment (Diessner, 1992; Persson and Wittgren, 2003), since Total Suspended Solids (TSS) is commonly correlated with other stormwater pollutants (Strecker, 1994). In other words, a significant proportion of urban stormwater pollutants is transported in particulate form, while a much smaller fraction exists in dissolved form (Pitt & Amy, 1973; Diessner, 1992; Kadlec and Knight, 1996; Wong et al., 1998; Braskerud, 2001; WERF, 2005). In addition, different pollutants are normally associated with different
particle size ranges (e.g., heavy metals are predominantly attached to particles less than 63 µm, Sansalone et al., 1998; Deletic and Orr, 2003), suggesting that the particle size distribution (PSD) of TSS has a significant effect on the removal of associated contaminants and will consequently facilitate selection and design of appropriate treatment measures for targeted pollutants (Verstraeten and Poesen, 2001; WERF, 2005; Breen and Lawrence, 2006).

The particle size distribution (PSD) of solids in stormwater has been documented in a number of studies. For Australian catchments, suspended sediments conveyed by urban stormwater appear to have a particle-size range slightly finer than that of overseas catchments as shown in Figure 2-6 (Lloyd et al., 1998; Deletic et al., 2003; CRCCH, 2005). From the NURP data (USEPA, 1983), the PSD for urban runoff is reported towards smaller particle sizes, 55% of the particles by mass being equal to or less than 10 µm in diameter.
and over 70% being equal to or less than 20 µm. In general, particle sizes in urban stormwater runoff are quite small although the properties of particles may vary from site to site, as a function of loadings, hydrology and infrastructure (WERF, 2005).

Table 2-1: Summary statistics for typical stormwater pollutant concentrations (Source: Duncan, 1997).

<table>
<thead>
<tr>
<th>Type of pollutant</th>
<th>TSS</th>
<th>TP</th>
<th>TN</th>
<th>COD</th>
<th>BOD</th>
<th>Total Lead</th>
<th>Total Zinc</th>
<th>Total Copper</th>
</tr>
</thead>
<tbody>
<tr>
<td>Land use</td>
<td>Geometric mean (mg/l)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Roads</td>
<td>2.1</td>
<td>72</td>
<td>0.22</td>
<td>0.81</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>High Urban roads</td>
<td>257</td>
<td></td>
<td>17</td>
<td>0.47</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Low urban roads</td>
<td>69</td>
<td>0.13</td>
<td></td>
<td>0.2</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>High Urban</td>
<td>155</td>
<td>2.6</td>
<td>14</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Low urban</td>
<td>34</td>
<td>3</td>
<td>0.14</td>
<td>0.2</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Residential</td>
<td>0.4</td>
<td></td>
<td>0.045</td>
<td>0.16</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Non-resid. high urban</td>
<td>0.32</td>
<td>78</td>
<td>0.32</td>
<td>0.062</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Industrial</td>
<td></td>
<td></td>
<td>166</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Roofs</td>
<td>35</td>
<td>0.26</td>
<td></td>
<td>0.021</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Agricultural</td>
<td>186</td>
<td>0.54</td>
<td>3.9</td>
<td>0.024</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Forest</td>
<td>79</td>
<td>0.072</td>
<td>0.83</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Zinc roofs</td>
<td></td>
<td></td>
<td>3.7</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Non-zinc roofs</td>
<td></td>
<td></td>
<td>0.16</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Figure 2-6: Comparison of particle size distributions of suspended solids in urban stormwater (data published in Lloyd et al., 1998; Deletic et al., 2003; CRCCH, 2005).
2.2.2 The key treatment processes in constructed stormwater wetlands

Enhancement of water quality in constructed stormwater wetlands and ponds can be achieved by an array of complex processes, based on physical, chemical and biological treatment mechanisms. During storm events, a large amount of pollutants are transported and physical processes of sedimentation and filtration are dominant. Whilst during inter-event (dry weather) periods, chemical and biological processes may be more important (Breen, et al., 2006). Understanding pollutant behaviours in the treatment systems is a fundamental key for predicting and enhancing the performance of the treatment systems. A brief description of the three key treatment processes, i.e. physical, chemical and biological processes, which affect the removal and transformation of pollutants in constructed wetlands, is provided below.

1. Physical processes

**Physical sedimentation** is the preliminary mechanism for pollution removal in ponds and wetlands (Reinelt and Horner, 1995; Liebig et al., 1999; Wong et al., 1999a; Strecker, et al, 2001; Walker, 2001; WERF, 2005; NCHRP, 2006; Kutzner et al., 2007). A particle settles from water to the bed of the treatment system because of a difference in density from water (Kadlec and Knight, 1996; Minto, 2005), only if the water velocity is low enough to provide a sufficient time for this to occur (this time is known as detention time). Removal under quiescent conditions is generally a function of particle density, particle size, and fluid viscosity, but dynamic removal under turbulent conditions is usually dependent upon surface hydraulic loading, particle settling velocities, and shear stress (Urbonas, 1995; WERF, 2005). Suspended solids and attached pollutants are primarily associated with this removal mechanism. The literature suggests that the efficiency of sedimentation is generally a function of detention time and hydraulic loading rate (Hickok et al., 1977; Barten, 1987; Meiorin, 1989; Walker, 2001; Kadlec, R.H., 2003; Huber et al., 2006).

**Re-suspension** is the process where the settled particles are dislodged when near-bed shear
stress exceeds a critical value (Evans, 1994; Kadlec and Knight, 1996). The critical shear stress is a function of the properties of the sediments (James et al., 2003). A variety of factors such as water turbulence, wind-driven turbulence, water level fluctuations (e.g., drawdown), animals of all types and sizes, and gas lift etc. can cause re-suspension, which may in turn increase turbidity, light attenuation and algal growth. In the Hampton Park Wetland study, it was found that the re-suspension of fine particulates affects the dry weather performance for TSS and TP (Fletcher et al., 2004). Effler and Matthews (2004) pointed out that sediment re-suspension is probably an important component of deposition and a source of inorganic particles in many surface water systems. It is likely that there is an equilibrium between deposition and re-suspension at any time, and that this equilibrium varies with flow conditions (turbulence, shear stress) and characteristics of the sediment (size, shape, specific gravity).

**Filtration** is another key physical process in the surface treatment systems associated with colloidal substances, which take long time to settle. The wetland macrophytes with fine but dense stem structures in the treatment systems can create abundant plant surfaces, which (along with the biofilms attached to them) act as filters to provide efficient sites for colloidal agglomeration and adhesion as flow passes through macrophytes (Wong et al., 1999a).

2. **Chemical processes**

Chemical processes are often involved in the removal of soluble pollutants. Chemical uptake mechanisms are associated with the adsorption process, in which some soluble materials are readily adsorbed by suspended solids and epiphytes on the macrophytes during storm events and inter-event periods. Particle size distributions and local soil characteristics significantly influence the effective interception of soluble materials. The sediment uptake of soluble phosphorus is a dominant mechanism transforming soluble phosphorus into particulate form (Breen and Lawrence, 2006). The adsorbed soluble pollutants can then be precipitated via sedimentation.

3. **Biological uptake and transformation**
Biological removal of organic and inorganic constituents (e.g. nutrients and metals) by living organisms (aquatic vegetation, algae and microbes) is also an important process in surface flow systems, particularly during dry weather periods (WERF, 2005). It is the sole way for some soluble nutrients like nitrate to be intercepted and transformed. Epiphytic biofilms attached on the surface of emergent aquatic vegetation convert soluble pollutants into biomass, e.g. algal/plant and bacterial particulate matter which in sequence can settle on to the sediments (Breen et al., 2006). The biological uptake rate of dissolved nitrogen depends on density of epiphytes and thus on available sub-stratum surface area (Taylor, 2006). Once nutrients and organic matters are trapped in the sediments, they may be decomposed by bacteria and removed from the system.

2.3 Modelling performance of constructed stormwater wetlands and ponds

Constructed stormwater wetlands are principally designed to serve for water quality improvement and quantity management (Trepel et al., 2000). Efficient evaluation of the performance of the constructed wetlands is crucial to their planning, design and management under a variety of conditions. Models have generally been used to provide necessary information and suggestions for designing and maintaining a constructed wetland or pond.

Models are defined by the US EPA as processes or tools that are used to increase the level of understanding of (natural or constructed) systems and ways in which they react to varying conditions (McAlister et al., 2006). In addition to the simulation of key water quality and quantity processes, models are also valuable and widely used for enhancing planning and design efficiencies, and for real-time control (Phillips and Thompson, 2002; Novotny, 2003; McAlister et al., 2006; Wong et al., 2006). In particular, mathematical stormwater models can be used to assist in meeting hydraulic, hydrologic, engineering and economic requirements. The following subsections provide explanations of existing modelling approaches for stormwater wetlands and ponds.
2.3.1 General approaches in stormwater modelling practice

Many efforts have been made to develop mathematical models for stormwater treatment in the past. However, it is very difficult to find a way to classify stormwater treatment models, because they are multi-dimensional and multi-process oriented, as well as multi-purpose based.

Models must fit the specific purpose for which they have been developed. Urban stormwater models can be used as operational, planning or design tools. In some cases their use may be restricted for research purposes. The fundamental differences in the modelling approaches are the amount of data required, the information obtained from the model, the model complexity and the simulation period. The complexity is associated with required computational resources, model limitations and reliability (Zoppou, 2001). In practice, the reliability, robustness and complexity of models are crucial for their usefulness.

Models are often characterized as either stochastic or deterministic (Nix, 1994). Stochastic models have some uncertain variables (or at least the uncertainty is included explicitly in the model), and are described with a probability distribution. A deterministic model has all variables known with certainty (or at least assumed to be), and is based on well-established physical laws. Stochastic approaches are often used in model sensitivity and uncertainty analyses to provide a conceptually simple framework for representing heterogeneity (Grayson and Blöschl, 2001). Both stochastic and deterministic models may be further classified into either conceptual or empirical, depending on whether the model is based on experimental observations (empirical) (Zoppou, 2001) or theoretical analyses (conceptual). However, conceptual models may be refined using empirical data, and empirical models may also be incorporated with conceptual elements. In other words, these classifications are not binary, and models may reflect a combination of approaches.

Nix (1994) also classified models based on spatial differences such as distributed models (accounting for variations as a function of the position in the system) and lumped models
(assuming that all characteristics are constant over the system by neglecting spatial variability).

Models may also be classified in terms of their temporal representations. Event-based models are used to predict wetland performance for a given event (often called a “design event”), whilst continuous models use a time series of climate, inflow and concentration over the long term. Event-based models are suitable for the design of the hydraulic components of stormwater infrastructure and act as operational models. Continuous models simulate process behaviours over a long continuous period like months or seasons (including the duration between storm events) and form the basis of planning models for water resources.

2.3.2 Processes to be modelled

In general, mathematical descriptions for stormwater quality behaviour involve the simulations of flow hydrodynamics, since it influences the degree of pollutant removal (Wong et al., 2006). Basically, modelling the performance of constructed stormwater wetlands must consider at least the simulation of:

1. flow behaviour,
2. pollutant transport, and
3. pollutant removal.

Therefore, the hydrology and hydraulics of the treatment systems must be modelled in all cases. Without simulating flow (e.g. its depth and velocity) it is impossible to model pollutant treatment because of the importance of flow as a dominant mechanism for transporting pollutants. In addition, modelling pollutant transport is essential, since pollutant transport and flow are co-dependent. Ideally, sediment transport and flow should be modelled simultaneously (Zoppou, 2001). Finally, it is also important to include the pollutant removal processes outlined in Section 2.2.2 in the pollutant transport equation (Periáñez, 2005; WERF, 2005). Sustainable stormwater management often demands qualitative and quantitative predictions of the impacts of the pollutants on the receiving
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Water bodies, involving both flow behaviour and water quality. However, the combination of flow, pollutant transport and treatment processes in models is often only possible if the mathematical complexity of the model is low enough that data collection to support the model is possible (Trepel et al., 2000).

Therefore, in the next section, the methods for modelling flow will be first summarized. This is followed by a detailed review of methods for pollutant transport and of some existing models for estimating sediment trapping efficiency.

2.4 Modelling of hydraulics

Flow behaviour in a stormwater treatment system is highly dynamic, with low inflows during dry periods (known as the base flow) and high inflows during wet weather periods, as previously discussed in Section 2.1.2. In most of Australia, there are distinct dry and wet weather periods. Further to this, flow characteristics depend greatly on the geometry and characteristics of treatment systems.

Flow routing is a procedure to predict the spatial and temporal variations along a watercourse based on known upstream hydrograph and characteristics of the watercourse (Mays, 2005). Basic equations for fluid flow and transport are typically used; i.e. conservation equations for water mass, momentum and constituent mass (Martin and McCutcheon, 1999). Stormwater models can be categorized in several ways, e.g. according to their capability to cope with complex hydrological systems, or according to their solution methods (analytical and numerical methods), and so on. Routing models for stormwater flow are generally classified as either hydrologic models (routing by lumped system methods) or hydraulic models (routing by distributed system methods; Mays, 2005; Chin, 2006). Zoppou (2001) identified that the distinction between hydrologic and hydraulic models is determined by the process that is being modelled. Hydrological models usually only solve the continuity equation, but hydraulic models solve it along with the momentum or the energy equations in a coupling way to describe the spatial behavior of the flow process. Thus, the modelling approaches from the complex hydraulic models
(derived from Navier-Stokes equations) to the commonly-used simple hydrological models are discussed in next two sub-sections.

2.4.1 Complex hydraulic approaches

There are numerous hydraulic models used to assess the behaviours of stormwater treatment processes (i.e. to optimize the hydraulic efficiency), based on a form of Navier-Stokes (N-S) equation. Depending on the assumptions made and the choice of dependent variables, the N-S equations may vary in complexity and form. Numerical schemes such as finite differences, finite elements or the method of characteristics are used for complicated problems (Zoppou, 2001). A general overview of hydraulic models in 3-D, 2-D and 1-D for surface flows is given below.

A general *three-dimensional model* is very complex and less stable than simpler alternatives, but has become more likely with the rapid growth in computing capacity (Trepel *et al.*; 2000). It is expected that 3-D modelling will become more effective and commonly used in the future. The 3-D hydraulic models have a potential to simulate rapidly varying transient flows and wind-induced flows. They can provide better results where the topography is more complicated (and thus where the flow patterns are highly variable over small spatial scales). Wu *et al.* (2000) presented a 3-D numerical model for calculating flow and sediment transport in open channels, derived from solving the full Reynolds-averaged N-S equations with the k-ε turbulence model, but with the water level determined from a 2-D Poisson equation according to 2-D depth-averaged momentum equations. Sometimes the 3-D models have to be used to study the turbulent flows which often occur in the presence of vegetation. Based on laboratory experiment results, Schmid *et al.* (2005) simplified such a 3-D model and derived an approximate solution to a 2-D sediment transport equation to describe an analytical relationship between flow patterns and sediment deposition in the presence of vegetation in constructed wetland ponds.

The 3-D programs are obviously more time-consuming to run than the 2-D and 1-D models. Examples of 3-D packages are Fluent, CFX, Mike3, FIDAP and PHOENUCS; see Trepel,
The main weaknesses of 3-D models are that they are complicated to build and run due to the requirements for a great quantity of input data and the consequent difficulties in calibration. Therefore, they are often impractical for operational objectives, and are mainly used for research purposes.

**Two-dimensional models** assume either lateral or vertical mixing, in order to simplify the flow processes into two dimensions. Analytical solutions can only be obtained for a few specific cases involving regular and symmetric geometries. In most practical situations, the solutions have to resort to numerical techniques, even through approximation errors exist. Abdel-Gawad and McCorquodale (1984) developed a 2-D numerical scheme to solve the Navier-Stokes equations for the flow in a rectangular basin by using a combination of a strip integral method and the finite difference method. Similarly, Naser *et al.* (2005) used a steady, 2-D numerical model to study the hydrodynamics of a rectangular sedimentation basin under turbulent conditions.

Two-dimensional modelling approaches have been widely used for modelling river and coastal hydraulics, and recently have become a viable practical option for modelling urban stormwater systems at a range of temporal and spatial scales (Huber *et al.*, 2006). Based on a 2-D depth-averaged scheme, numerical methods used to study the 3-D fluid flow and constituent transport have become very popular. Models such as SWMM and its enhanced version XP-SWMM (US Environmental Protection Agency) use a depth-averaged approach for hydrologic simulation. However, their predictive ability may be limited where the system's geometry is complex, as pointed out by Trepel *et al.* (2000). SWMM also allows users to define whether flow is completely mixed or plug flow, so as to predict overall degree of treatment for a pond. The user does not always be in a position to make such a judgement for models. These models have complicated predictions of pollutant generation and hydraulics, but their descriptions of treatment behaviour are limited by available basic treatment methods and by treatment measures to which they can be applied (Wong *et al.*, 2006). To consider the effect of flow depth, a depth-integrated method can be used, such as is used in several commercial software
packages WASP/DYNHYDR5 (US Environmental Protection Agency), Mike21 (Danish Hydraulic Institute). These models are very complex and require extensive input data sets. Therefore, the model applications may be restricted in practice.

**One-dimensional models** ignore the hydrological variations so that the longitudinal direction is the sole dimension, with variations in the vertical and horizontal directions of the flow cross-section being completely disregarded. The results for fluid movement in one dimension can be obtained by solving the Navier-Stokes (N-S) equations in terms of the conservation of momentum. Numerous software packages have been developed for solving the 1-D N-S equation, which is suitable for some specific water systems such as rivers, channels, lakes and reservoirs. For example, the shallow water wave equations can be used to simulate 1-D flows in open channels with the bed slope and the frictional slope (Deletic, 2001), and the well-known kinematic and diffusion wave equations are simplified versions of the shallow water wave equations (Zoppou, 2001). Both methods neglect the local and convective acceleration terms, but the former also ignores a pressure term in the momentum equation. The dynamic wave model includes all acceleration and pressure terms (Mays, 2005).

The HEC-RAS software is one of the most widespread 1-D models, developed by the Hydrologic Engineering Centre of the United States Army Corps of Engineers to perform 1-D steady and unsteady flow hydraulic calculations for a full network of natural and constructed channels. All analysis components use a common geometric data representation and common geometric and hydraulic computation routines. The basic computational procedure is based on the solution of the 1-D energy equation. Energy losses are estimated by friction (Manning’s equation) and contraction/expansion (coefficient multiplied by the change in velocity head). The momentum equation is used when the water surface profile is rapidly varied. The HEC-RAS model solves the full 1-D Saint-Venant equations for unsteady open channel flow. The model is capable of simulating water surface profiles in subcritical, supercritical, and mixed flow regimes. Since HEC-RAS is used to calculate water surface profiles and energy grade lines with the
assumptions of 1-D (flow direction), steady-state (no time term is present), gradually varied flow (hydrostatic pressure is assumed) (Hydrologic Engineering Centre, 2002), it may not be useful to determine accurately flow characteristics such as steep slope, lateral spreading, hydraulic jump with distributed outflow. Other 1-D flow routing approaches such as Mike 11 and ISIS are based on the Shallow Water Equations or variations used in practical river engineering (Zoppou, 2001; Pappenberger et al., 2005).

The widespread usage of 1-D models in practice may be due to the fact that these approaches not only are simpler to use, but also require a minimal amount of input data and computing power (Hydrologic Engineering Centre, 2002). Furthermore, a lack of adequately accurate descriptions of boundary conditions and truncation errors may preclude the use of complex 2-D and 3-D models. For instance, the usage of the Manning’s equation in 1-D models is much simpler, but could be criticized for higher dimensional models (Pappenberger et al., 2005). Hence, even though all real physical systems are three-dimensional, sufficient accuracy may be achieved by using only 1-D models.

2.4.2 Simplified hydrological approaches

Hydrological approaches are generally based on the conservation of mass and ignore the spatial variations in the system (Zoppou, 2001). There are many methods available to solve the continuity equation. The most commonly used hydrological methods are the Puls method and the Muskingum method, which are outlined below.

In a simple routing method, the storage equation for the reservoir can be defined as (Mays, 2005)

$$\frac{dS}{dt} = Q_{in}(t) - Q_{out}(t)$$

(2.2)

where $S$ is the storage, $Q_{in}$ the inflow, $Q_{out}$ the outflow and $t$ the time.

The Puls method rewrites the storage equation expressed as, over a finite time interval, $\Delta t$,
\[
\frac{2S^{j+1}}{\Delta t} + Q_{out}^{j+1} = Q_{in}^{j} + Q_{in}^{j+1} + \frac{2S^{j}}{\Delta t} - Q_{out}^{j}
\]  (2.3)

where \(S^{j}, Q_{in}^{j}, Q_{out}^{j}\) and \(S^{j+1}, Q_{in}^{j+1}, Q_{out}^{j+1}\) are storage volumes, inflow rates, discharge rates at time \(j\) and \(j+1\), respectively.

This method describes the storage as a nonlinear function of outflow \(Q_{out}\) and requires the construction of two curves for \(S\) and \(2S/\Delta t + Q_{out}\) in terms of \(Q_{out}\). Therefore, the storage–discharge (S-Q) relationship for a treatment system can be calculated by this approach.

The Muskingum method is an approximation to the shallow water equations for the conservation of mass, based upon a variable discharge-storage relationship. This method assumes that storage changes linearly with inflow \(Q_{in}\) and outflow \(Q_{out}\) in the form

\[
S = KQ_{out} + \theta K(Q_{in} - Q_{out})
\]  (2.4)

where \(K\) is the storage/channel reach parameter (i.e. a lag time for the reach) and has the unit of time, \(\theta\) is a dimensionless weighting factor that ranges between 0 and 0.5 (in relation to attenuation of inflows within the reach).

It must be noted that hydrological models that neglect the momentum equation have a major drawback in that they cannot account for the downstream backwater effect (Akan and Houghtalen, 2003). However, although hydraulic models are generally considered more accurate, hydrological models are simple to use and have acceptable accuracy under some circumstances. They have thus been used over a long time for the management of surface flow systems (Chin, 2006). In operational contexts, they are generally more commonly used than are complex hydraulic models.

The CRC for Catchment Hydrology developed the Model for Urban Stormwater Improvement Conceptualisation (MUSIC). The model offers a decision supporting tool for the evaluation and comparison of various stormwater treatment strategies. Grass swales, wetlands, ponds and infiltration systems are considered to be a single continuum of
treatment, to which Universal Stormwater Treatment Model (USTM) can be applied. MUSIC adopted the Puls method to describe the hydrologic routing within a treatment system and the Muskingum method to compute the passage of a stormwater wave through a catchment, or between treatment nodes (CRCCH, 2005).

2.5 Modelling of pollutant transport

In stormwater modelling, pollutants are generally treated as neutrally buoyant material. The transport of pollutants is induced by (1) flow advection for the pollutants are conveyed by flowing water, (2) turbulent diffusion caused by non-uniformity of velocity in flow profiles, and (3) pollution molecular diffusion in describing the transportation of pollutants from high to low concentration areas (Zoppou, 2001). Therefore, the pollutants can spread throughout the fluid. Flow advection is the main process, and diffusion is a secondary one in stormwater treatment systems. Diffusion due to turbulence is normally two or more orders of magnitude larger than molecular diffusion, so that the latter can normally be neglected in transport. The combination of advection and diffusion produces pollutant dispersion in the water (Chadwick et al., 2004). The common basis of most pollutant transport models is the principle of mass continuity, known also as the transport equation. Numerical models were developed to model pollutant dispersion in multiple dimensions like complex 3-D approaches. However, as hydraulic analyses, most of them used in stormwater quality modelling are in one dimension such as plug flow, hydraulic efficiency and CSTRs model, which will be discussed further in the following subsections.

2.5.1 Complex approaches - models based on diffusion equations

As we know, the transport of pollutants involves two fundamental processes: advection and diffusion. Under the Cartesian coordinate system, the basic constituent transport equation is given by Martin and McCutcheon (1999) as
where $C$ is the concentration of a constituent, $V_x, V_y, V_z$ the flow velocities along $X, Y, Z$ directions, $D_x, D_y, D_z$ the diffusion coefficients and $S_k$ a source or sink related to physical, chemical and biological processes such as sedimentation, re-suspension and decomposition.

Periáñez (2005) presented a very complex 3-D hydrodynamic model to account for turbulence and the suspended matter equations to consider transport of sediments in the water column and settling of particles on to the sea bed. Four particle classes were considered simultaneously. Nadaoka et al. (1991) suggested a model to evaluate quasi-3-D suspended sediment transportation in a non-equilibrium state, based on a kind of weighted residual concept, assuming an exponential distribution of $C$ in the vertical direction to minimize its residual integrated over the water depth and to choose 1 and $C$ as weighting factors for determining other two unknown variables in the model.

Steady-state assumptions are often made to reduce model complexity and data requirements; the basic equation for the mass balance of any constituent $C$ can thus be expressed in a 1-D or 2-D form. If the distribution of the pollutants is not symmetrical within a simple flow field, a 2-D model can be used to consider the effects of turbulence, and the diffusion coefficients are directly related to shear velocity and water depth (Chadwick et al., 2004). Wu et al. (2000) suggested that the overall sediment transport in open channels can be governed by a 2-D equation, integrated over the water depth. This approach consists of a suspended-load and a bed-load model; the interaction between them occurs through the net deposition/entrainment flux of sediment at the top of the bed-load layer. Walker (1998) proposed a method that uses 2-D models to solve the depth-averaged flow and transport equations for estimating the long-term residence time distribution (RTD) in wetlands and wet ponds with permanent pools. Based on the assumptions of near-constant vertical velocity distribution in shallow flows through emergent vegetation,
Schmid et al. (2005) derived an approximate solution to the 2-D equation for the sediment deposition rate typically required to judge the trapping efficiency of constructed wetland ponds.

In the case of steady flow (i.e. the distribution of velocity and turbulence does not vary across the cross-section), the problem can be reduced to 1-D, that is, at time $t$, Equation 2.5 is reduced to

$$\frac{\partial C}{\partial t} + V \frac{\partial C}{\partial X} = D \frac{\partial^2 C}{\partial X^2} + S_k$$  \hspace{1cm} (2.6)

This form of the diffusion equation is used in computer models like WQRRS and HEC-5Q (Hydrologic Engineering Centre, 1978; 1986), which are also capable of assessing urban stormwater quality and quantity (Zoppou, 2001).

Nepf (1999) described diffusion in vegetated flow including both turbulent and mechanical diffusion. Turbulent diffusion is the result of wakes and eddies while mechanical diffusion arises from the physical obstruction to the flow by plant stems. The diffusion coefficient $D$ can be solved from a physically-based model for drag, turbulence and diffusion within emergent vegetation. Building on this approach, Wadzuk and Burke (2006) carried out a laboratory dye tracer study to determine the diffusion coefficient for a laminar flow in stormwater wetland and concluded that debris and bed shear stress affect diffusion and should be considered when computing the coefficient. It should be noted that if the wetland can be divided into two regions (main channel and storage zone) as done by Martinez and Wise (2003), another transport equation should be specified for the storage zone.

In these transport models, the flow behaviour is unaffected by the pollutant and there is no pollutant transformation. For sediment transport, the flow and sediment transport should be modelled as a coupled system (Zoppou, 2001). The solutions to the coupled hydrodynamic and pollutant transport model must resort to numerical methods. In addition, these deterministic models normally involve some model parameters. Also, they can only be applied to predict the system’s performance for a given event, rather than for a long period.
2.5.2 Simple approaches – the hydraulic efficiency model and CSTRs model

The mathematical descriptions of pollutant transport can be very complex as discussed above, when all aspects of flow conditions are considered. However, surface flows in constructed wetlands are somewhat simpler, being gradually varied flows on very mild slopes (Kadlec, and Knight, 1996). Thus, many sediment transport models use simplified diffusion equations. Due to the transient stormwater flow patterns, these models include series or parallel combinations of two ideal extremes of mixing: a plug flow reactor (i.e. totally unmixed zone) and completely mixed reactor as discussed below.

1. Plug flow model

Plug flow is based on the continuity equation and includes travel time in pollutant transport. Under ideal hydrodynamic flow conditions, there is a uniform velocity profile in one cross-section with no lateral mixing (WPCF, 1990). The treatment storage consists of a series of conceptual plugs and all fluid elements have the same practical detention time in the system. The detention time of plugs is determined by the ratio of the discharge rate to the storage volume as illustrated in Figure 2-7.

![Figure 2-7: Residence time distributions](Source: Persson et al., 1999).
2. Hydraulic efficiency model

Unfortunately, ideal plug flow conditions rarely happen in stormwater treatment systems, whose complex geometry can result in large variations in detention time. Re-circulating and stagnant zones shrink the engaged basin volume due to non-idealized geometry. Short-circuit flow paths result in a decrease in detention time, whilst stagnant zones result in an increase.

Optimizing detention time and minimizing flow short-circuiting are important objectives for a successful design to optimize water quality treatment. Persson et al. (1999) proposed a quantitative measure of system hydrodynamic behaviour for evaluating the hydraulic efficiency factor ($\lambda$) of a treatment facility, which consists of two components: the effective volume ratio $e_v$ that is related to how much of the pond is used by flows and the degree of mixing $N$. The $\lambda$ factor is simply defined as:

$$\lambda = e_v (1 - 1/N) = \frac{t_p}{t_n}$$

(2.7)

where $t_p$ is the actual detention time of the peak concentration through the facility, $t_n$ is the theoretical or nominal detention time, computed as the ratio between the total volume and the discharge rate ($S/Q$) based on the above plug flow model. $N$ is the number of continuously stirred tank reactors defined below. The effective volume ratio, $e_v$, is defined by the ratio between the mean detention time and the nominal detention time ($t_m/t_n$), and is affected by the length and width of the system and other factors such as wind (Persson, 2000). The mean detention time, $t_m$, is always less than the nominal detention, $t_n$, because the ideal plug flow condition on which the nominal detention time is calculated, does not occur in reality, and so the effective volume differs considerably between ponds (Persson, 2000). Under the plug flow conditions, the concentration distribution can be simplified as a spike with a very small standard deviation about the mean residence time, as illustrated in Figure 2-7 (Wong et al., 2001). Therefore, it is clear from Equation 2.7 that a larger $\lambda$ represents a stronger tendency towards plug flow over the full storage volume (i.e. little mixing or short-circuiting). A lower $\lambda$ indicates a completely well mixed reactor. Persson et al. (1999) investigated the influence of pond shapes with different inlet / outlet and length
to width configurations on the hydrodynamics of these systems as shown in Figure 2-8. The results of the corresponding values of effective volume $e_v$ and hydraulic efficiency $\lambda$ are presented in Table 2-2.

**Figure 2-8: Hydraulic efficiencies ($\lambda$) of detention systems** (Source: Persson et al., 1999).

**Table 2-2: Modelled hydrodynamic measures of hypothetical cases** (Source: Persson et al., 1999).

<table>
<thead>
<tr>
<th>Case</th>
<th>$S_{\text{short-circuiting}}$</th>
<th>$e_v$</th>
<th>$I-I/N$</th>
<th>$\lambda = t_p/t_n$</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>0.29</td>
<td>0.74</td>
<td>0.41</td>
<td>0.3</td>
</tr>
<tr>
<td>B</td>
<td>0.24</td>
<td>0.79</td>
<td>0.33</td>
<td>0.26</td>
</tr>
<tr>
<td>C</td>
<td>0.1</td>
<td>0.46</td>
<td>0.23</td>
<td>0.11</td>
</tr>
<tr>
<td>D</td>
<td>0.16</td>
<td>0.34</td>
<td>0.52</td>
<td>0.18</td>
</tr>
<tr>
<td>E</td>
<td>0.68</td>
<td>0.89</td>
<td>0.85</td>
<td>0.76</td>
</tr>
<tr>
<td>G</td>
<td>0.72</td>
<td>1</td>
<td>0.76</td>
<td>0.76</td>
</tr>
<tr>
<td>H</td>
<td>0.1</td>
<td>0.44</td>
<td>0.25</td>
<td>0.11</td>
</tr>
<tr>
<td>I</td>
<td>0.3</td>
<td>1</td>
<td>0.41</td>
<td>0.41</td>
</tr>
<tr>
<td>J</td>
<td>0.87</td>
<td>1</td>
<td>0.9</td>
<td>0.9</td>
</tr>
<tr>
<td>K</td>
<td>0.34</td>
<td>0.78</td>
<td>0.46</td>
<td>0.36</td>
</tr>
<tr>
<td>O</td>
<td>0.25</td>
<td>0.73</td>
<td>0.35</td>
<td>0.26</td>
</tr>
<tr>
<td>P</td>
<td>0.57</td>
<td>0.96</td>
<td>0.64</td>
<td>0.61</td>
</tr>
<tr>
<td>Q</td>
<td>0.5</td>
<td>0.93</td>
<td>0.64</td>
<td>0.59</td>
</tr>
</tbody>
</table>

The water movement within the wetland must involve some hydrologic parameters, i.e., residence time and hydraulic loading rate, which may be adequate to describe the on-site flow pattern and performance of the system (Kadlec, 1994). Werner and Kadlec (2000)
developed a computer model to reproduce the experimental wetland residence time distributions (RTDs) based on the concept that the non-ideal flow of constructed wetlands is a network of an infinite number of small stirred tanks distributed along a set of main plug flow channels, and the stirred tanks represent zones without mixing. The concept of stirred tanks is discussed further in the following section. Holland et al. (2005) noted that water level had a direct impact on the residence time distribution of a wetland, and therefore increasing the water depth draws out a decrease in hydraulic efficiency.

3. CSTRs model

A range of mechanistic methods have been developed to describe wetland hydrodynamics, such as plug flow and plug flow with dispersion as discussed above. The following approach of conceptualizing treatment systems as a series of continuously stirred tanks is widely applied in chemical engineering applications, and used in MUSIC to simulate wetland hydrodynamics (CRCCH, 2005), as explained below.

As stormwater flows through a surface flow wetland, flow hydrodynamics within the system may be modelled as a combination of plug flow and a number of continuously stirred tanks reactors (CSTRs) (i.e. totally mixed zones), according to Kadlec and Knight (1996), as shown in Figure 2-9.

![Figure 2-9: The continuously stirred tanks reactors model for mixing in a constructed wetland (Source: Kadlec and Knight, 1996).](image)

Using this approach, a wetland can be conceptually partitioned into a number of equal sized tanks; each of the CSTRs is a notional volume in which the inflow is immediately and completely mixed with the existing contents. As the number of CSTRs in series approaches
infinity, the residence time distribution approaches that of plug flow (Wong et al., 2001). The number of CSTRs \((N)\) can be approximately related to the hydraulic efficiency of the treatment facility as \(\lambda \approx 1 - \frac{1}{N}\). For some wetland geometries, it has been observed that the value of \(N\) ranges from 2 to 8, although the reasons for this small range are unknown (Kadlec and Knight, 1996; Lightbody et al., 2007). For the \(n\)-th CSTR cell, the mass balance equation of a given pollutant is

\[
Q_{n-1}C_{n-1} - Q_nC_n = M_{Rn} = kA_b \left( C_n - C^* \right)
\]  

(2.8)

where \(Q_{n-1}\) and \(C_{n-1}\) are the inflow rate and the influent concentration of the \(n\)-th tank, \(Q_n\) and \(C_n\) its outflow rate and the effluent concentration, \(M_{Rn}\) the reduction rate in pollutant, \(C^*\) the background concentration, \(A_b\) the bottom area of each tank and \(k\) a decay rate constant.

2.6 Modelling of sediment behaviour relevant to wetlands

This section reviews models which are relevant to modelling sediment behaviour in wetlands. It begins with explaining simple sediment removal models that have been specifically developed for wetlands. Next we discuss conceptual models for a suite of pollutants and sediment transformations. Physically-based models are then reviewed for two main physical processes involved in sediment behaviour: deposition and re-suspension. Some of these approaches have not been directly used in wetland modelling in the past, but are regarded as important building blocks for future model development.

2.6.1 Regression models

Regression models based on observed data are developed to meet the need for a simple, easy-to-apply tool; they attempt to develop simple relationships between water quality and relevant explanatory variables. These approaches rely on measured data, and not on an understanding of the physical processes in the treatment systems. In these models, the
system is treated as a “black box”, although some understanding of the key pollutant removal processes will be helpful in identifying the most important variables in the regression.

Duncan (1998) derived a number of empirical relationships for total suspended solids (TSS), total phosphorus (TP) and total nitrogen (TN), using regression-based studies of 76 wetlands around the world. For example, Figure 2-10 shows relationships of suspended solids output percent with hydraulic loading and input concentration. While the standard errors of these performance curves are large, they nevertheless provide a preliminary method for determining the wetland size to meet a pollutant removal target.

![Figure 2-10: Predicted outflow TSS concentrations as a percentage of inflow concentration from statistical analysis of Australian and overseas data (Source: Duncan, 1998).](image)

Strecker et al. (2001) reported that the ASCE project team had evaluated efficiency of Best Management Practices (BMPs) and BMP systems based on the monitoring data on BMP water quality. The data were analysed in two different ways; analysing systems individually, and then examining in groups by BMP type. The individual analysis included rigorous statistical analysis and graphical analysis. The individual analysis was conducted for each parameter and each BMP. The BMP group analysis looked at the results for all BMPs of a particular type such as wetland basins, or retention ponds. Strecker et al. (2001) found that the volumetric ratio between mean runoff and wetland storage was a key predictor of pollutant removal.
The primary limitation of the regression approach is that the relationships are developed from given sets of data, and the relationship may therefore have limited transferability to other systems or areas. They often have a great degree of unexplained variability, and thus cannot be used with confidence to aid design (e.g. Figure 2-10). This is particularly the case if the database used was not comprehensive, which is often the case due to high cost for data gathering. Regression models can probably be regarded as sufficient for planning purposes only (Driver and Tasker, 1988), but cannot work as a continuous modelling approach that is necessary to track the varying operating conditions over time.

2.6.2 Conceptual models

Conceptual models provide a “lumped” description of overall processes with some presumptions, and thus attempt to represent the complex interactions of physical and biochemical processes of the system in simple mathematical forms with a few calibration coefficients. These models normally treat some features of the system as a kinetic process, so as to identify their priorities for planning purposes. They may have some physical and chemical base, such as the conceptual stormwater management models XP-AQUALM and XP-SWMM. Although well-known techniques are available to calculate the effect of the residence time distribution on pollutant removal performance (Kadlec and Knight, 1996; Walker, 1998), these models have rarely been used in treatment wetland data analysis or design (Kadlec, 2000).

The first order kinetic (or k-C*) model has been one of the most used conceptual analysis and design models so far (Kadlec, 2000; CRCCH, 2005; Wong et al., 2006), despite its limitations. This model assumes steady and plug-flow conditions, and has generally been used in predicting the long-term averaging performance of wastewater treatment facilities (Kadlec and Knight, 1996; Wong et al., 1999b). In most cases, kinetic processes through a time averaging scheme are described by the first-order relationship of the form (Kadlec, 2000; Kadlec and Knight, 1996),

\[ q \frac{dC}{dy} = k(C - C^*) \] (2.9)
where $q$ is hydraulic loading (m/year), defined as the ratio of the inflow to the surface area of the system, $y$ the fractional distance from inlet to outlet, $C$ the time averaging pollutant concentration at the fractional distance $y$ (mg/L), $C^*$ the background concentration (mg/L), and $k$ the removal rate constant (m/year).

The $k$-$C^*$ model provides a suitable description of the long-term mean performance (Kadlec and Knight, 1996). Since it simulates overall behaviour of a treatment system in a period of time, it can serve as a planning model. In particular, the $k$-$C^*$ model can account for a wide range of pollutants such as total suspended solids (TSS), total nitrogen (TN), total phosphorus (TP), biochemical oxygen demand (BOD), turbidity, and even heavy metal through a series of treatment systems (CRCCH, 2005; Wong et al., 2006). Whilst the processes involved in the removal of each of these pollutants may be different, the resulting overall outcome may be relatively well described by the $k$-$C^*$ model.

The $k$-$C^*$ model has been adopted in the software MUSIC (CRCCH, 2005). In MUSIC it is incorporated into the Universal Stormwater Treatment Model (USTM), where it is combined with the previously-described CSTRs model. The solution to Equation 2.9 is expressed as follows (Wong and Geiger, 1997):

$$C_{out} = C^* + (C_{in} - C^*)e^{-k/q}$$

(2.10)

Where $C_{in}$ is the input concentration of the treatment system (mg/L), $C_{out}$ the output concentration of the system (mg/L), and $k$ the pollutant removal rate constant (m/year) that reflects the settling velocities of the targeted sediment size (for TSS) or the chemical decay rate (for TN, for example). A higher $k$ means a faster way to reach equilibrium and thus a higher treatment capacity. $C^*$ is a background concentration. When $C^* = 0$, combining the above equation with the removal efficiency formula provided by Fair and Geyer (1954) gives:

$$k = -q \ln \left[ \left( 1 + \frac{1}{n_t} \frac{V_s}{q} \right)^{-n_t} \right]$$

(2.11)

where $V_s$ is the settling velocity of particles, and $n_t$ the turbulence and short-circuiting
parameter that is a similar type of measure to the hydraulic efficiency $\lambda$ (CRCCH, 2005).

According to the $k$-$C^*$ theory (Kadlec and Knight, 1996), $C^*$ should include the effect of particle re-suspension. The original theory suggests that it should be less than the inflow concentration at any time. However, recent evidence has suggested that this need not be the case. For example, the dry-weather inflows of sediment may be lower than the background concentration in a wetland, particularly where wind-induced turbulence and re-suspension are significant. The parameter $k$ in the USTM has lumped together the influence of a number of predominantly physical factors. For modelling of sedimentation, it has been proposed that $k$ is equal to the solid settling velocity (Kadlec and Knight, 1996).

In practice, these parameters must be determined through a process of calibration and verification (McAlister et al., 2006) from data regression. However, studies have shown that the two parameters of this conceptual model, $k$ and $C^*$, actually vary with the hydraulic loading rate, inlet concentration and sediment particle size, meaning that they are not really constant even at one site (Wong & Geiger, 1997; Kadlec, 2000; Kutzner et al., 2004). A recent study of the $k$-$C^*$ model for a swale in Brisbane showed that for the same site $k$ varies with inflow rate (Deletic and Fletcher, 2006). Hence, these two parameters actually depend strongly on hydraulic loading ($k$ and $C^*$), inlet concentration and particle size distribution ($C^*$). Further work is therefore required to improve the model’s prediction ability, under variable operating conditions (Fletcher et al., 2004). A recent study by Kutzner et al. (2007) concluded that alternative approaches are necessary.

Additionally, a better understanding of internal hydraulics such as the effects of spatial distributions of vegetation density (which affects spatial distributions of flow resistance and treatment efficiency), is also necessary for the advancement of conceptual models for constructed wetland design (Kadlec, 2000).

### 2.6.3 Physically-based models

These models are a mix of deterministic and stochastic models since they are based to some extent on physical laws of hydrodynamics and pollutant transport (and thus could also be
called pseudo-deterministic models). Physically-based models use essential equations to represent physical processes with model parameters, rather than derived distribution functions. This type of model is intended to minimize the need for calibration by using relationships in which the parameters are, in principle, measurable physical quantities (Grayson and Blöschl, 2001). These parameters can be calibrated easily using monitoring data.

There are two main physical processes that need to be modelled in surface flow stormwater treatment systems: (1) sedimentation and (2) re-suspension.

It should be noted that pollutant removal from stormwater wetlands is affected by flow conditions, pollutant sources and states, residence time and season (Reinelt and Horner, 1995). In addition, dense vegetation is generally credited for promoting sedimentation by a combination of reducing turbulence and slowing water velocity (Barko et al., 1991; Sand-Jensen, 1998; Schmid, et al., 2005; WERF, 2005), increasing sediment adhesion (often called ‘enhanced sedimentation), and minimising wind-generated turbulence (through dampening wave activity, redirecting currents, and providing sediment stability). The combination of these effects will protect deposited sediment from re-suspension in shallow water systems (Wong et al., 1998; 2000; 2006; James et al., 2004). However, Fennessy (1994) demonstrated that vegetation was not important for sedimentation in a constructed freshwater wetland, with sediment removal being instead a function of hydrologic loading near the inlet. Based on field data on flow in a marsh, Leonard et al. (2002) suggested that differences in vegetation cover had no significant effect on flow regime, sediment transport and deposition patterns. Brueske and Barrett (1994) more specifically stated that vegetation patterns did not influence the deposition of sediments in a high hydraulic loading wetland (26.07 m/year), but had a significant effect in a low-loading wetland (3.13 m/year).

1. Sedimentation theory models

Sedimentation theory models always begin with an assessment of the settling velocity of particles. The most simplified approach for modelling of this process is based on Newton's

2-37
and Stokes’ sedimentation laws in view of force balance. The settling velocity for a specific particle that is freely falling through calm water is

\[ V_s = \frac{g}{18\mu} (\rho_s - \rho_w) d_s^2 \]  

(2.12)

where \( g = 9.8 \text{ m/s}^2 \) is the gravity acceleration, \( \rho_s \) the particle density (kg/m\(^3\)), \( \rho_w \) the water density (kg/m\(^3\)) and \( d_s \) the particle diameter (m), and \( \mu \) dynamic viscosity of water (kg/s/m) which is a function of water temperature.

It should be noted that Stokes’ law is valid for steady-state flow or in stationary fluid. The settling velocity is applicable if the Particle Reynolds Number \( Re_p = \rho_w V_s d_s / \mu \) is less than 1 and the particle diameter is equal to or less than 0.1 mm (Yang, 1996). For Reynolds Number up to 2, the Goldstein approximation should be used to modify the settling velocity \( V_s' \), that is, (Yang, 1996)

\[ V_s' = V_s / V_D \]  

(2.13)

where the velocity ratio \( V_D = (1 + \frac{3}{16} Re - \frac{19}{1280} Re^2 + \frac{71}{20480} Re^3 + ...) \)

For unsteady-state flows at relatively large Reynolds Number (up to \( 10^4 \)), Fair and Geyer (1954) provided an expression for sediment transport in a turbulent flow

\[ V_D = (1 + \frac{1}{8} \sqrt{Re} + \frac{17}{1200} Re ) \]  

(2.14)

Rubey (1933) introduced formulae for particle sizes larger than 1 mm. For quartz particles, the fall speed can be calculated by

\[ V_s = F[d_s g(\frac{\rho_s - \rho_w}{\rho_w})]^{1/2} \]  

(2.15)

where the parameter \( F = 0.79 \). For particle sizes greater than 2 mm, the settling velocity can be approximated by
Despite many efforts, the mathematical modelling of settling velocity is less than perfect. In general, the factors affecting the settling velocity are $\rho_s$, $\rho_w$, $\mu$, $d_s$, particle surface roughness, shape, suspended sediment concentration $C$ and strength of turbulence (Yang, 1996; Boogerd et al., 2001).

However, all these theories should be reviewed in relation to the stormwater characteristics presented in Section 2.2. The particles found in stormwater are mainly very small, with median particle size well below 0.1 mm (Figure 2-6). Hydraulic loading rates and therefore flow velocities in stormwater treatment systems are also very low. Finally, concentrations of particles in stormwater are relatively low. Therefore, it may be speculated that simple Stokes’ law may be good enough for prediction of settling velocity of particles in stormwater systems. However, it is wise to calculate the Particle Reynolds Number $R_{ep}$ and to make adjustments of the settling velocity accordingly (e.g. using one of the listed equations).

Once the settling velocity of a particle in question is determined, the task is to model its time of travel through the system. If the particle with a known settling velocity $V_s$ is located at a height $H$ from the pond base (Figure 2-11), the settling time for this particle is

$$t_s = \frac{H}{V_s} \quad (2.17)$$

Therefore the particle can be trapped only if the settling time is less than the residence time. For the plug flow approximation, the nominal residence time $t_a$ can be calculated by the storage volume and the mean flow rate of the treatment system. For a rectangular pond, the residence time is equal to $L/V$. Hence the determined physical variable, that is, the Particle Fall Number, is the ratio of the horizontal flow time to the vertical particle settling time (Deletic, 2000). Specially, the Particle Fall Number, $N_f$, is calculated as:
where \( L \) is the length of the system (m), \( H \) the mean depth of flow (m), \( V_s \) the particle settling velocity (m/s) and \( V \) the mean flow velocity (m/s).

\[
N_f = \frac{LV_s}{VH} \tag{2.18}
\]

Based on estimated settling velocities, a number of theories such as Hazen’s model (Hazen, 1904), numerical constitutive computational fluid dynamic models, and surface overflow theory, can be used to derive the sediment trapping efficiency through Type I (free falling) sedimentation for a specific treatment system, although Hazen’s model can account for non-ideal settling conditions (WERF, 2005). For example, sedimentation in a rectangular settling tank is shown in Figure 2-11. All particles with a settling velocity greater than the critical velocity \( V_{cs} = (H/L)V \) will settle out, and a fraction of the particles with a settling velocity \( V_s \) less than the critical velocity \( V_{cs} \) will settle out at a ratio \( V_s/V_{cs} \) (Kiely, 1997). To obtain the output pollutant concentration for a treatment system, the trapping efficiency of sediments for all particle diameters are summed, using the obtained formulation for sediment trapping efficiency \( Tr \). Thus, the total fraction that settles in the rectangular settling tank can be expressed as (Kiely, 1997)

\[
Tr = (1 - X_s) + \int_0^{X_s} \frac{V}{V_{cs}} dX \tag{2.19}
\]

where \((1 - X_s)\) is the fraction of particles with settling velocity greater than \( V_{cs} \) and the second term is for those particles with a velocity less than \( V_{cs} \).
This theory was expanded further and adopted for calculation of sediment trapping by grass filter strips used under rural runoff conditions. Tollner et al. (1976) carried out experiments on simulated unsubmerged grassed areas. Based on their results, they proposed the following empirical relationship for trapping efficiency \( T_r \) of a particle at a diameter \( d_s \), that is, Kentucky model

\[
T_r = \exp \left[ -1.05 \times 10^{-3} \left( \frac{VR_s}{\nu} \right)^{0.82} N_f^{-0.91} \right] 
\]

(2.20)

where \( \nu \) is the kinematic viscosity of sediment and water mixture (m\(^2\)/s), and \( R_s \) is spacing hydraulic radius (m). It is of interest to note that the Kentucky model took into account the effect of turbulent flow through the use of Reynolds Number. However, this model failed to predict performance of grass filter strips for urban stormwater conditions, since it was developed for much larger particles and higher concentrations of sediment than are usually found in urban stormwater (Deletic, 2005) and was never fully verified even for rural runoff conditions (Munoz-Carpena et al., 1999).

An extensive laboratory study on artificial grass was completed by Deletic (2001) in order to examine the processes of sediment transport in runoff over grass. This study resulted in the development of the Aberdeen model, for the assessment of TSS removal in overland flow over non-submerged grasses based on a simple empirical function (Deletic, 2001, 2005),

\[
T_r = \frac{N_f^{0.69}}{N_f^{0.69} + 4.95} 
\]

(2.21)

So far, the Aberdeen model has been successfully tested for stormwater swales in Australia, and a grass filter strip in Scotland (Deletic & Fletcher, 2004), showing promise for future applications in assessment of vegetated stormwater treatment systems. However, it was concluded that the model is not very reliable for particle less than 6 \( \mu \)m and larger than 100 \( \mu \)m in diameter (Deletic 2005). It is speculated that the Aberdeen model would not cope with fine particles well because it does not consider the effect of sediment re-suspension.
It is of interest that the $k$-$C^*$ model is also related to the fall number for sedimentation. It is noted that $k/q$ in Equation 2.10 is actually the Particle Fall Number as defined in Equation 2.18. In other words, by rearrangement of Equation 2.9 we can obtain the outlet concentration $C_{out}$ as follows

$$\ln\left(\frac{C_{out} - C^*}{C_{in} - C^*}\right) = -\frac{V_s L}{H V} = -N_f$$

Equation (2.22)

It is found that Equation 2.22 is equivalent to Equation (11-14) and Equation (11-15) in Kadlec and Knight (1996, p327), in which the Particle Fall Number is referred to as the Damköhler Number for TSS. If $C^* = 0$, the TSS trapping efficiency $Tr = (C_{in} - C_{out})/C_{in}$ can be obtained as

$$Tr = 1 - e^{-N_f}$$

Equation (2.23)

Thus the $k$-$C^*$ and $N_f$ models have similar roots in sedimentation theory.

2. Physical models of re-suspension

Physical models of re-suspension are rarely used in assessment of performance of stormwater surface systems. However, this process may be very important, as discussed in Section 2.2.2 and therefore, some of the most popular re-suspension models used in river morphology are discussed below, together with those used in surface flow wetlands.

Yang (1973) derived a formula for total sediment concentration in rivers using dimensional analysis, aiming to investigate the re-suspension of particles. The total sediment concentration was expressed in the following dimensionless form

$$C = \phi^\ast \left(\frac{V - V_{cr}}{V_s} \right) S_{slope} \left(\frac{U^*}{V_s}, R_e\right)$$

Equation (2.24)

where $S_{slope}$ is the slope, $V_{cr}$ the average critical stream velocity at incipient motion of sediments, $U^*$ the shear velocity for re-suspension, instead of the bulk or mean velocity. A specified form of the above expression is given in Yang (1996). The velocity distribution
along the water depth in open channel flow takes the form (Yang, 1996)

\[
\frac{V}{U^*} = 5.75 \ln \frac{h}{D_h} + R_b
\]  

(2.25)

where \( h \) is the distance from the bed, \( D_h \) the reference depth at which \( V = R_b U^* \) and \( R_b \) is the roughness, which depends on whether the boundary is in a hydraulically smooth, transition, or completely rough regime.

It is well recognized that the gradient in velocity can give rise to shear stress that can sustain the suspension sediments to delay their falling. Suspended sediment transport depends strongly on the ratio of the shear velocity to the settling velocity in terms of the Rouse equation (Rouse, 1937):

\[
\frac{C}{C_h} = \left( \frac{H - y}{y} - \frac{h}{H - h} \right)^z
\]  

(2.26)

where \( C \) and \( C_h \) are sediment concentrations by weight at distances \( y \) and \( h \) above the bed, respectively, \( H \) the mean water depth, \( z (= V_s/k' U^*) \) a function of particle settling velocity \( V_s \), shear velocity \( U^* \) and Prandtl-von Kármán universal constant \( k' (= 0.4 \) for clear water).

In addition, re-suspended particles have a short distance to settle compared with those in inflow. Their settling velocity can be five times faster than that based on sedimentation theory in lake water (Malmaeus and Hakanson, 2003). Markensten and Pierson (2003) presented a simple dynamic model to predict suspended inorganic particle movement in a lake basin based on variations in wind and river inflow. The model combined the effects of flow and wind driven sediment re-suspension based on measurements of wind speed and mean river discharge rate, while wind-induced re-suspension was related to the square root of wind speed.

Tsanis et al. (1998) gave explicit forms for sediment deposition and re-suspension in Cootes Paradise Marsh, the majority of which is open water surrounded by marshy areas
and woodland. The model accounted for contributions of bottom sediments due to carp activity and wind induced re-suspension. The specific forms for settled and re-suspended solids are given by the following two equations:

\[
M_{dep} = \frac{(1 - \tau_b) V_s C}{\tau_{cd} H}
\]

\[
M_{res} = -\frac{(1 - \tau_b) \epsilon}{\tau_{cr} H}
\]

where \(M_{dep}\) is the loss of suspended solids due to deposition, \(M_{res}\) the contribution of suspended solids due to re-suspension, \(H\) the water depth, \(\tau_b\) the bottom shear stress calculated by \(\tau_b = \rho_d U^2\), \(\tau_{cd}\) and \(\tau_{cr}\) the critical shear stresses for deposition and re-suspension, respectively; \(\epsilon\) the re-suspension rate. Both \(\epsilon\) and \(\tau_{cr}\) depend on the bed structure of the sediment. Based on this formulation, it was predicted that more sediments were re-suspended with larger bottom shear stresses.

Frehamann et al. (2005) developed a mathematical model for predicting sedimentation and remobilization in in-line storage sewers for stormwater treatment based on the results of pilot plant studies. The fall velocity was used as the dominant parameter for the sedimentation. The remobilization of settled particles was described by an exponential function, which is expressed by a combination of some parameters including the shear stress, the area covered by the sediment, and an empirical parameter.

Despite those efforts, there remain many discrepancies concerning the mechanisms of sediment re-suspension in surface-flow systems. Modelling by Tsanis et al. (1998) showed that the critical shear stress for re-suspension had a great impact on sediment behaviour in a natural wetland. However, Braskerud (2001) studied four constructed rural runoff wetlands and found that re-suspension had decreased approximately 40% and became negligible five years after a wetland had been constructed (presumably due to a coating of larger organic particles, as a result of decomposition within the wetland). This is consistent with the assertion by Kadlec and Knight (1996) that in the surface flow treatment wetland
environment, physical re-suspension is not a dominant process. To include the effect of re-suspension in assessing the sediment trapping efficiency, an alternative hypothesis could be that the parameters in physically-based models depend on the ratio of the shear velocity to the settling velocity of re-suspended particles. The form of the parameters for a given type of stormwater treatment systems could be determined through the field data. Obviously, the validity of this hypothesis would need to be tested.

## 2.7 Key research gaps

This literature review has identified a number of research gaps and uncertainties in sediment behaviour in constructed stormwater wetlands. These gaps will provide the basis for the work presented in the following chapters of this thesis.

A through understanding of the significance of the inter-correlation among water quality parameters is necessary before reliable methods to predict pollutant removal can be developed. These parameters could potentially be based on physical sedimentation process. As pointed out by Zoppou (2001), sediment models should be more physically-based. However, the reliability and certainty of current models are limited mainly due to their inconsistency with stormwater quantity and quality controlling processes, and the highly dynamic nature of stormwater events makes it difficult to develop complicated models.

Many efforts have been made to simplify the models of wetland sedimentation, so that the complex processes can be treated in a regression manner. Those simple models can fairly well predict the water quality for the purpose of wetland design, although they lack universality due to limited datasets. In Australia, the most widely used model, MUSIC, combines the simple first order kinetic model ($k$-$C^*$ model) with the CSTRs model to account for the flow hydrodynamics in the treatment systems. However, the model parameters, $k$ - rate constant and $C^*$ - background concentration, were found to vary strongly with hydraulic loading and inlet concentration, and therefore are changeable even within one site. Further work on simplified models is therefore required, and should focus on allowing the models to be validly used under a range of operating conditions.
The importance of vegetation in removal of sediment in wetlands has been widely disputed. As previously described, some studies showed that vegetation cover can minimize re-suspension, and maximize sedimentation, but others suggest that the difference in vegetation cover had no significant effect on flow regime, sediment transport or deposition patterns. Similarly, there was also a debate about re-suspension of sediments in surface flow wetlands.

In summary, although constructed wetlands for treatment of urban runoff have been extensively studied, the key research gaps still lie in:

1. The individual and combined effects of influencing factors such as hydraulic loading, vegetation density, and inflow concentration under wet and dry weather conditions on treatment performance are still not well understood. Those physical quantities should explicitly be studied in a quantitative way for their importance in sediment removal under various operation conditions.

2. The role of sediment re-suspension induced by flow under varying operating conditions is still unclear, and effective physically-based methods for predicting re-suspension of sediments in constructed stormwater wetlands are required.

3. Many models are based on or derived from studies of wastewater treatment systems. As previously discussed, wastewater systems differ fundamentally from stormwater wetlands. For example, the original Hazen model has been widely used for combined sewer overflows (CSO) and waste water treatment systems, but it has not been fully examined for stormwater wetlands.

4. Existing models suffer from many deficiencies, such as low reliability, extensive input data and difficulties in calibration. In general, there is a lack of simple but robust physically-based models with relatively constant model parameters for continuous prediction of the performance of constructed stormwater wetlands.

5. Sensitivities of a proposed model (e.g. $k-C^*$ model) have to be investigated completely, prior to its engineering applications. This step is very important in order to validate any proposed model.
2.8 Specific objectives of this thesis

Based on the main aim of the project and the key research gaps identified in this chapter, the objectives of the research are:

1. Provide insights into important factors and processes which influence sediment particle trapping efficiency in wetlands, using controlled laboratory experiments in mesocosm stormwater wetlands and ponds;
2. Refine existing understanding of sedimentation and re-suspension processes in constructed stormwater wetlands under both wet and dry weather conditions;
3. Develop a new approach for prediction of trapping efficiency of sediment particles and attached pollutants (such as total phosphorus and heavy metals) in surface-flow constructed stormwater wetlands through experimental and field data.

The modelling method will aim to strike a balance between ease of use and complexity of the system considered. The main attributes of the proposed model should be:

- ease of use and widespread applicability,
- physically-based,
- containing a minimum number of calibration parameters, which are relatively constant across a number of constructed stormwater wetlands without any further calibration or tuning.

4. Evaluate the most popular method for modelling sediment removal in wetlands (the $k-C^*$ model); and
5. Test the proposed methods using field monitoring data collected from an operational wetland, and compare model results with those generated using the $k-C^*$ model.

2.9 Conclusions

Constructed stormwater wetlands and ponds are becoming increasingly more important in
urban catchments in order to improve stormwater quality. In particular, free surface flow wetlands can provide an integrated system for urban stormwater treatment, as well as recreation and landscape amenity. In these wetlands, water quality can be controlled not only by a complex array of physicochemical and biological treatment processes, but also by physical sedimentation, which in turn are influenced by flow hydrodynamics and sediment transport. Previous studies, based on experimental, field and numerical analyses, have indicated that sedimentation is the primary mechanism for pollution removal in ponds and wetlands. Modelling sedimentation of suspended solids is therefore essential to wetland design. In this chapter, the knowledge gathered so far on the three aspects associated with sediment removal has been critically reviewed: (1) the characteristics of wetland flow hydraulics, (2) the key mechanisms in pollutant transport, and (3) the key processes involved in sediment removal in constructed wetlands.

Flow routing models used in stormwater can range from simple 1-D hydrological routing methods to complex 3-D Computational Fluid Dynamics approaches which solve the N-S equations. Although hydrological models have inherent drawbacks to describe transient flow, their simplicity in use and often acceptable accuracy make them become popular in wetland flow hydraulics simulations under most circumstances.

Pollutant movement in stormwater either by flow advection or in terms of diffusion can be summarised in a transport equation, with a source term for pollutant sedimentation or re-suspension. Complex models focus on the localised variations of pollutant concentrations in wetlands. They are time-consuming to develop and run, and involve many parameters to be calibrated. However, a steady-state assumption and 1-D simplification are widely used to make the problems more tractable. The hydraulic efficiency factor in relation to the plug-flow estimate can be easily incorporated into a CSTRs approach for more complex hydraulic and geometric situations.

Sedimentation in constructed stormwater wetlands with complex flow conditions and planted vegetation is much more difficult to predict than the case where sediments fall freely in stationary ponds. To avoid the complexity, regression models and conceptual
models have been developed. However, these models rely greatly on the accuracy of the lumped model parameters, since they may vary with flow conditions and initial pollutant concentrations. For example, the first-order kinetic $k-C^*$ model belongs to this category. The physically-based models seem more reliable than others, since they start from basic sedimentation theory. Although some influencing factors are missed in these models, they can be applied to a wide range of cases without modification of model parameters. The paradox of simplicity versus accuracy seems to be an ongoing theme of the modelling of stormwater wetlands. One non-linear $N_r$ model has been successfully verified (without any further calibration) on two field stormwater grass filters, and seems to have potential for bridging the divide between simplicity and accuracy.

It is appropriate to hypothesise that a similar methodology (i.e. the non-linear $N_r$ model) can also be developed for constructed stormwater wetlands. The remainder of this thesis will thus examine the potential suitability of this model for modelling wetlands, using a range of field and laboratory-based data, and will compare the model with other commonly-used approaches.
Chapter 3

Laboratory Study of Sediment Behaviour in Constructed Stormwater Wetlands

3.1 Introduction

3.1.1 Background

The main aim of this study was to identify and assess the most important processes and factors that impact upon sediment behaviour in stormwater wetlands. Stormwater treatment systems are subjected to very uneven flows resulting from the stochastic nature of storm events (Werner and Kadlec, 2000; Pontier et al., 2004). In particular, stormwater wetlands are subjected to prolonged periods of much lower inflow, following storm events with very high hydraulic loading and pollutant loads (Wong and Breen, 2002; Wong et al., 2006). Owing to the complex characteristics of wetlands, their performance under dry weather conditions may be greatly different than that during wet weather periods. Therefore, it was important to study sediment behaviour during both:

(1) wet weather (during storm event), and

(2) dry weather (between storm events).

Due to the complexity of the processes involved in sediment and associated pollutant removal in wetlands (Fennessy et al., 1994), it was decided to begin with by conducting controlled laboratory experiments in mesocosm stormwater wetlands. The outcomes of experiments were then used to develop mathematical models with a minimum number of calibration parameters for predicting the sediment removal rate in constructed stormwater wetlands.

This chapter reports on the framework of laboratory experiments undertaken to investigate
Laboratory mesocosm wetland study

the treatment of suspended solids in mesocosm wetlands. The main hypotheses of the laboratory study are first outlined, and an overview of the experimental program is provided. Development of an experimental rig, including the scaling issues for meeting hydraulic characteristics of physical models, is then described. The experimental procedure and programme are followed with detailed elucidation. Experimental results are subsequently presented and discussed, based on the different experimental conditions, i.e. wet weather and dry weather. Finally, some conclusions are drawn.

3.1.2 Hypotheses tested

As outlined in the literature review, sediment behaviour in ponds and wetlands may include particle deposition, re-suspension of particles from the system’s bed, and wash-off of particles that are suspended in the pool of water (Kadlec and Knight, 1996). The following hypotheses were assumed and then tested in this study:

**Hypothesis 1** - Re-suspension of particles from the system’s bed due to flow turbulence\(^1\) can be neglected during both wet and dry weather. Usually the re-suspension rate is regarded to be a function of particle size, \(d_s\), and shear velocity, or simply a function of Shear Velocity Reynolds Number, \(R_e^*\):

\[
R_e^* = \frac{U^* d_s}{\nu}
\]  
\[\text{(3.1)}\]

where \(U^*\) is the shear velocity (m/s), and \(\nu\) the kinematic viscosity (m\(^2\)/s). The shear velocity, \(U^*\), can be found by assuming a logarithmic velocity \((V)\) distribution along water depth \(H\) (Nakayama, 1999; Marsh et al., 2004):

\[
V = 2.5U^* \ln(U^* H / \nu) + 3U^*
\]  
\[\text{(3.2)}\]

**Hypothesis 2** - Deposition of particles is the main process that governs sediment behaviour

\(^1\)This does not include re-suspension due to wind or birdlife.
Laboratory mesocosm wetland study during both wet and dry weather.

During *wet weather*, sediment deposition can be modelled as a simple function of the Particle Fall Number, $N_f$, defined by Equation 2.18 in Chapter 2. Furthermore, $N_f$ can be expressed as

$$N_f = \frac{xV_s}{HV} = \frac{V_s}{q}$$

(3.3)

where $x$ is the distance of the system (m), $V_s$ the particle setting velocity (m/s) and $q$ the hydraulic loading (m/s), which is calculated by

$$q = \frac{Q}{A}$$

(3.4)

where $A$ is the treatment surface area (m$^2$).

If the particle falls freely as a discrete sphere, the particle settling velocity $V_s$ is given by Stokes’ law, as expressed by Equation 2.12.

During *dry weather*, sediment deposition of particles is not only a function of the Particle Fall Number $N_f$, but also of a time ratio parameter $t^*$, that is defined as the ratio between the actual time since the last storm event and the mean detention time in the wetland, given by

$$t^* = \frac{t}{t_d}$$

(3.5)

where $t_d$ is the mean wetland detention time, i.e., the average time that inflow remains in the treatment system, defined as

$$t_d = \frac{S}{Q}$$

(3.6)

where $S$ is the volume of the treatment system (m$^3$).

**Hypothesis 3** - Wash-off$^2$ of fine particles that never settle in the pool of water (i.e. particles

2 Washoff that is caused solely by flow turbulence is examined.
that are in constant suspension) does not play an important role during wet and dry weather. For the purpose of testing this hypothesis, washoff is modelled as a function of particle size and Root Mean Square (RMS) Turbulent Reynolds Number, $R_{eT}$, an indicator of the level of turbulence of the flow in vegetated channels. $R_{eT}$ is given as follows (Tollner et al., 1976):

$$R_{eT} = \frac{VR_s}{\nu} \tag{3.7}$$

where $R_s$ is the spacing hydraulic radius (m), defined as

$$R_s = \frac{W_{ss}H}{2H + W_{ss}} \tag{3.8}$$

where $W_{ss}$ is the distance between two plant stems (m).

In non-vegetated systems, this number becomes Flow Reynolds Numbers, $R_e$, defined as

$$R_{eT} = R_e = \frac{VR}{\nu} \tag{3.9}$$

where $R$ is the flow hydraulic radius (m), defined as

$$R = \frac{BH}{2H + B} \tag{3.10}$$

where $B$ is the flow width of the treatment system (m).

### 3.1.3 Outline of the experimental methodology

To test the hypotheses listed above, two different types of experiments had to be conducted:

1. **Wet weather experiments** with a specific aim to examine processes during storm events, as well as to test whether or not the modified Hazen model (the non-linear $N_f$ model, Equation 2.21 in Section 2.6.3) is able to model reliably sediment trapping efficiency during storm events.

2. **Dry weather experiments** with an aim to study sediment behaviour during dry spells (focused in particular on Hypothesis 3) and if possible to develop and calibrate an analytical model for sediment trapping efficiency under dry weather conditions.
Laboratory mesocosm wetland study

In general, a series of experiments was carried out under the following conditions:

- Well-established emergent macrophyte cells with different densities were used to mimic real constructed stormwater wetlands.
- Steady-state conditions in water flow and TSS concentration were applied.
- No infiltration was allowed, in order to focus on sediment deposition and re-suspension.
- Natural sediment, collected from a nearby stormwater retarding basin, was sieved through a 300 µm sieve and used as the wet weather input sediment. The sediment particle size distribution was within the natural range of stormwater.

3.2 Experimental installation

Stormwater wetlands are rather large structures, and if we are to study their performance in laboratory conditions, we have to use physical models of the real systems. Such physical modelling relies on ‘scaling theory’, which has been used for long time in hydraulic research. Therefore, the first job was to design a rig which could represent a model of actual wetlands (following the rules of scaling theory), and would allow for investigations of a range of different wetland design characteristics.

Due to resource limitations, an existing laboratory rig with some modest alterations (housed in the Hydraulics Laboratory of the Civil Engineering Department, Monash University, Australia) was utilized. The rig was previously designed and used by Taylor (2006) for studying nitrogen behaviour in constructed stormwater wetlands, and was therefore a logical choice. The first step was to modify this rig keeping in mind that the scaling rules have to be followed. This section explains the rig, starting with a discussion on how the scaling was undertaken.

3.2.1 Scaling issues in physical modelling of stormwater wetlands and ponds

The main principle of physical modelling of hydraulic processes is to keep a similarity of the
main non-dimensional parameters that govern the studied processes. In our cases, the ranges of the following non-dimensional numbers must at least be as close as those of real systems,

1. Shear Velocity Reynolds Number, $R_{e*}$,
2. Particle Fall Number, $N_f$, and
3. Turbulent Reynolds Number, $R_{eT}$.

Therefore, the first task was to determine the ranges of these three numbers under real operational conditions, and then to construct a mesocosm system capable of replicating these values in laboratory.

Field data collected at the Hampton Park wetland, Melbourne (Fletcher et al., 2004; Taylor et al., 2006) were used to provide a rough estimation of these three non-dimensional numbers in real systems (the size of vegetated and non-vegetated ponds of this wetland are shown in Table 3-1). The range of hydraulic loading from 50 m/y to 100 m/y covered the possible inflow rates over the dry weather period, and that from 2000 m/y to 7000 m/y was used for the wet weather period (Duncan, 1998), while the flow depths were the same as measured in the field for the chosen inflow rates. The typical range of particle sizes in the stormwater inflow was assumed to be from 1 to 300 µm, while their density varied from 2300 to 2750 kg/m$^3$ (Deletic and Orr, 2003). The vegetation densities ranged from 500 to 3000 culms/m$^2$. Table 3-1 shows the three non-dimensional numbers as found in the Hampton Park vegetated and non-vegetated ponds based on particle size of $d_s = 40$ µm (assessed to be median for stormwater in Melbourne, adopted from CRCCH, 2005), particle density of 2520 kg/m$^3$, and vegetation density of 590 culms/m$^2$ (averaged for the Hampton Park wetland).

The size of the mesocosms and the experimental inflow rates were then determined, so that the laboratory system can generate the similar values of $R_{e*}$, $N_f$ and $R_{eT}$ found in the field. As shown in Table 3-1, similarity was achieved for $R_{e*}$ and $N_f$ in both vegetated and non-vegetated systems, if the mesocosm was 1.5 m long, 0.25 m wide and 0.04 - 0.05 m deep under both dry and wet weather conditions. Unfortunately, the mesocosm sizing was constrained to a great extent by the limited resources for modification or reconstruction. The
length of the mesocosms was therefore fixed at 1.5 m.

During wet weather experiments, the Turbulent Reynolds Number, $R_{eT}$, was reasonably similar between the mesocosm and real vegetated wetlands (although somewhat lower in the mesocosm, but still in the range of number found in real wetlands). For the non-vegetated mesocosm pond, it was impossible to achieve replication of real $R_{eT}$ (that in this case was the Flow Turbulence Number, $R_e$, Equation 3.9) while also maintaining similarity for $N_f$ and $R_e^*$ at the same time. For non-vegetated ponds, it was possible to scale for only two of the three numbers within the appropriate range (e.g. if sediment size and density was also scaled, we could achieve similarity in $N_f$ and $R_{eT}$, but not in $R_e^*$). It was therefore decided to build the mesocosms as specified in Table 3-1, and to concentrate on similarity of $N_1$ and $R_{eT}$ in the wet weather period, since the literature reviews have shown that the processes of deposition and re-suspension are the most important in this period. The wash-off process could also be studied in the scaled vegetated ponds to some extent since at least $R_{eT}$ was in the same order of magnitude. However, to study this process in non-vegetated ponds separate experiments were required. This was therefore undertaken in pilot scale open-water pond experiments (see Chapter 4).

During dry weather experiments, the Turbulent Reynolds Number, $R_{eT}$, was lower at the mesocosm than in the real vegetated wetlands. For the non-vegetated mesocosm pond, it was also impossible to achieve replication of $R_{eT}$. However, inflow particles are typically very fine during the dry weather period and it may be more important to examine the effect of wash-off of fine particles. It was therefore decided to increase the hydraulic loading in the mesocosms (from an original range of 50 – 300 m/y to 100 – 500 m/y) to obtain the same ranges of the Turbulent Reynolds Number $R_{eT}$ as in reality. Furthermore, it was necessary to increase the hydraulic loading to the level to be able to sample for sediment concentrations, since sufficient water volume must be collected within a certain time. In essence, the area of each mesocosm was only 0.375 m$^2$ (1.5 m $\times$ 0.25 m). Limited by the existing rig, the outflow rate based on the dry weather hydraulic loading (i.e. 50 – 100 m/y) was very low, and thus, a higher flow rate (in the range of 100 – 500 m/y) was required in the laboratory.
Table 3-1: Hampton Park wetland characteristics and mesocosm characteristics for particle size, $d_s = 40 \mu m$ and density, $\rho_s = 2420 \ kg/m^3$.

<table>
<thead>
<tr>
<th>Weather conditions</th>
<th>Dry weather</th>
<th>Wet weather</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hydraulic loading $q$ (m/y)</td>
<td>50</td>
<td>100</td>
</tr>
<tr>
<td><strong>Vegetated systems: Vegetation density = 590 culms/m$^2$</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Hampton Park Wetland</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Size: length, width and depth (m)</td>
<td>30, 77, 0.19</td>
<td>30, 77, 0.21</td>
</tr>
<tr>
<td>Particle Shear Velocity Reynolds Number $R_e^*$</td>
<td>1.54E-03</td>
<td>2.31E-03</td>
</tr>
<tr>
<td>Particle Fall Number $N_f$</td>
<td>725</td>
<td>362</td>
</tr>
<tr>
<td>Turbulent Reynolds Number $R_{eT}$</td>
<td>5.1</td>
<td>8.9</td>
</tr>
<tr>
<td>Flow rate of each cell (mL/s)</td>
<td>0.6</td>
<td>1.2</td>
</tr>
<tr>
<td>Size: length, width and depth (m)</td>
<td>1.5, 0.25, 0.04</td>
<td>1.5, 0.25, 0.04</td>
</tr>
<tr>
<td>Particle Shear Velocity Reynolds Number $R_e^*$</td>
<td>7.59E-04</td>
<td>1.13E-03</td>
</tr>
<tr>
<td>Particle Fall Number $N_f$</td>
<td>730</td>
<td>365</td>
</tr>
<tr>
<td>Turbulent Reynolds Number $R_{eT}$</td>
<td>0.8</td>
<td>1.7</td>
</tr>
<tr>
<td><strong>Non-vegetated systems</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Hampton Park Wetland</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Hydraulic loading $q$ (m/y)</td>
<td>50</td>
<td>100</td>
</tr>
<tr>
<td>Size: length, width and depth (m)</td>
<td>120, 70, 0.94</td>
<td>120, 70, 1.03</td>
</tr>
<tr>
<td>Particle Shear Velocity Reynolds Number $R_e^*$</td>
<td>9.76E-04</td>
<td>1.50E-03</td>
</tr>
<tr>
<td>Particle Fall Number $N_f$</td>
<td>830</td>
<td>415</td>
</tr>
<tr>
<td>Turbulent Reynolds Number $R_{eT}$</td>
<td>185</td>
<td>370</td>
</tr>
<tr>
<td>Flow rate of each cell (mL/s)</td>
<td>0.6</td>
<td>1.2</td>
</tr>
<tr>
<td>Size: length, width and depth (m)</td>
<td>1.5, 0.25, 0.04</td>
<td>1.5, 0.25, 0.04</td>
</tr>
<tr>
<td>Particle Shear Velocity Reynolds Number $R_e^*$</td>
<td>7.07E-04</td>
<td>1.04E-03</td>
</tr>
<tr>
<td>Particle Fall Number $N_f$</td>
<td>836</td>
<td>418</td>
</tr>
<tr>
<td>Turbulent Reynolds Number $R_{eT}$</td>
<td>1.8</td>
<td>3.6</td>
</tr>
</tbody>
</table>
3.2.2 Experimental rig

The experimental rig cross-section and plan view are shown with all dimensions in Figure 3-1. The rig contained a 10 KL tank with a regulated discharge rate, and four experimental cells (flumes that represent wetlands). An emergent macrophyte, *Baumea articulata*, capable of standing upright in the chosen range of hydraulic loadings, was well established in a 0.4 m thick sandy loam layer, with daylight lamps to support plant growth (Figure 3-2). Each cell had a different density of vegetation (Figure 3-3):

1. no vegetation,
2. low-density = 590 culms/m²,
3. medium-density = 1620 culms/m², and
4. high-density = 2936 culms/m².

The relatively high plant density was used for simulating the effects of dense emergent macrophyte bands in real wetlands. In addition, higher vegetation densities can magnify their effects in the laboratory tests. V-notch weirs were installed at the inlet of each cell for accurate measurement of flow (Figure 3-4). Removable gates at the outlet of the cells were installed to adjust the water depth, targeting a depth of *h* = 0.05 - 0.06 m. An outlet weir was also constructed at the end of each cell for sample collection as shown in Figure 3-5.

To assure that a constant concentration of sediment can be achieved in the inflow into each cell, sediment feeders were designed as shown in Figure 3-4. They were built from four cups, each installed in front of one V-notch weir to allow continuous manual injection of slurry into inflow. The idea was to inject consistently slurry of the same concentration at the same short intervals using this simple system, and therefore assure that inflow concentration be as constant as possible (something that has been a challenge in many studies on sediment transport). Mixing of inflow water and sediment was helped by funnels that directed inflow onto the cell front wall (Figure 3-4). Although this substantially reduced the turbulence created at the start of the cell by incoming water (i.e. remove un-realistic laboratory conditions), baffles were placed close to the inflow for further turbulence reduction, as shown in Figure 3-1-b and Figure 3-4.
Tubes for siphoning water samples were installed along the cells at 0.5 m and 1 m from the inlet weirs as shown in Figure 3-1 and Figure 3-6 for wet weather experiments. The point at 0.5 m from the inflow was taken as ‘the starting point’ of the experimental flume, allowing the first 0.5 m for flow mixing (establishment of uniform flow and sediment profiles). The samples were only collected at the outflow weirs for dry weather experiments. This was due to the fact that the dry weather flow rates were so low that extracting intermediate samples might result in inadequate outflow at outlet for sampling and probably affect sedimentation in the flumes (e.g. causing re-suspension) in such a shallow and low flow. A similar technique has been tested and used in the past (e.g. Deletic, 2005).
Figure 3-2: Daylight lamps for supporting plant growth.

Figure 3-3: Four mesocosm cells with different vegetation densities.

Figure 3-4: 30° inlet V-notch weirs for controlling inflows, cups for manual injection of slurry every two minutes, semi-funnels for mixing and directing flow into cells, and baffles for reduction of inflow turbulence.
3.3 Experimental procedure and programme

The experimental procedure and programme for both wet and dry weather experiments are explained in this section. The methods for analysis of samples are also presented, along with sample measurement uncertainties.

3.3.1 The experimental procedure

Sediment preparation

The sediment used in the experiments was collected from a nearby stormwater retarding
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basin and sieved through a 300 µm sieve to remove gross pollutants. This was done in wet to obtain slurry, and then the slurry was diluted to a given TSS concentration as the experimental input. In Figure 3-7, the mean PSD curve of samples collected at the starting point (0.5 m from the inflow) can be seen to be very close to the typical PSD of developing catchments in Melbourne and Brisbane, (CRCCH, 2005), and to the PSD measured in stormwater in Scotland (Deletic and Orr, 2003). The input median particle size, \( d_{50} \), varies from 20 to 56 µm, and therefore is within the natural range of stormwater sediment. Similarly, the density of the particles is within 2490 to 2540 kg/m³ (similar to the values found in field, Deletic and Orr, 2003).

![Figure 3-7: The comparison of the mean particle size distribution used in the mesocosm experiments with documented PSDs from the literature.](image)

**Wet weather experiments**

Wet weather experiments were accomplished first to achieve steady-state conditions for better simulating the real constructed stormwater wetlands in the rig set as shown in Figure 3-1. At the start of each wet weather experiment, steady flow was established prior to injection of sediment. This was done by controlling the valve located between the constant-head tank and inlet pool to keep water level constant in the inlet pool. The mean flow depth and the flow rate were measured for each cell. A known mass of slurry of known TSS concentration was added every two minutes over 2 hours for low flows, and over 1.2
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hours for high flows. Water sampling was started only after steady sediment concentrations were established along the cells (determined in a pilot study). Samples were then taken every 15 minutes for the low flows and 10 minutes for the high flows. Samples were collected by siphoning tubes at 0.5, 1.0 m from the inlet weirs, and at the outlet weirs (2 m) by simple collection of the outflow. Two bottles of samples were collected at each location and time; one for TSS analysis and another for PSD analysis.

Dry weather experiments

Each dry weather experiment was undertaken immediately following the preceding wet weather experiment (except for the first pilot wet weather experiment). After stopping the continuous dosing of slurry, the inflow rate was immediately reduced to a low flow rate in accordance with dry weather programme. It took about two minutes to bring the water level down to the steady dry weather depth in the flumes. Only clean water was thereafter introduced into the flumes and the injection of clean water lasted four hours under steady flow conditions. At the beginning of dry weather experiments, each flume contained the suspended sediments remaining in the water column and pre-deposited sediments on the bed from the previous wet weather experiments. The first dry weather sample was taken five minutes later after the start of injection of clean water at a distance of 2.0 m to the inlet (i.e. the outlet weir). Afterwards, samples were taken at 10, 20, 30, 45, 60, 100, 160, 240 minutes. Water samples were analysed for both TSS concentration and PSD. The actual flow rate of each cell could be obtained by measuring outflow volume in a certain time. The flow depths were measured at several positions along each flume in order to calculate the mean depth.

3.3.2 The experimental programme

Wet weather experiments

Table 3-2 summarizes the wet weather experimental variables selected for the experimental programme: two different hydraulic loading rates (i.e. flows), four vegetation densities, and three sediment concentrations at the starting point (0.5 m from the inlet), classified based on the recorded variables for all wet weather experiments in Appendix A1. It should be noted
that the hydraulic loading rates were at the upper bound of the ranges found in real conditions (Duncan, 1998). This was mainly due to the limitations of the existing rig, as previously described (it was impossible to regulate reliably flows lower than those used), but also because we wanted to test the system for the “worst” conditions. Up to three experiments were carried out for each combination of similar experimental variables. It was, however, very difficult to replicate sediment concentrations at the starting point due to unpredictable sedimentation at the upstream of this point. In total, 40 experiments were carried out, grouped in five classes according to hydraulic loading and inflow concentration: LF-LC, LF-MC, HF-LC, HF-MC and HF-HC (where L stands for Low, M for Medium, H for High, F for Flow, C for Concentration, as explained in Table 3-2).

Table 3-2: Wet weather experimental variables and number of experiments done for each combination of variables.

<table>
<thead>
<tr>
<th>Numbers of Experiments</th>
<th>Vegetation density (culms/m²)</th>
<th>Sediment concentration at 0.5 m from the inflow (mg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>LC = 30-110</td>
</tr>
<tr>
<td>LF = 2000</td>
<td>High = 2936</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>Medium = 1620</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>Low = 590</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>No vegetation = 0</td>
<td>2</td>
</tr>
<tr>
<td>HF = 7000</td>
<td>High = 2936</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>Medium = 1620</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>Low = 590</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>No vegetation = 0</td>
<td>1</td>
</tr>
</tbody>
</table>

Dry weather experiments

Table 3-3 summarizes the dry weather experimental variables (see further details in Appendix A2) selected for the experimental programmes: two different hydraulic loading rates, four vegetation densities, and three mean wet weather sediment concentrations in the flumes produced from the preceding wet weather experiments. In total, 36 valid dry weather experiments were completed, grouped in five classes according to hydraulic loading and mean wet weather sediment concentrations in the flumes: LF-LC, LF-MC, HF-LC, HF-MC and HF-HC.
Table 3-3: Dry weather experimental variables and number of experiments done for each combination of variables (including preceding mean wet weather sediment concentration in the flume).

<table>
<thead>
<tr>
<th>Number of experiments</th>
<th>Vegetation density (culms/m²)</th>
<th>Mean wet weather sediment concentration in the flume (mg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>LC = 30-100</td>
</tr>
<tr>
<td>LF = 300</td>
<td>High = 2936</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>Medium = 1620</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>Low = 590</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>No vegetation = 0</td>
<td>1</td>
</tr>
<tr>
<td>HF = 500</td>
<td>High = 2936</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>Medium = 1620</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>Low = 590</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>No vegetation = 0</td>
<td>1</td>
</tr>
</tbody>
</table>

3.3.3 Sample analysis and measurement uncertainties

Sediment concentration measurement

All samples were analysed for TSS using standard methods (APHA/AWWA/WPCF 1998). The apparatus used for the analysis in the lab is shown in Figure 3-8. A well-mixed portion of a water sample with a known volume was filtered through a dried and pre-weighed glass-fibre filter paper. This siphoned filter was dried at 103 – 105 °C overnight and its weight then was measured several times by a balance so as to find an averaged weight (the error is less than 0.0001 g). Since the sample volume was recorded prior to drying, the concentration could be determined by dividing the measured weight by the fluid volume, that is,

$$ C = \frac{(U_{sample} - U_{filter})}{U_{volume}} \text{ (mg/L)} \quad (3.11) $$

where $U_{sample}$ is the mass of the filter and residue (mg), $U_{filter}$ the mass of the filter (mg), and $U_{volume}$ the volume of sample (L).

There are some uncertainties affecting the measurement accuracy of TSS concentrations,
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which can be assessed using standard laboratory protocol. The sources for relative errors might well come from the following four important factors (Hines, 2005):

1. Mass \( U_{mass} \) – mass of the filters weighed by an analytical 4-decimal place balance with ±0.03 mg linearity,
2. Volume \( U_{volume} \) – sample volume measured using a cylinder,
3. Method recovery \( U_{rm} \) – variations in sample (sediment) recovery from the method used, and
4. Sample homogeneity \( U_{dup} \) – variations in measuring the same water sample.

These factors combine with each other to create an overall measurement accuracy. The contribution of individual factor to measurement uncertainties has been discussed by Hines (2005) for the above measurement procedure. In particular, for the specified concentration \( C_m \), the individual uncertainty \( U_{cm} \), is given by the sum of relative standard deviations as follows,

\[
U_{cm} = C_m \sqrt{\left(\frac{U_{mass}}{mass}\right)^2 + \left(\frac{U_{volume}}{volume}\right)^2 + \left(\frac{U_{rm}}{rm}\right)^2 + \left(\frac{U_{dup}}{dup}\right)^2} \tag{3.12}
\]

Where mass and volume refer to sample’s mass and volume; \( rm \) and \( dup \), both equal to unity, are recovery of method and sample homogeneity. This equation provides a tolerance for measurement accuracy. It has been found by Hines (2005) that after combining all sources of uncertainties, the overall uncertainty of TSS of a sample is ± 3.8% of the measured concentration (including a coverage factor of 2), giving an approximately 95% confidence interval.

Results were also corrected for all recognized systematic biases. The relative error was determined using the pre-calibrated macerated filtered paper as an internal reference material. For each group of twenty samples or less, two blanks, two duplicate samples and two internal reference materials were measured as well. Only if the measured TSS for the reference material lay within ± 3.8% of the given concentration, were the results accepted. In addition, duplicate samples were analysed to check if the relative error was too large. The
following equation was used to calculate the difference between two results in the percentage form,

\[ P_m = \left[ 100 - \frac{100(m_1 - m_2)}{m} \right] \% \]  (3.13)

where \( m_1 \) and \( m_2 \) are the duplicate results and \( m \) the mean of the two results. The results were accepted if for low-level duplicates (<10 mg/L) \( P_m \) lay between 25 and 125%; and for high-level duplicates (>10 mg/L) \( P_m \) lay between 90 and 110%.

Sampling uncertainties

The accuracy of TSS measurements depended on the analytical method used and the uniformity of the sediment concentrations across a cell profile, since some samples were siphoned from a single point in one cross-section.

However, the uncertainty due to the fact that some samples were extracted from a single point in the cross-sections (always in the centre of the profile) had to be investigated independently. For five different combinations of the experimental variables (LF-LC, LF-MC, HF-LC, HF-MC and HF-HC) of each cell, samples were siphoned simultaneously from five points of a cross-section at 0.5 and 1 m from the inlet weir. They were analysed for TSS, and the mean TSS of this cross-section was calculated. For all cases of the point at 1 m from the inlet, the mid-point TSS (i.e. the sample taken at the middle of the cross-section,
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which was to be used in normal sampling) was always within ±10% in variance of the mean TSS concentration across the section. For those at 0.5 m from the inlet, their relative error was less than 15%, except for the cell with high vegetation density, (the mid-point TSS was around 23% higher than the mean value for all five experiments). This was likely due to the fact that the vegetation in the upstream up to the 0.5 m point was creating a certain flow pattern to produce consistently this relative error.

It was concluded that for all cases except that at 0.5 m from the inlet in the high-density vegetation cell, TSS measurement from the mid-point of a cross-section was adequately accurate. However, it was decided that for the high-density vegetation cell, the sediment concentrations at 0.5 m point were adjusted to 77% of the measured, to account for the systematic bias.

**Particle size analysis**

Particle sizes were measured using a Malvern MasterSizer/E particle sizer. Particle size analysis was only undertaken if the sample concentration was within an acceptable range for the instrument (some concentrations were too low to be measured). The principles of measurement of the instrument are based on the Mie theory of laser light scattering in the presence of particles in water. The main component of MasterSizer/E is the optical measurement unit, which includes (Figure 3-9):

- a transmitter (housing the laser, its power supplies and the beam expanding optics that creates
- the analyser beam),
- a receiver (housing the range lens, the detector and associated electronics and computer interfaces), and
- a sample presentation unit.

The Mie theory assumes that particles are spherical, and therefore the method has some built-in inaccuracies (although the manufacturer claims ±2% accuracy on volume median diameter, Malvern Ins Ltd, 1990). The instrument can measure particles in diameter from 0.1
µm to 600 µm in three ranges (0.1 – 80 µm, 0.5 – 180 µm and 1.2 – 600 µm) by using different range lens (45 mm, 100 mm and 300 mm). To cover the size range of all particles in samples (0 – 400 µm), the middle range lens was used. This might have caused a significant error in measurement of particles below 2 µm. However, to maximize the accuracy of measurements, at least three readings were taken for each sample, and their mean PSD was calculated.

Particle density measurement

Particle density was measured using a fully-automatic gas displacement AccuPyc 1330 Pycnometer for a small amount of dry sample with known mass. The pycnometer is comprised of a keypad, a display area, and an analysis cell chamber. It determines density and volume by measuring the pressure change of helium in a calibrated volume. Samples taken at 0.5 m from the inlet weir in four mesocosm flumes were dried in the oven overnight. Several grams (1 – 100 cm$^3$) of each sample were pre-weighed and loaded into the cell chamber. The instrument measures sample volume, from which density can be derived automatically by the pycnometer once the sample weight has been entered. The AccuPyc 1330 Pycnometer is controlled by commands entered through the keypad. Its unique feature may increase the precision of analysis results by reporting data from five consecutive runs that are within a user-specified tolerance (Micromeritics Ins Corp, 1992). The manufacturer
declares the error of the instrument is within 0.03% of reading plus 0.03% of the sample capacity.

### 3.3.4 Data analysis

Data collected from the laboratory experiments during both the wet and dry weather periods were analysed graphically and statistically in this study.

#### General trends

Since the sediment concentrations and the particle size distributions were measured along the mesocosm flumes over time for both wet and dry weather experiments, changes in TSS and PSD along the cells were plotted over distance and time.

Mean TSS concentrations and PSDs were calculated at each measuring point for each wet weather experiment. Variances around the TSS means were also calculated to evaluate whether or not steady-state conditions were achieved for wet weather experiments.

The measured mean TSS and PSD at each point were used to calculate concentrations of five different particle size fractions (i.e. 0 - 6, 6 - 21, 21 - 46, 46 - 124 and 124 - 404 µm) for further analyses. It was assumed that the particle density was the same for all fractions in one sample (as in Deletic, 2005), and thus, the fractional particle concentration at each sampling point was calculated by multiplying the observed sediment concentration by the particle size distribution measured at the same location and time.

#### Determination of significant variables

The aim of data analysis was to determine the key variables that impact on processes in both wet and dry weather. During wet weather experiments, the trapping efficiency of particle size fraction \( s \), \( T r_s \), was calculated for three segments along the cell in terms of the sampling points, that is, between (1) 0.5 m and 1 m, (2) 0.5 m and 2 m, and (3) 1 m and 2 m from the inlet weir, using the following equation:

\[
T r_s = (C_{in,s} - C_{out,s}) / C_{in,s}
\]

(3.14)
where $C_{in,s}$ is the input concentration of fraction $s$ into the segment (mg/L), $C_{out,s}$ the output concentration of fraction $s$ from the segment (mg/L).

For **dry weather experiments**, since steady-state conditions had been achieved during the preceding wet weather experiment, the measured mean values of TSS and PSD at each sampling point were used to calculate the mean wet weather concentrations of five different particle size fractions, as the initial condition for each flume. Since wet weather samples along the flumes were first taken at a distance of 0.5 m from the inlet weirs to minimize the effect of the inflow turbulence, it was assumed that sediment and flow were mixed thoroughly in the first-half meter zone so that sediment concentrations inside it were considered to be the same as that at the half-meter point. In addition, the measured wet weather sediment concentrations were assumed to vary linearly between two measuring points for each particle size fraction, and then the mean wet weather sediment concentration of each particle size fraction, $C_{\text{mean(wet)},s}$, could be worked out by summing the weight of sediments over the flume and dividing it by the flume length ($= 2$ m). The measured dry weather TSS concentrations and PSDs at the outlet at each time were also used to calculate the dry weather output sediment concentration $C_{\text{out(dry)},s}$ of each size fraction. The trapping efficiency of fraction $s$, $Tr_s$, in dry weather systems was defined as

$$Tr_s = \frac{(C_{\text{mean(wet)},s} - C_{\text{out(dry)},s})/C_{\text{mean(wet)},s}}{1 - \frac{C_{\text{out(dry)},s}}{C_{\text{mean(wet)},s}}}$$

(3.15)

Relationships were examined between particle trapping efficiency and following influencing factors:

- hydraulic loading,
- sediment particle size,
- flow depth,
- flow velocity,
- shear velocity,
- detention time,
- inflow concentration,
Laboratory mesocosm wetland study

- plant density,
- three non-dimensional numbers: $R_e^*$, $N_f$ and $R_e r$. (as defined in Equation 3.1, Equation 3.3, Equation 3.7, respectively), and
- for dry weather data, the mean wet weather sediment concentration in the flume (i.e. the mean TSS concentration in the flume prior to dry weather experiments) and the time ratio parameter $t^*$ (defined in Equation 3.5).

Statistical analysis used Spearman correlation coefficients, $r_s$, and corresponding $p$-values (Hinton et al., 2004) to identify the key factors affecting sediment trapping efficiency and test the hypotheses. Spearman’s correlation was used in preference to Pearson’s correlation to overcome the effects of irregularly distributed data. The importance of re-suspension was also analysed using the well-known Modified Shields Diagram for sediment mobilisation in flow (Govers, 1987).

3.4 Results and discussion

The total results of measured TSS concentration over time for all ten groups of experiments (including 40 wet weather experiments and 36 dry weather experiments, although Experiment 1 had no records under dry weather) are provided in Appendix A3. The available measured cumulative PSDs for the wet and dry experiments are presented in Appendices A5 and A6, respectively, showing variations of observed TSS and PSD along the cells over time. It should be noted that the flow rate (hydraulic loading) of each flume was slightly different across four mesocosms owing to the different positioning of V-notch weirs.

3.4.1 Wet weather experiments

General trends

The mean TSS concentrations and PSDs were obtained for each measuring point of each experiment. A small variance around the mean TSS concentration indicates that steady-state conditions were achieved during the experiment. The change in TSS over the duration of Experiment 9 for LF–MC Experiment (see Table 3-2 for the experiment description) is presented in Figure 3-10 for the four mesocosm cells. It is clear that steady-state
concentrations have been achieved. This is also obvious in other experiments as shown in Appendix A3, because the variance of the mean TSS of each experiment is within 3 to 18% of the mean values of all experiments. It is evident that TSS concentrations decrease significantly over distance along the cells, showing that deposition is an important process.

Figure 3-10: TSS concentrations versus time and sampling distance for Experiment 9 (LF-MC): (a) Cell with high vegetation density (b) Cell with medium vegetation density (c) Cell with low vegetation density (d) Cell without vegetation.

Figure 3-11 presents examples of the measured cumulative PSD curves at different sampling points and times in four mesocosm cells for Experiment 9, confirming the attainment of steady-state conditions (variations in PSD with time at any point along the cells are small). The cumulative PSD curves also shift to the left with distance, showing a decrease in particle sizes along the cell associated with deposition. This is noticeable in all measured cumulative PSD graphs shown in Appendix A5. As expected, along each cell, the larger particles are removed at a higher rate than the smaller particles, and the median particle size $d_{50}$ thus
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gradually decreases over distance. The range of particle sizes, which is wide at the distance of 0.5 m from the inlet weir, is reduced to a narrow distribution containing only small particles at the outlet (2.0 m).

Since steady-state conditions are achieved, it is safe to average measured TSS and PSD over each experiment and then use the mean concentrations (of the individual particle size fractions and total TSS) in further analyses. For example, the mean fraction concentrations during Experiment 9 in four mesocosms are plotted against sampling distance in Figure 3-12. It is clear that there is an exponential decrease in concentrations along each flume (it follows the first-order decay model), with the decay coefficient being highly dependent on particle size. Similar trends can be found in other experiments as shown in Appendix A4. It can be concluded that the distance and particle size are important factors in sediment transport. These findings are entirely consistent with previous studies (Kadlec and Knight, 1996; Wong et al., 2006).

Figure 3-11: The cumulative PSD curves over time and sampling distance during Experiment 9 (LF-MC) in four mesocosm cells.
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Experiment 9 (LF-MC) High-vegetated Cell
- 0-6 µm
- 6-21 µm
- 21-46 µm
- 46-124 µm
- 124-404 µm

Experiment 9 (LF-MC) Medium-vegetated Cell
- 0-6 µm
- 6-21 µm
- 21-46 µm
- 46-124 µm
- 124-404 µm

Experiment 9 (LF-MC) Low-vegetated Cell
- 0-6 µm
- 6-21 µm
- 21-46 µm
- 46-124 µm
- 124-404 µm

Experiment 9 (LF-MC) Non-vegetated Cell
- 0-6 µm
- 6-21 µm
- 21-46 µm
- 46-124 µm
- 124-404 µm

Figure 3-12: Results from Experiment 9 (LF-MC): the mean fraction concentrations over experiment versus distance in four mesocosm cells.

Determination of significant variables

Table 3-4 presents the results of the Spearman correlation analyses between the trapping efficiency, $Tr_s$, and explanatory variables. In total 465 data points have been used in this analysis. In this table, the variables are listed according to the values of absolute $r_s$ in a descending order, in other words, a decreasing importance of the variable for the particle trapping process. For example, Particle Fall Number (No. 1) has $r_s = 0.92$ and therefore explains 92% of the variation in observed TSS concentrations, while vegetation density (No. 13) does not significantly influence $Tr_s (r_s = -0.01)$. The low $p$-value (i.e. indicating a high level of significance) observed for all correlations but vegetation density, is in part a function of the amount of points used in the analysis (465). Results from the non-vegetated cell need to be regarded with some caution since typical values of $R_{eT}$ were not replicated in the mesocosm.
Table 3-4: Spearman correlation statistics between the trapping efficiency, $Tr_s$, and the tested variables based on 465 measured data points.

<table>
<thead>
<tr>
<th>No</th>
<th>Variable</th>
<th>Correlation Coefficient, $r_s$</th>
<th>p-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Particle Fall Number, $N_f$</td>
<td>0.92</td>
<td>0</td>
</tr>
<tr>
<td>2</td>
<td>Particle fall velocity</td>
<td>0.86</td>
<td>0</td>
</tr>
<tr>
<td>3</td>
<td>Particle size at the input</td>
<td>0.86</td>
<td>0</td>
</tr>
<tr>
<td>4</td>
<td>Particle Shear Velocity Reynolds Number $R_e$*</td>
<td>0.77</td>
<td>0</td>
</tr>
<tr>
<td>5</td>
<td>Hydraulic loading</td>
<td>-0.31</td>
<td>0</td>
</tr>
<tr>
<td>6</td>
<td>Detention time</td>
<td>0.28</td>
<td>0</td>
</tr>
<tr>
<td>7</td>
<td>Input sediment concentration</td>
<td>-0.28</td>
<td>0</td>
</tr>
<tr>
<td>8</td>
<td>Flow rate</td>
<td>-0.19</td>
<td>0</td>
</tr>
<tr>
<td>9</td>
<td>Water depth</td>
<td>-0.18</td>
<td>0</td>
</tr>
<tr>
<td>10</td>
<td>Flow velocity</td>
<td>-0.16</td>
<td>0.001</td>
</tr>
<tr>
<td>11</td>
<td>Shear velocity</td>
<td>-0.15</td>
<td>0.001</td>
</tr>
<tr>
<td>12</td>
<td>Turbulent Reynolds Number, $Re_T$</td>
<td>-0.12</td>
<td>0.008</td>
</tr>
<tr>
<td>13</td>
<td>Vegetation density</td>
<td>-0.01</td>
<td>0.835</td>
</tr>
</tbody>
</table>

The most important findings from this analysis are summarised below:

(1) Vegetation density does not play any role in total sediment deposition in wetlands during wet weather. As discussed in the introduction, this has been found previously for freshwater wetlands (Brueske and Barret, 1994; Fennessy et al., 1994; Leonard et al., 2002), but has been disputed for stormwater wetlands (Wong et al., 2000; WERF, 2005) and constructed wetland ponds (Schmid, et al., 2005). Some studies suggested that an increase in vegetation density may remove finer particles more efficiently in the long term due to filtration and other physical, chemical and biological processes (Breen, 1990; WERF, 2005). Unfortunately, we were not able to quantify the effect of vegetation on these very fine particles, because they presented only a very small fraction of the TSS, and the inherent experimental error therefore prevented us from assessing accurately their concentrations.

(2) Flow characteristics, such as flow velocity, depth and rate, have some influences, e.g., for higher flow rates yield a lower $Tr_s$. Finally, hydraulic loading (a combined measure of flow rate and length available for deposition), has a relatively weak influence ($r_s = -0.31$), with a similar influence of detention time (the ratio between
pool volume and flow rate).

(3) Input concentration has a relatively small influence ($r_s = -0.28$) on sediment trapping efficiency, $Tr_s$, despite contrary predictions in earlier studies (Duncan, 1998; WERF, 2005).

(4) Particle size is clearly the most important single variable, accounting for 86% of the variance in observed TSS concentration.

(5) Of the three non-dimensional numbers studied, $Re_T$ had no significant influence while $Re^*$ and $N_f$ were both highly significant.

Testing the hypotheses for wet weather events

**Hypothesis 1** - The strong effect of $Re^*$ on the trapping efficiency ($r_s = 0.77$) could be entirely due to the importance of particle size ($r_s = 0.86$), because the shear flow velocity, $U^*$, was not very significant ($r_s = -0.15$). Since particle size is so important for the deposition process, conclusions about the relevance of the re-suspension process could not be easily reached. It was therefore decided to evaluate the importance of this process further, by using the general sediment transport theory and in particular Shield’s diagram (Govers, 1987; Yang, 1996).

To broaden the scope of the lab study, $Re^*$ was calculated for operational conditions which are typically designed for wetlands in Australian practices. According to current Australian design guidelines, the design mean flow velocity, $V$, is below 0.05 m/s, while during wet weather it can range from 0.02 m/s to 0.1 m/s (Wong and Breen, 2002). The water depth, $H$, in wetlands is typically between 0.15 m and 0.75 m, while for ponds it is between 1 m and 2 m (Kadlec, 2005). Using these conditions, $Re^*$ values were calculated for a range of stormwater sediment particles (the same range as used in the scaling of the mesocosms).

These values are then plotted as against their matching Densimetric Froude Number $\rho_w U^2 / \rho g d_s$, as shown in Figure 3-13. It is clear that almost all calculated points were below the modified Shield’s critical curve, with only few points at a particle size of 1 µm and under the highest allowed velocity (0.1 m/s) being above the critical curve. This clearly indicates that re-suspension is unlikely to be a governing process in real wetlands.
From both the laboratory study and the Shield’s diagram it was concluded that re-suspension of particles from the system’s bed can be neglected, even for very high flow rates that could occur in well designed wetlands during wet weather. However, it is possible that re-suspension can be important in wetlands and ponds with no bypass under large floods (i.e. where velocities are uncontrolled), which is beyond of our scope. In practice, a variety of other factors than flow can cause sediment re-suspension in wetlands, such as wind-driven turbulence, water level fluctuations (e.g., drawdown), animals, and gas release from the bed (unfortunately it is very hard to model these activities in a laboratory system).

**Hypothesis 3** – The wash-off process appears not to play a big role during wet weather since flow turbulence does not significantly influence sediment transport (i.e. Turbulent Reynolds Number $R_{e^T}$ explained only 12% of the variance in TSS concentrations). This supports our third hypothesis, although the results should be taken with caution for non-vegetated systems; since we were unable to meet the scaling requirement for $R_{e^T}$ in the mesocosm non-vegetated pond (this could further be re-examined in Chapter 4).

**Hypothesis 2** - Deposition has been proven to be the main process that governs sediment behaviour in wetlands. The Particle Fall Number, $N_f$, had the far highest $r_s$ in all studied variables (0.92). This is in agreement with the Hazen settling theory (Hazen, 1904), as well
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as findings from a similar laboratory study on stormwater sediment behaviour in grass filters (Deletic, 2005).

### 3.4.2 Dry weather experiments

**General trends**

The change in TSS over the duration of Experiment 8 (LF-MC in Table 3-3) is shown in Figure 3-14 for all four mesocosm cells. The evolutions of TSS concentrations measured at outlet weirs for the dry weather experiments, are presented in Appendix A3.

![Figure 3-14: TSS concentrations versus time and sampling distance for Experiment 8 (LF-MC) (dry weather experiment started at time \( t = 125 \) min).](image)

It was found that the outlet TSS concentrations decrease rapidly in an exponential manner with time during dry weather experiments. In addition, the sediment concentrations at the outlets in most cases were reduced to less than 1.0 mg/L in 100 minutes after starting the dry weather experiments, regardless of the level of the input sediment concentrations inherent
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from wet weather experiments. This demonstrates a significant removal of sediment associated with deposition. On the other hand, the effect of sediment re-suspension on trapping efficiency under dry weather flow condition is largely insignificant since the concentration of fine particles also decreases with time as found in Figure 3-14.

Figure 3-15 shows an example of the measured cumulative PSD curves at different sampling points and times for both the preceding wet weather and the following dry weather experiments (after 100 minutes the sediment concentrations were too low to measure PSD) for Experiment 6 (HF-HC) in the highly vegetated flume. The accumulated PSD curves at the outlet (2 m) under dry weather conditions shift to the left with time due to sediment deposition. This implies that the median particle sizes $d_{50}$ at the outlet are gradually reduced with time as the removal of the large-size particles. In addition, the PSDs in other three different flumes have similar but slightly varied responses, as shown in Appendix A6.

Figure 3-15: The cumulative PSD curves over time and sampling distance for high-vegetated cell during Experiment 6 (HF-HC); Wet-wet experiment; Dry-dry experiment; X-distance from the inflow V-notches; t-dry weather sampling time.

Determination of significant variables

The results of statistical analysis are presented in Table 3-5, based on 929 valid sample points. In this table, the variables are listed in order according to the descending absolute value of $r$. Similar to wet weather experiments, results from the non-vegetated cell need to
be treated with caution since typical values of $R_{cT}$ were not replicated. The main findings from this statistical analysis are summarized below:

(1) Wet weather input particle size has a significant influence on TSS concentrations, with $r_s = 0.72$ and $p < 0.001$. It is clearly demonstrated that inflow sediment particle size is the most important variable, accounting for 72% of the variance in observed dry weather TSS concentrations. Related particle fall velocity is a crucial factor to particle settling, and thus greatly affects the sediment behaviours under both wet and dry weather conditions. This is consistent with the reported results in previous studies.

(2) Flow characteristics such as hydraulic loading, are not strongly correlated with sediment trapping efficiency. Hydraulic loading has a correlation coefficient $r_s$ of 0.08, and a $p$ value of 0.013, as in Table 3-5. It is also shown that flow rate, flow velocity, mean water detention time and water depth have no meaningful influence on trapping efficiency ($r_s = -0.08, 0.07, -0.07, 0.05$ respectively), even though some of their $p$-values are less than 0.05. The possible explanation for this is that very low hydraulic loadings and flow rates under dry weather conditions provide adequate detention times for particle settling. In other words, detention times are at their maximum (Wong and Breen, 2002). Thus detention time, flow velocity and water depth become unimportant for sediment deposition under dry weather conditions.

(3) Despite the statistically insignificant correlation between observed dry weather sediment trapping and wet weather inflow sediment concentration ($r_s = -0.04, p = 0.266$), negative correlations are noticed for some variables such as wet weather mean sediment concentration ($r_s = -0.12, p < 0.001$).

(4) Based on the results of $r_s = 0.02, p = 0.563$, the effects of vegetation density do not give rise to significant difference in dry weather sediment trapping efficiency, the same observation as in the wet weather experiments. Meanwhile, the Turbulent Reynolds Number also does not prove to be of significant influence on sediment trapping efficiency as assumed, since its correlation coefficient is small ($r_s = 0.01$), and the $p$ value is 0.865.

(5) The time ratio $t^*$ reflects the ideal estimation of water replacement in the system with
time. The statistical analysis shows that $t^*$ plays a significant role on trapping efficiency ($r_s = 0.56, p < 0.001$) and is one of the key factors controlling the sediment trapping under dry weather conditions.

(6) Another strongly correlated variable is found to be the Particle Fall Number with a correlation test result of $r_s = 0.70$ and $p < 0.001$.

Table 3-5: Spearman correlation statistics between the trapping efficiency, $Tr$, and the tested variables based on 929 measured data points.

<table>
<thead>
<tr>
<th>No</th>
<th>Variable</th>
<th>Correlation Coefficient, $r_s$</th>
<th>p-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Particle fall velocity</td>
<td>0.72</td>
<td>0</td>
</tr>
<tr>
<td>2</td>
<td>Wet weather particle size at the input</td>
<td>0.72</td>
<td>0</td>
</tr>
<tr>
<td>3</td>
<td>Particle Shear Velocity Reynolds Number, $Re^*$</td>
<td>0.71</td>
<td>0</td>
</tr>
<tr>
<td>4</td>
<td>Particle Fall Number, $N_f$</td>
<td>0.7</td>
<td>0</td>
</tr>
<tr>
<td>5</td>
<td>Time ratio, $t^*$ (Time / Mean detention time)</td>
<td>0.56</td>
<td>0</td>
</tr>
<tr>
<td>6</td>
<td>Hydraulic loading</td>
<td>0.08</td>
<td>0.013</td>
</tr>
<tr>
<td>7</td>
<td>Flow rate</td>
<td>0.08</td>
<td>0.013</td>
</tr>
<tr>
<td>8</td>
<td>Flow velocity</td>
<td>0.07</td>
<td>0.037</td>
</tr>
<tr>
<td>9</td>
<td>Shear velocity</td>
<td>0.06</td>
<td>0.052</td>
</tr>
<tr>
<td>10</td>
<td>Water depth</td>
<td>0.05</td>
<td>0.139</td>
</tr>
<tr>
<td>11</td>
<td>Vegetation density</td>
<td>0.02</td>
<td>0.563</td>
</tr>
<tr>
<td>12</td>
<td>Turbulent Reynolds Number, $Re_{Te}$</td>
<td>0.01</td>
<td>0.865</td>
</tr>
<tr>
<td>13</td>
<td>Wet weather input sediment concentration</td>
<td>-0.04</td>
<td>0.266</td>
</tr>
<tr>
<td>14</td>
<td>Detention time</td>
<td>-0.07</td>
<td>0.044</td>
</tr>
<tr>
<td>15</td>
<td>Wet weather mean sediment concentration</td>
<td>-0.12</td>
<td>0</td>
</tr>
</tbody>
</table>

However, it has to be noted again that this study was performed under somewhat idealized conditions, where real life factors, like birdlife disturbance of sediment or wind-induced water mixing, have not been considered. Therefore the findings are relevant for processes related to water flow only.

Testing the hypotheses for dry weather events

Hypothesis 1 –The importance of $Re^*$ for the trapping efficiency ($r_s = 0.71$) appears to be largely or entirely due to the importance of particle size ($r_s = 0.72$), because the shear flow velocity, $U^*$, is not very important ($r_s = 0.06$). As particle size associated with sediment deposition dominates over other processes, the re-suspension process can be neglected as
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found for wet weather cases.

**Hypothesis 3** – The wash-off process does not play a role during dry weather, since flow turbulence does not significantly influence sediment transport (i.e. Turbulent Reynolds Number $R_{er}$ explains only 1% of the variance in TSS concentrations). This is somewhat surprising, since very often outflow concentrations in real wetlands can be high during dry weather spells (Fletcher *et al.*, 2004). However, the results suggests that it is unlikely that the re-suspension is caused by processes related to flow through the system, but rather due to other disturbance, such as bird activity or wind.

The findings should also be accepted with caution for non-vegetated systems, since we were unable to obtain the same $R_{er}$ in the non-vegetated mesocosm pond as in reality.

**Hypothesis 2** - Deposition is the main process that governs sediment behaviour in wetlands. Based on the results of statistical analysis, the two dimensionless parameters, $N_f$ and $t^*$, have much higher values of $r_s$ than other studied variables (0.70, and 0.56, respectively). It must be noted that these two parameters involve many other influencing factors, such as Particle Fall Number, particle size, particle settling velocity, flow velocity, hydraulic loading, water detention time, particle settling time, geometry of the treatment system, even vegetation density, and $t^*$ (time and detention time during dry weather period). Therefore the particle trapping efficiency under dry weather conditions can be described as a function of $N_f$ and $t^*$ based on the results of statistical analyses.

### 3.5 Conclusions

The primary aim of this chapter was to examine sediment behaviour in constructed stormwater wetlands in order to gain further insights into the relationship between the variables of the treatment system and sediment trapping efficiency. Three mesocosm stormwater wetlands (vegetated with a well-established canopy of different densities) and one non-vegetated mesocosm pond were used, all scaled to achieve the assigned values of Particle Fall Number $N_f$, and Particle Shear Velocity Reynolds Number, $R_{er}$, found in real-world wetlands. The mesocosm vegetated systems also had values of Turbulent
Reynolds Numbers \((R_e_T)\) similar to those found in full-scale systems. To study the wash-off process in non-vegetated ponds, however, needs additional experiments with similar ranges of \(R_e_T\), and the results will be presented in Chapter 4.

In total, ten groups of steady-state experiments (including 40 wet weather experiments and 36 dry weather experiments) were carried out, at different hydraulic loadings and sediment inflow concentrations (also maintained within the ranges found in real systems during wet and dry weather). Samples were taken along the mesocosms and analysed for Total Suspended Solids concentrations (TSS) and Particle Size Distribution (PSD).

It has been confirmed by the experimental results that sediment concentration decreases exponentially over distance, with a decay coefficient being highly dependent on particle size. Particle trapping efficiency is independent of the density of vegetation, flow turbulence \((R_e_T)\) or Particle Shear Velocity Reynolds Number \((R_e^*)\) under both wet and dry weather conditions. Flow characteristics do have a significant influence on sediment removal during wet weather period (sediment trapping efficiency decreases with an increase in hydraulic loading), so do inflow concentrations to some extent (which was not expected). However, the particle size (or settling velocity) has been found, not surprisingly, to have the dominant influence. Therefore it is concluded that the Particle Fall Number is strongly correlated with the particle trapping efficiency. The time ratio \(t^*\) has a significant effect on particle trapping efficiency and is considered as another key factor that controls sediment trapping under dry weather conditions.

It was also concluded from the experimental results that re-suspension of bed sediment, and wash-off of suspended particles can be neglected in the laboratory tests, implying that deposition is the most important process. It should be emphasized that the findings obtained are only valid for flow-affected sediment removal, and the environmental factors such as bird activity and wind-driven turbulence have not been taken into account in the laboratory experiments.
Chapter 4

Laboratory Study of Sedimentation in Non-vegetated Ponds

4.1 Introduction

4.1.1 Background

This chapter presents a laboratory study undertaken to examine sediment behaviour in non-vegetated stormwater ponds, based on the findings of Chapter 3. The laboratory mesocosms were not fully suitable for studying sediment behaviour in non-vegetated stormwater ponds. Indeed only a full-scale model of a wetland can assure that the scaling theory is fully satisfied; i.e. the three dimensionless numbers that govern the processes are the same in the physical model as they are in real systems. As explained in Section 3.2.1, the non-vegetated mesocosm pond (one of four cells of the laboratory rig shown in Figure 3-1) was able to achieve a realistic Particle Fall Number $N_f$ and Shear Velocity Reynolds Number $R_e^*$ only, while Turbulent Reynolds Number $R_{eT}$ (calculated using Equation 3.9) was not within the realistic range. Hence, the findings reported so far on sediment behaviour in non-vegetated ponds require further examination.

This chapter therefore provides an overview of the laboratory non-vegetated stormwater pond experiments. It begins with a description of the experimental installation, including experimental scaling requirements and rig set-up. Following this, the experimental procedure and programme are outlined. The subsequent sections summarize and discuss the experimental results. Finally, the conclusions are presented.
4.1.2 Outline of the experimental methodology

The study was undertaken in an open flume under strictly controlled conditions, which was much larger in size than the mesocosm pond used in Chapter 3. Two different types of experiments were conducted in this open flume, as carried out for the previous mesocosm wetland experiments:

1. **Wet weather experiments** in order to test whether or not the findings in Chapter 3 are capable of being applied to open-water ponds during storm events.

2. **Dry weather experiments** which focus on dry weather sediment behaviour in open-water ponds.

Additionally, laboratory experiments were set up to meet the following conditions:

- Flow through the open channel mimics that in constructed stormwater non-vegetated ponds.
- A steady state in water flow and TSS concentration is reached.
- No infiltration is allowed, in order to examine sediment deposition and re-suspension solely.
- Artificial plastic granules are used as both input sediments and the bed load in the wet weather experiments to meet scaling requirements.

4.2 Experimental installation

The first step was to design the experimental rig, as well as the programme of experiments, taking into account the scaling requirements as discussed below.

4.2.1 Scaling issues

As in the mesocosm study, field data collected at the Hampton Park wetland were first used to estimate the realistic ranges of the three dimensionless numbers, as discussed in Section 3.2.1. Hydraulic loadings between 50 and 7000 m/y were used to cover the possible inflow range under wet and dry weather conditions. Table 4-1 shows the three
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Non-dimensional numbers as found in the Hampton Park’s non-vegetated ponds for particle size of $d_s = 69 \, \mu m$ (the median particle size of stormwater in Melbourne and Brisbane catchments as shown in Figure 3-7 and Figure 4-3 below, Lloyd et al., 1998; CRCCH, 2005) and particle density of 2520 kg/m$^3$.

Since it is only possible to achieve similarity in two out of the three dimensionless numbers at any scaled model of non-vegetated ponds (Section 3.2.1), it was decided that in this study sediment behaviour would be examined using a different set of dimensionless numbers than was used in the mesocosm study. In this way we can confirm that the findings of the mesocosm are not simply artifacts of ‘scaling’ issues in the mesocosm system. In the non-vegetated mesocosm ponds (Chapter 3), appropriate Particle Fall Number $N_f$ and Shear Velocity Reynolds Number $Re^*$ were achieved, while similarity for Turbulent Reynolds Number $Re_T$ was not reached. Therefore in this study, it was decided to investigate sediment behaviour with a set-up that has realistic $N_f$ (its importance was demonstrated in Chapter 3) and $Re_T$ numbers.

An existing flume (located in the Hydraulics Lab at Monash University) was modified and used in this study. This rig has been regularly used to study sediment transport in open channels in the past. The size of the flume is given in Table 4-1 and a photograph shown in Figure 4-1. The experimental variables, (i.e. flow rate and sediment characteristics) were determined (Table 4-1) to achieve appropriate values for $N_f$ and $Re_T$. In order to achieve an appropriate $Re_T$, the hydraulic loading rates in the laboratory system had to be about 12 times higher than those in real systems (i.e., increasing flow rate so as to achieve a higher value of $Re_T$). Meanwhile, appropriate values of $N_f$ could be achieved for these high flow rates by increasing sediment particle settling velocities by 12 times. Therefore, an artificial sediment, which has very different properties than natural stormwater sediments, had to be used.

A type of plastic granules of approximately 1.8 mm in diameter and about 1095 kg/m$^3$ in density was tested for their ability to meet the particle settling velocity requirements. The settling velocities of 800 plastic granules were measured in a 1m-long cylinder. The results
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are shown in Figure 4-2. The comparison of the settling velocity distribution of the plastic granules with those of stormwater particles adopted from the literature is presented in Figure 4-3. It is clear that the plastic granules’ velocity distribution is within the velocity range of stormwater particles. Moreover, their median settling velocity $V_{s0}$ is 0.047 m/s, which is about 12 times that of a possible $V_{s0}$ for Melbourne and Brisbane catchments ($= 0.004$ m/s at an equivalent particle diameter $d_{50} = 69$ µm), and therefore appropriate $N_f$ values have been achieved. In conclusion, by increasing both hydraulic loading rates and particle settling velocities to 12 times of their observed values, the Flow Turbulence Number $R_{ef}$ and the Particle Fall Number $N_f$ in the laboratory well match with those in real systems (Table 4-1).

As expected, $Re^*$, which so far has been demonstrated not to be important in well designed stormwater wetlands and ponds (see Chapter 3), has far higher values in the laboratory than in the real systems (Table 4-2). To investigate the importance of $Re^*$, its properties are discussed below.

In the transition regime where $Re^*$ are within the range between 1.2 and 70, Yang (1996) suggested that the particle incipient motion criteria can be determined by the following equation,
\[
\frac{V_{cr}}{V_s} = \frac{2.5}{\log(R_e^*) - 0.06} + 0.66
\] (4.1)

where \(V_{cr}\) is the average critical flow velocity at incipient motion, and \(V_s\) the particle fall velocity.

![Figure 4-2: Experimental plastic granule setting velocity distribution.](image)

Figure 4-2: Experimental plastic granule setting velocity distribution.

![Figure 4-3: Particle settling velocity distribution in urban stormwater (after Lloyd et al., 1998; Deletic and Orr, 2003; CRCCH, 2005).](image)

Figure 4-3: Particle settling velocity distribution in urban stormwater (after Lloyd et al., 1998; Deletic and Orr, 2003; CRCCH, 2005).

Using Equation 4.1, the average critical flow velocity \(V_{cr}\) in the laboratory flume can be
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calculated as provided in Table 4-1. It shows that when simulating hydraulic loading rates are greater than 5000 m/y (which in our flume corresponds to rates of above 61720 m/y), the flow velocities in the flume are higher than their incipient motion criteria. This indicates that re-suspension and remobilization of bed load may occur in the flume at these high hydraulic loading rates. The direct implication for our laboratory study is that it probably is safe by confining $R_e^*$, at hydraulic loading rates in real systems less than 5000 m/y; i.e. only rates less than 61720 m/y in the laboratory flume can keep $R_e^*$ on the 'safe side'. This had to be taken into account in the experimental design, as discussed below.

4.2.2 Experimental rig

The cross-section and plan views of the experimental rig are shown with all dimensions in Figure 4-4. It contained a 1.2 m wide, 0.5 m deep, 12 m long flume (or an effective length 11.5 m for representing the non-vegetated pond) with a constant-head inflow tank, an outlet tank, flow controlling valves, flow meters and a water-recirculation system (including pump, underground sink, filter and cycling pipes). A 0.25 m-thick layer of the plastic granules (same as the input sediment) pre-saturated with water covered the bottom of the flume to serve as bed loads (Figure 4-5).

Water was injected from the storage (underground sink) to the inflow tank by a pump. The flow rate was controlled by a valve and measured by a flow meter installed in the inflow pipe (Figure 4-6). A perforated baffle covered with very fine metal mesh was placed between the upstream end of flume and the inflow tank for dissipating energy, distributing flow evenly, and intercepting plastic granules to maintain the cleanliness of the inflow water (Figure 4-7). A 1.2 m-long platform was installed under the perforated baffle at the start of the flume to reduce the turbulence to the bed sediment created by incoming water (i.e. remove boundary conditions introduced by the laboratory set-up) (Figure 4-4). A crest weir was also placed at the outlet of the flume for an easy collection of particles at any time. The removable gate at the outlet of the flume allowed the water depth in the flume to be adjusted, targeting a depth of $H = 0.15$m for wet weather conditions (Figure 4-8).
Table 4-1: Hampton Park non-vegetated pond characteristics for particle size, $d_s = 69 \mu m$ and density, $\rho_s = 2520 \text{ kg/m}^3$ and pilot-scaled non-vegetated flume characteristics for particle size, $d_s = 1.8 \text{ mm}$ and density, $\rho_s = 1095 \text{ kg/m}^3$.

<table>
<thead>
<tr>
<th>Flow conditions</th>
<th>Dry weather</th>
<th>Wet weather</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hampton Park wetland</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Hydraulic loading (m/y)</strong></td>
<td>50</td>
<td>1200</td>
</tr>
<tr>
<td>Size (m): length</td>
<td>120</td>
<td>120</td>
</tr>
<tr>
<td>width</td>
<td>70</td>
<td>70</td>
</tr>
<tr>
<td>depth</td>
<td>0.94</td>
<td>1.03</td>
</tr>
<tr>
<td><strong>Flow velocity (m/s)</strong></td>
<td>0.0002</td>
<td>0.0004</td>
</tr>
<tr>
<td>Particle Fall Number $N_f$</td>
<td>2469</td>
<td>1234</td>
</tr>
<tr>
<td>Flow Reynolds Number $Re_T$</td>
<td>185</td>
<td>370</td>
</tr>
<tr>
<td>Particle Shear Velocity Reynolds Number $Re^*$</td>
<td>0.001</td>
<td>0.002</td>
</tr>
<tr>
<td>Pilot scale system</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Hydraulic loading (m/y)</strong></td>
<td>630</td>
<td>1270</td>
</tr>
<tr>
<td>Size (m): length</td>
<td>11.5</td>
<td>11.5</td>
</tr>
<tr>
<td>width</td>
<td>1.2</td>
<td>1.2</td>
</tr>
<tr>
<td>depth</td>
<td>0.15</td>
<td>0.15</td>
</tr>
<tr>
<td><strong>Flow velocity (m/s)</strong></td>
<td>0.0002</td>
<td>0.0003</td>
</tr>
<tr>
<td>Particle Fall Number $N_f$</td>
<td>2358</td>
<td>1170</td>
</tr>
<tr>
<td>Flow Reynolds Number $Re_T$</td>
<td>184</td>
<td>370</td>
</tr>
<tr>
<td>Particle Shear Velocity Reynolds Number $Re^*$</td>
<td>0.26</td>
<td>0.46</td>
</tr>
<tr>
<td>Critical flow velocity for incipient motion (m/s)</td>
<td>–</td>
<td>–</td>
</tr>
</tbody>
</table>
Figure 4-4: Laboratory open-water pond experiment rig: (a) Plan view, (b) Cross-section ($S_n$ – sampling point).

Figure 4-5: 0.25 m-thick plastic granules acting as bed sediment in the flume.

The tubes for siphoning water samples were installed along the flume at distances of 0.7 m, 1.7 m, 2.7 m, 4.7 m, 6.7 m and 8.7 m downstream from the dosing point. The outflow of each siphoning tube was filtered through a sieve to capture particles as shown in Figure 4-9. Meanwhile, the suspended particles in the outflow were fully collected at the outlet weir within a given sampling time by a large textile-mesh tray (Figure 4-8).
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Figure 4-6: Flow meter and valve for measuring and controlling inflow rate.

Figure 4-7: Perforated baffle covered with fine mesh for generating a uniform flow and keeping the inflow clean.

Figure 4-8: Removable gate for controlling the water depth in the flume and large textile-mesh trap for collection of outflow particles.
4.3 Experimental procedure and programme

This section provides the details of the procedure and programme for the non-vegetated pond experiments undertaken for both wet and dry weather conditions. The methods for measurement of the required data are also described.

4.3.1 The experimental procedure

Two types of experiments (wet and dry weather) were carried out to investigate sediment behaviour under different hydraulic conditions in non-vegetated ponds. As discussed above, the flow rate and velocity had to be increased in order to achieve the appropriate values of $Re_T$ consistent with real field conditions. However, this resulted in a much higher value of $Re^*$ in the laboratory flume, as well as a higher flow velocity. The high velocity can result in incipient motion of bed sediments and force the bed to shift (see Equation 4-1).

We therefore had to test firstly the bed-load transport rate at different hydraulic loadings to confirm whether or not an increase in flow velocity would affect the results of wet weather experiments. The experimental study included the following three types of experiments: (1) bed load, (2) wet weather, and (3) dry weather experiments, as discussed below.
Bed-load transport tests

At the beginning, bed-load transport tests were carried out in the rig shown in Figure 4-4 at different hydraulic loading rates for wet weather conditions. The surface was prepared as smooth as possible to produce a uniform 0.25 m-thick bed layer, and then the flume was slowly filled up with water overnight so that plastic granules could be saturated to form a wet bed as in reality. During these experiments, only clean water was pumped into the flume via the constant-head inflow tank. A steady inflow (set up by the inflow valve and the flow meter) was established so that the water depth was fixed at 0.15 m at the start of each experiment (controlled by the outlet removable gate). Several samples were taken at the outlet weir by collecting all the particles in outflow within a given time period (1 to 15 minutes depended on the flow rate) to determine the mean outflow particle concentration.

Wet weather experiments

Wet weather experiments were undertaken through continuously dosing of plastic granules into the flume over a given duration (1 hour) after steady-state flow conditions were reached.

Before starting the tests, the dry plastic granules with a known mass were saturated with water, to be used as input sediments. As explained in Section 4.2.1 and shown in Figure 4-3, the measured plastic granules’ median settling velocity was 0.047 m/s, well located within the range of stormwater particle settling velocities. Since the granules’ velocities varied in a very narrow range (Figure 4-2), they could be reasonably assumed to be the median settling velocity.

At the beginning of each wet weather experiment, a steady flow and a constant water depth (= 0.15 m) were established prior to the direct manual injection of sediments into the upstream of the flume. Meanwhile, the flow discharging rates through six siphon pipes at intermediate sampling positions were measured separately so that the particle concentrations could be calculated based on the measured flow rates and the sampling period. After calibration, bottles of wet plastic granules were manually added into the inflow water every two minutes across the flume width so as to mix thoroughly just after
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the perforated baffle (Figures 4-4 and 4-7). Sample collection started after steady cross-sectional sediment concentrations were reached along the flume (normally 2-3 times of detention time based on the flow rate). Wet weather sampling lasted 30 minutes. Two ways were used for collection of samples. Samples were taken by siphon tubes at six points along the flume (inlets of tubes were placed at the middle points of each cross-section), and plastic granules were collected by the sieves for the entire sampling period (Figure 4-9). Also, outflow sediments were collected at the outlet weir through a textile-mesh tray (Figure 4-8). The collected plastic granules were weighted after drying (in an oven at 65°C for 48 hours) in order to calculate the sediment particle concentrations.

**Dry weather experiments**

Dry weather experiments started instantly after the preceding wet weather experiment by stopping adding inflow sediment and reducing the flow rate (through the valve and flow meter installed in the inlet pipe) in line with dry weather conditions. The sediment particles were only collected at the outlet of the flume by a textile-mesh tray.

**4.3.2 The experimental programme**

**Bed-load transport tests**

Table 4-2 summarizes the six selected hydraulic loading rates used in bed-load transport tests. They were chosen in this way to achieve the realistic values for $R_e T$. In addition, the experiments were named in the table for reference in presenting results and discussion.

**Wet and dry weather experiments**

For wet weather experiments, three hydraulic loadings (in accordance with 2000, 5000, and 7000 m/y in non-vegetated ponds) and one input sediment concentration (around 300 mg/L) were used for three sampling runs.

Dry weather experiments were carried out after wet weather experiments. The hydraulic loading rate was 1270 m/y (corresponding to 100 m/y in wetlands) for dry weather experiment, as summarized in Table 4-3.
Table 4-2: Bed-load transport experimental variables and name of experiments in laboratory non-vegetated flume.

<table>
<thead>
<tr>
<th>Hydraulic loading rate (m/y)</th>
<th>Values simulated in real wetlands</th>
<th>Appropriate lab flume values</th>
<th>Exp. Name</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2000</td>
<td>24900</td>
<td>BL1</td>
<td></td>
</tr>
<tr>
<td>3000</td>
<td>37260</td>
<td>BL2</td>
<td></td>
</tr>
<tr>
<td>4000</td>
<td>49530</td>
<td>BL3</td>
<td></td>
</tr>
<tr>
<td>5000</td>
<td>61720</td>
<td>BL4</td>
<td></td>
</tr>
<tr>
<td>6000</td>
<td>73810</td>
<td>BL5</td>
<td></td>
</tr>
<tr>
<td>7000</td>
<td>85890</td>
<td>BL6</td>
<td></td>
</tr>
</tbody>
</table>

Table 4-3: Non-vegetated pond wet and dry weather experiments for different flow conditions.

<table>
<thead>
<tr>
<th>Type of experiments</th>
<th>Hydraulic loading rate (m/y)</th>
<th>Exp. Name</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>the simulated in real wetlands</td>
<td>the actual in the lab flume</td>
</tr>
<tr>
<td>Wet weather</td>
<td>2000</td>
<td>24900</td>
</tr>
<tr>
<td></td>
<td>5000</td>
<td>61720</td>
</tr>
<tr>
<td></td>
<td>7000</td>
<td>85900</td>
</tr>
<tr>
<td>Dry weather</td>
<td>100</td>
<td>1270</td>
</tr>
<tr>
<td></td>
<td>100</td>
<td>1270</td>
</tr>
<tr>
<td></td>
<td>100</td>
<td>1270</td>
</tr>
</tbody>
</table>

It should be mentioned that for both wet and dry weather experiments, sediments were collected by siphon tubes and sieves at six points along the flume. However, their weights were so light that they could be neglected. Only a small number of particles were captured by siphon tubes, even though a large number of particles were observed to move in the water under the high flow conditions. This may be due to the uneven distribution of large particle granules (1.8 mm in diameter) across a cross-section and the limited size of the inlet of siphon tubes (which is too small to catch particles, but needs to be small enough to avoid extracting large amount of water so as not to affect flow conditions in the flume). In all experiments, the samples collected at the intermediate locations along the flume therefore failed to represent the effective particle concentration variations. Hence, only the measured outflow particle concentrations (all particles in outflow collected at the outlet weir) were used in the experimental data analyses.
4.4 Data analysis

Data collected from the bed-load transport tests and the wet and dry weather experiments, were plotted and analysed in a statistical manner. The calculation of mean sediment concentration was based on the division of the measured mass of dry plastic granules by the recorded flow volume (equal to the product of flow rate and sampling time). In the experiments, the flow rate and the elapsed time were recorded for each measurement. Therefore, mean outflow particle concentration could be calculated.

Changes of TSS concentration in the outflow were plotted against the hydraulic loading rate in order to determine the critical flow velocity beyond which re-suspension cannot be neglected in non-vegetated ponds. The measured outflow TSS concentrations in wet weather experiments were compared with those obtained from the preceding bed-load tests (under the same flow conditions) to examine the importance of sediment deposition and re-suspension in non-vegetated ponds.

4.5 Results and discussion

This section provides the results for both wet and dry weather experiments, together with the bed-load transport tests. In particular, the effects of increasing flow velocity on sediment re-suspension are discussed based on the experimental results. Finally, re-examination of the three main hypotheses given in Chapter 3 is presented.

4.5.1 Bed-load transport tests

The measured experimental results (such as hydraulic loading rates and mean outflow particle concentrations) and the derived parameters (such as $R_{eT}$ and flow velocities) are presented in Table 4-4 for bed-load transport tests. The measured mean bed-load sediment discharges (by dry weight) at the outlet weir of the flume for the six experiments are plotted against the hydraulic loading in Figure 4-10. It shows clearly that an increase in hydraulic loading will increase the discharge of sediment bed-load when the hydraulic loading is higher than 50046 m/y (equivalent to a rate of 4000 m/y in real wetlands), corresponding to
the flow velocity 0.12 m/s. It was therefore clear that re-suspension and remobilization of bed load occurred in the flume during high hydraulic loadings (i.e. high flow velocity). Conversely, when the hydraulic loading was less than 50046 m/y (i.e. the flow velocity was less than 0.12 m/s), very few particles were transported in the flume (in this case, the outflow bed-load discharge was less than 10 mg/L). This implies that re-suspension and bed-load transport can be neglected in the flume when the flow velocity was less than 0.12 m/s.

Table 4-4: Bed-load transport experimental variables

<table>
<thead>
<tr>
<th>Exp. name</th>
<th>BL 1</th>
<th>BL 2</th>
<th>BL 3</th>
<th>BL 4</th>
<th>BL 5</th>
<th>BL 6</th>
</tr>
</thead>
<tbody>
<tr>
<td>Real hydraulic loading (m/y)</td>
<td>2000</td>
<td>3000</td>
<td>4000</td>
<td>5000</td>
<td>6000</td>
<td>7000</td>
</tr>
<tr>
<td>On the model hydraulic loading in the lab flume (m/y)</td>
<td>25137</td>
<td>37249</td>
<td>50046</td>
<td>61929</td>
<td>73813</td>
<td>85924</td>
</tr>
<tr>
<td>Flow velocity (m/s)</td>
<td>0.06</td>
<td>0.09</td>
<td>0.12</td>
<td>0.15</td>
<td>0.18</td>
<td>0.2</td>
</tr>
<tr>
<td>Flow Reynolds Number $R_e$</td>
<td>7333</td>
<td>10867</td>
<td>14600</td>
<td>18067</td>
<td>21533</td>
<td>24800</td>
</tr>
<tr>
<td>Measured mean outflow particle concentration (mg/L)</td>
<td>0.1</td>
<td>3.2</td>
<td>7.8</td>
<td>30</td>
<td>175</td>
<td>449</td>
</tr>
</tbody>
</table>

Figure 4-10: Hydraulic loading versus outflow mean bed-load discharge.

In fact, these results are very consistent with Yang’s findings (Equation 4.1). The critical flow velocities for particle incipient motion shown in Table 4-1 for different hydraulic loading rates are obviously in agreement with the results for bed-load tests (Figure 4-10).
When the flow velocity exceeded the critical value for incipient motion (such as in Experiments BL4, BL5 and BL6), bed sediment particles started to move and re-suspension became significant. On the other hand, it is detected in Figure 4-10 that re-suspension can be neglected for Experiments BL1, BL2 and BL3 when the hydraulic loading rate is less than 50046 m/y, since the actual mean flow velocities in the flume $V$ are much lower than their critical velocities $V_{cr}$. Therefore, the experimental results suggest that the critical flow velocity for bed sediment particle re-suspension in non-vegetated ponds is around 0.12 m/s. This critical flow velocity is roughly consistent with current Australian design guidelines, which require that the designing mean flow velocity should be less than 0.05 m/s, and during wet weather it must be within the range between 0.02 m/s to a maximum of 0.1 m/s (Wong and Breen, 2002).

### 4.5.2 Wet weather experiments

The results for wet weather experiments at three hydraulic loading rates (simulating 2000, 5000, and 7000 m/y) are presented in Table 4-5.

<table>
<thead>
<tr>
<th>Wet weather experiment variables</th>
<th>Exp. Name</th>
</tr>
</thead>
<tbody>
<tr>
<td>Simulated hydraulic loading (m/y)</td>
<td>W1</td>
</tr>
<tr>
<td></td>
<td>W2</td>
</tr>
<tr>
<td></td>
<td>W3</td>
</tr>
<tr>
<td>Actual hydraulic loading in the lab flume (m/y)</td>
<td>2000</td>
</tr>
<tr>
<td></td>
<td>5000</td>
</tr>
<tr>
<td></td>
<td>7000</td>
</tr>
<tr>
<td>Actual hydraulic loading rate</td>
<td>24452</td>
</tr>
<tr>
<td></td>
<td>61929</td>
</tr>
<tr>
<td></td>
<td>85924</td>
</tr>
<tr>
<td>Flow velocity (m/s)</td>
<td>0.06</td>
</tr>
<tr>
<td></td>
<td>0.15</td>
</tr>
<tr>
<td></td>
<td>0.21</td>
</tr>
<tr>
<td>Inflow sediment concentration (mg/L)</td>
<td>306</td>
</tr>
<tr>
<td></td>
<td>300</td>
</tr>
<tr>
<td></td>
<td>360</td>
</tr>
<tr>
<td>Measured mean outflow particle concentration (mg/L)</td>
<td>0.3</td>
</tr>
<tr>
<td></td>
<td>39</td>
</tr>
<tr>
<td></td>
<td>450</td>
</tr>
</tbody>
</table>

It was found that the bed particles did not move in the case of W1 (low hydraulic loading rate 24452 m/y, equivalent to 2000 m/y in real wetlands, with a flow velocity of 0.06 m/s) and most inflow plastic granules settled within 0.40 m from the starting point as demonstrated in Figure 4-11. The average travel distance was 0.20 m; some of them could transport as far as 0.4 m by flow advection as shown in Figure 4-12. According to the Hazen settling theory, the average travel distance $L$ is determined by the particle settling velocity $V_s$, flow velocity $V$ and water depth $H$, through $L = (H/V_s)V = 0.20m$. Thus, the
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The experimental result is in good agreement with the Hazen model under steady-state laminar flow conditions. It should be also noted that this distance of 0.2 m in the scaled physical model represents 1.97 m in real systems. In this case, a small portion of particles (total dry mass 6.04 g) was gathered at the outlet of the flume after 30 minutes, which is equal to a particle concentration 0.3 mg/L in the water flow. This indicates that virtually no re-suspension happens in the flume.

Nevertheless, there were significant changes in outflow concentration when the hydraulic loading rate was increased to a higher level (e.g., 0.15 m/s and 0.21 m/s of the flow velocity in Table 4-5). With increasing flow velocities, bed particles were observed to slide, roll or jump along the bed, as well as the bed movement induced patterns such as ripples and dunes (Figure 4-13). Consequently, more bed particles were discharged from the flume. It should be noted that the actual water depths gradually became deeper than 0.15 m as more granular particles were released from the bed into the water flow over time.

The results for experiment W2 (medium hydraulic loading rate 61929 m/y corresponding to 5000 m/y in the wetland) without serious bed load transport showed that input sediment particles traveled further away from the inlet because of flow advection. There was an increase in the observed outflow particle concentration (39 mg/L) compared with that from the preceding mean bed-load discharge (30 mg/L at the same hydraulic loading rate). This increase in particle concentration may be attributed to the input sediment particles discharged in the outflow. However, the input sediment concentration was 300 mg/L, much larger than the outflow concentration. The result also indicates that a large fraction of the inflow particles have settled within the flume.

In the case of the W3 (high hydraulic loading rate 85924 m/y for simulating 7000 m/y in the wetland), a change in bed surface from flat plane to dunes was clearly detected (Figure 4-13). The observed outflow particle concentration (450 mg/L) was very close to the mean bed-load discharge (449 mg/L) obtained by the preceding bed-load transport test, even though the inflow particle concentration was 360 mg/L. This shows a high deposition rate and a dynamic balance between deposition and re-suspension in the flume at high flow.
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velocities. The reason may be that at high flow velocity, due to particle interaction and bed surface roughness, a sediment particle can be transported alternatively as bed-load or as suspended load due to flow turbulence. For example, the continued loss of bed materials from the flume resulted in an increased water depth, which reduced the flow velocity locally, thus enhancing the sediment deposition rate.

Figure 4-11: Particles settled out within 0.4 m from the input point when hydraulic loading was 24453 m/y in the flume (corresponding to simulating wetland hydraulic loading of 2000 m/y).

Figure 4-12: The average particle travel distance was 0.2 m when simulating wetland hydraulic loading of 2000 m/y.
In conclusion, the results presented so far in this section have shown that, in non-vegetated ponds, flow conditions play a key role in sediment transportation and deposition when the flow velocity is greater than 0.12 m/s.

4.5.3 Dry weather experiments

In total three dry weather experiments were carried out, each started immediately after a wet weather experiment. No particles were collected at the outlet weir. Under dry weather conditions, the suspended plastic granules in the water column deposited quickly and could not travel as far as the outlet owing to the high particle settling velocity and the low flow velocity. This has also verified that the Particle Fall Number $N_f$ is an important parameter for sediment deposition in non-vegetated ponds.

4.5.4 Testing the hypotheses for both wet and dry weather conditions

The above results have further confirmed the hypotheses for estimation of sediment trapping efficiency in wetlands and ponds.

_Hypothesis 1_ - The importance of $R_{e,*}$ for sediment trapping efficiency is based on the flow conditions. When the flow velocity is greater than its critical flow velocity (0.12 m/s in
Laboratory non-vegetated pond study

non-vegetated ponds), re-suspension becomes significant, otherwise, it can be neglected as shown in Table 4-1 (Yang’s theory) and observed in bed-load transport tests (Figure 4-10).

**Hypothesis 2** - Deposition is the main process that governs sediment behaviour in non-vegetated ponds and wetlands. The results on sediment travelling distance are in full agreement with the Hazen settling theory (Hazen, 1904).

**Hypothesis 3** – The wash-off process appears not to play an important role during wet weather when flow velocity is less than the critical flow velocity, since flow turbulence does not significantly influence sediment transport as observed in dry weather experiments.

### 4.6 Conclusions

In summary, in order to test the previous three hypotheses of Chapter 3 in constructed stormwater non-vegetated ponds, experiments were carried out in an open-water flume. The experimental system was scaled to meet two of the three numbers \((N_f \text{ and } R_{e*} \text{ or } N_f \text{ and } R_{eT})\) which are obtained from a real system. To achieve this, higher hydraulic loading and larger particle sizes had to be used. It has been demonstrated in experimental results that deposition is the main process and re-suspension can be neglected in the non-vegetated ponds when the flow velocity \(V\) is less than the critical flow velocity \(V_{cr} (= 0.12 \text{ m/s})\). This has also further confirmed the three hypotheses presented in Chapter 3. Conversely, when flow velocity exceeded its critical value, bed sediments started to move and the bed plane became rugged to facilitate the formation of local turbulent flow. It is too complex to predict the deposition rate and re-suspension rate in this case. Also, it was difficult to separate bed load and suspended load in experiments, and the experimental results showed that deposition was no longer necessarily the dominant process. Thus, the proposed hypotheses and sediment behaviour model become invalid for the cases where deposition induced by flow no longer controls the overall sediment behaviour. However, it must be remembered that current Australian wetland design guidelines recommend that the mean flow velocity should be limited to no more than 0.1 m/s for wet weather conditions (Wong
and Breen, 2002), with wetland geometry and bypass structures being modified to meet this criterion. Unexpected rapid flow conditions are beyond the scope of this study anyhow, and it may be concluded that the hypotheses demonstrated in the vegetated mesocosm wetlands are also valid for non-vegetated ponds under well-controlled flow conditions (i.e. mean flow velocity less than 0.12 m/s).
Chapter 5

Studying the Properties of the $k$-$C^*$ Model Using Laboratory Experimental Data

5.1 Introduction

The first order kinetic $k$-$C^*$ model (Equation 2.9) is one of the most popular conceptual models to date for modelling stormwater treatment (Kadlec, 2000; CRCCH, 2005; Wong et al., 2006), as discussed in Chapter 2. This two-parameter model describes pollutant removal by an exponential decay function towards an equilibrium value or background concentration ($C^*$) with an exponential rate constant ($k$) (Kadlec and Knight, 1996). The parameters $k$ and $C^*$ are lumped parameters representing the overall effect of a number of pollutant removal mechanisms (Wong et al., 2006). The parameter $k$ reflects how quickly the concentration of a pollutant in the treatment system approaches its equilibrium concentration $C^*$ (provided that $C^*$ is less than the inflow concentration), while as an asymptote, $C^*$ controls the value the concentration can be reduced to (Wong et al., 2006).

As discussed in Chapter 2, many studies have demonstrated that these two parameters vary with hydraulic loading rates, inflow concentrations and sediment particle sizes (Kadlec, 2000; CRCCH, 2005). Despite this, default values in the software MUSIC (Model for Urban Stormwater Improvement Conceptualisation) are very often used in practice for these two parameters (e.g. $k = 1500$ m/y, $C^* = 6$ mg/L for TSS) with no knowledge available on how changes in these two parameters impact on modelled results. It is also fair to say that the model is rarely validated in practice.

The aim of this chapter is to:

(1) assess reliability of the $k$-$C^*$ model using the data collected in the mesocosm lab
5.2 Calibration of the $k-C^*$ model

The $k-C^*$ model can be expressed algebraically in the following form (i.e. Equation 2.10), which will be used in the calculations performed on a spreadsheet:

$$ C_{out} = C^* + (C_{in} - C^*)e^{-k/q} $$

This section first provides a method for categorizing the laboratory wet weather mesocosm data. Based on the data available, $k$ and $C^*$ can be calibrated. The calibration method used is primarily dependent on data analysis for finding the optimum $k$ and $C^*$ parameter set. Finally, results are provided with detailed discussion on the conclusions between the predicted and observed TSS data.

5.2.1 Methods

Preparation of calibration data

As explained in Chapter 3, the laboratory study was conducted in three mesocosm wetlands (vegetated with different densities) and one mesocosm non-vegetated pond. Samples were taken along the mesocosm cells at 0.5m, 1.0m and 2.0m from the inlet weir and analysed for Total Suspended Solids concentrations (TSS) and Particle Size Distributions (PSDs). The results of measured TSS concentration for all 40 experiments are provided in Appendix A3. The variations of measured cumulative PSDs at the three sampling points with time are presented in Appendix A5. In summary, the wet weather
Application of the k-C* model using experimental data

Experimental results in these three sections (i.e. between (1) 0.5 m and 1 m, (2) 0.5 m and 2 m, and (3) 1 m and 2 m from the inlet weir) are re-sorted into different categories according to the levels of vegetation density, hydraulic loading rate, input sediment concentration and input sediment median particle size, as shown in Table 5-1. There are 107 measured points in total that were used in the model application.

Table 5-1: Mesocosm experimental variable categories and available data points.

<table>
<thead>
<tr>
<th>Experimental variable</th>
<th>Data category</th>
<th>No. of data points</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vegetation density (culms/m²)</td>
<td>High = 2936</td>
<td>27</td>
</tr>
<tr>
<td></td>
<td>Medium = 1620</td>
<td>27</td>
</tr>
<tr>
<td></td>
<td>Low = 590</td>
<td>25</td>
</tr>
<tr>
<td></td>
<td>No Vegetation = 0</td>
<td>28</td>
</tr>
<tr>
<td>Hydraulic loading (m/y)</td>
<td>Very high = 20000-30000</td>
<td>18</td>
</tr>
<tr>
<td></td>
<td>High = 8000-15000</td>
<td>29</td>
</tr>
<tr>
<td></td>
<td>Medium = 4000-8000</td>
<td>30</td>
</tr>
<tr>
<td></td>
<td>Low = 1900-4000</td>
<td>30</td>
</tr>
<tr>
<td>Input sediment concentration (mg/L)</td>
<td>High = 500-900</td>
<td>29</td>
</tr>
<tr>
<td></td>
<td>Medium = 110-500</td>
<td>51</td>
</tr>
<tr>
<td></td>
<td>Low = 30-110</td>
<td>27</td>
</tr>
<tr>
<td>Input sediment median particle size (µm)</td>
<td>Large = 45-56</td>
<td>17</td>
</tr>
<tr>
<td></td>
<td>Medium = 30-45</td>
<td>43</td>
</tr>
<tr>
<td></td>
<td>Low = 18-30</td>
<td>47</td>
</tr>
</tbody>
</table>

Calibration Procedure

In general, model calibration is an iterative process which involves using and optimizing an objective function that represents the errors between the observed data and the model predictions. The essence of the approach is to find out the highest peak of the response surface in the parameter space defined by the objective function (Beven, 2001). The shape of the response surface is strongly subject to the kind of objective function, which is chosen to evaluate the model performance (Pappenberger et al., 2005). Various objective functions can be used. However, the selection of an appropriate objective function is crucial for model calibration (Diskin and Simon, 1977; Yu and Yang, 2000).

The optimum parameters set for the model can be assessed using Root Mean Square Error (RMSE), defined below, as an objective function that minimizes the sum of the squared
errors, or by applying the commonly-used model efficiency $E$ as an objective function that maximizes the sum of error variance between observed and predicted data (Beven, 2001). In particular, the Root Mean Square Error (RMSE) gives the standard error of the estimate in absolute values and is defined as

$$RMSE = \sqrt{\frac{1}{n} \sum_{i=1}^{n} (O_i - P_i)^2} \quad (5.2)$$

where $n$ is the number of observations, $O_i$ the observed value and $P_i$ the predicted value.

The model efficiency ($E$) of Nash and Sutcliffe (1970) is defined as the proportion of the initial variance accounted for the model (i.e. which is a normalised value of $RMSE$):

$$E = 1 - \frac{\sum_{i=1}^{n} (O_i - P_i)^2}{\sum_{i=1}^{n} (O_i - O_{mean})^2} \quad (5.3)$$

where $O_{mean}$ is the mean observed value. A perfect fit should have $E$ approaching 1, while negative $E$ value indicates that the model produces more variations than the observed data. Comparative studies have suggested that these two objective functions emphasize the high and low value ranges separately, which are suitable for characterizing the features around the peak value (Yapo et al., 1996; Yu and Yang, 2000).

Another technique to evaluate modelling performance is the coefficient of determination (cross-correlation coefficient), $R^2$, which is defined as

$$R^2 = \left( \frac{\sum_{i=1}^{n} (O_i - O_{mean})(P_i - P_{mean})}{\sqrt{\sum_{i=1}^{n} (O_i - O_{mean})^2 \sum_{i=1}^{n} (P_i - P_{mean})^2}} \right)^2 \quad (5.4)$$

where $P_{mean}$ is the mean predicted value. Its value approaching unity means a perfect fit, and the fit is very poor if it is around zero. This regression method tends to match the entire range of data set rather than the peak values, that is, the regression result is a more uniform
distribution of residuals.

For this thesis, the model efficiency $E$ was considered to be appropriate due to the fact that the peaks in flow rates and TSS concentrations are probably most important in urban stormwater quantity and quality (i.e. these “large events” carry a large proportion of the mean annual pollutant load).

The $k-C^*$ model depends on the choices of the parameters $k$ and $C^*$. The calibration of these two parameters was then undertaken against the measured data to achieve the best fit using the objective function of maximizing the model efficiency $E$. The calibration procedure is elucidated as follows.

Firstly, all data gathered from all four mesocosms (including non-vegetated flume) were used to find the optimum parameter set for $k$ and $C^*$. It was hypothesized that this universal parameter set would be able to characterize the four mesocosms under various experimental variables. Therefore, all the measured 107 points were used in this calibration.

Secondly, each data category grouped based on experimental variables (as outlined in Table 5.1) was used to obtain an optimum parameter set of $k$ and $C^*$ individually, to investigate the effects of variables on the model performance. The number of points used for each category is given in Table 5.1 and varies with the experimental variables such as:

1. vegetation density,
2. hydraulic loading,
3. input sediment concentration, and
4. input median particle size.

Data Analysis

Using the calibrated parameter sets, the predicted TSS concentrations at two downstream sampling points in the mesocosm flume could be calculated. Based on these outputs, the predicted TSS data were then plotted against the observed, as well as their ±25% and ±50% error bands. Meanwhile, the values of model efficiency $E$ were also calculated to
The optimum $k$ and $C^*$ values for each data category were used to calculate their mean and variances so as to examine model sensitivity against the experimental variables. To analyse the varying trends of those two parameters with experimental variables, these optimum results were plotted in histograms and initially analysed in a visual manner.

5.2.2 Results and discussion

Based on all available data (107 points), the $k$-$C^*$ model parameters were calibrated and the modelling results are presented along with the measured data in Figure 5-1. It was found that the optimal $k$ value was 8004 m/y and the $C^*$ value was 58 mg/l. It is clear in Figure 5-1 that the predicted TSS concentrations match well the observed values. The model efficiency $E$ is 0.88 and cross-correlation coefficient $R^2$ is 0.89, with the majority of the concentrations located well within ±50% error bands. On the other hand, this perhaps also indicates that the quality of experimental data is satisfactory. For each data category with an optimum parameter set, the modelled TSS concentrations are plotted against the measured in Appendix B1. It is also found that the modelled TSS concentrations are largely within ±50% bands of the observed.
Application of the k-C* model using experimental data

data category along with model efficiency $E$ and $R^2$ to study how experimental variables impact on the model parameters. The values of $E$ and $R^2$ between the measured and modelled were greater than 0.8 in most cases except for high input sediment concentration cases ($E = 0.57$ is still acceptable in this case).

The coefficient of variation of the mean (= standard deviation / mean) for all data sets is 52% and 106.6% for $k$ and $C^*$, respectively. For each category, it ranges from 21.2% to 52% for $k$ and from 26.7% to 110.0% for $C^*$ as presented in Table 5-2. The large variations in the coefficients possibly indicate that $k$ and $C^*$ are dependent on some experimental variables (e.g., hydraulic loading, inflow sediment concentration, and input sediment particle size), although the $k$-$C^*$ model performed well for most simulations. On the other hand, it appears that vegetation density has relatively less influence on the optimum values of $k$ and $C^*$, since the coefficient of variation of the means is 21.2% and 26.7% for $k$ and $C^*$, respectively.

However, how $k$ and $C^*$ vary with these factors is unclear based on the limited experimental data points (Table 5-1). In Figure 5-2, we cannot find consistent trends for the variations of $k$ and $C^*$, except that $k$ increases with hydraulic loading rate as shown in Figure 5-2-b and $C^*$ increases with inflow sediment concentrations as in Figure 5-2-c. When the input median sediment particles are large (over 45 µm in the experiments), the optimum $C^*$ value will become zero, since large particles settle quickly and completely.

The recommended $k$ and $C^*$ values for TSS in wetlands in MUSIC are 500 - 5000 m/y and 5 - 6 mg/l, respectively (CRCCH, 2005). Using the theoretical values in MUSIC of $k = 5000$ m/y and $C^* = 6$ mg/L, it was found that $E$ was 0.79 and $R^2$ was 0.91. The modelling results can be regarded as good compared with experimental data. If the value of $k$ is reduced, $E$ will decrease as shown in Figure 5-3. The value of $E$ can become negative at the lower bound of the recommended range of $k$. In addition, $E$ does not change with $C^*$ since the suggested range is extremely narrow. It seems that the suggested values for two parameters in MUSIC are lower than the calibrated ones based on the experimental data. This is probably due to the higher hydraulic loading rates in the section between 0.5 - 1.0 m
from the inlet generating higher $k$ values. However, the designed experimental hydraulic loading rates were based on the effective length of the full mesocosm cell (i.e. 1.5 m). They higher values of $k$ and $C^*$ may also be due to the higher steady input concentrations in the mesocosms in contrast to the intermittent nature of real-world input concentrations.

Table 5-2: The optimum $k$ and $C^*$ parameter sets for each data category ($k$: m/y; $C^*$: mg/L).

<table>
<thead>
<tr>
<th>Experimental variable</th>
<th>Data category</th>
<th>Model parameter $K$</th>
<th>$C^*$</th>
<th>$E$</th>
<th>$R^2$</th>
<th>RMSE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vegetation density</td>
<td>High = 2936</td>
<td>7635</td>
<td>64</td>
<td>0.87</td>
<td>0.88</td>
<td>51.66</td>
</tr>
<tr>
<td></td>
<td>Medium = 1620</td>
<td>10435</td>
<td>72</td>
<td>0.85</td>
<td>0.87</td>
<td>53.47</td>
</tr>
<tr>
<td></td>
<td>Low = 590</td>
<td>7172</td>
<td>37</td>
<td>0.96</td>
<td>0.96</td>
<td>29.68</td>
</tr>
<tr>
<td></td>
<td>No Vegetation =0</td>
<td>6644</td>
<td>52</td>
<td>0.9</td>
<td>0.91</td>
<td>47.38</td>
</tr>
<tr>
<td>Coefficient of variation (%)</td>
<td>21.2</td>
<td>26.7</td>
<td>5.2</td>
<td>4.3</td>
<td>23.9</td>
<td></td>
</tr>
<tr>
<td>Hydraulic loading (m/y)</td>
<td>Very high=20000-30000</td>
<td>11266</td>
<td>30</td>
<td>0.86</td>
<td>0.86</td>
<td>64.38</td>
</tr>
<tr>
<td></td>
<td>High = 8000-15000</td>
<td>6381</td>
<td>40</td>
<td>0.87</td>
<td>0.87</td>
<td>42</td>
</tr>
<tr>
<td></td>
<td>Medium = 4000-8000</td>
<td>6811</td>
<td>42</td>
<td>0.91</td>
<td>0.91</td>
<td>32.31</td>
</tr>
<tr>
<td></td>
<td>Low = 1900-4000</td>
<td>3346</td>
<td>22</td>
<td>0.76</td>
<td>0.76</td>
<td>14.35</td>
</tr>
<tr>
<td>Coefficient of variation (%)</td>
<td>47.0</td>
<td>28.1</td>
<td>7.3</td>
<td>7.3</td>
<td>54.5</td>
<td></td>
</tr>
<tr>
<td>Input sediment concentration (mg/l)</td>
<td>High = 500-900</td>
<td>20843</td>
<td>274</td>
<td>0.57</td>
<td>0.57</td>
<td>62.59</td>
</tr>
<tr>
<td></td>
<td>Medium = 110-500</td>
<td>6981</td>
<td>72</td>
<td>0.9</td>
<td>0.9</td>
<td>24.52</td>
</tr>
<tr>
<td></td>
<td>Low = 30-110</td>
<td>13898</td>
<td>18</td>
<td>0.83</td>
<td>0.84</td>
<td>6.55</td>
</tr>
<tr>
<td>Coefficient of variation (%)</td>
<td>49.8</td>
<td>110.0</td>
<td>22.9</td>
<td>23.0</td>
<td>91.7</td>
<td></td>
</tr>
<tr>
<td>Input median sediment particle size (µm)</td>
<td>Large = 45-56</td>
<td>3995</td>
<td>0</td>
<td>0.91</td>
<td>0.92</td>
<td>48.55</td>
</tr>
<tr>
<td></td>
<td>Medium = 30-45</td>
<td>8924</td>
<td>71</td>
<td>0.83</td>
<td>0.85</td>
<td>61.37</td>
</tr>
<tr>
<td></td>
<td>Small = 18-30</td>
<td>4566</td>
<td>36</td>
<td>0.95</td>
<td>0.95</td>
<td>19.28</td>
</tr>
<tr>
<td>Coefficient of variation (%)</td>
<td>46.3</td>
<td>99.2</td>
<td>6.6</td>
<td>5.7</td>
<td>50.1</td>
<td></td>
</tr>
<tr>
<td>All wet weather experimental data</td>
<td></td>
<td>8004</td>
<td>58</td>
<td>0.88</td>
<td>0.89</td>
<td>50.28</td>
</tr>
<tr>
<td>Coefficient of variation (%) for all categories</td>
<td>52.0</td>
<td>106.6</td>
<td>10.9</td>
<td>10.9</td>
<td>45.3</td>
<td></td>
</tr>
</tbody>
</table>

### 5.3 Sensitivity analysis

In general, methods for calibrating a model look for an optimum parameter set that is at the peaks of its response surface, regardless of the weighted influence of each parameter on the results. However, the performance of the model may be much more sensitive to changes in the values of some parameters than others (Beven, 2001). Therefore, it is important to evaluate the uniqueness of the model parameters through sensitivity analyses. Once the importance is known for all parameters, the results can subsequently be applied for various
Application of the $k$-$C^*$ model using experimental data

(a) The optimum $k$ and $C^*$ values based on vegetation density categories.

(b) The optimum $k$ and $C^*$ values based on hydraulic loading categories.

(c) The optimum $k$ and $C^*$ values based on input sediment concentration.

(d) The optimum $k$ and $C^*$ values based on input median sediment particle size.

Figure 5-2: The optimum $k$ and $C^*$ values based on the experimental variables.
Application of the $k$-$C^*$ model using experimental data

Figure 5-3: Variations of model efficiency $E$ with $k$ and $C^*$ values recommended in MUSIC.

purposes, such as for assessing changes in the system’s responses because of parameter variations, for ranking the individual parameters in order based on their relative importance to the response (Cacuci, 2003), and for enhancing the efficiency of parameter calibration (Beven, 2001). In order to investigate the sensitivity of the $k$-$C^*$ modelling results to its parameters, the methods for studying behaviour of the objective functions and local sensitivity are presented in this section, followed by the results and related discussion.

5.3.1 Methods

Accordingly, two things have to be done:

(1) behaviour of the objective function: the response surface of the objective function to evaluate the model performance on the parameter space, and

(2) local sensitivity: the sensitivity to the changes in the model parameters that have a value of $E$ within 5% around the maximum.

Behaviour of the objective function

A common method for sensitivity analysis is to make sufficient runs by adjusting the model parameters in an ordered manner so as to plot the response surfaces (such as contours) of a given objective function (i.e. this graphically presents how goodness of fit changes with the
Application of the $k$-$C^*$ model using experimental data

model parameters). If the response surface behaves at smooth slopes towards the global minima, the optimum parameter set will be clearly identified. In this case, it signifies that the parameters are unique on the model results; this is also known as a "Hill-climbing" algorithm.

In this study, the $k$-$C^*$ model parameters were assigned independently for all available wet weather experimental data and each data category (listed in Table 5-1). A range of values for each parameter ($k$ and $C^*$) with a specified increment was used to define a parameter space. In particular, the selected values of $k$ and $C^*$ are defined in the following way:

\[
\begin{align*}
k & : \text{from 200 m/y to 25000 m/y at an increment of 200 m/y;} \\
C^* & : \text{from 0 mg/l to 300 mg/l at an increment of 2 mg/l.}
\end{align*}
\]

A combination matrix of the parameters $k$ and $C^*$ for each data case therefore comprises 18750 sets to form the parameter space. The $k$-$C^*$ model was run for each parameter set to obtain the values of objective functions (e.g. the model efficiency $E$). The resulting values were then plotted to define a response surface for further identifying the sensitivity to each parameter.

Local sensitivity analysis

The objective of local sensitivity analysis is to study the behaviour of the model results in response to changes in the model parameters around a chosen point such as the minima or the maxima. Parameter sensitivity is normally evaluated in the immediate area of a best estimated parameter set or an identified optimum parameter set after a model calibration (Beven, 2001). The basic approach for evaluating the local sensitivity of each parameter is the "dotty plot" technique (Wagener et al., 2003). From the calculations of the objective function, the best performing parameter values (i.e. top 5% of the maxima) from 18750 parameter sets were selected and presented in the form of dotty plots. If these dots are highly concentrated in a narrow area with clear optima in the dotty plots, it indicates the model performance is sensitive to the parameter. Conversely, if these dots are scattered in a widespread range of a parameter without optimum values, this suggests that this parameter...
Application of the k-C* model using experimental data

has no influential role in the model performance, because equally good performances are distributed widely over the parameter space.

Since the k-C* model has two parameters to be calibrated, the dotty plots in one plot cannot interpret the degree of sensitivity of each parameter. A frequency analysis has to be conducted to compare the sensitivity of the two parameters. For example, the frequency was calculated for those dots to yield an objective function $E$ greater than 0.80 (as the maximum $E$ was 0.88).

In particular, for all parameter sets with $E$ greater than a certain value (e.g. 0.80), the range of each parameter is segmented into some sections. For a parameter $c$, its value is normalized in the following way so that the dimensionless number ranges from 0 to 1. Especially, the formula for defining the non-dimensional parameter $c*$ is:

$$c* = \frac{c - c_{\text{min}}}{c_{\text{max}} - c_{\text{min}}} \quad (5.5)$$

where $c_{\text{min}}$ is the lower bound of the parameter range with the higher $E$, and $c_{\text{max}}$ the upper bound.

We can count the number of runs that produce a higher $E$ for a section of $c*$. Hence, the frequency for this section is defined as the fraction of the total runs which generate the higher $E$.

In the same way, the frequency analyses based on each data category for various experimental variables were undertaken for those parameter sets to yield an objective function $E$ within 5% around the maximum.

According to previous studies (e.g. Haydon and Deletic, 2007), the frequency distribution for insensitive parameters should yield a very flat line, while for sensitive parameters a curved distribution is detected with a clear optima. In other words, an insensitive parameter will produce similar model results across a wide range of the parameter’s values, while a sensitive parameter will give the best performance at specified values.
5.3.2 Results and discussion

Parameter response surfaces

Figure 5-4 shows that the objective function $E$ varies with the two parameters $k$ and $C^*$ for all data (18750 runs). The global maxima of $E$ are encompassed within the innermost contour line (around 0.87) and the contour line of 0.83 (the dashed line) demonstrates the best 5% fits. This type of simple hill contours indicates that the effects of the changes in the two parameters' ($k$ and $C^*$) on the modelling results are significant.

![Figure 5-4: The response surface for the two parameters $k$ and $C^*$ with objective function $E$ for all experimental data.](image)

The forms of the response surface for each data category are presented in Appendix B2. The forms of simple hills are globally identifiable within all response surfaces for four types of data category. It may thus be concluded that the findings of an optimum parameter set for each data category described in the previous section are acceptable and the simulation results are sensitive to the two parameters $k$ and $C^*$ in terms of their respective response surfaces in the parameter spaces.

Local sensitivity analysis

The probabilities for each parameter to produce a top 5% value of the objective function $E$ within 18750 parameter sets as a function of $k$ or $C^*$ are presented in the form of dotty plots in Figure 5-5 for various data categories. It is clear that the objective function $E$ changes
with \( k \) and \( C^* \) in these dotty plots since they all have an obvious range.

It can be detected from these plots that there are clear optima for both \( k \) and \( C^* \) and the values of \( E \) scatter in a wide range for all data categories. This means that the model performance is sensitive to the two parameters under different experimental conditions. It is found from Figure 5-5 that to some extent, \( C^* \) is very sensitive to inflow TSS concentrations and input particle sizes (the optimum values range from 18 to 274 mg/L, and 0 to 72 mg/L, respectively), but relatively insensitive to vegetation density and hydraulic loading rates (the optimum values of \( C^* \) varied in a narrow range from 42 to 72 mg/L, and 22 to 58 mg/L, respectively).

On the other hand, the optimum values of the parameter \( k \) broadly differ from 3346 m/y to 11266 m/y for different hydraulic loading rates (Figure 5-5-b), and this may indicate the importance of hydraulic loading on the best-fitting value of parameter \( k \). Similar results can be obtained for its dependence on input sediment concentration and input sediment particle size (Figures 5-5-c and 5-5-d). Conversely, it appears that the variations in vegetation density (Figure 5-5-a) do not prominently affect \( k \) as much as other variables.

The same findings are confirmed in the frequency analyses. Figure 5-6 presents the frequency plot for the two parameters with counted runs producing a model efficiency \( E \) greater than 0.8 in the case of all observed data. It clearly demonstrates that the model results are sensitive to both \( k \) and \( C^* \), and more sensitive to \( k \) than to \( C^* \). The same trend can also be found in some other frequency plots in Appendix B3 for the two parameters, which generate a model efficiency \( E \) within the top 5% of the best for each data category. Figure 5-7 shows four examples for various experimental variables. However, the results are more sensitive to \( C^* \) than \( k \) in the cases of high and low input sediment concentrations. In both experiments and real-world wetlands, high-concentration inflows can produce a higher apparent background concentration, which leads to a stronger influence on the model performance. The same conclusion has been found by Walsh et al. (1997) and Wong et al. (2006). On the other hand, the background concentration in storage becomes more important to concentration results for low-concentration inflows.
Application of the $k$-$C^*$ model using experimental data

Figure 5-5: Dotty plot showing objective function $E$ versus the two parameters $k$ and $C^*$, respectively, for each data category.
Figure 5-6: Parameter frequency within the model parameter space with the value of objective function $E$ greater than 0.8 for all data.

Figure 5-7: Parameter frequency for the model parameter sets which have a value of $E$ within 5% around the maximum for different data categories.
5.4 Conclusions

In this study, the $k$-$C^*$ method was used to calculate the sediment concentration based on optimum values of the two parameters. The parameter calibration was carried out based on the Nash-Sutcliffe coefficient of efficiency ($E$) and $R^2$ acting as objective functions. It can be concluded that the $k$-$C^*$ model performed well to fit all the wet weather experimental data, with a high model efficiency $E = 0.88$ and a high cross-correlation coefficient $R^2 = 0.89$, based on the optimal values of two parameters $k = 8004 \text{ m/y}$ and $C^* = 58 \text{ mg/l}$. From this, it may be confirmed that the experimental data are generally of good quality. The determined values of two parameters under different experimental conditions are also provided. It was also found that the choices of the upper bounds of the recommended values by MUSIC can also give rise to reasonably acceptable model efficiency.

Furthermore, sensitivity analyses were conducted for the model results. By constructing a parameter space, sufficient parameter sets were used. All available results and those that had a value of $E$ within 5% around the maximum were employed for global and local sensitivity studies. It was found that the model predictions are sensitive to both $k$ and $C^*$, although the model’s sensitivity to $k$ is stronger in most cases (except the very high or low input sediment concentration conditions). It is also apparent that the two parameters are not invariant with respect to hydraulic loading, inflow sediment concentration and input sediment particle size, confirming the results of previous studies. In particular, it seems that $k$ increases with hydraulic loading rate and $C^*$ increases with inflow sediment concentrations. Further research is needed to investigate how and to what degree these factors may influence the values of $k$ and $C^*$. 
Chapter 6
Development of a New Physically-Based Model

6.1 Introduction

As discussed in Chapter 2, there is a need to develop a new wetland model that could readily be transferred from one system to another with minimal required calibration. It is more likely that physically-based models can achieve this. An effort is made here in light of the laboratory experimental findings. Basically, the model should include two components, a wet weather module and a dry weather module, so as to provide a better description of sediment behaviour under different flow conditions in the constructed stormwater wetlands. The wet weather module is based on an existing model developed by Deletic (2001, 2005), while the dry weather one is independently developed from the laboratory study presented in the previous two chapters.

This chapter firstly describes the development of the wet weather model. This is followed by a detailed account of the development of the dry weather model. As an important step in the development of the model, this chapter provides a detailed description of the new physically-based model, including its model structure, governing equations, input data requirements and model parameters that need to be calibrated.

6.2 The wet weather model

As discussed in Chapter 3, four mesocosm wetlands (with different vegetation densities) were used to examine sediment behaviour under both wet and dry weather conditions. It has been confirmed that sediment concentration decreases exponentially over distance,
with a decaying exponent being highly dependent on particle size. In particular, a single parameter, \textit{Particle Fall Number}, $N_f$, is strongly correlated with the particle trapping efficiency, as determined by the statistical analysis of experimental data, and is capable of characterising the sedimentation process under wet weather conditions. It is therefore decided to make use of the predictive capability of the Particle Fall Number in the wet weather model.

### 6.2.1 Modelling methodology

Based on the main conclusions from the experimental results presented in Section 3.4.1, as well as previous modelling studies of sediment behaviour in grass filters (Deletic, 2001, 2005), the following non-linear relationship between the particle trapping efficiency of particle size fraction $s$, $Tr_s$, and the Particle Fall Number, $N_f$, was proposed, where $a$ and $b$ are regression parameters:

$$Tr_s = \frac{N_f^a}{N_f^a + b}$$  \hspace{1cm} (6.1)

It was assumed that particles behave independently from each other (no interaction between the particles) and that the particle size distribution (PSD) of stormwater can be adequately represented by five size fractions covering five continuous particle size ranges (i.e. $0-6$, $6-21$, $21-46$, $46-124$ and $124-404$ µm as used in Section 3.4.3 for experimental data analysis). The observed trapping efficiency $Tr_s$ of particle size fraction $s$ was calculated using Equation 3.14.

The two-parameter non-linear $N_f$ model (Equation 6.1) was applied to experimental data based on the above five fractions with the corresponding median diameters: $d_{50} = 3, 13.5, 33.5, 85, \text{ and } 264 \mu m$. Using Equation 6.1, the trapping efficiency for each fraction was predicted at measured points along the cells. The regression parameters $a$ and $b$ were then calibrated through the best fit of the predictions of trapping efficiency against all available measured data. Considering five fractions for each measurement, the number of trapping efficiency measurements provided by experiments was in total 465 points. To find out the
model parameters in the optimisation method, the Nash-Sutcliffe coefficient of model efficiency $E$ was used as the objective function, defined by Equation 5.3, which had already been used in the calibration of $k-C^*$ model, as discussed in Chapter 5. The correlation coefficients between the measured and modelled trapping efficiencies were also calculated, in line with the approach used in preceding chapters.

In Chapter 5, it has been found that hydraulic loading rate and inflow sediment concentration both influence the parameters of the $k-C^*$ model. It was important to examine whether or not those operational variables also impact on the model parameters ($a$ and $b$ in Equation 6.1) in the $N_f$ model. To verify this, the calibration was carried out for the experimental data, regrouped in terms of the magnitude of the selected operational parameter or in a random way, as listed below:

1. Hydraulic loading (see Appendix A1 for details on these experiments):
   - Experiments with low inflow (> 23.5 mL/s): Exp. 1, 2, 7, 8, and 9,
   - Experiments with high inflow (> 78.6 mL/s): Exp. 3, 4, 5, 6 and 10,
   - Experimental data with low hydraulic loading rate (1900 – 4000 m/y),
   - Experimental data with medium hydraulic loading rate (4000 – 80000 m/y),
   - Experimental data with high hydraulic loading rate (8000 – 15000 m/y),
   - Experimental data with very high hydraulic loading rate (20000 – 30000 m/y).

2. Inflow sediment concentration:
   - Experiments with low to medium input sediment concentration: Exp. 1, 2, 5, 7, 8 and 10,
   - Experiments with medium input sediment concentration: Exp. 2, 5, 8 and 9,
   - Experiments with medium to high input sediment concentration: Exp. 3, 4, 5, 6, 8 and 9.

3. Vegetation density:
   - Experimental data collected from non-vegetated mesocosm cell,
   - Experimental data collected from low-vegetated mesocosm cell,
   - Experimental data collected from medium-vegetated mesocosm cell,
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- Experimental data collected from high-vegetated mesocosm cell.

(4) Random selection of six experiments:

- Experiments: 1, 2, 5, 7, 9 and 10,
- Experiments: 1, 3, 6, 7, 8 and 10,
- Experiments: 1, 4, 5, 6, 7 and 9,
- Experiments: 1, 2, 5, 6, 7 and 9,
- Experiments: 1, 3, 4, 5, 7 and 8,
- Experiments: 1, 3, 4, 6, 7 and 9,
- Experiments: 1, 2, 4, 7, 8 and 10,
- Experiments: 1, 3, 4, 6, 7 and 8,
- Experiments: 1, 4, 5, 7, 9 and 10.

The values of the two model coefficients \( a \) and \( b \) for each group were derived using the same optimisation method as described above for all available measured trapping efficiencies. The number of observed trapping efficiencies varied with each data set, depending on the cases studied. The mean and variance of the two coefficients for each data set were also calculated to assess model sensitivity against the experimental variables (e.g., hydraulic loading rate, inflow sediment concentration and vegetation density), and the varying trends in these coefficients were identified from the graphs as did in Chapter 5.

Additionally, the reliability of the model was assessed using the coefficient of model efficiency \( E \) for each group of experiments as listed above. The corresponding modelled TSS was also plotted against the measured TSS, to discover if the predictions were located within \( \pm 25\% \) and \( \pm 50\% \) error bands.

6.2.2 Results and discussion

The best fit for the regression model described by Equation 6.1 is presented along with the all available experimental data in Figure 6-1. The values of the regression parameters are: \( a = 0.43 \) and \( b = 1.42 \). Both the model efficiency \( E \) and the cross-correlation coefficient \( R^2 \) between the measured and the modelled are 0.82.
Figure 6-1: Fitting of the curve based upon Equation 6.1 for wet weather experimental results at five different ranges of particle sizes.

For each group of different flow rates, hydraulic loadings, vegetation densities and input sediment concentrations and for any of six random experiments, the best fits for the model coefficients are listed in Table 6-1, along with the model efficiencies $E$ and the correlation coefficients $R^2$. Studies were also carried out for all experimental runs. The variations in $a$ and $b$ are also plotted in Figure 6-2 for different cases. No consistent trends in the variations of $a$ and $b$ are evident, for any of the tested groups. The coefficients of variation are 7.8 and 10.9% of the means for $a$ and $b$, respectively, with $R^2$ always greater than 0.76. Therefore, the model parameters are located in a small band and can be regarded as independent of the key experimental parameters (e.g. inflow rates, inflow sediment concentrations, vegetation density etc).

It should be noted that for the smallest particle size fraction (below 6 µm), the scattering degree of the observed data in Figure 6.1 is higher than the modelled. This is probably due to the proportionally higher uncertainty in PSD measurements for small particles. The model slightly underestimates $T_r$, of the biggest fraction (124 – 404 µm); this may also be due to the uncertainty associated with measuring the typically very low concentrations of large particles.

Compared to the strong variations in two parameters of the $k-C^*$ model assessed in Chapter...
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5 (see Table 5-2), the ranges of the parameters of the $N_f$ model are far smaller. This therefore indicates that the $N_f$ model, being physically-based, is more likely to be readily transferable from the laboratory study to the field conditions (i.e., the parameters should be the same across different wetlands).

Table 6-1: Wet weather model coefficients $a$ and $b$ and the statistics of the model fit.

<table>
<thead>
<tr>
<th>Case No.</th>
<th>Experiment No. (data category)</th>
<th>Model Coefficients</th>
<th>$E$</th>
<th>$R^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>All experiments: 1,2,3,4,5,6,7,8,9,10</td>
<td>0.43 1.42</td>
<td>0.82</td>
<td>0.82</td>
</tr>
<tr>
<td>2</td>
<td>Low Flow: 1,2,7,8,9</td>
<td>0.44 1.32</td>
<td>0.82</td>
<td>0.83</td>
</tr>
<tr>
<td>3</td>
<td>High Flow: 3,4,5,6,10</td>
<td>0.41 1.49</td>
<td>0.81</td>
<td>0.81</td>
</tr>
<tr>
<td>4</td>
<td>Low hydraulic loading (1900-4000m/y)</td>
<td>0.43 1.18</td>
<td>0.84</td>
<td>0.84</td>
</tr>
<tr>
<td>5</td>
<td>Medium hydraulic loading (4000-8000m/y)</td>
<td>0.43 1.25</td>
<td>0.82</td>
<td>0.83</td>
</tr>
<tr>
<td>6</td>
<td>High hydraulic loading (8000-15000m/y)</td>
<td>0.47 1.84</td>
<td>0.84</td>
<td>0.84</td>
</tr>
<tr>
<td>7</td>
<td>Very high hydraulic loading (20000-30000m/y)</td>
<td>0.32 1.63</td>
<td>0.76</td>
<td>0.76</td>
</tr>
<tr>
<td>8</td>
<td>Low Conc.-Medium Conc.: 1,2,5,7,8,10</td>
<td>0.44 1.45</td>
<td>0.82</td>
<td>0.84</td>
</tr>
<tr>
<td>9</td>
<td>Medium Conc.: 2,5,8,9</td>
<td>0.44 1.43</td>
<td>0.83</td>
<td>0.83</td>
</tr>
<tr>
<td>10</td>
<td>Medium Conc.-High Conc.: 3,4,5,6,8,9</td>
<td>0.43 1.39</td>
<td>0.83</td>
<td>0.83</td>
</tr>
<tr>
<td>11</td>
<td>All experiments for non-vegetated cell</td>
<td>0.47 1.72</td>
<td>0.82</td>
<td>0.82</td>
</tr>
<tr>
<td>12</td>
<td>All experiments for low-vegetated cell</td>
<td>0.45 1.38</td>
<td>0.89</td>
<td>0.89</td>
</tr>
<tr>
<td>13</td>
<td>All experiments for medium-vegetated cell</td>
<td>0.37 1.18</td>
<td>0.81</td>
<td>0.81</td>
</tr>
<tr>
<td>14</td>
<td>All experiments for high-vegetated cell</td>
<td>0.45 1.47</td>
<td>0.82</td>
<td>0.83</td>
</tr>
<tr>
<td>15</td>
<td>Random experiments No: 1,2,5,7,9,10</td>
<td>0.43 1.4</td>
<td>0.81</td>
<td>0.82</td>
</tr>
<tr>
<td>16</td>
<td>Random experiments No: 1,3,6,7,8,10</td>
<td>0.42 1.3</td>
<td>0.81</td>
<td>0.83</td>
</tr>
<tr>
<td>17</td>
<td>Random experiments No: 1,4,5,6,7,9</td>
<td>0.46 1.48</td>
<td>0.84</td>
<td>0.85</td>
</tr>
<tr>
<td>18</td>
<td>Random experiments No: 1,2,5,6,7,9</td>
<td>0.48 1.6</td>
<td>0.85</td>
<td>0.85</td>
</tr>
<tr>
<td>19</td>
<td>Random experiments No: 1,3,4,5,7,8</td>
<td>0.45 1.49</td>
<td>0.84</td>
<td>0.85</td>
</tr>
<tr>
<td>20</td>
<td>Random experiments No: 1,3,4,6,7,9</td>
<td>0.45 1.39</td>
<td>0.83</td>
<td>0.83</td>
</tr>
<tr>
<td>21</td>
<td>Random experiments No: 1,2,4,7,8,10</td>
<td>0.42 1.4</td>
<td>0.81</td>
<td>0.81</td>
</tr>
<tr>
<td>22</td>
<td>Random experiments No: 1,3,4,6,7,8</td>
<td>0.45 1.44</td>
<td>0.84</td>
<td>0.84</td>
</tr>
<tr>
<td>23</td>
<td>Random experiments No: 1,4,5,7,9,10</td>
<td>0.42 1.31</td>
<td>0.81</td>
<td>0.82</td>
</tr>
</tbody>
</table>

| Coefficient of variation (%) | 7.8 10.9 2.7 2.8 |

The predicted TSS concentrations – composed of the sums of the sediment weights of five fractions - are very close to the observed concentrations (see Figure 6-3). The $E$ and $R^2$ coefficients are 0.91 and 0.92, respectively, with all modelled concentrations being well within $\pm 50\%$, and most within $\pm 25\%$ range of the observed concentrations.
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(a) Optimum $a$ and $b$ values based on hydraulic loading rate categories.

(b) Optimum $a$ and $b$ values based on input sediment concentration categories.

(c) Optimum $a$ and $b$ values based on vegetation density categories.

(d) Optimum $a$ and $b$ values based on hydraulic loading rate, input sediment concentration and six random experiments (Case Number refers to Table 6-1).

Figure 6-2: Optimum $a$ and $b$ values based on hydraulic loading rate, input sediment concentration, vegetation density and six random experiments.
It should be noted that the optimum model parameters \((a = 0.43 \text{ and } b = 1.42)\) are quite different from the corresponding values found for grass swales and filter strips \((a = 0.69 \text{ and } b = 4.95, \text{ Deletic, 2005})\). This indicates that, although the proposed regression model (Equation 6.1) may be universally used for different stormwater treatment systems (e.g. wetlands, swales and filter strips), the model parameters are dependent on the system type.

Further testing of the model against field data is still needed. If additional research can show that the parameters are constant for any constructed stormwater wetland, as was the case for swales and filter strips (Deletic and Fletcher, 2006), then the model may be confidently used for wetland design.

![Figure 6-3: Predicted versus observed mass of sediment for all wet weather laboratory mesocosm experiments, conditions of which are given in Appendix A1.](image_url)

### 6.3 The dry weather model

#### 6.3.1 Modelling methodology

As described in Chapter 3, nine groups of laboratory experiments were carried out directly following the wet weather experiments in four mesocosm stormwater wetlands, to investigate sediment behaviour under dry weather conditions. In these experimental studies, the controlling factors (i.e. hydraulic loading, input sediment concentration and
vegetation density) were adjusted separately. As expected, the outflow sediment concentrations decreased exponentially with time under dry weather conditions. In addition, our analyses of the experimental results (Section 3.4.2) identified the governing and secondary factors controlling sedimentation under dry weather conditions. For example, inflow sediment particle size had a significant effect on sediment trapping efficiency. Vegetation density did not exhibit as an important factor for sediment removal during dry weather periods. Some flow characteristics, including hydraulic loading, flow velocity, water depth and detention time, did not demonstrate meaningful influence on sedimentation under laboratory experimental conditions.

Based on the statistical analyses of the dry weather experimental results in Section 3.4.2, it was noted that the time ratio $t^*$ (i.e. the ratio between the actual time $t$ since the last storm event and the mean detention time $t_d$ in the wetland as defined in Equation 3.5) also had a significant effect on particle trapping efficiency. These results should be taken into account as another key factor that determines the sediment trapping efficiency under dry weather conditions. It was thus expected that dry weather sediment trapping efficiency $Tr_s$ (defined in Equation 3.15) can be taken as a functional form of the Particle Fall Number $N_f$ and the non-dimensional time $t^*$,

$$Tr_s = f(N_f, t^*) \quad (6.2)$$

The relationship between trapping efficiency $Tr_s$ and the two non-dimensional numbers $N_f$ and $t^*$ was established in the following way.

As the wet weather flow model, it was assumed that the measured particle size distribution of stormwater can be adequately represented by five size fractions, covering five continuous particle size ranges (i.e. 0 - 6, 6 - 21, 21 - 46, 46 - 124 and 124 - 404 µm). The experimentally observed, mean wet weather sediment concentrations in each flume (i.e., mean TSS concentration in the flume at the start of the dry weather experiment), $C_{mean(wet),s}$ and dry weather outflow sediment concentrations, $C_{out(dry),s}$, were also determined for these five fractions with the following median diameters: $d_{50} = 3, 13.5, 33.5, 85$ and 264 µm,
using the method explained in Section 3.3.4.

The trapping efficiency of each size fraction can be evaluated separately, without considering the interaction between the particles, consistent with the assumptions used for the wet weather model. The trapping efficiency of fraction $s$, $T_r_s$, in the dry weather system was previously defined in Equation 3.15. To find the relationship between $T_r_s$ and two controlling variables $N_J$ and $t^*$, the observed non-dimensional outflow concentrations of each fraction, at each sampling time (defined as $C_{out(dry)}/C_{mean(wet)}$), were plotted against $N_J$ in different colours based on the range of $t^*$ value (0 -0.3, 0.3 – 0.7, 0.7 – 1.0, 1.0 – 2.0 and 2.0 – 3.0). They were then plotted against $t^*$, based on the range of $N_J$ value (0 -50, 10 -50, 50 -100, 100 – 1000 and 1000 – 3800) for all 929 points obtained from the experimental results. It should be borne in mind that the aim of the modelling approach is to use parameters as few as possible. In the end, the most appropriate mathematical equation was selected based on the findings from the graphs.

The selected equation was then applied to the five particle size fractions to predict the dry weather outflow trapping efficiency at each time. The model coefficients were calibrated against all available 929 data points by maximizing the objective function $E$ (the coefficient of model efficiency) for the variation between the modelled and observed outflow particle trapping efficiencies of the five fractions. The correlation coefficient between the measured and modelled results was also calculated.

Similar to the approach for the wet weather model, the effect of experimental variables (e.g., hydraulic loading rate, mean wet weather sediment concentration and vegetation density) on the model parameters was examined using chosen groups of experiments, as listed below:

(1) Hydraulic loading (see Appendix A2 for details on these experiments)
   - Experiments with low hydraulic loading: Exp. 2, 7, 8, and 9,
   - Experiments with high hydraulic loading: Exp. 3, 4, 5, 6 and 10.

(2) Mean antecedent wet weather sediment concentration in the flume:
• Experiments with low mean wet weather TSS concentration: Exp. 7 and 10,
• Experiments with medium mean wet weather TSS concentration: Exp. 2, 5, 8 and 9,
• Experiments with high mean wet weather TSS concentration: Exp. 3, 4, 6.

(3) Vegetation density:
• Experimental data collected from non-vegetated mesocosm cell,
• Experimental data collected from low-vegetated mesocosm cell,
• Experimental data collected from medium-vegetated mesocosm cell,
• Experimental data collected from high-vegetated mesocosm cell.

(4) Random selection of five experiments:
• Experiments: 2, 4, 6, 8 and 10,
• Experiments: 2, 3, 5, 7 and 10,
• Experiments: 3, 4, 5, 7 and 8,
• Experiments: 4, 5, 7, 8 and 10,
• Experiments: 3, 4, 8, 9 and 10,
• Experiments: 3, 6, 7, 8 and 9,
• Experiments: 2, 3, 4, 5 and 6,
• Experiments: 5, 6, 7, 8 and 9,
• Experiments: 6, 7, 8, 9 and 10,
• Experiments: 2, 3, 8, 9 and 10.

The values of model coefficients for each data set were derived from the same methodology as described above for maximizing the objective function $E$, based on the predicted and measured trapping efficiencies. Each random selection of experiments contained a different number of observed trapping efficiencies. The mean and variance of the optimum parameters $A$, $B$ and $C$ for each data set were calculated to assess model sensitivity to the experimental variables. They were also plotted against each experimental variable (as was the case in wet weather), to identify possible trends.

Ultimately, the modelled TSS concentration at each time and location was obtained by summing the concentrations of all five size fractions. Predicted and observed TSS...
concentrations were also compared to evaluate the model's reliability. The modelled TSS was also plotted against the measured TSS, as well as ± 25% and ± 50% error bands as did in the wet weather analysis.

6.3.2 Results and discussion

The non-dimensional outflow concentration, expressed as a proportion of the wet weather concentration, $C_{\text{out(dry)}}/C_{\text{mean(wet)}}$, is plotted against the Particle Fall Number $N_f$ on a log scale, for the five ranges of $t^*$, as shown in Figure 6-4. It is evident that the concentration ratio decreases with $N_f$. However, the decreasing rate depends strongly on the time ratio ($t^*$). For small values of $t^*$ the decrease is rapid, while for larger values of $t^*$ the process is far slower (again demonstrating an exponential form to reduction over time). It is also noted that there is a wider distribution of the data for smaller values of $N_f$ rather than for larger values.

It is seen from Figure 6-4 that the relationship between $C_{\text{out(dry)}}/C_{\text{mean(wet)}}$ and $\ln N_f$ seems to be almost linear for all but the lowest value of $t^*$.

The dimensionless concentration, $C_{\text{out(dry)}}/C_{\text{mean(wet)}}$, is also plotted against $t^*$ for the above five ranges of the Particle Fall Number $N_f$, as shown in Figure 6-5. It is clear that there is an exponentially decaying relationship between $C_{\text{out(dry)}}/C_{\text{mean(wet)}}$ and $t^*$ for all values of $N_f$.

Based on the experimental results presented in Figures 6-4 and 6-5, the particle trapping efficiency $T_{rs}$ for dry weather conditions can be expressed as a function of $\ln N_f$ and $e^{-t^*}$. A simple regression is then established to describe these trends;

$$
T_{rs} = \begin{cases} 
1 - Ae^{Bt^*} \left( C - \ln N_f \right) & \text{if } N_f \leq 800 \\
1 & \text{if } N_f > 800
\end{cases}
$$

(6.3)

where $A = 0.1$, $B = -1.87$ and $C = 7.29$.

The coefficient of model efficiency for this relationship, $E$, is 0.84, when all 929 measured points are used. This is very similar to the findings for the wet weather relationship and is a
Table 6-2 presents the best fits for the model coefficients $A$, $B$ and $C$, along with the model efficiencies $E$ and the correlation coefficients $R^2$, for all cases studied. The experimental runs were also grouped according to their values of hydraulic loading, mean wet weather concentration in the flume, vegetation density, along with a group of random selections of
five runs to test model sensitivity. The detailed variations of $A$, $B$ and $C$ are plotted in Figure 6-6 for different cases. As the wet weather experiments, there are no apparent trends in the model coefficients. The coefficient of variation is 6.59, 7.54 and 2.31% of the means for $A$, $B$ and $C$, respectively, with $R^2$ always greater than 0.81. Therefore, similar to the wet weather model, these regression parameters can be regarded as independent from the key variables of flow, concentration and vegetation, making the model readily transferable between sites.

Table 6-2: Dry weather model coefficients $A$, $B$ and $C$ and the statistics of the model fit.

<table>
<thead>
<tr>
<th>Case No.</th>
<th>Experiment No.</th>
<th>Model coefficients</th>
<th>$E$</th>
<th>$R^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>$A$</td>
<td>$B$</td>
<td>$C$</td>
</tr>
<tr>
<td>1</td>
<td>All: 2,3,4,5,6,7,8,9,10</td>
<td>0.10</td>
<td>-1.87</td>
<td>7.29</td>
</tr>
<tr>
<td>2</td>
<td>Low Flow: 2,7,8,9</td>
<td>0.10</td>
<td>-1.61</td>
<td>7.44</td>
</tr>
<tr>
<td>3</td>
<td>High Flow: 3,4,5,6,10</td>
<td>0.10</td>
<td>-2.07</td>
<td>7.17</td>
</tr>
<tr>
<td>4</td>
<td>Low mean wet weather Conc.: 7,10</td>
<td>0.10</td>
<td>-1.79</td>
<td>7.44</td>
</tr>
<tr>
<td>5</td>
<td>Medium mean wet weather Conc.: 2,5,8,9</td>
<td>0.10</td>
<td>-1.87</td>
<td>7.00</td>
</tr>
<tr>
<td>6</td>
<td>High mean wet weather Conc.: 3,4,6</td>
<td>0.10</td>
<td>-2.07</td>
<td>7.23</td>
</tr>
<tr>
<td>7</td>
<td>All experiments for non-vegetated cell</td>
<td>0.09</td>
<td>-2.10</td>
<td>7.24</td>
</tr>
<tr>
<td>8</td>
<td>All experiments for low-vegetated cell</td>
<td>0.11</td>
<td>-1.88</td>
<td>6.76</td>
</tr>
<tr>
<td>9</td>
<td>All experiments for medium-vegetated cell</td>
<td>0.11</td>
<td>-2.31</td>
<td>7.08</td>
</tr>
<tr>
<td>10</td>
<td>All experiments for high-vegetated cell</td>
<td>0.10</td>
<td>-1.92</td>
<td>7.12</td>
</tr>
<tr>
<td>11</td>
<td>Random experiments No: 2,4,6,8,10</td>
<td>0.11</td>
<td>-2.01</td>
<td>7.12</td>
</tr>
<tr>
<td>12</td>
<td>Random experiments No: 2,3,5,7,9</td>
<td>0.10</td>
<td>-2.00</td>
<td>6.95</td>
</tr>
<tr>
<td>13</td>
<td>Random experiments No:3,4,5,7,8</td>
<td>0.10</td>
<td>-1.94</td>
<td>7.02</td>
</tr>
<tr>
<td>14</td>
<td>Random experiments No: 4,5,7,8,10</td>
<td>0.10</td>
<td>-1.93</td>
<td>7.07</td>
</tr>
<tr>
<td>15</td>
<td>Random experiments No: 3,4,8,9,10</td>
<td>0.09</td>
<td>-1.85</td>
<td>7.07</td>
</tr>
<tr>
<td>16</td>
<td>Random experiments No: 3,6,7,8,9</td>
<td>0.09</td>
<td>-1.96</td>
<td>7.20</td>
</tr>
<tr>
<td>17</td>
<td>Random experiments No: 2,3,4,5,6</td>
<td>0.11</td>
<td>-2.16</td>
<td>6.94</td>
</tr>
<tr>
<td>18</td>
<td>Random experiments No: 5,6,7,8,9</td>
<td>0.09</td>
<td>-1.98</td>
<td>7.19</td>
</tr>
<tr>
<td>19</td>
<td>Random experiments No: 6,7,8,9,10</td>
<td>0.09</td>
<td>-1.91</td>
<td>7.28</td>
</tr>
<tr>
<td>20</td>
<td>Random experiments No: 2,3,8,9,10</td>
<td>0.10</td>
<td>-1.86</td>
<td>7.05</td>
</tr>
</tbody>
</table>

Coefficient of variation (%) 6.59 7.54 2.31 3.07 3.75
Figure 6-6: Optimum A, B and C values based on different flow rates, input sediment concentrations and five random experiments (Case Number refers to Table 6-2).
To test the proposed model further, as explained in Section 6.3.1, Equation 6.3 was used to predict concentrations for the five size fractions, and these concentrations were then summed to calculate the total TSS concentration for all measured points. The modelled versus predicted TSS concentrations are shown in Figure 6-7. The predicted total TSS concentrations match the observed concentrations well; with the coefficient of model efficiency $E = 0.91$. The correlation coefficient $R^2$ between the measured and the modelled results is also very high ($R^2 = 0.93$). Figure 6-7 shows the agreement between the measured and modelled TSS concentrations, and the scattered points are located within ± 50% error bands. It should be noted that the dispersion of the data for fine particles may be largely, if not completely, attributed to the uncertainty and inaccuracy in the PSD measurements as discussion in Section 3.3.3.

![Figure 6-7: Predicted versus observed mass of sediment for all dry weather laboratory mesocosm experiments whose conditions are given in Appendix A2.](image)

The results indicate that the proposed model performs better in predicting the trapping efficiency for the whole particle size range than for the individual size fraction. However, this is not surprising, since it is very likely that the errors in the measurement of each particle fraction combine to produce a smaller relative overall error (Bertrand-Krajewski, 2004).
6.4 The new physically-based model of sediment behaviour in wetlands

Treatment of sediments by a free surface flow wetland involves deposition, re-suspension, filtration and infiltration of particles, as shown in Figure 6-8. The proposed model assumes that filtration and infiltration can be merged with deposition that is the main process considered in this model. It was asserted in Chapters 3 and 4 that the contribution of sediment re-suspension to TSS due to water flows in operational ranges (i.e. when a bypass exists to convey the high flows) is negligible, and therefore the re-suspension due to flow does not need to be taken into account. There may be significant sediment re-suspension due to other environmental sources such as birdlife and wind, which has not been studied in the current laboratory experiments (the study of such unpredictable factors would be very difficult). Thus, these processes, or the inability of wetland to settle all the particles, are represented using a lumped parameter, named “background concentration”, C*. In other words, it is assumed that due to unpredictable environmental influences, the outflow concentrations will never be lower than C*.

![Figure 6-8: Processes involved in sedimentation in surface flow wetlands.](image)

The model uses separate equations for wet and dry weather flow regimes, accounting for the difference in the treatment mechanisms during storm events and inter-event periods. Therefore the model includes a switch between wet and dry weather, depending on whether the inflow rate $Q(t)$ is above a threshold value, named here “base flow”.

6-17
This section explains the model structure and governing equations, as well as numerical methods of the new physically-based model. The input data requirements and the model parameters are also discussed.

### 6.4.1 Model structure and governing equations

It is assumed that a constructed stormwater wetland can be conceptually simplified as a series of prismatic pools (i.e. cells) that have their equivalent length \(L\) and width \(W\), depth \(H\) and storage volume \(S\) according to the actual system layout. This means that the volume and area of the system remain as constructed, but the shape is simplified. Each cell is a totally mixed water body and is thus modelled separately. The outflow rate and the sediment concentration from the upstream cell, become the input for the immediate downstream cell as shown in Figure 6-9.

![Figure 6-9: Conceptualised cell chain for free surface flow wetlands.](image)

In addition, the modelling approach assumes that the total sediment can be represented by five fractions and each fraction behaves independent of the other as shown in Figure 6-10. As a result, deposition and re-suspension due to environmental impacts are modelled for each fraction, and the fractions' outflow concentrations are eventually summed to obtain the outflow sediment concentration. Sediment trapping efficiency for each fraction under wet and dry weather conditions is calculated based on the results already presented in this chapter (i.e. Equations 6.1 and 6.3). Re-suspension due to environmental factors is modelled here by introducing a lumped parameter - the background concentration, \(C^*\). This parameter needs to be calibrated using field data. The processes of sedimentation and re-suspension are thus combined in Equation 6.7 in Table 6-3 to calculate the outflow TSS concentrations from the cell for both wet and dry weather conditions.
Figure 6-10: Components of the sediment mass and their balance for a treatment cell. \( f_r \) represents the particle fraction \( s \). \( C_{in}, C_{out} \) and \( C^* \) represent the inflow, outflow and background sediment concentrations, \( C_{in,s}, C_{out,s} \) and \( C^*_s \) the inflow, outflow and background concentrations for particle fraction \( s \), PSD\text{\_wet} and PSD\text{\_dry} the inflow wet weather PSD and dry weather PSD.

For each wetland cell, the model is applied in the manner as illustrated in Figure 6-11. The input data including geometric parameters, PSDs and flow characteristics (to be discussed in the following sub-section), need to be specified for a cell. In particular, the proposed model for sediment trapping has to be coupled with an appropriate hydraulic model to account for the unsteady nature of stormwater flow through the real treatment systems. The flow characteristics (i.e. water depth, velocity, storage volume and detention time) can be calculated using any suitable flow model, depending on the user preferences. For example, a simple flow routing method (such as the Puls method used in MUSIC) can be used, or a complex 3-D flow model.

The continuously stirred tanks reactors (CSTRs) model (explained in Section 2.5.2) has been adopted here to account for the flow hydrodynamics within a wetland. The treatment wetland is represented by an appropriate number of CSTR cells. Each cell is assumed to be totally mixed. Thus, some features associated with this assumption must be taken into account in the calculations. The mean sediment concentration \( C_{mix,s} \), after fully mixing for particle fraction \( s \) in Equation 6.7, is used to replace the input TSS concentration in Equation 3.14 for wet weather conditions. Also, for dry weather conditions, the time step \( \Delta t \) is used to replace \( t \) in the calculation of \( t^* \) (i.e., \( t^* = t/t_d = \Delta t/t_d \)) since the sediments in each cell are fully mixed at the start of each time step.
The new physically-based model therefore comprises the following sequential modules as presented in the flow chart in Figure 6-11 with all the equations listed in Table 6-3.

(1) **Sediment input data module**: TSS input concentrations during either wet or dry weather are separated into five fractions based on the supplied particle size distributions (PSDs). In the same way, the background concentration $C^*$ is also separated into five $C_{s*}$ representing the background concentration for each fraction (the parameter $C^*$ will be discussed below);

(2) **Sediment mixing module** that adopts the concept of the CSTRs model to account for the sediment dilution process in the cell: The inflow sediments instantly mix completely with those in the storage of the cell. It is assumed that there is no sediment deposition or water discharge during the time of mixing;

(3) **Sediment trapping module**: Sediment trapping is modelled separately for wet and dry weather cases, following the methods developed in the first part of this chapter. The equation for calculating the trapping efficiency for each fraction, $Tr_s$, is chosen based on the flow regime;

(4) **Outflow sediment concentrations module**: For each fraction, the outflow concentration $C_s$ is calculated taking into account both the deposition ($Tr_s$) and re-suspension (through the background concentration, $C_{s*}$). These outflow sediment concentrations are determined as a function of the mean sediment concentration of the fraction $C_{mix,s}$ in the cell;

(5) **The total outflow TSS concentration module**: A sum of the concentrations of all five fractions simply gives the total outflow TSS concentration. In this way, the model produces a TSS pollutograph at the outlet cross-section of each CSTR cell in a wetland.
Model development

Conceptualisation of the surface-flow system:
Number of cells and their layout

Input data
L, W, H, C_{in}(t), PSD_{wet}, PSD_{dry}

Flow model
h(t), V(t), S(t), t_d(t)

The New physically based N_{f} model

Input TSS fractions:
\[ C_{in,s} = C_{in} \times \begin{cases} \text{PSD}_{wet,s} & \text{if } Q(t) > Q_{\text{baseflow}} \\ \text{PSD}_{dry,s} & \text{if } Q(t) \leq Q_{\text{baseflow}} \end{cases} \]
\[ C_s^* = C^* \times \text{PSD}_{dry,s} \]

Sediment mixing:
\[ \frac{d(SC_s)}{dt} = Q_{in}C_{in,s}(t) \Rightarrow C_{mix,s}(t) \]

Sediment trapping:

Wet weather: If \( Q(t) > Q_{\text{baseflow}} \):
\[ T_{rs} = \frac{N_f^{0.43}}{N_f^{0.43} + 1.42} \]

Dry weather: If \( Q(t) \leq Q_{\text{baseflow}} \):
\[ T_{rs} = \begin{cases} 1 - 0.1 e^{-1.871(7.29 - \ln N_f)} & \text{if } N_f \leq 800 \\ 1 & \text{if } N_f > 800 \end{cases} \]

Outflow TSS fractions:
\[ C_{out,s} = C_s^* + (1 - T_{rs})(C_{mix,s} - C_s^*) \]

Total outflow TSS concentration \( C_{out}(t) \):
\[ C_{out}(t) = \sum_{i=1}^{5} C_{out,s}(t) \]

Figure 6-11: Model structure and key equations.
### Table 6-3: The governing equations of the physically-based N_r wetland model.

<table>
<thead>
<tr>
<th>Model equation</th>
<th>Comment</th>
<th>Eq. No.</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Input TSS fraction module</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>$C_{in,s} = C_{in} \times \begin{cases} \text{PSD}<em>{wet,s} &amp; \text{if } Q(t) &gt; Q</em>{baseflow} \ \text{PSD}<em>{dry,s} &amp; \text{if } Q(t) \leq Q</em>{baseflow} \end{cases}$</td>
<td>Wet or dry weather inflow TSS concentration of fraction $s$</td>
<td>6.4</td>
</tr>
<tr>
<td>$C_s^* = C_s^* \times \text{PSD}_{dry,s}$</td>
<td>Background concentration of fraction $s$</td>
<td>6.5</td>
</tr>
<tr>
<td><strong>Sediment mixing module</strong></td>
<td>Calculating the mean concentration of fraction $s$ after mixing</td>
<td>6.6</td>
</tr>
<tr>
<td>$\frac{d(SC_s)}{dt} = Q_{in}C_{in,s}(t)$</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Sediment trapping module</strong></td>
<td>Wet weather trapping efficiency of fraction $s$</td>
<td>6.1</td>
</tr>
<tr>
<td><strong>Wet weather process</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>$T_r = \frac{N_f^{0.43}}{N_f^{0.43} + 1.42}$</td>
<td>Particle Fall Number</td>
<td>3.3</td>
</tr>
<tr>
<td>in which $N_f = \frac{LV_s}{HV}$</td>
<td>Particle falling velocity: Stoke’s law</td>
<td>2.12</td>
</tr>
<tr>
<td>$V_s = \frac{g}{18\mu}(\rho_s - \rho_w) d_s^2$</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Dry weather process</strong></td>
<td>Dry weather trapping efficiency of fraction $s$</td>
<td>6.3</td>
</tr>
<tr>
<td>$T_r = \begin{cases} 1 - 0.1e^{-1.87t^*} (7.29 - \ln N_f) &amp; \text{if } N_f \leq 800 \ 1 &amp; \text{if } N_f &gt; 800 \end{cases}$</td>
<td>Time ratio</td>
<td>3.5</td>
</tr>
<tr>
<td>in which $t^* = \frac{t}{t_d}$</td>
<td>Particle Fall Number</td>
<td>3.3</td>
</tr>
<tr>
<td>$N_f$</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Outflow TSS fraction module</strong></td>
<td>Outflow TSS concentration of fraction $s$</td>
<td>6.7</td>
</tr>
<tr>
<td>$C_{out,s} = C_s^* + (1 - T_r)(C_{mix,s} - C_s^*)$</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Total outflow TSS module</strong></td>
<td>Total outflow TSS concentration</td>
<td>6.8</td>
</tr>
<tr>
<td>$C_{out}(t) = \sum_{s=1}^{S} C_{out,s}(t)$</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

$Q_{in}$ the inflow rate  
$Q_{baseflow}$ the base flow in the wetland  
$C_{in}$ and $C_{out}$ the inflow and outflow sediment concentrations, respectively  
$C_{in,s}$ and $C_{out,s}$ the influent and effluent concentrations for particle fraction $s$, respectively  
$C_s^*$ and $C_s^*$ the background concentration and background concentration for particle fraction $s$, respectively  
$\text{PSD}_{wet,s}$ and $\text{PSD}_{dry,s}$ the percents of particle fraction $s$ of total weight based on the wet weather inflow particle size distribution and dry weather particle size distribution, respectively  
$C_{mix,s}$ the flow-weighted mean sediment concentration of fraction $s$ after inflow sediments mixing totally with those ones in the storage of the cell
S the water storage of the cell
N the Particle Fall Number
L the length of the cell
H the mean water depth in the cell
V the mean flow velocity
V the falling velocity of particle s
d, the diameter of the particle s
ρs and ρw the densities of the particles s and water, respectively
µ the dynamic viscosity of water
Tr.s the trapping efficiency of particle fraction s
t* the time ratio
t the time since the last storm event (or time step Δt)
td the mean detention time in the cell

6.4.2 Numerical solution method

A simple finite difference method is used to solve the key model equations in Table 6-3. The element of the numerical scheme is presented in Figure 6-12, which in fact presents one CSTR cell. In the figure, j is an index of time step, Δx the length of the cell and Δt the time step are both constant for the entire calculation regime. The flow rate Q and the sediment concentration of each fraction s entering and leaving the cell are assigned to the four points of the element at a time step, referring as Q\textsubscript{in}\textsuperscript{j} and Q\textsubscript{in}\textsuperscript{j+1}, Q\textsubscript{out}\textsuperscript{j} and Q\textsubscript{out}\textsuperscript{j+1}, C\textsubscript{in,s}\textsuperscript{j} and C\textsubscript{in,s}\textsuperscript{j+1}, C\textsubscript{out,s}\textsuperscript{j} and C\textsubscript{out,s}\textsuperscript{j+1}, while S\textsuperscript{j} and S\textsuperscript{j+1} are the storage volumes of the cell at times j and j+1, respectively, in Figure 6-12.

Figure 6-12: A single CSTR cell modelling scheme.

The equations are solved using the method outlined below.

(1) Sediment input data module: The background concentration and the inflow
concentrations are separated into five fractions and calculated in a straightforward manner, that is,

\[ C_s^* = C^* \times PSD_{\text{dry},s} \]  

\[ C_{in,s}^j = C_{in}^j \times \begin{cases} 
PSD_{\text{wet},s} & \text{if } Q_{j+1} > Q_{\text{baseflow}} \\
PSD_{\text{dry},s} & \text{if } Q_{j+1} \leq Q_{\text{baseflow}} 
\end{cases} \]  

where \( Q_{j+1} \) is the mean flow rate in the cell during the time increment \( j \) to \( j+1 \), which is calculated as

\[ Q_{j+1} = \frac{Q_{in}^j + Q_{in}^{j+1} + Q_{out}^j + Q_{out}^{j+1}}{4} \]  

(2) **Sediment mixing module:** For the time increment from \( j \) to \( j+1 \), Equation 6-6 is rewritten as

\[ S_{mix,s}^{j+1} - S_j C_{out,s}^j = \bar{Q}_{in} C_{in,s} \]  

where \( C_{mix,s}^{j+1} \) is the flow-weighted mean sediment concentration for fraction \( s \) after fully mixing. The mean input mass flow rate of fraction \( s \) is given by

\[ Q_{in} C_{in,s} = \frac{Q_{in}^j C_{in,s}^j + Q_{in}^{j+1} C_{in,s}^{j+1}}{2} \]  

Considering the water mass balance in the time increment according to the Puls method, we have

\[ S_{j+1} = S_j + (\bar{Q}_{in} - \bar{Q}_{out}) \Delta t \]  

where \( \bar{Q}_{in} = (Q_{in}^j + Q_{in}^{j+1})/2 \) and \( \bar{Q}_{out} = 0 \) at the mixing stage.

Combining Equations 6.11 and 6.13, \( C_{mix,s}^{j+1} \) in the time increment \( j \) to \( j+1 \) is then given by,

\[ C_{mix,s}^{j+1} = \frac{S^j C_{out,s}^j + Q_{in} C_{in,s} \Delta t}{S^j + Q_{in} \Delta t} \]
At the start of the calculation \((j = 0)\), \(C_s^{j}\) of each CSTR should be \(C_s^*\).

(3) **Sediment trapping module:** The dimensionless Particle Fall Number \(N_f\) for fraction \(s\) in each CSTR cell during the time increment \(j\) to \(j+1\) is calculated as

\[
N_{f}^{j+1} = \frac{\Delta x V_s}{H^{j+1} V^{j+1}} = \frac{V_s}{q^{j+1}}
\]  
(6.15)

where \(H^{j+1}\) and \(V^{j+1}\) are mean water depth and flow velocity in the cell in this time increment, respectively. The mean hydraulic loading rate \(q^{j+1}\) for this time increment is calculated as

\[
\frac{q^{j+1}}{A} = \frac{Q^{j+1}}{A}
\]  
(6.16)

where \(A\) is the surface area of the notional CSTR cell \((m^2)\), which is assumed to be time independent.

The calculations are carried out at a fixed time step \(\Delta t\), and the value of \(N_f\) is updated at every time step for each CSTR cell. In particular, the \(N_f\) value that is quoted for a storage cell (i.e. full length \(\Delta x\) of the cell) assumes a steady-state treatment over the full detention time. During the wet weather period, when the selected time step is shorter than the detention time, the effective treatment length is only a fraction of the full length \(\Delta x\). As a result, a time factor \(t^*\), which is the ratio of the time step \(\Delta t\) to the mean detention time \(t_{d}^{j+1}\) in the time increment \(j\) to \(j+1\), must be applied to scale the \(N_f\) value for that time step. On the other hand, if the time step is longer than the mean detention time, the time factor \(t^*\) is larger than unity. This implies that the water does not stay in the CSTR cell for the full time step. The treatment performance, however, should be the same as that within the detention time. Consequently, the time factor \(t^*\) should not be greater than unity.

Thus, under wet weather conditions, the trapping efficiency \(T_{r_s}^{j+1}\) of particle fraction \(s\) during the time increment \(j\) to \(j+1\) is determined by,
\[
Tr_s^{j+1} = \frac{(N_f^{j+1} \times t^*)^{0.43}}{(N_f^{j+1} \times t^*)^{0.43} + 1.42} \left( \frac{V_s}{q_s^{j+1} \times t_d^{j+1}} \right)^{0.43} + 1.42
\]

(6.17)

where the mean detention time \( \bar{t}_d^{j+1} \) in the time increment \( j \) to \( j+1 \) is given by

\[
\bar{t}_d^{j+1} = \frac{S^{j+1}}{Q^{j+1}}
\]

(6.18)

in which the mean storage volume \( S^{j+1} \) in the time increment \( j \) to \( j+1 \) is calculated by

\[
S^{j+1} = \frac{S^j + S^{j+1}}{2}
\]

(6.19)

Under dry weather conditions, the trapping efficiency \( Tr_s \) is a function of \( N_f \) and \( t^* \) that has already taken into account the effects of the time step in the calculations. The formula is

\[
Tr_s^{j+1} = \begin{cases} 
1 - 0.1e^{-1.87t^*^{j+1}} (7.29 - \ln N_f^{j+1}) & \text{if } N_f \leq 800 \\
1 & \text{if } N_f > 800
\end{cases}
\]

(6.20)

in which \( t^*^{j+1} = \frac{\Delta t}{t_d^{j+1}} \)

(4) Outflow sediment concentrations module: The following expression is used for both wet and dry weather conditions in calculating the effluent TSS concentration of particle fraction \( s \) of each cell at the time \( j+1 \),

\[
C_{out,s}^{j+1} = C_s^* + (C_{mix,s}^{j+1} - C_s^*)(1 - Tr_s^{j+1})
\]

(6.21)

(5) The total outflow TSS concentration module: The total TSS concentration of the outlet can be calculated as a sum of the concentrations for all five fractions, that is,

\[
C_{out}^{j+1} = \sum_{i=1}^{5} C_{out,s}^{j+1}
\]

(6.22)
6.4.3 The input data

The model requires the following four different types of input data:

- The equivalent length, depth, and width of each wetland cell. As explained, the wetland is conceptualised as a number of simplified cells, which have the same surface area and volume as the specified part of the wetland.

- Inflow (into the wetland) wet weather and dry weather sediment particle size distributions. Preferably these should be measured, but can also be assumed using previously reported values.

- Time series of inflow rates into the first wetland cell. Usually these are measured or modelled using a suitable rainfall/ runoff model.

- Time series of inflow TSS concentrations into the first wetland cell. Again, these can be measured or modelled using a suitable stormwater quality model.

However, flow characteristics have to be modelled using a hydrodynamic model, and they can therefore be regarded as input data for the sediment model.

6.4.4 Model parameters

The only model parameter that has to be calibrated in the Nr treatment model is the background concentration \( C^* \), which accounts for the all environmental impacts that may disturb sedimentation on a specific site, since the effects of wind and birdlife vary at different sites. Measurement of \( C^* \) is relatively straightforward (being able to be done during dry weather periods), or once more, its reported values can be used as a number of studies on background concentrations in wetlands exist (Walsh et al., 1997; Fletcher et al., 2004; Wong et al., 2006).

Furthermore, under unsteady flow conditions, the Nr model is coupled with a CSTRs model to account for flow hydrodynamics and sediment trapping simultaneously, that is, it has to be applied to each of these CSTR cells in series. The number of CSTRs \( N \) becomes another parameter for the combined model. Each CSTR cell can be assumed to have \( \frac{1}{N} \) of the total volume of the real treatment storage. To achieve the best model performance, the
number of CSTRs is also required to be calibrated.

6.4.5 Model coding and testing

The model algorithm is straightforward as presented in Figure 6-11. In particular, the time series flow rates and storage volume of each CSTR cell can be created from a flow model, one-by-one, along the cell chain. All input data are prepared for the inlet of the wetland, such as the sediment particle concentration of each fraction. The \( N_r \) model is used to calculate the output concentrations of each cell for each particle size fraction in a time step \( \Delta t \). The obtained results at the outlet of a cell are transferred to the inputs of the immediately downstream cell for next time step. It should be noted before concluding this time step, the sum of output concentrations of the last cell is the outflow TSS of the wetland. The above processes are repeated for each time step until the assigned time is reached and the outflow pollutograph of the wetland is finally generated.

The about algorithm was implemented in a simple Excel spreadsheet. The code was tested using a check on mass balance among inflow, outflow, and deposited sediments in each cell, for each time increment.

6.5 Conclusions

In summary, based on sedimentation theory and analyses of the experimental data, a simple non-linear regression algorithm was developed for the prediction of the trapping efficiency of sediments and attached pollutants in constructed stormwater wetlands. Sediment trapping efficiency is a function of the dimensionless Particle Fall Number \( N_r \), under wet weather conditions (Equation 6.1) and is a function of both \( N_r \) and the normalized time ratio \( t^* \), under dry weather conditions (Equation 6.3). The wet and dry weather model parameters are insensitive to hydraulic loading rate, inflow sediment concentration, and vegetation density. The \( N_r \) model thus seems to be applicable across a number of stormwater wetlands without further calibration. The proposed method for calculation of TSS was of high accuracy with model efficiency \( E \) greater than 0.90 and majority of
concentrations predicted fall well within ± 50% of the observed values under both wet and dry weather conditions, based on the laboratory experiment data.

Integrating both wet and dry weather features in the $N_f$ model, makes it capable of continuously assessing the long-term performance of stormwater constructed wetlands during each storm event and over inter-event dry periods. It was also found that the approach has some limitations for very fine particles less than 6 µm in diameter because of the low accuracy for their concentration measurement.

A modelling algorithm, incorporated with the $N_f$ model for long-term assessment of the wetlands, is elucidated. The integrated $N_f$ model in conjunction with an appropriate hydraulic model and the CSTRs model can account for the unsteady nature of stormwater flow through the real wetlands. Only the apparent background concentration $C^*$ and the number of CSTRs $N$ need to be calibrated using the field data. This new physically-based $N_f$ model will be tested in the following chapters.
Chapter 7

A Field Study at the Ruffey’s Creek Wetland

7.1 Introduction

The aim of the work explained in this chapter is to collect field data for testing both the $k-C^*$ model in Chapter 5 and the newly proposed model discussed in Chapter 6. One of the main objectives for this thesis is to examine whether or not the new model that was fully developed from laboratory data can be applied to a constructed stormwater wetland without any further calibration and tuning.

Collecting field data from a stormwater system is highly complicated and costly. Therefore it was decided to use existing field data if possible, with limited additional field measurements. Fortunately such a data set was available and easily accessible. In 2003, Monash University was involved in a monitoring program of the Ruffey’s Creek Wetland, as a part of the work with the Cooperative Research Centre for Catchment Hydrology (CRCCH, Australia). There was good ‘in-house’ knowledge of the monitoring program of this urban stormwater wetland that could provide the data set required for the testing of the models.

The Ruffey’s Creek Wetland was designed to control and treat stormwater inflows from a residential catchment in Melbourne. The aims of the water monitoring project undertaken by the CRCCH were to (Fletcher, 2001):

- Monitor the fate of pollutants entering the Ruffey’s Creek Wetland under varying hydraulic loading regimes.
- Provide data to enhance existing relationships between contaminant removal,
design parameters and pollutant characteristics.

- Refine model capabilities to predict constructed stormwater wetland performance.

This chapter firstly describes the layout of the Ruffey’s Creek Wetland, and then reports the intensive monitoring program and water quality and quantity data collected during eight storm events and baseflow conditions. However, the existing data set does not include all data imperative for the model testing, and therefore some additional field work had to be carried out to complete it. For instance, the particle size distribution (PSD) of sediment was not given by the old data collection, and therefore the key objective of new field work is to collect the PSD data for both dry and wet weather cases. The details for the additional field work carried out by the author are also discussed in this chapter.

7.2 Site description

The Ruffey’s Creek Wetland (RCW), located in the Bonview sub-catchment in Doncaster, Melbourne, Australia, was constructed in 1997. The wetland configuration was retrofitted in 2001 to enable better control of the hydrologic regime and to increase the detention time for enhancing pollutant removal performance (Fletcher, 2001). A photograph of the wetland is shown in Figure 7-1, while Figure 7-2 presents its plan and longitudinal cross-sectional views. The RCW receives runoff from a 114.5 ha sub-catchment of the Yarra River. The sub-catchment includes a range of land uses such as residential, commercial activity, park and recreation, public education, local government and road. The impervious area is approximate 51% of the total sub-catchment (Taylor, 2006). The mean annual rainfall range of the catchment is about 700-800mm (Bureau of Meteorology, 2005).

The RCW receives approximately 92% runoff of the sub-catchment via a main drainage pipe and the remaining 8% runoff via a small drainage pipe (Taylor, 2006). The wetland drains are discharged into the Ruffey’s Lake, which was designed to function as a retarding basin receiving water from the wetland and adjacent catchments.

This free surface flow constructed wetland consists of four treatment cells – an inlet
sedimentation pond, an ephemeral cell, and two macrophyte cells as shown in Figure 7-1. A suite of vegetation species such as trees, shrubs, grasses, herbs, rushes and sedges were incorporated into the multiple vegetation zones. Among the more plentiful species were *Bulboschoenus medianus*, *Poa ensiformis*, *Schoenoplectus validus*, *Poa labillardieri*, *Microlaena stipoides* and *Carex appressa* (Taylor, 2006). These species have been segmented into specified zones throughout the wetland, and plant density was the greatest in the ephemeral and macrophyte cells (Figure 7-4 and Figure 7-5).

Macrophyte cell 2 and the Ruffey’s Lake were not considered in the wetland monitoring regime (an embankment separates Macrophyte Cell 1 and Macrophyte Cell 2), and thus were not monitored in the CRCCH project. The schematics of the monitored wetland cells are illustrated in Figure 7-2.

![Figure 7-1: Ruffey's Creek Wetland, Melbourne, Australia.](image)

**Inlet Sedimentation Pond**

Stormwater enters the inlet pond via a 1600mm diameter pipe and a 600 mm diameter pipe (Figure 7-2 and Figure 7-3-a). The inlet sedimentation pond has a permanent volume of water (the normal operating water depth is 1.2m) for providing pre-sedimentation of coarse
Field study to medium-sized particles before they enter the macrophyte zone of the wetland (Figure 7-2). The wetland has two bypass mechanisms; one occurs when flows of excessive rates enter the inlet zone, and the other occurs when the macrophyte zones are fully inundated.

The extended water depth of the inlet pond is 0.7 m, so that flow volumes in excess of 0.7 m deep can drain from the inlet pond via a bypass spillway passing around the downstream macrophyte zones, in order to maintain appropriate hydrologic control, and to allow sufficient time for residential water to be treated. The invert of the bypass spillway is set to the same level as the obvert of the riser outlets (which is 0.25m below the obvert of the culvert) in Macrophyte Cell 1 to allow feedback-control of water level in the wetland (Fletcher, 2001) as shown in Figure 7-2. Two 750mm culverts (grade 1 in 60) are located at the downstream end of the inlet pond for conveying flow through to the ephemeral cell, and the entry of the culverts was configured to deflect low to medium flows into one culvert (Figure 7-3-b).

**Bypass Spillway**

The bypass spillway, able of conveying the 100 year ARI flow, is positioned at the downstream end of the inlet pond (Figure 7-2) to protect the downstream macrophyte zones from high flow velocities in order to avoid vegetation damage, sediment re-suspension and scouring. As previously described, the spillway operation is triggered under either of two conditions:

- when the inlet flow rate is higher than the discharge capacity of two downstream culverts connecting the inlet pond to the ephemeral cell, i.e. operating under inlet control conditions; and
- when the ephemeral cell – the macrophyte zone, is full, and thus the inlet pond is subject to the downstream water depth, i.e. operating under backflow conditions.

**Ephemeral Cell**

The ephemeral cell was designed to promote trapping and aerobic breakdown of organic matter. The stormwater flows into the ephemeral cell via the two culverts, and the invert of
culvert outlet is placed to the same level as the base of the upstream end of the ephemeral cell (Figure 7-4-a). Flow in the culverts is thus affected by downstream conditions. A sinuous low flow channel was built in the middle of the ephemeral cell surrounded by dense macrophytes, to conveying dry weather low flow during inter-event periods (Figure 7-4-b). Hence the ephemeral cell has a hydrologic operation different from the macrophyte cell and experiences regular inundation (wetting and drying cycles).

**Macrophyte Cells**

Macrophyte Cell 1 was designed for enhancing sedimentation and filtration, further trapping pollutants associated with fine suspended particles and soluble stormwater pollutants. It consists of a permanent pool and an extended detention storage (extended water depth 1.0m). The ephemeral cell and Macrophyte Cell 1 are separated by a porous rock embankment to distribute flow evenly to the downstream cell and to dissipate high-energy flow during the filling phase of the cell (Figure 7-5-a). Two riser outlets (each consisting of a 750mm diameter upstanding pipe with three 100mm orifices at different levels, designed to maintain a uniform detention time regardless of stage height; Fletcher, 2001) were installed at the downstream end of Macrophyte Cell 1 for outflow control to achieve relatively constant detention time, as shown in Figure 7-5-b.

The inlet sedimentation pond and wetland permanent pool surface areas, volumes, and extended water depths (based on the water surface level of permanent pool) are given in Table 7-1.

**Table 7-1: Physical characteristics of the Ruffey’s Creek Wetland.**

<table>
<thead>
<tr>
<th>Cell</th>
<th>Permanent pool surface area (m²)</th>
<th>Permanent pool volume (m³)</th>
<th>Extended water depth (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Inlet pond</td>
<td>724</td>
<td>458</td>
<td>0.7</td>
</tr>
<tr>
<td>Macrophyte cells</td>
<td>1250</td>
<td>561</td>
<td>1</td>
</tr>
</tbody>
</table>
Figure 7-2: Schematics of the monitored Ruffey's Creek Wetland.
Figure 7-3: Inlet sedimentation pond at the Ruffey's Creek Wetland (a) inlet pipe (1600 mmØ) (b) twin outlet culverts (750 mmØ).

Figure 7-4: Ephemeral Cell (a) inlet twin culverts (750 mmØ) (b) the low flow channel.

Figure 7-5: Macrophyte Cell 1 (a) a porous rock embankment at boundary between cells for distributing flow evenly (b) two riser outlets with glory holes.
7.3 Details of existing data set

As discussed above, a monitoring program for the system was carried out between the main inlet pipe and Macrophyte Cell 1 by CRCCH, and it recorded flow and water quality. The details of this monitoring study were published by Taylor et al. (2005) and Taylor (2006).

7.3.1 Monitoring setup and operation

Flow monitoring equipment and automatic water quality sampling stations were installed at the RCW to enable flow and pollutant concentrations to be measured. Three flow and four water quality monitoring stations were set up along the wetland. The locations and functions of sampling equipments are illustrated in Figure 7-2-a.

7.3.1.1 Flow monitoring

Inflow of the inlet pond

A Sigma 950 automated flowmeter was installed in the 1600mm main inlet pipe (Site 1, Figure 7-2) to measure water depth and velocity, which are recorded to a data-logger for subsequent inflow analysis. Data was logged every two minutes and calibrated using a rating curve based on height, velocity and cross-section measurements. Since the flow sensor was placed at approximately 65mm above the invert of the inlet pipe to avoid disturbance of the backwater from the inlet pond, the flow rate that potentially cannot be measured, was estimated to be approximately 30 L/s (Taylor, 2006).

The unmonitored inflow from the secondary 600mm inlet pipe that drains a small sub-catchment adjacent to the Bonview catchment was estimated to contribute an additional 8% flux to the inflow volume. Thus, the total inflow entering the inlet pond was estimated to be 1.08 times of the measured flux at the main inlet pipe (Taylor, 2006).

Overflow at the bypass spillway

A depth probe was installed (Figure 7-2-a) to monitor the water level through the bypass at a two-minute time step (Mindata logger 3500 series) and the flow rate was measured based
on a rating curve.

**Discharge of Macrophyte Cell 1**

Another depth probe (Greenspan) and Mindata 3500 series logger were installed at the outlet of Macrophyte Cell 1 as shown in Figure 7-2 (Site 4). The wetland discharge rates were calculated using a rating curve derived for the two riser outlets.

### 7.3.1.2 Water quality monitoring

**Sample collection**

Four Sigma 950 automatic water quality sampling stations (referring to as sites 1 - 4 in Figure 7-2-a) were established along the wetland, capable of collecting 24 discrete samples for a storm event at each station in order to record spatial and temporal changes in pollutant concentrations. Stormwater sampling was activated in a variety of ways. Sampling at inlet (Site 1) was triggered once a pre-determined flow rate (120L/s was chosen after several trials) was reached. Samplers at sites 2, 3 and 4 were controlled using float-switches set to the levels that ensured the auto-samplers to be triggered in a sequential order moving downstream. Once triggered, samples were taken at a regular time interval to fill the 24 bottles in each sampler. Samples were configured to be taken at specified frequencies based on flow-weighted time intervals during storm events, that is, the wet weather sampling frequency was matched as good as possible to the 'typical' flow hydrograph, with more intensive sampling during the first part of an event (Taylor, 2006).

For a series of regular (1 to 2 months) dry weather (inter-event), grab samples at four wet weather monitoring sites were taken as well to track an entire change of the water quality within the wetland.

**Chemical analyses of collected water samples**

Samples were analysed for nutrients via chemical analysis, as well as for nitrogen, Total Phosphorus (TP) and Total Suspended Solids (TSS), by the NATA accredited Water Studies Centre at Monash University, Melbourne (Taylor, 2006). Concentrations of TSS
were analysed using standard methods (APHA/AWWA/WPCF 1998) only for eight storm events at some sites since the CRCCH program was more focused on the analysis of changes in nutrients. Moreover, particle size distribution analyses were not undertaken in the CRCCH program.

7.3.2 Use of existing field data in this study

Twelve events were monitored at the RCW from March 2003 to October 2003, but the first three events were discarded due to incomplete hydrograph data. Thus, the data of nine storm events were recorded for both water quantity and quality, but TSS data were not recorded for Event 5. A summary for the details of recorded flow and TSS data for eight storm events are shown in Table 7-2, along with the total volumes of inflow and outflow, the total TSS load in inflow and outflow for each event.

In general, the following data sets measured were utilized in our modelling study:

1. Flow data monitored at three sites during storm events
   - inflow (total inflow of inlet sedimentation pond),
   - outflow (discharge from Macrophyte Cell 1), and
   - bypass overflow (discharge from the bypass spillway when it happened).

2. TSS data collected at four sites during storm events
   - inflow TSS (Site 1)
   - outflow TSS (Site 4), and
   - intermediate TSS (Site 2 and Site 3).

The flow rates measured for eight storm events, along with available corresponding TSS data were also presented as a set of hydrographs and pollutographs for each storm event in Appendix C1, where each curve corresponds to a monitoring site (e.g., Site 1 represents the inlet of the inlet pond, and Site 4 the outlet of Macrophyte cell 1, while Site 2 was the inlet of Ephemeral cell and Site 3 the outlet of Ephemeral cell).
Table 7-2: Summary of water quantity and TSS data collected from the RCW for eight storm events.

<table>
<thead>
<tr>
<th>Storm event No.</th>
<th>4</th>
<th>6</th>
<th>7</th>
<th>8</th>
<th>9</th>
<th>10</th>
<th>11</th>
<th>12</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sampling time</td>
<td>22/07/03</td>
<td>12-13/03</td>
<td>23-24/03</td>
<td>30/08/03</td>
<td>15-16/09/03</td>
<td>10/11/10/03</td>
<td>15-16/10/03</td>
<td>28-29/10/03</td>
</tr>
<tr>
<td>Inflow</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Max inflow (m³/s)</td>
<td>0.322</td>
<td>0.436</td>
<td>0.617</td>
<td>0.397</td>
<td>0.132</td>
<td>1.254</td>
<td>1.435</td>
<td>0.194</td>
</tr>
<tr>
<td>Total inflow volume (m³)</td>
<td>1735</td>
<td>3529</td>
<td>3528</td>
<td>1398</td>
<td>1034</td>
<td>4651</td>
<td>2214</td>
<td>1280</td>
</tr>
<tr>
<td>Max inflow TSS concentration (mg/L)</td>
<td>320</td>
<td>108</td>
<td>91</td>
<td>91</td>
<td>69</td>
<td>53</td>
<td>440</td>
<td>270</td>
</tr>
<tr>
<td>Min inflow TSS concentration (mg/L)</td>
<td>7.8</td>
<td>18</td>
<td>7.4</td>
<td>8</td>
<td>5</td>
<td>2.9</td>
<td>5.4</td>
<td>6</td>
</tr>
<tr>
<td>Total inflow TSS load (g)</td>
<td>64640</td>
<td>107671</td>
<td>100985</td>
<td>49899</td>
<td>17966</td>
<td>–</td>
<td>265790</td>
<td>45944</td>
</tr>
<tr>
<td>Overflow</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Max overflow (m³/s)</td>
<td>0.009</td>
<td>0.037</td>
<td>0.072</td>
<td>0.039</td>
<td>0.005</td>
<td>0.153</td>
<td>0.17</td>
<td>0.014</td>
</tr>
<tr>
<td>Total overflow volume (m³)</td>
<td>17</td>
<td>280</td>
<td>207</td>
<td>67</td>
<td>16</td>
<td>285</td>
<td>173</td>
<td>33</td>
</tr>
<tr>
<td>Max overflow TSS concentration (mg/L)</td>
<td>–</td>
<td>63</td>
<td>39</td>
<td>44</td>
<td>45</td>
<td>–</td>
<td>380</td>
<td>52</td>
</tr>
<tr>
<td>Min overflow TSS concentration (mg/L)</td>
<td>–</td>
<td>15</td>
<td>21</td>
<td>38</td>
<td>34</td>
<td>–</td>
<td>30</td>
<td>19</td>
</tr>
<tr>
<td>Total overflow TSS load (g)</td>
<td>–</td>
<td>6891</td>
<td>6806</td>
<td>161</td>
<td>182</td>
<td>–</td>
<td>21633</td>
<td>844</td>
</tr>
<tr>
<td>Outflow</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Max outflow (m³/s)</td>
<td>0.016</td>
<td>0.082</td>
<td>0.093</td>
<td>0.063</td>
<td>0.027</td>
<td>0.112</td>
<td>0.079</td>
<td>0.043</td>
</tr>
<tr>
<td>Total outflow volume (m³)</td>
<td>506</td>
<td>2138</td>
<td>2729</td>
<td>1144</td>
<td>319</td>
<td>3653</td>
<td>1616</td>
<td>755</td>
</tr>
<tr>
<td>Max outflow TSS concentration (mg/L)</td>
<td>24</td>
<td>32</td>
<td>34</td>
<td>31</td>
<td>19</td>
<td>75</td>
<td>79</td>
<td>73</td>
</tr>
<tr>
<td>Min outflow TSS concentration (mg/L)</td>
<td>16</td>
<td>12</td>
<td>13</td>
<td>14</td>
<td>13</td>
<td>10</td>
<td>15</td>
<td>14</td>
</tr>
<tr>
<td>Total outflow TSS load (g)</td>
<td>8165</td>
<td>41288</td>
<td>42080</td>
<td>18179</td>
<td>7784</td>
<td>–</td>
<td>64165</td>
<td>23352</td>
</tr>
<tr>
<td>TSS removal rate (%)</td>
<td>–</td>
<td>55</td>
<td>52</td>
<td>63</td>
<td>56</td>
<td>–</td>
<td>68</td>
<td>47</td>
</tr>
</tbody>
</table>
Field study

The observed inflow and outflow hydrographs showed that the wetland was effective for peak flow attenuation during storm events, and the bypass system for protecting the downstream vegetated cells from high flows could reduce re-suspension of sediment and could prevent scouring of epiphytes and deposited fine particles.

Monitoring of the sediment TSS concentrations through the wetland demonstrated that TSS concentrations mostly decreased with distance during storm events, especially comparing the inflow and outflow TSS concentrations, but the overall sediment removal rate was greatly dependent on the inflow rates. The varying trends of inflow and outflow flow rates and TSS concentrations can also be seen from figures presented in Table 7-2.

It should be noted that there are some inadequacies in the TSS data due to the difficulties in monitoring the stochastic nature of storm events. The inflow (Site 1) water quality sampling was triggered when reaching a pre-determined flow (120 L/s), and the baseflow was determined to be 15 L/s (Taylor, 2006). It was likely that the peak inflow TSS concentration was missed if it occurred before the start of sampling and was between 15 and 120 L/s. For example, Figure 7-6 shows the measured hydrographs and pollutographs for storm event 8. There was a small peak inflow (<120 L/s) before the first inflow sample was taken. Therefore, the preceding inflow TSS concentration may be higher than the measured peak TSS concentration (91 mg/L) due to loss of some important data. Furthermore, if the storm event lasts for a very long time and have a few peak inflows, for instance, like Event 10 (Figure 7-7), the limitation in the maximum number of samples taken in one storm event (especially when sampling intervals were more than 30 minutes) may possibly result in missing some peak inflow TSS concentrations from the rising limb or the peak of an storm event hydrograph. Therefore, sampling needs to be more frequent to catch all peak inflow concentrations in theory. However, this may be limited in practice due to both logistics (the autosamplers are limited to taking 24 samples) and cost.
3. Dry weather TSS data

Dry weather TSS data collected by grab sampling during inter-event from May 2003 to May 2004 (some data were collected from another CRCCH project in 2004) are given in Table 7-3.
Table 7-3: Observed dry weather TSS concentrations at four sites throughout the RCW.

<table>
<thead>
<tr>
<th>Sampling time</th>
<th>TSS concentration (mg/L)</th>
<th>Site</th>
<th>Mean TSS for all sites (mg/L)</th>
<th>Coefficient of variation (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>1</td>
<td>2</td>
<td>3</td>
</tr>
<tr>
<td>8/05/2003</td>
<td>8.8</td>
<td>12</td>
<td>10</td>
<td>28</td>
</tr>
<tr>
<td>3/07/2003</td>
<td>9.1</td>
<td>15</td>
<td>12</td>
<td>28</td>
</tr>
<tr>
<td>4/08/2003</td>
<td>7.1</td>
<td>20</td>
<td>20</td>
<td>14</td>
</tr>
<tr>
<td>10/10/2003</td>
<td>13</td>
<td>40</td>
<td>22</td>
<td>8.8</td>
</tr>
<tr>
<td>6/11/2003</td>
<td>8.8</td>
<td>22</td>
<td>15</td>
<td>22</td>
</tr>
<tr>
<td>19/03/2004</td>
<td>17</td>
<td>42</td>
<td>25</td>
<td>33</td>
</tr>
<tr>
<td>7/04/2004</td>
<td>17</td>
<td>30</td>
<td>21</td>
<td>14</td>
</tr>
<tr>
<td>5/05/2004</td>
<td>10</td>
<td>25</td>
<td>9</td>
<td>5</td>
</tr>
<tr>
<td>Long term mean TSS at each site (mg/L)</td>
<td>19</td>
<td>26</td>
<td>17</td>
<td>19</td>
</tr>
<tr>
<td>Long term mean TSS for all sites (mg/L)</td>
<td></td>
<td></td>
<td></td>
<td>20</td>
</tr>
</tbody>
</table>

The coefficients of variation for the mean dry weather TSS concentration at each site during the monitoring period and within the wetland at each sampling time are also presented in Table 7-3. The strong variations of these coefficients possibly indicate variability in dry weather TSS concentrations of one site at different times and at different sites for the same time. The long-term mean TSS concentration within the wetland during the whole sampling period was around 20 mg/L. The coefficient of variation of this value is about 19%, which can be regarded as being very low, since combined sampling and laboratory analysis errors of TSS are typically around 30% (Bertrand-Krajewski, 2004).

### 7.4 Field data collected within this study

#### 7.4.1 Methods

For application of the $N_f$ model, the particle size distribution (PSD) of sediment in inflow is a required input. For testing of this model, it was thus desirable to have PSD data at all points where TSS was measured, and most importantly for the outflow. Unfortunately, PSDs were not measured at the RCW in the CRCCH project. It it was therefore necessary to conduct an additional project on monitoring sediments in this wetland to determine
both typical dry and wet weather PSDs.

The monitoring was conducted in two sampling rounds (No 1 and No 2), and in each round, two types of samples were collected; during (1) storm events and (2) inter-event (dry) periods. Table 7-4 summarises the two sampling rounds. The times, places and numbers of samples taken are listed, together with the instruments used for PSD measurement. The details of each sampling round are given below.

<table>
<thead>
<tr>
<th>Table 7-4: Summary of sampling times and corresponding samples for two grab sampling rounds at the RCW.</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Sampling No.</strong></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td>1</td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
</tbody>
</table>

The first sampling round

The first sampling was carried out in May 2007 for wet weather conditions. Samples were taken manually every ten minutes at each site (Site 1 to 4, Figure 7-2) throughout the wetland. Five samples were taken at each site during the whole storm event. In addition, five dry weather samples were also collected at the same site during dry weather periods (six days after the storm event). In total 20 wet weather samples and 19 dry weather samples were collected, and analysed using a Mastersizer 2000 particle size analyser (which is the latest laser diffraction system from Malvern Instruments) at HRL Technology Pty Ltd. This type of Malvern instrument can measure particles from 0.02µm to 2000µm with ± 1% accuracy on the Dv50 using the Malvern Quality Audit Standard claimed by the manufacturer (Malvern Ins Ltd, 2005). It should be noted that the laboratory analysis was undertaken without the use of sonication or any other means of disaggregation of particles, except for gentle shaking.
The second sampling round

The second sampling was undertaken in October 2007. Wet weather samples were taken at Site 1, 2 and 4 every 15 minutes during a storm event. Four samples were collected at each site. Nine dry weather samples were also collected at these three sites just one day after the storm event under baseflow conditions.

To cross-check the results of the first round, the collected samples were sent to the School of Geography & Environmental Science at Monash University and were analysed using a laser diffraction particle size analyser, the Bechmann Coulter LS100, which measures the size distribution of particles in suspension in the range of 0.4µm – 1000µm. Again no disaggregation was performed prior to measurement.

7.4.2 Results and discussion

Table 7-5 summarizes all measured PSD data from the two samplings with $d_{10}$, $d_{50}$ and $d_{90}$, respectively.

Table 7-5: Summary of measured wet and dry weather PSD data for two sampling rounds at the RCW.

<table>
<thead>
<tr>
<th>Sampling No.</th>
<th>Weather condition</th>
<th>Wet weather</th>
<th>Dry weather</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>PSD ($\mu$m)</td>
<td>Site 1  2  3  4</td>
<td>Site 1  2  3  4</td>
</tr>
<tr>
<td>1</td>
<td>$d_{10}$</td>
<td>1.6  1.9  2.6  2.8</td>
<td>5.7  6.0  7.1  7.5</td>
</tr>
<tr>
<td></td>
<td>$d_{50}$</td>
<td>6.2  6.7  8.7  7.5</td>
<td>17.1 17.6 22.9 21.8</td>
</tr>
<tr>
<td></td>
<td>$d_{90}$</td>
<td>33.1 37.0 37.0 26.3</td>
<td>103.0 98.4 87.7 86.5</td>
</tr>
<tr>
<td>2</td>
<td>$d_{10}$</td>
<td>0.9  0.9  –  1.1</td>
<td>1.1  1.1  –  0.8</td>
</tr>
<tr>
<td></td>
<td>$d_{50}$</td>
<td>6.3  5.0  –  6.5</td>
<td>9.6  10.7  –  7.3</td>
</tr>
<tr>
<td></td>
<td>$d_{90}$</td>
<td>166.5 122.1  –  122.6</td>
<td>91.8 99.8  –  96.6</td>
</tr>
</tbody>
</table>

The first sampling round

The measured wet weather PSD curves at four sites for the first-round sampling are presented in Figure 7-8. There were variations in PSD curves with time at Site 1, 2 and 3, but the PSD curves seemed to be time independent at the outlet (Site 4). This may indicate that the particle size distributions in the discharge of the wetland were relatively
steady after treatment since the larger particles were trapped in Macrophyte Cell 1. On the other hand, it is hard to find a trend of PSD over the length along this wetland. The measured particle sizes at downstream sampling points sometimes were larger than the upstream points in contrast to the expected trend of decreasing median size over distance. This probably is the result of re-suspension of sediments, due to turbulence caused by wind and birdlife (some birds and ducks were observed in the wetland). This can occur particularly in the inlet pond, since the high-flow bypass is located at the downstream of this zone, leaving it subject to uncontrolled turbulent flow.

Figure 7-9 illustrates the dry weather PSD curves measured at four sites in the RCW for the first-round sampling. Figure 7-10 presents the comparisons of the mean wet weather PSD curves at each sampling site with the mean dry weather one measured by the Mastersizer 2000. Again some unexpected results were found, e.g., the dry weather particle sizes were larger than the wet weather ones in the wetland. This observation is difficult to explain. Maybe this could have happen due to re-suspension caused by wind-driven turbulence, the activity of birds and animals (such as the many birds and ducks that were observed during monitoring period in the wetland), thermal effects, and even gas release from the bed of the wetland during a long dry period. The effect of bird life is the most likely cause, particularly given that the density of birdlife was observed to be much greater during dry weather periods than during stormflows.

The second sampling round

Figures 7-11 and 7-12 show the results of the measurement for wet weather PSD and dry weather PSD in the second-round sampling, respectively. The comparisons of the mean wet weather PSD at each sampling site and the mean dry weather one measured by the Bechmann Coulter LS100 are illustrated in Figure 7-13. This time the PSD along the wetland did not change a lot for both wet and dry events (Table 7-5 clearly shows no changes). However, similar to the first round, the dry weather particle size on average was bigger than the wet weather one, again showing the likely effect of physical disturbance and re-suspension caused by birdlife.
Figure 7-8: The measured cumulative wet weather PSD curves from four monitoring sites (Site 1 - 4, Figure 7-2) in the first wet weather sampling at the RCW (five samples were taken at each site during the storm period, where the first number “1” in the sample ID stands for the first sampling round, W for wet weather sample, S for site, the number after “S” for the number of sampling site, the last number after hyphen for the order (i.e. time) of the sample collected).

Figure 7-9: The measured cumulative dry weather PSD curves from four sites (Site 1 – 4) in the first dry weather sampling at the RCW (five samples were taken at each site during storm period, where the first number “1” stands for the first sampling round, D for dry weather sample).
Figure 7-10: The comparison of the mean wet weather PSD and the mean dry weather PSD at each sampling site measured by Mastersizer 2000.

Figure 7-11: The measured cumulative wet weather PSD curves from three sites (Site 1, 2 and 4) in the second wet weather sampling at the RCW (four samples were taken at each sampling site during the storm-event period, where the first number “2” in the sample ID stands for the second sampling round, W for wet weather sample, S for site, the number after “S” for the number of sampling site, the last number after hyphen for the order of the sample collected at the same site).
Figure 7-12: The measured cumulative dry weather PSD curves from three sites (Site 1, 2 and 4) in the second dry weather sampling at the RCW (three samples were taken at each site during inter-event period, where the first number “2” stands for the second sampling round, D for dry weather sample).

Figure 7-13: The comparisons of the mean wet weather PSD of each sampling site with the mean dry weather PSD of each site measured by the Coulter LS100.

Comparisons of the measured PSDs from two sampling rounds

Figure 7-14 illustrates all PSD curves measured at the inlet of the wetland (Site 1) during
Field study

the wet weather period for the two sampling rounds. The median particle sizes ($d_{50}$) are below 10 µm (ranged from 4.2 to 9.9 µm), while 10% and 90% of the particles are below 2.5 and 195 µm, respectively. This may be due to the fact that most runoff flows through a grass filter in this sub-catchment before being drained to the RCW. As a consequence, the large particles are probably intercepted by the grass.

Figure 7-15 presents the comparison of the mean dry weather PSD curves at each sampling site for two dry weather sampling rounds. The median particle sizes are below 28.7 µm, while the sizes of 10% and 90% of all particles are below 7.4 and 99.8 µm, respectively. The PSD curves from the first-round sampling are above those from the second-round. It should be noted that the first-round dry weather sampling was undertaken six days after the storm event, while the second-round was carried out just one day after the antecedent storm event. Since the second-round sampling was carried out in a much shorter dry period (one day after the storm as compared to one for the first-round) and the water level in the wetland was higher, re-suspension of sediments of a large size was not as significant as the first-round.

![Ruffey's Creek Wetland Inlet PSD (Site 1)](image)

Figure 7-14: The comparison of wet weather PSD curves at the inlet (Site 1) from two wet weather sampling rounds.
Figure 7-15: The comparisons of dry weather PSD curves from two dry weather sampling rounds (each site PSD curve was represented by the averaged PSD curve measured from that site during each dry weather sampling period).

7.5 Conclusions

The primary aim of this Chapter was to gather field data to facilitate the assessment of the $k-C^*$ model and the $N_r$ model outlined in Chapters 5 and 6 respectively. An intensive monitoring program at the RCW carried out by CRCCH provided time series data for flow rates and TSS, but the required PSD in stormwater for both wet and dry weather cases had to be obtained with newly collected samples from the wetland in two storm events and two dry periods following those storm events.

Based on the data and statistical analysis of flow rates and TSS, it showed that the wetland was effective for peak flow attenuation during storm events, and the bypass system for protecting the downstream vegetated cells from high flows could prevent re-suspension of sediments, scouring of epiphytes and deposited fine particles. Furthermore, TSS concentrations mostly decreased over distance during storm events, especially comparing the inflow and outflow TSS concentrations. The overall sediment removal rate for the system was greatly dependent on the inflow rates. On the other hand, the measured dry
Field study

weather mean particle sizes were higher than those under wet weather conditions in the wetland. This implies that sediment re-suspension can be caused primarily by environmental factors such as wind and birdlife in the shallower water during dry weather periods, rather than by flow processes.

More importantly, long and contiguous series of data with high temporal resolution (i.e. sampling at shorter time intervals) are necessary to generate continuous pollutographs with acceptable accuracy, for testing of the proposed models. The recorded flow rates and TSS concentrations along the wetland at every time step satisfy the data requirements for model calibration.
Chapter 8

Testing of the $k-C^*$ Model Using the Ruffey’s Creek Wetland Field Data

8.1 Introduction

One of the most comprehensive stormwater modelling software packages used in Water Sensitive Urban Design in Australia is MUSIC (the Model for Urban Stormwater Improvement Conceptualisation), developed by the Cooperative Research Centre for Catchment Hydrology, as part of their Catchment Modelling Toolkit. It was developed in a somewhat modular form, allowing the latest technologies and research findings to be incorporated readily. A pilot version of MUSIC was released in March 2001 and the latest Version (3.01) was released in May 2005.

The first-order kinetic decay ($k-C^*$) model for stormwater treatment has been adopted in MUSIC, combined with the continuously stirred tank reactors (CSTRs) model to account for both pollutant treatment and flow hydrodynamics simultaneously, in a range of stormwater treatment systems. The main aim of this chapter is to apply the MUSIC model to the field data collected at the Ruffey’s Creek Wetland (RCW) described in Chapter 7, and therefore to test its reliability.

Although the $k-C^*$ model is widely used in Australia, very few calibration studies have been undertaken (e.g., the work by Wong et al. (2006) is a rare paper that shows how the model performs against measured data). Further to this there are no studies done on the model’s sensitivity to the key calibration parameters. Therefore this application study includes two tasks: (1) to calibrate $k$ and $C^*$ for a constructed stormwater wetland using field data, and (2) to provide more insight into sensitivity of the model to the $k$ and $C^*$
parameters. The results from this work will be used, in Chapter 10, to compare the two models studied in this thesis, the $k-C^*$ model and the new proposed $N_t$ model.

### 8.2 The MUSIC software

This section first provides an overview of the MUSIC software, followed by the model structure, input data requirements and model parameters. In the following section, the modelling procedure used in assessing wetland performance for TSS is described in details, and other components of this complex software are also briefly mentioned.

MUSIC is an urban stormwater quantity and quality model that was designed to operate at a range of temporal and spatial scales; e.g., catchment areas from 0.01 km$^2$ to 100 km$^2$ and modelling time steps ranging from 6 minutes to 24 hours with regard to match the catchment scale. MUSIC can simulate the following pollutant-associated processes on a single event or continuous basis: (1) generation of catchment runoff and pollutants; (2) transport of stormwater and pollutants; and (3) storage and treatment of stormwater pollutants (including routing of both flows and pollutants, through drainage network and through treatment systems). It should be noted that MUSIC is a conceptual design tool rather than a detailed design tool. It enables users to evaluate and prioritize conceptual designs and stormwater treatment strategies based on the prediction of hydrology and water quality, but it does not contain the algorithms necessary for detailed design of stormwater quantity/quality facilities (CRCCH, 2005).

In MUSIC, a range of stormwater treatment measures such as grass swales, wetlands, ponds, sedimentation basins and infiltration systems, are modelled using a unified model (including the Puls hydrologic routing method, the $k-C^*$ pollutant decay model and the CSTRs hydrodynamics model, all described in Chapter 2). It is hypothesised that the treatment systems can be considered as a single continuum, described by the following two fundamental processes as stormwater passes through these treatment measures: (1) flow attenuation and detention, and (2) the sum of all processes involved in pollutant removal (this includes a lumped conceptual approach to simulate complex physical, biological and
chemical pollution removal processes). Although the appearance and layout of these treatment systems is different, the basic physical processes occurring in each of these treatment measures are hypothesised by MUSIC to be the same. Therefore, the unified model is used in MUSIC to describe these physical processes, as well as the overall pollutant removal behaviour (chemical and biological processes also contribute to pollutant removal) (Wong et al., 2006). The unified model is referred to as the **Universal Stormwater Treatment Model (USTM)** in MUSIC.

### 8.2.1 Model structure

MUSIC is a standard “node and link” model. A network of nodes represents catchments or sub-catchment, stormwater treatment measures and junctions or receiving waters. The nodes are connected by links representing the drainage pathways (which may or may not have flow and pollutant routing, dependent on the user). MUSIC provides a user interface to allow complex stormwater management scenarios to be quickly and efficiently created and the results to be viewed using a range of graphical and tabular formats. Figure 8-1 shows an example of the window interface of MUSIC simulating the performance of a group of stormwater management measures, which are configured in series or in parallel to form a “treatment train”.

![Figure 8-1: MUSIC user interface.](image)

The basic MUSIC model structure for stormwater treatment is illustrated in Figure 8-2 and
Application of MUSIC using the field data is explained in details in the following.

Figure 8-2: MUSIC model structure (fundamental components include Source Node, Treatment Node, Receiving Node and Drainage Links).

1. Source Nodes

Source Nodes characterize sub-catchment areas. There are several types of source nodes including urban, forest and agricultural nodes, as shown in Figure 8-3, with the only difference being their default pollutant export characteristics. A rainfall-runoff model based on the SimHyd model developed by Chiew et al. (1997) is adopted in MUSIC to estimate streamflow from rainfall based on catchment characteristics (such as impervious area and soil moisture storage). Streamflow quantities and qualities are normally determined by creating source nodes after selecting an appropriate climate condition (rainfall and evapotranspiration) under which to run the model. Stochastic pollutant concentrations of the streamflow are generated from user-supplied or default statistical distributions for the specified catchment region.

Furthermore, MUSIC also provides an Imported Data Node to allow users to import time-series data (e.g. flow, TSS, TP, TN and gross pollutants) for creating customer-made source nodes, or treatment nodes.
2. Treatment Nodes

Stormwater treatment systems are simulated using Treatment Nodes in MUSIC. A range of treatment measures are available in MUSIC, such as wetlands, ponds, sedimentation basins, infiltration systems, gross pollutant traps, buffer strips, bioretention systems, vegetated swales, rainwater tanks and finally a generic treatment node which allows users to define “transfer functions” for flows and water quality for those stormwater quality treatment measures that are not explicitly modelled in MUSIC, as shown in Figure 8-4.
(1) **Flow routing** to simulate the movement of water through the treatment system, which is modelled based on the *Puls Method* for reservoir routing, as described in Section 2.4.2. Basically, the method calculates the outflow hydrograph from assuming a horizontal water surface, at given inflow hydrograph and storage-discharge (S-Q) relationship. The S-Q relationship at an outlet is a key input to this hydrologic routing routine. In MUSIC, from a practical point of view, the S-Q curve is derived using the outlet structure, i.e., a riser outlet treated as an orifice of equivalent diameter or a weir outlet which is specified by the users. Therefore, the notional equivalent pipe diameter may not be the diameter of the pipe itself, but the equivalent diameter of its orifices (CRCCH, 2005). In addition, the equivalent pipe diameter and the orifice discharge coefficient need to be calibrated using observed outflow data.

(2) **Pollutant routing** to simulate sediment transport within the treatment system using the *continuously stirred tank reactor (CSTR) model* as discussed in Section 2.5.2. The degree to which flow hydrodynamic conditions in a treatment system can be modelled relies on appropriate selection of the number of CSTRs, based on the shape, bathymetry, inlet and outlet configurations, and vegetation characteristics of the system (Wong, *et al.*, 2006). The number of CSTRs (N) can be determined according to the hydraulic efficiency λ of the treatment facility. A range of typical treatment system layouts and their relevant hydraulic efficiencies λ, effective volume ratio $e_v$, and equivalent number of CSTRs $N$ are illustrated in Figure 2-8.

(3) **Pollutant removal** to simulate water quality behaviour using the *first-order decay model – k-C* model* in the form of Equation 2.10 as discussed in Section 2.6.2. The parameters $k$ and $C*$ are lumped parameters representing the combined effects of a number of pollutant removal mechanisms. Thus, the $k-C*$ model provides a lumped method to describe the overall pollutant movement towards an
equilibrium. It should also be noted that the $k-C^*$ model adopted in the unified approach (USTM) is currently strictly applicable only during storm event operation (CRCCH, 2005).

3. Drainage Link

The links connecting sources, treatment and junction nodes may represent pipes, open channels or natural watercourses, as shown in Figure 8-2. The passage of stormwater and pollutants through a link can be specified by users as having no routing, translation only, or full Muskingum-Cunge routing, as discussed in Section 2.4.2.

8.2.2 Input data requirements

1. Climate data

MUSIC simulations are based on a “meteorological template” which can be of any duration, with a time step ranging from six minutes to 24 hours. Climate templates can also be created from local rainfall and evapotranspiration files. MUSIC has pre-loaded rainfall and evapotranspiration files for a range of Australian locations.

2. Source Node properties

In creating a Source Node, the following data need to be specified by the users:

- catchment area and impervious area,
- pervious area rainfall-runoff properties (default values are provided for Australian capital cities, but users are expected to supply values based on local calibration), and
- event mean and dry weather pollutant concentrations (default values are provided, but the User Manual encourages the collection of local calibration data).

Alternatively, node data can be imported by a file of flow and pollutant concentration data directly measured at a site (i.e. using an Imported Data Node). This can bypass the entire Source Node simulations. This method is used in Section 8.3.1 for MUSIC application.
3. Treatment Node properties

The design properties of a given treatment node such as the inlet, storage and outlet features should be specified by the users. Advanced parameters can also be accessed by users to modify the default modelling parameters such as the $k$ and $C^*$ values in the USTM.

4. Drainage Link properties

The routing properties (used for the Muskingum-Cunge method) of each link can be specified by the users in order to enable more accurate simulations of the hydraulic behaviour of the drainage network.

8.2.3 Model parameters

The USTM approach describes the flow hydrodynamics in treatment facilities in terms of a series of CSTRs, where the $k$-$C^*$ model is applied through each of the CSTRs in turn, to express the behaviour of the pollutant removal simultaneously. The success of the USTM in various stormwater treatment systems can be achieved by changing the following four main inputs:

- the storage-discharge ($S$-$Q$) relationship (which in MUSIC is actually determined by the notional equivalent pipe diameter and the orifice discharge coefficient for wetlands),
- the number of CSTR cells $N$,
- the background concentration $C^*$, and
- the exponential decay coefficient $k$.

These principal parameters in MUSIC are required to be calibrated using local or nearby data whenever possible to achieve the best model performance. MUSiC defaults to five CSTR cells for constructed wetlands ($N = 5$), which presents the number of notionally well-mixed storage cells in the presence of flow short-circuiting and turbulence.

The $k$-$C^*$ model can be applied to a wide range of pollutants in a variety of stormwater treatment systems, but appropriate $k$ and $C^*$ values need to be selected to match the
characteristics of each type of treatment facility (Wong et al., 2006). Table 8-1 presents the summary of theoretical and recommended \( k \) and \( C^* \) values provided in the MUSIC Version 3 User Manual. The default values in MUSIC are selected from these recommended ranges.

Table 8-1: Theoretical and recommended \( k \) and \( C^* \) values for MUSIC V 3.0 (\( k - \text{m/y}, C^* - \text{mg/L} \)) (Source: CRCCH, 2005).

<table>
<thead>
<tr>
<th>Treatment measure</th>
<th>TSS</th>
<th>TP</th>
<th>TN</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>( k )</td>
<td>( C^* )</td>
<td>( k )</td>
</tr>
<tr>
<td>Wetland</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Theoretical</td>
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<td>6</td>
<td>2800</td>
</tr>
<tr>
<td>Recommended</td>
<td>500-5000</td>
<td>5-6</td>
<td>300-2800</td>
</tr>
<tr>
<td>Pond</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Theoretical</td>
<td>1000</td>
<td>12</td>
<td>500</td>
</tr>
<tr>
<td>Recommended</td>
<td>200-1000</td>
<td>12-15</td>
<td>150-500</td>
</tr>
<tr>
<td>Infiltration system</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Theoretical</td>
<td>1000</td>
<td>12</td>
<td>500</td>
</tr>
<tr>
<td>Recommended</td>
<td>200-1000</td>
<td>12-15</td>
<td>150-500</td>
</tr>
<tr>
<td>Sedimentation basin</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Theoretical</td>
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<td>12000</td>
</tr>
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<td>Recommended</td>
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<td>3000-12000</td>
</tr>
<tr>
<td>Swale</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Theoretical</td>
<td>15000</td>
<td>30</td>
<td>12000</td>
</tr>
<tr>
<td>Recommended</td>
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<td>10-30</td>
<td>3000-12000</td>
</tr>
<tr>
<td>Bioretention system</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Theoretical</td>
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<td>12000</td>
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<tr>
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<td>3000-12000</td>
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<tr>
<td>Recommended</td>
<td>200-1000</td>
<td>12-15</td>
<td>150-500</td>
</tr>
</tbody>
</table>

8.2.4 Modelling procedure for wetlands

In MUSIC, a wetland is abstractly simplified as a conceptual model which can be treated by the USTM. A typical conceptual diagram for the wetland properties in MUSIC is presented in Figure 8-5. This wetland system consists of two cells: an inlet pond and the main wetland or macrophyte cell. The macrophyte cell has a permanent volume of water with a low-level outlet pipe, above which there is an extended detention volume. The system is modelled on the assumption of a riser outlet, with the permanent pool volume reflecting the invert of the lowest orifice. Stormwater can bypass the whole system when it exceeds or falls below the predefined flow rates. An overflow weir has an elevation equal to the extended detention depth above the standing water level of the permanent pool, as shown in Figure 8-5.
The initial wetland properties used in the dialogue box of MUSIC for describing the basic characteristics of a wetland system are provided in Fig. 8-6.

1. Inlet properties

The inlet properties define the physical characteristics of the inlet pond. MUSIC assumes that the macrophyte cell is preceded by a separate inlet pond (users can bypass this assumption by using the Pond Node rather than the Wetland Node, but modifying the other parameters to match that of a wetland). The inlet pond in wetland systems provides a pre-treatment of the stormwater, which includes sedimentation of coarse- to medium-sized particles and hydrologic control by facilitating a bypass of flow around the macrophyte zone when the water level in the macrophyte zone has reached its maximum extended detention depth. MUSIC models these pre-treatment functions using the inlet pond volume, which is assumed to be a permanent pool.
Flow is hydrologically routed through the wetland using the Puls method, according to the characteristics defined by the users. When the stormwater inflow rate exceeds the user-defined *high flow-bypass rate* all of the flow volume in excess will bypass the wetland and will not be treated by the wetland. On the other hand, all of the stormwater that enters the wetland below the user-defined *low flow-bypass rate* will also bypass the wetland. Thus, only the flow above the low flow-bypass rate and below the high flow-bypass rate will enter into and be treated by the wetland. Bypassing of the macrophyte cell will also occur, as described previously, when the water level is such that the macrophyte cell is full. Inflows will then be discharged over the overflow weir, which is assumed to be located at the downstream end of the inlet zone.

2. Storage properties

The storage properties define the physical characteristics of the macrophyte cell of the
Surface area

The hydrologic routing analysis calculates the volume of water in storage during a storm event depending on the depth of water in the cell and the surface area.

Extended Detention Depth

The extended detention depth is the depth below the top water level and above the permanent pool level. Detention time of the extended detention volume will be determined by its volume and the outlet properties (i.e., the equivalent diameter of the outlet orifices). Water volume in excess of the extended detention depth will be discharged from the wetland via the overflow weir.

Permanent pool volume

The wetland system has a permanent volume of water, which does not affect the hydrologic routing of storm water through the system, but does affect the hydraulic retention time during a storm event. Therefore, it has an effect on the treatment of pollutants.

Vegetation cover

This parameter is disabled in Version 3.0. In subsequent versions, it will impact on treatment performance.

Seepage

Seepage from the permanent pool into the underlying soil can be modelled by defining the seepage rate. The water that seeps from the permanent pool of the wetland is assumed to be lost from the catchment and does not re-enter the system downstream.

Evaporative Loss

Evaporation from the permanent pool can be modelled by defining the evaporative loss rate, as a percentage of the daily potential evapotranspiration data contained in the Meteorological Template.

3. Outlet properties
The outlet properties define the physical characteristics of the outlet pipe and weir.

**Equivalent Pipe Diameter & orifice discharge coefficient**

MUSIC assumes that water from the wetland is discharged through a simple outlet whose equivalent pipe diameter is related to corresponding estimated outflow rates. Hence, the equivalent pipe diameter may not be the diameter of the pipe itself, but the equivalent diameter of its orifices.

**Notional detention time**

MUSIC automatically calculates and displays the notional detention time with changing the equivalent pipe diameter.

**Overflow weir width**

Overflow weir width of the outlet usually has little effect on performance as large flows rarely happen. The overflow weir (assumed to be located at the downstream end of the inlet zone) will start to discharge water once the depth in the pond reaches the extended detention depth defined above.

4. **Advanced wetland properties**

The advanced properties define the parameters that describe the hydraulic characteristics for the outflow structure (e.g. orifice discharge coefficient and weir coefficient), and the parameters that describe the treatment process in the wetland such as the number of CSTR cells, the values of $k$ and $C^*$, etc.

**8.3 Methods**

The methods used for modelling the Ruffey's Creek Wetland (RCW) in MUSIC are explained in this section. Firstly, the MUSIC model was set up to represent this wetland, using the defined storage-outflow ($S - Q$) curve based on the observed inflow and outflow data. The key model parameters, that is, the parameters $k$ and $C^*$ and the number of CSTRs $N$, were then calibrated using all available data. Finally, sensitivity analyses of the key model parameters ($k$ and $C^*$) was carried out.
8.3.1 Model set-up for the Ruffey’s Creek Wetland

Flow and TSS data of the RCW were recorded for eight storm events as discussed in Chapter 7, but the data for inter-event periods were unavailable. Thus, the MUSIC model had to be set up individually for each monitoring storm event rather than continuously simulating all events at the same time. In other words, eight event-based MUSIC models were set up to calibrate the performance of the model in this study. An Imported Data Node was selected for each model instead of using general source nodes, so that the inflow and inflow TSS data monitored at the RCW could be imported directly as the downstream treatment node input. A Wetland Treatment Node was also selected and linked with the Imported Data Node for simulating the performance of the RCW during each storm event. Figure 8-7 shows an example of the MUSIC model interface for Event 4 (explained in Chapter 7). The methodology used for the detailed MUSIC model set-up is explained in the following.

![Figure 8-7: MUSIC model for Event 4 at the Ruffey’s Creek Wetland.](image)

The first task was to prepare observed input data (i.e. inflow rates and TSS of inflows received by the RCW) and to import them into the Imported Data Node for each event-based model. After that, the RCW system had to be conceptualized and the required data for wetland properties were then imported into the Wetland Treatment Node.

It should be noted that a six-minute time step was chosen for modelling the performance of
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the RCW due to the intermittent nature of the stormwater flows in wetlands, and the relatively small size of the wetland being modelled. Since the time step should be comparable to the interval in which processes are occurring or changing in the treatment system, the shortest time step (six minutes) allowed in MUSIC was thus used for this study.

1. Flow and TSS input data

The available flow data for the RCW were inflow rates (measured at Site 1), outflow rates (measured at Site 4, Figure 7-2) and overflow rates if occurring, together with water quality data at four sites throughout the wetland. In this application, the time-series inflow rate and inflow TSS data monitored were imported directly into the Imported Data Node.

The two-minute inflow reading data (e.g., collected at Site 1, Figure 7-2) were aggregated into those at six-minute time step by combining or interpolating monitored data if necessary. Inflow TSS concentrations were assumed to change linearly from one measured point to the next within an event, to produce a continuous pollutograph, and then were averaged over each six-minute time step in the same way to ensure the data format to match the form specified by MUSIC. In this manner, inflow rates and TSS concentrations from the storm events 4, 6, 7, 8, 9, 10, 11 and 12 reported in Chapter 7 were imported into the Imported Data Nodes for each MUSIC model.

2. Wetland input properties data

The RCW has a typical conceptual layout in line with the wetland conceptual model in MUSIC (Figure 8-5), which comprises two cells: the inlet pond and the macrophyte cell (the ephemeral cell can be treated as a part of the macrophyte cell, see Figure 7-2 for more details). How this wetland is represented in MUSIC by two cells is shown in Figure 8-8.

The information on wetland characteristics as conceptualized above was fed into MUSIC through the Wetland Treatment Node in the following way.

*Inlet properties*: The value of the low flow by-pass rate is given by the default value of 0 m³/s, since low flow cannot bypass the wetland.
As described in Chapter 7, stormwater bypasses the RCW only via the overflow weir located at the downstream end of the inlet pond when either (1) flows exceed the discharge capacity of the two culverts connecting the inlet pond to the macrophyte cells, or (2) the macrophyte cell reaches its extended detention water level. The invert of the overflow weir is set to the same level as the extended detention water level. Thus, the occurrence of high flow bypass is actually influenced by the water level in the inlet pond rather than by one specified inflow rate for all events. As shown in Appendix C1, the inflow rates at which overflow started varied widely over the eight storm events. The high flow by-pass rate for each storm event was determined in the following way in order to describe more accurately the occurrence of high flow bypass for each event: (1) the volume of measured overflow during an overflow period was first calculated (some events may have a few separate overflow periods, see Appendix C1); (2) the observed inflow hydrograph was cut from the top within the same overflow period by the same volume as the calculated overflow volume to find out the equivalent peak inflow rate; and then (3) the equivalent inflow rate with the maximum overflow volume was set as the required high flow-bypass rate for this event in the MUSIC wetland model.
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There are two different ways to model an inlet pond of a wetland in MUSIC: (1) the inlet pond may be modelled separately from the main wetland as an individual sedimentation basin, but is only operated under inlet control and the water in it freely drains to the wetland; (2) the inlet pond is modelled integrally within the main wetland (i.e. the macrophyte zone) as it operates under outlet control (being influenced by the backwater of the macrophyte zone). In fact, flow conditions at the RCW are very complex and vary with time. At the beginning of a storm event, the wetland is operated under inlet control since the water level is low. As water level increases with time, the macrophyte cell is full and backflow occurs during most of the storm event period in the wetland, while the water in the inlet pond cannot freely drain to the downstream macrophyte zone. Furthermore, it should be acknowledged that the discharge rates of the two culverts (i.e. the inflow of the macrophyte zone) during storm events are very difficult to be determined accurately. It was thus decided to model the inlet pond integrally within the wetland. The inlet pond volume was found to be 458 m$^3$ based on the physical characteristics of the wetland (Table 7-1).

It should also be noted that the hydraulics of the Wetland Treatment Node in MUSIC is based entirely on the storage and outlet properties specified in the wetland dialogue box (Figure 8-6), but not on the inlet properties. The inlet pond provides no flow routing but does offer both mixing and $k\cdot C^*$ treatment for pollutants. Thus, the inlet pond modelled integrally within the wetland in the MUSIC simulation is essentially equivalent to one CSTR cell.

Storage properties: In order to estimate more accurately the actual surface areas for different storm events, the mean wetted surface areas for each storm event were derived from the outcomes of previous HEC-RAS modelling simulations, undertaken by Taylor (2006) as part of the storm event monitoring campaign. The mean wetted surface areas used for input data in MUSIC for eight storm events are shown in Table 8-2.

According to the site construction of the RCW, the normally operating water depth within the macrophyte zone is one metre. Thus, the extended detention depth of the storage was assumed to be 1 metre. A permanent pool volume of 1250 m$^3$ was used, based on the
Application of MUSIC using the field data

geometry of the wetland cells. It was assumed that there was no seepage from the wetland (it is a compacted clay-lined system), but there was assumed evaporation from the wetland, equal to the default of 125% of PET defined for the catchment (as is recommended in the MUSIC User Manual; CRCCH, 2005).

<table>
<thead>
<tr>
<th>Storm event number</th>
<th>4</th>
<th>6</th>
<th>7</th>
<th>8</th>
<th>9</th>
<th>10</th>
<th>11</th>
<th>12</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean wet weather surface area (m$^2$)</td>
<td>1324</td>
<td>1941</td>
<td>1928</td>
<td>1839</td>
<td>1419</td>
<td>1825</td>
<td>2116</td>
<td>1535</td>
</tr>
</tbody>
</table>

Outlet properties: The modelled outflow rate must match well with the measured outflow at the wetland. Thus, it was necessary to calibrate the equivalent pipe diameter and the orifice discharge coefficient to obtain the best fit with the observed outflow hydrograph.

The width of the Overflow weir was 10 metres, which allowed weir flow to occur freely for strong storm events.

8.3.2 Calibration procedure

Calibration of the MUSIC model using a real stormwater treatment data set has to be carried out using an iterative approach, but unfortunately there are no parameter optimization tools built into the software package (neither can the software be easily linked with standard optimization tools). Calibration was therefore carried out by manually adjusting the five model parameters (i.e. the notional equivalent pipe diameter and the orifice discharge coefficient for the outlet of the wetland, the number of CSTRs, the background concentration and the exponential decay rate) to maximize the model efficiency coefficient $E$ for the TSS concentrations at the wetland’s outlet (Site 4, Figure 7-2). Owing to the complexity of hydraulics inside the wetland during the storm events, the data measured at the intermediate points (Sites 2 and 3) were not used in the calibration.

The calibration methodology for this study was composed of the following two tasks, as
1. Calibration of flow model parameters

The aim is to optimize the notional equivalent pipe diameter and the orifice discharge coefficient in order to define storage – discharge \((S - Q)\) relationship for the wetland. The MUSIC model was run for each of eight events individually to obtain the optimum values of these two parameters. This was achieved by changing the values of equivalent pipe diameter and weir discharge coefficient in the Advanced Wetland Properties to match the observed outflow data, and the objective function \(E\) was calculated for each tested parameter set. The weighted average equivalent pipe diameter was then calculated by the mean of all events, weighted by the model efficiency \(E\) for each event (if \(E < 0\), assuming to be 0). This averaged diameter was then used as the equivalent pipe diameter for all events in other parameters calibrations. A similar method was used for the calibration of the orifice discharge coefficient.

It should be noted that MUSIC’s Observed Data facility only receives observed data at any point displayed in time series graphs in order to be tested against the modelled data in a visual or graphical manner. However, the observed data in MUSIC cannot be used directly for statistical analysis. Thus, statistical analyses were done outside MUSIC (which was a very time demanding job), for instance, the modelled flows at outlet of the wetland were exported from MUSIC flux files into Excel spreadsheets so that they were compared with the observed outflows to calculate the objective function \(E\).

2. Calibrations of TSS model parameters

The other three principal parameters that have to be calibrated are the number of CSTRs \((N)\), the background concentration \((C^*)\), and the corresponding exponential decay parameter \((k)\).

Calibration was done by maximising \(E\) value between measured and modelled outflow TSS concentrations (Site 4). For a predefined value of \(N\), which ranges from 1 to 5, the MUSIC model was run for each event in the selected parameter space for the parameters \(k\).
and $C^*$ outlined below:

$C^*$: for all tested $N$ values, this parameter was varied between 1 mg/L to 30 mg/L at increments of 1 or 2 mg/L; and

$k$: for $N = 1$, from 100 m/y to 7000 m/y at an increment of 100 m/y, and from 7000 m/y to 15000 m/y at an increment of 1000 m/y (1794 parameter sets are used in this case); for $N = 2$, from 200 m/y to 4000 m/y at an increment of 200 m/y and from 4000 m/y to 10000 m/y at an increment of 1000 m/y (350 sets are used); and for $N = 3, 4,$ and $5$, from 500 m/y to 10000 m/y at an increment 200 m/y or 1000 m/y (308 sets are used).

In total the model was run for 3068 combinations of the three parameters, and for each combination the model coefficient $E$ was calculated. The model performance was therefore fully evaluated in a rather large parameter space (again this was very labour-intensive task since all the calculations of $E$ had to be done outside MUSIC software, and each run had to be individually undertaken, given MUSIC's inability to receive batch commands).

### 8.3.3 Parameter sensitivity

Sensitivity analysis was done only for $k$ and $C^*$, which are regarded as the key parameters of the treatment model in MUSIC. The analysis was undertaken only for TSS, since the aim of this thesis is to model sediment behaviour. This analysis was based on the variations of $E$ value within the tested parameter space. As explained above, MUSIC was run for a large number of parameter sets, and $E$ values were calculated using the measured and modelled outflow TSS data at the wetland outlet. Using these data, the contours of $E$ were constructed for various sets of $k$ and $C^*$, evaluating their response surfaces (i.e. the goodness of fit).

Local sensitivity analysis was carried out for the model parameter sets with the value of objective function $E$ greater than -0.5. The change in $E$ was presented as a function of each of the two parameters ($k$ and $C^*$) for different values of $N$, and in this way, dotty plots were
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generated to evaluate the importance of each parameter.

Furthermore, the same methodology described in Section 5.3.1 was used here for the frequency analysis. For the best value of $N$ (that was $N = 1$, see below), the frequency for both $k$ and $C^*$ were estimated using the parameter sets, which produce a value of $E$ greater than 0.40, since the maximum $E$ was 0.47. The frequency plots can tell us whether or not the model is sensitive to a particular parameter; if the parameter frequency plot is very flat (i.e. any value achieves fit of similar goodness) the model is not very sensitive to that parameter, while if the frequency curve is changing very sharply (i.e. a good fit is achieved only for a very small range of the parameter values) the model is very sensitive to the parameter.

In addition, the results of optimum parameter sets and sensitivity analyses in the field applications are compared with the findings obtained from laboratory experimental data in Chapter 5 to evaluate the variations of the optimum parameter sets ($k$ and $C^*$) under different environmental conditions (i.e. laboratory and real world).

## 8.4 Results and discussion

### 8.4.1 Model parameters calibration

1. Calibration of the flow model

The results of the optimum equivalent pipe diameters and orifice discharge coefficients for eight calibration events are listed in Table 8-3, together with the corresponding model efficiencies $E$.

<table>
<thead>
<tr>
<th>Storm event number</th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Optimum equivalent pipe diameter (mm)</td>
<td>81</td>
<td>207</td>
<td>217</td>
<td>215</td>
<td>125</td>
<td>236</td>
<td>240</td>
<td>122</td>
</tr>
<tr>
<td>Optimum orifice discharge coefficient</td>
<td>0.7</td>
<td>0.7</td>
<td>0.7</td>
<td>0.7</td>
<td>0.7</td>
<td>0.7</td>
<td>0.7</td>
<td></td>
</tr>
<tr>
<td>Model efficiency $E$</td>
<td>-0.18</td>
<td>0.71</td>
<td>0.82</td>
<td>0.68</td>
<td>0.2</td>
<td>0.89</td>
<td>0.85</td>
<td>0.2</td>
</tr>
</tbody>
</table>
Based on eight single event simulations, the **weighted average equivalent pipe diameter was 214mm**. This notional diameter was then used as a unified equivalent pipe diameter for all events.

It was found that the modelled outflow hydrographs were generally consistent with the observed outflow hydrographs, except for some low flow-rate events (such Event 4, 9 and 12). For instance, Figure 8-9 illustrates the time series hydrographs for Event 7 from the MUSIC simulations (*Flow In* refers to measured inflow rate at the inlet pond, *Flow Out* refers to modelled outflow rate including the weir overflow rate, and *Observed outflow* refers to measured outflow at the outlet of the macrophyte cell). The comparisons of the modelled and observed outflow hydrographs (at Site 4) for all eight storm events are presented in Appendix D1.

The model efficiency $E$ and cross-correlation coefficient $R^2$ are 0.74 and 0.88, respectively, for all eight events. This can be regarded as very satisfactory, since the flow model in MUSIC used is rather simple.

![Figure 8-9: The time series hydrographs for Event 7 in the MUSIC simulations.](image)

2. Calibrations of the TSS model

Figure 8-10 gives an example of the $k-C^*$ response surface when $N = 1$, while the response
surfaces for the other values of $N$ are given in Appendix D2. The comparisons of the modelled and observed outflow TSS concentrations for all eight events are presented in Appendix D3.

The peak of $E$ for each plot also gives the optimum values of the two parameters $k$ and $C^*$, and it clear that this global optimum can be identified for each value of $N$. The optimum values of $k$ and $C^*$ parameter sets are summarized in Table 8-4 for different numbers of CSTRs $N$, together with corresponding model efficiency $E$, cross-correlation coefficient $R^2$, and Root Mean Square Error (RMSE).

![Figure 8-10: The response surface for two parameters $k$ and $C^*$ with objective function $E$ for 1 CSTR in the MUSIC simulation.](image)

Table 8-4: The optimum $k$ and $C^*$ parameter sets based on different values of $N$.

<table>
<thead>
<tr>
<th>Parameters</th>
<th>Values</th>
</tr>
</thead>
<tbody>
<tr>
<td>Number of CSTRs $N$</td>
<td>1</td>
</tr>
<tr>
<td>Background concentration $C^*$</td>
<td>20</td>
</tr>
<tr>
<td>Pollutant decay rate $k$</td>
<td>3200</td>
</tr>
<tr>
<td>Model efficiency $E$</td>
<td>0.47</td>
</tr>
<tr>
<td>$R^2$</td>
<td>0.48</td>
</tr>
<tr>
<td>$RMSE$</td>
<td>11.40</td>
</tr>
</tbody>
</table>

It is clear from Table 8-4, that the highest value of the model efficiency $E = 0.47$, as well
as the highest $R^2 = 0.48$ (i.e. the best fit), was achieved for the following parameter set:

$$N = 1, \ C^* = 20 \ \text{mg/L and } k = 3200 \ \text{m/y}.$$ 

As addressed above, it should be noted that the inlet pond was modelled integrally within the wetland in the MUSIC simulation, and that in this way, it acted equivalent to one CSTR cell. Thus, the actual optimum number of CSTRs for the whole wetland used in the simulation was 2 CSTRs.

Figure 8-11 shows the predicted outflow TSS concentrations plotted against the observed outflow TSS concentrations at all measured points for eight storm events. The plot also includes ± 25% and ± 50% error bands. The modelled concentrations are largely within ± 50% range of the observed.

![MUSIC predicted vs observed outflow TSS (2 CSTRs)](image)

Figure 8-11: MUSIC predicted versus observed outflow TSS concentrations at measured points for all eight storm events when $N = 2$ (including 1 CSTR provided by the inlet pond).

It can be concluded that the model achieved rather modest accuracy, since $E < 0.5$ for all cases. It is interesting to note that for the recommended number of CSTRs $N = 5$, the model had the worst performance of $E = 0.21$. This is probably due to the fact that the monitored area only consists of a part of the wetland, because Macrophyte Cell 2 of the wetland is not included. However, the default value of $N$ in MUSIC is recommended for a whole wetland,
rather than some portions. Based on the recommended $k$ and $C^*$ values for TSS (500 - 5000 m/y and 5 - 6 mg/l) in wetlands in MUSIC, it was found that the best model efficiency $E$ was 0.21 ($k = 900$m/y, $C^* = 6$mg/L), as shown in Figure 8-12.

![Figure 8-12: Variations of Model efficiency $E$ with $k$ and $C^*$ values recommended in MUSIC using the RCW field data.](image)

### 8.4.2 Parameter sensitivity

The results presented above (in particular the plots of $E$ in Appendix D2) suggest that both $k$ and $C^*$ are important. Further analyses are given by the dotty plots in Figure 8-13 for the impact of each of the two parameters on $E$ (only the results obtained for the combinations of $k$ and $C^*$ with the $E$ value greater than -0.5 are presented in these plots). It appears that there are clear optima of $E$ for both $k$ and $C^*$ at different numbers of CSTRs, and the model is more sensitive to the background concentration $C^*$ than it is to the pollutant decay rate $k$.

![Objective function $E$ versus $C^*$ for 1 CSTR](image)

![Objective function $E$ versus $k$ for 1 CSTR](image)

(a) The number of CSTRs $N = 1$. 

8-25
Figure 8-13: Dotty plot showing objective function $E$ versus the two parameters $k$ and $C^*$, respectively, when using different number of CSTR cells (ranging from 1 to 5) in the MUSIC simulations.

The frequency of each parameter for the data fit to yield an objective function $E$ greater...
than 0.40 (the maximum $E$ was 0.47) when $N = 1$ is presented in Figure 8-14. It shows that the decay rate $k$ has a relatively flat distribution, while the background concentration $C^*$ has a curved distribution with clear maxima. This means that the model results are more sensitive to the parameter $C^*$ in the RCW simulations.

![Figure 8-14: Parameter frequency within the model parameter space with the value of objective function $E$ greater than 0.4 for 1 CSTR simulation.](image)

8.4.3 Comparison of the findings from the laboratory and field application studies

Table 8-5 summarizes the optimum $k$ and $C^*$ parameter sets from the laboratory and field application studies, together with the recommended ranges for $k$ and $C^*$ by MUSIC. The corresponding values of the model efficiency coefficient $E$ and the correlation coefficient $R^2$ are also presented. It is evident that the predictions of the $k$-$C^*$ model can match the laboratory experimental data very well. This is perhaps not surprising, since the $k$-$C^*$ model is, in theory, based on steady-state and plug-flow conditions, which are indeed provided by the laboratory study. On the other hand, the $k$-$C^*$ model seems less capable of predicting the results for the field data, which occur under highly stochastic conditions. As opposed to findings for the experimental data, the value of $k$ found for the field data is within the suggested range by MUSIC. However, the value of $C^*$ is still higher than the
recommended maximum. It is of interest and encouraging to note that the calibrated value of \( C^* \) at the RCW is similar to the measured long-term mean dry weather TSS concentration in wetland, as given in Table 7-3.

Table 8-5: Summary of the optimum \( k \) and \( C^* \) parameter sets in lab and field application studies.

<table>
<thead>
<tr>
<th>Optimum parameter</th>
<th>( k ) (m/( y ))</th>
<th>( C^* ) (mg/L)</th>
<th>( E )</th>
<th>( R^2 )</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Lab experimental data</td>
<td>8004</td>
<td>58</td>
<td>0.88</td>
<td>0.89</td>
<td></td>
</tr>
<tr>
<td>Field data at RCW (( N = 1 ))</td>
<td>3200</td>
<td>20</td>
<td>0.47</td>
<td>0.48</td>
<td></td>
</tr>
<tr>
<td>MUSIC recommended values</td>
<td>500 - 5000</td>
<td>5 - 6</td>
<td>0.79*</td>
<td>0.91*</td>
<td>((\text{highest } E \text{ for lab data}))</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>0.21*</td>
<td>0.25</td>
<td>((\text{highest } E \text{ for field data}))</td>
</tr>
</tbody>
</table>

The results of sensitivity analyses in the field application are somewhat inconsistent with the laboratory experimental findings (where modelled results are sensitive to both the parameters \( k \) and \( C^* \) but more sensitive to \( k \) than \( C^* \), except for high input sediment concentration cases). However, this intuitively makes sense since the background concentrations and the storages in the experimental cells were completely different from those in the RCW. In the laboratory experiments, clear water was injected into the mesocosm cells to establish steady flow conditions before adding sediments. The actual TSS background concentration in the mesocosm cells should be zero, and thus it causes relatively less change on the model results. In addition, the very small storage and the fixed surface area of the mesocosm cell with very little flow attenuation provide an environment more likely to be sensitive to the decay rate, \( k \) (the treatment rate parameter \( k \) perhaps can be visualised as the hydraulic loading). On the other hand, when stormwater passes through the real wetland, a large volume of existing water in storage (e.g. the permanent pool) will mix with the inflow, and flow attenuation in the presence of extended detention storage (when flow rate increases, the actual treatment surface area expanding to the extended detention storage will reduce the actual hydraulic loading), can increase the sensitivity to \( C^* \), but decrease the sensitivity to \( k \). However, it should be noted that the model results are also sensitive to the large value of \( k \) as shown in Figure 8-13, since both flow attenuation and the treatment surface area are limited during a large storm event (e.g., the water level exceeds the extended detention water depth).
8.5 Conclusions

This chapter reports the application of MUSIC to predict outflow concentrations of TSS from a full-scale stormwater wetland. In addition, a rigorous sensitivity analysis of the $k - C^*$ model was carried out based on the eight storm-event data reported in Chapter 7.

The calibration of MUSIC parameters requires good quality and continuous field data, i.e., high temporal resolution inflow and outflow data, including both flow and pollutant concentrations. The predicted outflow hydrographs were generally consistent with the observed (except for some low flow rate events) after the calibration of the notional equivalent pipe diameter for the storage-discharge relationship. Whilst there may be some uncertainties in the observed TSS data (such as some missing data as discussed in Section 7.3.2, and using the TSS concentration measured at one point to represent the mean across the whole cross-section), the predicted TSS concentrations were largely within ± 50% range of the observed (Figure 8-11) with the model efficiency $E = 0.47$ when $N = 1, C^* = 20 \text{ mg/L}$ and $k = 3200 \text{ m/y}$ (optimum values). However, considering the inlet pond as the equivalent of 1 CSTR, there were effectively 2 CSTRs in the simulation. The calibrated background concentration $C^* (= 20 \text{ mg/L})$ coincided with the measured long-term mean dry weather TSS concentration in the wetland (Table 7-3). The optimum value of $k$ was within the range of the recommended $k$ value in MUSIC Version 3.0 ($500$ – $5000 \text{ m/y}$). Taking into account the consistency and consecution of the available TSS data, it can be concluded that the MUSIC model shows promise in the prediction of the performance of the Ruffey’s Creek Wetland.

Meanwhile, the results of sensitive analysis indicated that the model results were sensitive to both the parameters $k$ and $C^*$, but more sensitive to $C^*$ than $k$, possibly resulting from the storage effects in real wetlands (a high background concentration in the storage during dry weather periods and flow attenuation during storm events).
Chapter 9

Testing of the \( N_f \) Model Using the Ruffey’s Creek Wetland Field Data

9.1 Introduction

Based on sedimentation theory and the findings from sediment behaviour in laboratory stormwater wetlands and ponds under both wet and dry weather conditions, a physically based method (named the \( N_f \) model) was developed for long-term continuous predictions of trapping efficiency of sediment particles (see Chapter 6). The aim of this chapter is to test whether or not the proposed \( N_f \) model could be used for real stormwater wetlands with sufficient accuracy so as to apply it confidently in stormwater management practice. Testing of the model is undertaken using the data collected from the Ruffey’s Creek Wetland (described in Chapter 7), consistent with the approach used for testing the \( k-C^* \) model.

As explained in Chapter 6, the \( N_f \) model needs to be coupled with an appropriate hydraulic model. Therefore the simple Puls routing model that is one of the most commonly used hydrological methods was selected for linking with the \( N_f \) model. This is consistent with our aim to make a simple modelling approach, which doesn’t impose excessive calibration and input data collection requirements on the user. It is also consistent with the approach used in the method for testing the \( k-C^* \) model (Chapter 8), allowing an objective comparison between the two models.

The flow modelling is described at the start of this chapter, followed by the sediment modelling. Results of the model calibration (i.e. the model has one parameter, \( C^* \), which needs to be calibrated) are presented, along with examination of the model’s reliability and sensitivity to its input PSD data. This was regarded as a very important part of the study,
since the PSD data, which are the key input of the model, are very hard to collect and are not always reliable. Meanwhile, the model sensitivity to its calibration parameter \( C^* \) is undertaken (in a similar way as for the parameters of the \( k-C^* \) in Chapter 8).

## 9.2 Modelling flows

This section explains the method used for routing flow through the Ruffey’s Creek Wetland, followed by the modelling results.

### 9.2.1 Methods

A simplified hydrological approach for reservoir routing, the Puls method, is outlined in Sections 2.4.2, 8.2.1 and 8.3.2. This method is simple to use and requires a minimal amount of input data and computing effort. This is a very important factor in determining the useability of the overall combined model. Furthermore, this method has been adopted in MUSIC, where it is coupled with the \( k-C^* \) model for predictions of treatment performance of wetlands. Therefore, the decision to couple this model with the \( N_f \) model is logical since it allows a direct comparison of the two testing sediment models (Chapter 8 presents testing of the coupled \( k-C^* \) and Puls methods within MUSIC).

The same method as explained in Chapter 8 was used (i.e. the flow modelling was done in MUSIC). Therefore the details on the methods can be found in Section 8.3.2. It should be noted that the inlet pond was assumed to provide no flow routing (the same as in MUSIC), that is, inlet pond outflow rates equal the inflow rates at every time step. In addition, to calculate hydraulic loading, the inlet pond was assumed to have a depth of 2.0 meters (based on survey data), so that the surface area could be calculated from the storage volume.

### 9.2.2 Results and discussion

The results of the MUSIC flow model have already been presented in Section 8.4.1, and the comparisons of the modelled and observed outflow hydrographs for all eight storm events are presented in Appendix D1. As already discussed, the model efficiency \( E \) and the
Testing of the $N_f$ model using RCW field data

cross-correlation coefficient $R^2$ were 0.74 and 0.88 for all eight storm events, respectively, which can be regarded as very high. This showed that sufficient accuracy could be achieved in the treatment system by using the simple MUSIC model, even though the real physical system is three-dimensional.

9.3 Modelling TSS removal

The $N_f$ model, fully explained in Chapter 6, was simply applied to the RCW, by coupling it with the Puls routing flow model. This section begins with the methods for model application and calibration against the field data, followed by the methods for parameter and input PSDs sensitivity analyses. After that, the results will be discussed.

9.3.1 Methods

9.3.1.1 Model application

Since the inlet pond and the main wetland (a macrophyte zone) were modelled as two individual CSTR cells in the MUSIC simulation of the RCW (see Chapter 8), the same conceptualisation was applied in this modelling exercise. The $N_f$ spreadsheet model was therefore set up for 2 CSTRs.

**Input data:** Data on wetland geometry, inflow rates and inflow concentrations (that were the same as used for the application of MUSIC – see Chapter 8) and PSDs are also needed. Since currently there is no model available to predict the dynamics of PSD in constructed wetlands, it has to be assumed that the inflow PSD is constant over the wet weather period and the dry weather PSD is also steady and uniform in the baseflow storage across the wetland (as initial conditions). Additionally, it was assumed that the particle size distribution of stormwater can be represented by five fractions (i.e. $0 - 4$, $4 - 10$, $10 - 20$, $20 - 100$ and $100 - 710$ µm) with different median diameters $d_{50}$ (i.e. 2, 7, 15, 60, and 405 µm) based on the measured PSDs at the RCW (see Section 7.4.2). The measured mean dry weather PSD (Figure 7-15) was used as the initial PSD in the storage throughout the wetland and the inflow PSD during dry weather periods.
Since the measured wet weather inflow PSDs and dry weather ones varied over time (see Figure 7-14 and 7-15) it was decided that impact of using constant (mean) values on modelled results should be evaluated. The minimum, mean, maximum PSDs for all measured nine wet weather inlet (Site 1) PSD curves shown in Figure 7-14 were calculated, as well as the minimum, mean, maximum PSDs for seven measured dry weather PSDs across the wetland (Site 1, 2, 3 and 4) shown in Figure 7-15. These minimum, mean, maximum wet and dry weather PSDs then produced nine different combinations of wet and dry input PSD data set to investigate the effects of the input PSDs.

**Model parameters:** The baseflow threshold, needed for the selection in between wet and dry weather equations, was set to 15 L/s for the RCW. This value has been suggested as the criterion between stormwater events and dry weather flows by the researchers who conducted the monitoring programme at the wetland (Taylor, 2006). The only model parameter that has to be calibrated is the background concentration $C^*$ (see Table 6-3 in Chapter 6).

9.3.1.2 Modelling and calibration process

The modelling was undertaken at a six-minute time step, to match the resolution of the flow model. The procedure used for calibration and testing of the $N_r$ model is explained below as a step-by-step procedure.

1. The input data were prepared and entered into the $N_r$ spreadsheet: (1) the MUSIC flow modelling results (see Chapter 8) included time-series of inflow rates, outflow rates and storage volumes for each of the two CSTR cells, (2) the inflow TSS concentrations were prepared as time series at the same time step, the values of which in between the measured were simply interpolated using a linear approach (3) information on five particle size fractions were entered using the wet and dry PSD data as discussed above;

2. The background concentration $C^*$ was assumed; and the $N_r$ model was applied to produce the outflow pollutograph of TSS. The model efficiency $E$ (i.e. the objective function, Equation 5.3) was calculated using the modelled and observed outlet (Site...
4) TSS concentrations for all eight events.

3. The above steps were repeated 41 times, by varying the \( C^* \) value between 0 and 40 mg/L. In this way it is possible to find the optimum value for \( C^* \) that provides the maximum \( E \), i.e. it produces the 'best fit' for the wetland.

As outlined in the above sub-section, the calibration was done for nine different combinations of wet and dry PSD inputs in separate modelling exercises (i.e. optimum \( C^* \) and its corresponding \( E \) was determined for each combination of PSD inputs).

Based on the calibrated background concentration \( C^* \), the predicted outlet (Site 4) TSS concentrations were plotted against the observed at measured points, as well as their ± 25% and ± 50% error bands. Meanwhile, the values of model efficiency \( E \) and the cross coefficient \( R^2 \) were also calculated to evaluate the model performance.

9.3.1.3 Parameter sensitivity analysis

The aim of sensitivity analyses is to evaluate whether or not the model is sensitive to variations in its sole calibration parameter, \( C^* \). This was done by calculating \( E \) values (using measured and modelled outflow TSS) with varying \( C^* \) within the range 1 mg/L to 40 mg/L, at an increment of 1 mg/L. In other words, the \( N_r \) model was run for each of the 40 values of \( C^* \) for all events using the measured mean wet and dry weather PSD curves as inputs.

The dotty plot between the objective function \( E \) and background concentration \( C^* \) was then obtained for assessing the model sensitivity to this parameter.

9.3.1.4 Model sensitivity to PSD input data

The errors in input data may propagate through models to trigger significant errors in the model results. Therefore, how sensitive the \( N_r \) model is to the input PSD data for both wet weather and dry weather conditions should be evaluated.

Based on the mean value of the optimum background concentration \( C^* \) for different input PSD combinations, the \( N_r \) model was applied to the same monitored data set for eight storm events using a range of possible values of PSD. Because the highest value of model
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efficiency $E$ was achieved based on the combination of the minimum wet weather PSD ($d_{50} = 4.1 \mu m$) and maximum dry weather PSD ($d_{50} = 29 \mu m$), as provided in Table 9-1 below, the range of wet weather PSD was assumed to be from -70% to 300% of the measured minimum inflow PSD so that the median particle sizes varied from 1.2 to 16.4 $\mu m$ in a sufficient range. Similarly, the range of dry weather PSD was assumed to change from -90% to 200% of the measured maximum dry weather PSD (i.e. the median particle sizes varied from 2.9 to 86 $\mu m$ in a large range). A PSD parameter space was then formed based on the above wet and dry weather PSD ranges at an increment of $\pm$ 10% to form 1140 possible PSD sets. The model performance and the importance of each type of the input PSD (i.e. wet and dry weather) were then evaluated in the parameter space by producing response surface and dotty plots, respectively.

In order to compare the sensitivity to the wet and dry weather PSD data with $C^* = 20$ mg/L, the frequencies of the PSDs changing in the percentage form of the minimum wet weather and the maximum dry weather PSDs, were estimated. The range of PSDs (i.e. -70% to 300% for wet PSDs and -90% to 200% for dry ones) is segmented into some sections so as to define a dimensionless number between 0 and 1 using the same method described in Section 5.3.1. The number of PSD sets to yield an objective function $E$ greater than 0.40 (as the maximum $E$ was 0.48) can be counted for calculating the frequency. The frequency was also expressed as a fraction of total runs that gave rise to a higher $E$ (>0.4).

9.3.2 Results and discussion

9.3.2.1 Model performance

Table 9-1 summarises the results of the optimum values of background concentration $C^*$, together with model efficiency $E$, correlation coefficient $R^2$, and Root Mean Square Error $RMSE$ for different input PSD combinations based on the measured PSDs for wet weather inflow and dry weather. For the input PSD data based on the measured mean wet and dry weather data, the best fit to the observed outflow TSS concentrations was achieved with background concentration $C^* = 20$ mg/L, with $E = 0.43$, $R^2 = 0.49$ and $RMSE = 11.9$. 9-6
For different input PSD combinations, the values of the optimum background concentration $C^*$ varied in a very narrow range (19 to 23 mg/L) with an average value of 20 mg/L and the coefficient of variation equal to 5.5%. However, the model efficiency $E$ ranged from 0.31 to 0.48, which indicates that the model may be very sensitive to PSD even though it is not very sensitive to $C^*$. It was also found that the combination of the measured minimum wet weather PSD and maximum dry weather PSD gave the best fit with $E = 0.48$, $R^2 = 0.50$ and $RMSE = 11.39$, and in this case the optimum value of $C^*$ was 20 mg/L.

Table 9-1: The summary of the optimum $C^*$ values with different combinations of measured wet weather inflow PSD and dry weather PSD.

<table>
<thead>
<tr>
<th>Input particle size distribution</th>
<th>Optimum $C^*$ (mg/L)</th>
<th>$E$</th>
<th>$R^2$</th>
<th>RMSE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wet weather inflow</td>
<td>Dry weather</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Minimum</td>
<td>Maximum</td>
<td>20</td>
<td>0.48</td>
<td>0.50</td>
</tr>
<tr>
<td>Minimum</td>
<td>Mean</td>
<td>19</td>
<td>0.47</td>
<td>0.49</td>
</tr>
<tr>
<td>Minimum</td>
<td>Minimum</td>
<td>20</td>
<td>0.46</td>
<td>0.48</td>
</tr>
<tr>
<td>Mean</td>
<td>Maximum</td>
<td>20</td>
<td>0.44</td>
<td>0.50</td>
</tr>
<tr>
<td>Mean</td>
<td>Mean</td>
<td>20</td>
<td>0.43</td>
<td>0.49</td>
</tr>
<tr>
<td>Mean</td>
<td>Minimum</td>
<td>21</td>
<td>0.41</td>
<td>0.47</td>
</tr>
<tr>
<td>Maximum</td>
<td>Maximum</td>
<td>21</td>
<td>0.35</td>
<td>0.49</td>
</tr>
<tr>
<td>Maximum</td>
<td>Mean</td>
<td>21</td>
<td>0.34</td>
<td>0.48</td>
</tr>
<tr>
<td>Maximum</td>
<td>Minimum</td>
<td>23</td>
<td>0.31</td>
<td>0.44</td>
</tr>
<tr>
<td>Coefficient of variation (%)</td>
<td></td>
<td>5.5</td>
<td>15.5</td>
<td>3.8</td>
</tr>
</tbody>
</table>

Figure 9-1 presents the predicted outflow TSS concentrations plotted against the observed ones at measured points for all eight storm events along with ±25% and ±50% error bands using the measured minimum wet and maximum dry weather PSDs. The modelled concentrations are largely located within ±50% bands, with Event 11 being poorly assessed.

The best model efficiency $E$ is 0.48 (in Table 9-1), which is not very satisfactory but could be considered as acceptable, given the large uncertainty in input data and initial conditions. Some of the uncertainties in the inflow TSS data have been discussed in Section 7.3.2. Moreover, the PSD data were not collected simultaneously with TSS during the eight storm events for the CRCCH project as mentioned in Chapter 7, with a separate campaign being
required to collect PSD data. It is possible that the PSD behaviour at the time of the eight monitored storms differed from that during the recent PSD sampling campaign.

![N$_1$ model predicted vs observed outflow TSS](image)

**Figure 9-1:** The predicted versus observed outflow TSS concentrations at measured points for all eight storm events with $E = 0.48$, $R^2 = 0.50$, and $RMSE = 11.4$ when using the measured minimum wet and maximum dry weather PSDs.

Previous studies also mentioned that the sources of uncertainty in the modelling processes include errors in input data, initial and boundary conditions, calibration data, and model structure, which can tend to induce uncertainty in the model predictions (Beven, 2001; Butts *et al.*, 2004; Haydon and Deletic, 2006). For instance, some assumptions in the modelling processes such as using the constant surface area of each cell to calculate hydraulic loading rate $q$, may cause errors in the model results. This occurred even though the different values of the surface area of the macrophyte zone were used for each storm event in the calculations based on the results adopted from previous HEC-RAS modelling simulations undertaken by Taylor (2006) as part of the storm event monitoring campaign, largely because the actual surface area of the macrophyte zone in the RCW dramatically changed in time during storm events.

### 9.3.2.2 Parameter sensitivity analysis

Figure 9-2 shows how the objective function $E$ changes with the model parameter $C^*$, in the
Testing of the N_f model using RCW field data

form of dotty plot. A clear optimum in the plot means that the performance of the model is sensitive to the parameter $C^*$. Therefore, it suggests that the background concentration $C^*$ needs to be calibrated using some local field data. However it would be interesting to test the model for a range of wetlands and examine how $C^*$ varies from one site to another.

![Dotty plot showing objective function $E$ versus background concentration $C^*$ using mean wet and dry weather PSDs.](image)

Figure 9-2: Dotty plot showing objective function $E$ versus background concentration $C^*$ using mean wet and dry weather PSDs.

9.3.2.3 Model sensitivity to PSD input data

The contour plot for response surface is presented in Figure 9-3. The nearly vertical contours imply that the dry weather PSD has insignificant effects on the model results. On the other hand, the model results seem to be more sensitive to the wet weather inflow PSD.

Figure 9-4 shows the changes (presented in the dotty plots) of the objective function as a function of input PSDs for wet and dry weather conditions, respectively, at $C^* = 20$mg/L. It also clearly shows that the model results are very sensitive to the input wet weather PSD because there is a clear optimum in Figure 9-4-a, and the results become more sensitive for large input wet weather PSDs. On the other hand, no clear optima are formed in the dotty plot for dry weather PSD (Figure 9-4-b). This again indicates that the $N_f$ model is not sensitive to the dry weather PSD.
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Figure 9-3: The response surface for input wet and dry weather PSDs with objective function $E$.

Figure 9-4: Dotty plot showing objective function $E$ versus wet weather PSD (a) and dry weather PSD (b), respectively, when $C^* = 20$mg/L.

These trends can also be detected in Figure 9-5 which provides the frequency plots with the objective function $E$ greater than 0.4 for both wet weather and dry weather PSDs, respectively. The sharp changes with wet weather PSD again confirm the importance of wet weather PSDs. However, it seems that the model results are insensitive to the small input wet weather PSDs. In fact, this may be caused by the limitation of the $N_r$ model for very fine particles as discussed in Chapter 6.

Therefore, it may be concluded that the input wet weather PSD is very critical for the $N_r$ model results, and sufficient PSD data sets should be collected for storm events to achieve optimal model performance.
9.4 Conclusions

The main aim of this chapter was to test the $N_f$ model against the field data collected at the Ruffey’s Creek Wetland so as to gain some confidence that the proposed model could be used in real wetlands with adequate accuracy. The Puls method used in MUSIC was adopted here for routing flows to generate time-series outflow and storage data, which were exported directly from MUSIC as obtained in Chapter 8. The $N_f$ model was then coupled with the CSTRs model as described in Chapter 6 to consider both flow hydrodynamics and sediment trapping simultaneously.

The best model efficiency $E$ was 0.48 compared with the field TSS data, which was considered acceptable, given uncertainties in the input data (e.g. TSS data and PSD data). Taking into account those uncertainties, it can be concluded that the proposed model shows promises for predicting treatment performance with reasonable accuracy. On the other hand, the sensitivity analyses of PSD data sets and the parameter $C^*$ have shown that the input wet weather PSD and background concentration $C^*$ are critical for the $N_f$ modelling results. Hence, sufficient on-site PSD data should be collected for storm events and the background concentration $C^*$ needs to be calibrated using local field data, in order to achieve optimal modelling results.
Chapter 10

Discussion of Findings and Limitations of Analysis

10.1 Introduction

This chapter discusses the major strengths and weaknesses of the work undertaken in this thesis, and compares the $N_f$ model with the $k-C^*$ model based on their performances.

This chapter will first discuss the laboratory study, and then the field data and the modelling work are followed with detailed discussion. Afterwards, the comparisons between the $N_f$ and $k-C^*$ models are presented, which are based on the models’ performances, their input data requirements and calibration requirements. Eventually, the models’ practical applications are addressed.

10.2 The laboratory study

In order to investigate sediment behaviours in constructed stormwater wetlands, data was collected from well-controlled laboratory experiments that were used to simulate the performance of real wetlands, under different inflow hydraulic loadings and inflow sediment concentrations. However, as all experiments, there are uncertainties in the data, relating to scaling, boundary conditions, sampling, handling, laboratory analysis, etc (Melching, 1995), which may impact on the results.

The uncertainties in the presented laboratory study are governed by the following key experimental characteristics; (1) the hydrodynamic scaling and (2) the experimental procedure and programme, as discussed below.
10.2.1 Hydrodynamic scaling issues

Normally, a laboratory study can be conducted at reduced scales, provided that its necessary conditions and generated solutions can mimic physical models (Hughes, 1993). Ideally, a properly designed laboratory experiment should behave in all respects like a miniature version of the real world. However, it is extremely difficult to simulate fully the behaviours of actual (real world) stormwater treatment facilities in the laboratory due to the fact that the myriad factors that are experienced in the field cannot be all replicated in one laboratory test. One of the accepted laboratory experimental methods is to use a scaling approach, which is based on dimensional analysis. It requires that a complete set of dimensionless products (constructed from the pertinent process variables) in the physical model have very close values as in the real world (Hughes, 1993).

At the start of this thesis, four mesocosm wetlands with different well established vegetation densities were used to replicate three dimensionless numbers associated with typical real-world wetlands: Particle Fall Number, $N_f$, Particle Shear Velocity Reynolds Number, $R_{e*}$, and Turbulent Reynolds Number, $R_{er}$. The field data gathered from a real wetland (the Hampton Park Wetland) was used to provide estimates for these key parameters to represent the “real world” under operational conditions. Similarity was achieved for $R_{e*}$ and $N_f$ in both vegetated and non-vegetated mesocosms. However, due to the limitation of an existing laboratory rig, the similarity in the Turbulence Reynolds Number, $R_{er}$, could not always be achieved. This number was reasonably similar between the mesocosm and real vegetated wetlands; although somewhat lower in the mesocosm than in the reality, it was well within the range of real wetlands. Unfortunately, for the non-vegetated mesocosm, it was impossible to achieve replication of $R_{er}$ in the laboratory, although the other two dimensionless numbers ($N_f$ and $R_{e*}$) were well scaled. The analyses showed that for open-water systems, we need full-scale modelling (i.e. the physical model has to be of the same size as the real system) to achieve similarity of all three numbers at the same time.

To overcome the above problem, further study was undertaken for pilot-scale open-water
Discussion of findings

pond experiments using an existing open channel (Chapter 4), by maintaining similarity for the $N_t$ and scaling for Flow Turbulence Number, $R_{eT}$, while not for $R_e^*$. Unfortunately, to achieve this, the flow rates (e.g., flow velocity) in the laboratory had to be much larger than in reality causing re-suspension during very high flow rates. Therefore the data collected at very high flow rates were not applicable to this study since they were related to conditions that should never occur in well designed wetlands (these flows are generally bypassed).

It may be concluded that the sediment transport in vegetated ponds was studied under adequate scaling consideration. Similar approaches have been widely used in physical modelling of many hydraulics studies (Hughes, 1993). For example, Frehmann et al. (2005) carried out a model study using a pilot plant model scaled 1:13 in accordance with the Froude Law of Similarity, and Wagner and Harvey (1997) presented a method for stream tracer experiments using a dimensionless Damkohler number as an indicator of the reliability with which the storage exchanging parameters can be estimated. However, some findings on the sediment behaviour in open-water ponds require further confirmation, due to the difficulties in meeting the scaling requirements for all three governing dimensionless numbers at the same time.

In addition, it cannot be denied that some other important processes, other than physical transport of sediment by water flow, could also be important for sediment behaviour in wetlands. For example, biological factors that may be associated with sedimentation in real wetlands, may not have been fully recreated in the experiments (although an emergent macrophyte, $Baumea articulata$ was well established in the laboratory mesocosms). Also, the presence of a prone litter layer (almost always thicker than five centimetres in real wetlands) and wetland-generated micro-debris probably were missed in the laboratory environments. Actually, bacteria, algae and micro-invertebrates can generate considerable TSS, which in the real world would contribute to wash-out. Furthermore, some studies showed that over the long term, wetland vegetation and organic biomass can enhance sedimentation and filtration mechanisms (i.e. removing the finer particles more efficiently) (Breen et al., 2006; WERF, 2005). Again, we were unable to quantify the effect of
vegetation on these very fine particles due to the short detention time in the existing experiment rig (even in dry weather experiments).

Therefore, the findings have to be confined to understanding of physical processes of sedimentation and re-suspension in wetlands under typical flow conditions.

10.2.2 The experimental procedure and programme

On experimental procedure

The experiments were conducted under steady state conditions. It is usually very hard to achieve steady input of sediments in experiments (i.e. Tollner, et al., 1976; Abt et al., 1994; Verstraeten and Poesen, 2001; Deletic, 2005). However, almost constant inflow sediment concentrations have been achieved in all experiments due to very laborious method used for sediment introduction. This is one of the key strengths of the experimental procedure.

In addition, tubes for siphoning water samples were installed along the mesocosm cells at 0.5 m and 1 m from the inlet weirs for wet weather experiments, and samples were extracted from a single point in the cross-sections (always in the centre of the cross-sectional profile), which may cause some sampling uncertainties. Therefore, some pilot experiments were carried out to assess and account for this uncertainty (Section 3.4.3). Furthermore, multiple samples at each point over a run time were tested for the wet weather experiments and their average was used to minimize the data uncertainties. On the other hand, it was very difficult to predict and replicate sediment concentrations at a distance of 0.5 m from the inflow weirs due to unpredictable sedimentation behaviour in upstream zone of this point (the point at 0.5 m from the inlet had to be taken as ‘the starting point’ of the cell to allow the first 0.5 m zone for flow mixing in order to establish a uniform flow and sediment profile). This can make the obtained inflow TSS concentrations and PSDs at the first sampling points in the mesocosms less than the expected.

The measurement errors in TSS were rather low (standard methods for TSS analysis were
used and the overall uncertainty of TSS of a sample was $\pm 3.8\%$ of the measured concentration as explained in Section 3.3.3), while PSD measurements were conducted using laser diffraction system from Malvern instruments which is regarded as very reliable. However, the experimental measurement errors for very fine and coarse particles were reasonably high (for similar reasons discussed by Deletic 2005). PSD measurements are not very accurate for very fine particles which coupled with very small concentrations results in yielding high relative errors. Very little could be done to resolve these problems as discussed by Deletic (2005).

**On extent of experimental programme**

The idea for experimental work is to obtain the results for all cases that we can have in reality. An attempt was made to achieve this by conducting experiments for a range of key variables that may impact on the sediment behaviour: four vegetation densities, two different hydraulic loading rates (i.e. flows) and three inflow sediment concentrations. The ranges of these variables were endeavoured to be similar to the ranges found in practice, and the experiments were carried out for each of their possible combinations. Although, the range covered is very large, some limitations should be acknowledged as discussed below.

The relatively high plant density was used for simulating the most densely planted systems. Although the vegetation density higher than in reality could magnify the plant effects, it is believed that this is not important since almost no plant effect was found. In other words, the fact that we investigated for a wider range of plant densities than expected and found that this variable has no effect, could be regarded as the strength of the programme.

The input sediment concentrations were expected to vary within a natural range including the extremely high inflow concentrations that may happen on bare soils during intense storm events. We attempted to work out the influence of the highly stochastic nature of hydraulic loading in stormwater wetlands. However, the hydraulic loading rates used were close to the upper bound of their ranges found in real wetlands. This was mainly due to the
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Limitations in the existing rig (it was impossible to reliably regulated flows lower than those used), but also because we wanted to test the system for the ‘worst’ conditions. On the other hand, there were large gaps in the range of hydraulic loadings (between 2000 m/y and 7000 m/y) at a limited cost. Uncontrolled flow rate or excessively high flow velocity was not taken into account in the laboratory tests, but it should be noticed that the settled particles can be remobilized and scoured away by the faster flow and they contribute to TSS in the outflow when infrequent storms occur in the real wetlands in the presence of unexpected flow short-circuiting and/or turbulence (even the use of bypass for protecting the macrophytes zone is adopted).

The experiments were carried out in the laboratory environment without environmental interactions. This could ignore some important influencing factors on sedimentation that are expected to exist in the real wetlands. For instance, wind-induced re-suspension was hard to model in the laboratory. Therefore, one of the key conclusions was that these environmental impacts (e.g. wind or birdlife driven re-suspension) should be incorporated in any future models for sediment behaviour in wetlands, if possible.

10.3 The field data

Calibrations of the proposed \( N_r \) model require time series data of both flow and pollutant concentrations, but there are very few such datasets available in Australia. The data collected from the Ruffey’s Creek Wetland (RCW) provided an opportunity for calibrating and testing those models. Furthermore, inter-event (i.e. dry weather) TSS data were also gathered regularly to provide a whole description of the water quality within the wetland.

One of the main strengths of field data used in verification of the \( N_r \) model is that the laboratory experiments that were used for model development have not been associated with the RCW; this gives an independent means of verification.

The extent to which the dataset can represent the performance of the wetland depends on the data quantity and quality. A long contiguous series of data with high spatial and


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Temporal resolution are required to achieve optimal testing and calibration of the model. Some uncertainties in the TSS data, such as missing peak values during storm events can limit the availability of the data in model calibration. It should also be noted that the measured TSS value at a single sampling point was used to represent the mean value for the whole cross section along the wetland, which should be considered as an influencing factor on data uncertainty. In addition, some uncertainties may exist in the measured flow data. Inflow rate was only monitored in the 1600 mm main inlet pipe and the inflow from the secondary 600 mm inlet pipe that drains an adjoining small sub-catchment was not monitored and roughly estimated to contribute an additional 8% to the inflow volume. Furthermore, the flow sensor was placed approximately 65 mm above the invert of the inlet pipe to avoid the disturbance caused by the backflow from the inlet pond, and therefore the potentially unmonitored inflow rate that cannot be measured was approximately estimated to be 30 L/s and the baseflow was estimated to be 15 L/s. This may give rise to some uncertainty in flow data.

Since the particle size distribution (PSD) data at the RCW were not collected at the same time as the flow and pollutant concentration data in the CRCCH project, this introduces a source of uncertainty. There may have been some changes in PSD over the four years between the CRCCH and the present study. In addition, the limited PSD data from two round samplings measured by two different particle size analysers (Mastersizer 2000 and Bechmann Coulter LS100) also showed some variations, which may also introduce systematic errors in PSD data.

10.4 Development of the N_f model

Two formulae for assessment of sediment trapping efficiency under wet and dry weather conditions, respectively, were developed by regression analyses of the measured particle trapping efficiencies by the functional forms of the non-dimensional numbers. The results from the regression analyses were derived based on 465 and 929 data points for wet and dry weathers under different flow conditions, respectively, which provide a large enough
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database for accurate and reliable statistical analyses. Thus, the sedimentation behaviours under wet and dry weather conditions can be regarded as fully investigated. However, it should be noted that the wet weather regression model was not very accurate for very fine particles, since the results in Figure 6-1 were not reliable for particle sizes less than 6 µm. This may be also one of the reasons why a lower value of model efficiency $E$ was obtained in the testing of the model against the field data collected from RCW (the measured inflow median particle sizes were below 10 µm, ranging from 4.2 to 9.9 µm).

To overcome the lack of consideration of ‘environmental conditions’ by the laboratory study, a lumped parameter - a background concentration $C^*$ - has been introduced into the model to account for wash-off and particle re-suspension caused by environmental factors such as wind, birdlife, and etc. How the background concentration $C^*$ varies (or not) across wetlands needs further investigations in the future.

10.5 Comparisons between the $N_f$ model and the $k-C^*$ model

The proposed $N_f$ model was compared with the $k-C^*$ model in terms of their ability to predict wetland treatment performance, their input data requirements and calibration requirements.

10.5.1 Model performance

The data collected from both the laboratory experiments and the field study at the RCW were used to test the proposed $N_f$ model and the standard $k-C^*$ model in Chapters 5, 6, 8 and 9, respectively. It should be noted that for the field application, the $N_f$ model requires calibration of one parameter ($C^*$), while the two parameters of the $k-C^*$ model are also calibrated against the same data set. The performance of the models was assessed using the same objective functions like model efficiency $E$, cross-correlation coefficient $R^2$, and Root Mean Square Error ($RMSE$), which are summarized here in Table 10-1.
It is clear that, with calibrated parameters, both models gave excellent predictions on sediment trapping efficiency for the laboratory experiments (e.g. both $E$ and $R^2$ values were around 0.9). The calibration of the $N_r$ model was marginally more successful than the $k-C^*$ model. In summary, it may be concluded that both models are capable of giving accurate prediction for sediment trapping efficiency under steady flow conditions.

Table 10-1: Model statistics for the $N_r$ model and the $k-C^*$ model using the laboratory experimental data and the field data at the RCW.

<table>
<thead>
<tr>
<th>Source of data</th>
<th>Type of model</th>
<th>$E$</th>
<th>$R^2$</th>
<th>RMSE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Laboratory</td>
<td>$N_r$</td>
<td>0.91</td>
<td>0.92</td>
<td>41.8</td>
</tr>
<tr>
<td></td>
<td>$k-C^*$</td>
<td>0.88</td>
<td>0.89</td>
<td>50.3</td>
</tr>
<tr>
<td>Field</td>
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<td>0.48*</td>
<td>0.50</td>
<td>11.4</td>
</tr>
<tr>
<td></td>
<td>$k-C^*$</td>
<td>0.47</td>
<td>0.48</td>
<td>11.4</td>
</tr>
</tbody>
</table>

* This value was based on the measured minimum wet weather PSD and maximum dry weather PSD which gave the best fit. For mean wet and dry weather PSD, the value of $E$ was 0.43.

On the other hand, the predictions obtained for both the $N_r$ and $k-C^*$ models were not so good for the field data sets collected from the RCW ($E$ values were 0.48 and 0.47, respectively), especially compared with their excellent performance for the laboratory data sets. This probably was due to unsteady intermittent nature of stormwater flow and pollutant loading through the real wetland. The models had to be coupled with the flow-hydrodynamic (CSTRs) model that can account for the variable conditions in real stormwater wetlands. However, the CSTRs model assumes complete mixing during each time step in one CSTR cell, and the water and sediment is routed from the upstream to the downstream cell with equal rates over the entire width and depth of the cells in series. This assumption of well-mixed water and sediment movement has been considered as a typical model for transport and conversion (Kadlec, 1994). In reality, re-circulating and stagnant zones can reduce the engaged water volume and significantly affect the treatment efficiency in consequence of complex geometries (Wörman and Kronnäs, 2005). The RCW consists of a combination of treatment systems with complex shapes. The ephemeral cell particularly has a distinct short circuit during low to medium flows (Figure 7-2) caused by the presence of a low flow channel. The overall hydraulic efficiency was estimated to be of the order of 0.37 (Taylor, 2006). Thus, the RCW was actually modelled using two...
Discussion of findings

CSTRs. The assumption of completely mixing in each CSTR cell, however, could lead to some uncertainties in the model results, especially for the cell with permanent storage since the water depth has a direct impact on hydraulic efficiency (Holland et al., 2004).

In general, despite the fact that there were some evident differences between the predicted TSS concentrations at outlet (Site 4) using both the $k-C^*$ model and the $N_f$ model and the RCW field data, the general agreement could be observed as shown in Figure 8-11 and Figure 9-1. The model efficiency values ($\geq 0.47$) for both of the two models were just acceptable, taking into account the uncertainties of the available data (see Section 10.3).

It may be concluded that in comparison to $k-C^*$ model, the $N_f$ model provided similarly reliable predictions of the sediment trapping efficiency in laboratory mesocosm wetlands. For the field conditions the predicting ability of both models is not great, but can be regarded as acceptable.

10.5.2 Input data requirements

In general, input data are required for model development, calibration, validation, and implementation and use. Data used for these purposes should be obtained from independent sources such that the data sets are distinct from each other. Input data requirements for modelling differ greatly, depending on the type of model being developed and the overall complexity of the system being modelled.

The required input data for the proposed $N_f$ model are as follows,

- inflow rates,
- inflow TSS concentrations,
- inflow wet weather and dry weather sediment particle size distributions, and
- geometry of the treatment system (such as length, depth, width, etc.).

The $k-C^*$ model requires the following input data,

- inflow rates,
Both models need inflow rates, inflow TSS concentrations, and geometry of the treatment system as input data. The \( N_r \) model also necessitates the particle size distributions of sediments in the storm-event and dry weather inflows (initial conditions). For example, five particle fractions are used in our analyses and their individual sediment trapping efficiency is calculated prior to obtaining the total sediment trapping efficiency. On the other hand, in the \( k-C^* \) model, the settling velocities for different sizes of sediment particles are reflected by the notional value of the areal removal rate \( k \) that needs to be calibrated using local data (Wong and Geiger, 1997; CRCCH, 2005).

The volume of the input data required for the model may have some effects on the application of these models since all model parameters are calibrated on the basis of available data. However, many WSUD designers often have to use assumed values as input data for their designs in practice owing to the fact that the required input data are often not available at the location of interest and must be obtained by transfer from the nearby or similar observations. In fact, the inflow rate, TSS and PSD are highly variable between the differing land uses, catchment geologies, and between areas with differing climatic conditions. The use of local stormwater quantity and quality data (i.e. flow, TSS and PSD data) becomes very important provided that the proposed model is of reliable input. Especially, care needs to be taken in ensuring that appropriate wet weather PSD is used to achieve better model performance, since the proposed \( N_r \) model is sensitive to the wet weather inflow PSD, but not to the dry weather PSD as shown in Section 9.3.2.3.

In comparison to the \( N_r \) model, the \( k-C^* \) model needs less input data (i.e. no PSD data), but leaves more parameters to be calibrated locally as discussed in the following section.

**10.5.3 Model parameters**

In general, hydrological and treatment models should be calibrated and verified against available observed data (Novotny and Olem, 1994; McAlister et al., 2006). This can be
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explained by the fact that the empirical or semi-empirical water-quality deterministic models require knowledge of a set of coefficients and reaction rates to describe the water quality of the treatment system, since they can only be obtained by extensively calibrating the model against measured data (Novotny, 2003).

Two calibration parameters exist in the $k-C^*$ model:

- the apparent background concentration $C^*$, and
- the areal decay rate $k$.

The optimum values of $C^*$ and $k$ for the laboratory mesocosms and the RCW are listed in Table 8-5, together with the recommended values provided in the MUSIC user manual. It showed that the values of the two parameters varied widely in different cases.

The previous studies have also shown that these two parameters are strong functions of the hydraulic loading, inlet concentration and sediment particle size (Kadlec, 2000; CRCCH, 2005). Furthermore, it should be noted that the distinction between true background concentration and apparent background concentration must be clarified. The apparent background concentration is the calibrated value that serves the purpose of allowing deviations from a strict exponential decline in sediment concentration through the wetland, while the true background concentration is the result of atmospheric and groundwater chemical additions, chemical speciation, and the biogeochemical cycle (Kadlec, 2000). Only in some cases, these two concentrations are found to be identical. Hence, satisfactory performance of the $k$-$C^*$ model highly depends on the appropriate $k$ and $C^*$ values that should be calibrated using local field data. Some previous studies (e.g. Deletic and Fletcher, 2006) have shown that both of these parameters can change, even at one site, making extrapolation to other sites very difficult.

In contrast, the model coefficients built in the $N_t$ model (Equations 6.1 and 6.3) were able to be kept constant for the field study, while only one model parameter needed to be calibrated, that is,

- background concentration $C^*$, which is a lumped parameter reflecting the overall
effects of wash-off and particle re-suspension in a water column.

Thus, the number of calibration parameters in the $N_r$ model is less than other similar models. The calibration process of the $N_r$ model in some aspects is simpler than that for the $k-C^*$ model. Since the calibration exercise of the RCW using different combinations of PSD in dry and wet periods, showed that $C^* = 20 \text{ mg/L}$ is a rather robust estimate, it may be hypothesised that this value may be rather universal. It would be interesting to study this model further by applying it to a number of wetlands, to test this hypothesis. If $C^*$ turns to be very constant (at around $20 \text{ mg/L}$), this would make the $N_r$ model easily transferable from site to site.

It is also important to note that the calibration of either the $k-C^*$ model or the proposed $N_r$ model has to be coupled with an appropriate hydraulic model and an effective volume transport model (such as CSTRs model in Chapter 8 and 9) to account for the unsteady nature of stormwater flow through constructed stormwater wetlands.

### 10.6 The model application in practice

It should be noted that all model calibrations and subsequent predictions will be subject to uncertainty (Beven, 2001). In many cases, the required calibration data is unavailable or the data collection from some areas is impractical (e.g. some new development areas), and even the input data have large uncertainties. Therefore one of the key sources of uncertainties is the model parameters that need to be calibrated; the more parameters we have to calibrate, the more difficult it is to establish the model and therefore the more unreliable our predictions are. For complex calibration parameter sets, large data sets may be required for a robust optimization (Beven, 2001). Therefore, the number of calibration parameters required by the model and the quality and quantity of the observed data can also significantly affect the modelling usefulness, efforts and success in practice.

This study confirmed that the main parameters in the $k-C^*$ model varied widely in the laboratory mesocosms and the RCW, and were different than the recommended values in MUSIC (especially the background concentration $C^*$). It seems that this model can work
Discussion of findings

only if it is calibrated using data collected at the wetland. This was also observed by other studies (Kadlec, 2003; Stein et al., 2006). Therefore it could be concluded that the $k-C^*$ model should not be used for detailed design of wetlands (Kadlec, 2000, no calibration data can be collected on systems that are yet to be built). However, this model is very simple to use and does not require lots of input data. Therefore the $k-C^*$ model can provide a quick assessment of loads reductions by building a 'conceptual' wetland (not only for TSS, but also for TP, TN). These predictions should be adequately accurate for the conceptual design and planning purposes. This is after all the main purpose of MUSIC (CRCCH, 2005).

In contrast, the $N_r$ model is a one-parameter model ($C^*$), but requires additional data on PSD. It is somewhat more complex. There are some indications that the parameter $C^*$ of this model may only vary within a narrow range, so further studies are needed to test this, and to test how transferable the model is. In any case, having only one calibration parameter makes the model more robust. Therefore, it is possible that the $N_r$ model could be even used for detailed design of wetlands if accurate data on inflow PSDs are available. For example, the input PSD data could be measured in the upstream pipes from a future wetland, and the model is then used to size the system. However, the model is still in need to be tested by applying it to a range of wetlands.
Chapter 11

Conclusions and Recommendations for Future Research

11.1 Conclusions

Constructed wetlands have been one of the most common measures for urban stormwater quality improvement in providing effective removal of suspended solids, nitrogen, phosphorus and heavy metals from stormwater. The effectiveness of sediment removal in constructed wetlands has been demonstrated in laboratory, field, and modelling studies carried out within this project. The key findings from each of these studies are summarized below.

1. Conclusions drawn from the laboratory study

The most important findings from the laboratory study in four mesocosm wetlands and an open-water pond are the following:

(1) During wet weather flows, TSS concentrations decrease exponentially along the wetland length, with a decay coefficient being highly dependent on sediment characteristics (e.g. particle size) and flow characteristics (e.g. flow velocity, rate and depth).

(2) During dry weather periods, the outflow TSS concentrations depend on particle size and flow characteristics, as well as on the time elapsed from the last wet weather event.

(3) Re-suspension due to flow has been found to be negligible during both wet and dry weather periods. This is only the case, however, for flows that are within ranges as suggested in current design guidelines (i.e. when a bypass is built to convey large
flows), while scouring will occur if flow velocities in the wetland are over 0.12 m/s. This has confirmed the appropriateness of current Australian guidelines on scour protection in wetlands.

(4) The effects of vegetation density do not give rise to significant differences in the total sediment deposition for both wet and dry weather experiments.

2. Conclusions drawn from the field data study

This study was built on a program of intensive monitoring of the Ruffey’s Creek Wetland for eight storm events and provided the following conclusions:

(1) The observed inflow and outflow hydrographs have shown that the wetland is effective for peak flow attenuation during storm events, and the bypass system (for protecting the downstream vegetated cells from high flows) can reduce re-suspension of sediment and prevent scouring of epiphytes and deposited fine particles.

(2) Monitoring of the sediment TSS concentrations through the wetland has demonstrated that TSS concentrations mostly decrease over distance during storm events, especially comparing the inflow and outflow TSS concentrations, but the overall sediment removal rate is significantly dependent on the inflow rates.

(3) The observed dry weather TSS concentrations by grab sampling for different times and at different sites in the wetland were highly variable. The measured dry weather particle sizes were larger than the wet weather particles in the wetland; this may be explained by the fact that re-suspension of sediment can be caused by environmental factors such as wind and birdlife in the shallower water, during dry weather periods.

3. Conclusions drawn from the development of the $N_f$ model

A simple model, named the $N_f$ model, has been developed to predict particle trapping efficiency in constructed stormwater wetlands. Its merits lie in its applicability to different flow conditions. The following are the key conclusions from this part of the study:
Conclusions

(1) During wet weather, the sediment trapping efficiency of a particle size fraction has been found to be a simple function of its Particle Fall Number, \( N_f \). This number is a ratio of the horizontal flow time to the vertical particle settling time. The following regression has been established using the laboratory data from four mesocosms:

\[
Tr_s = \frac{N_f^{0.43}}{N_f^{0.43} + 1.42}
\]

(2) During dry weather, the sediment trapping efficiency of a particle size fraction has been found to be a simple function of its Particle Fall Number, \( N_f \), and non-dimensional time ratio, \( t^* \). \( N_f \) is defined as for wet weather, while \( t^* \) is the ratio between the actual time since the last storm event and the mean detention time in the wetland. The following regression has been established using the laboratory data from four mesocosms:

\[
Tr_s = \begin{cases} 
1 & \text{if } N_f \leq 800 \\
1 - 0.1e^{-1.87t^*} (7.29 - \ln N_f) & \text{if } N_f > 800 
\end{cases}
\]

(3) The \( N_f \) model has been developed by incorporating the above two regressions into a simple framework that can also account for the flow hydrodynamics within a wetland. The integrated \( N_f \) model coupling with an appropriate hydraulic model and the continuously stirred tanks reactors (CSTRs) model is capable of continuously assessing the long-term performance of stormwater constructed wetlands during each storm event and over inter-event dry periods. Due to environmental factors not directly studied in this thesis (such as re-suspension due to birdlife or wind) the re-suspension is modelled by introducing a lumped parameter, named “the background concentration”, \( C^* \), which needs to be calibrated using readily-obtainable field data. The total TSS concentration of the outlet can be calculated as a sum of the concentrations for all fractions.

4. Conclusions drawn from the testing of the proposed \( N_f \) model and the \( k-C^* \) model

Both the \( k-C^* \) and \( N_f \) models were successfully calibrated against the lab data, with
coefficients of model efficiency being around $E = 0.9$. This is not surprising for the Nr model, since the regressions were developed using the same set of data.

The proposed Nr model, as well as the $k-C^*$ model, were coupled with the continuously stirred tank reactor (CSTR) model and were applied to the Ruffey's Creek Wetland. Both models had similar performance, with the model efficiencies being just below 0.5. It should be mentioned that the Nr model requires more input data than the conceptual $k-C^*$ model; the difference is in requirements for the PSD curves for sediment during both wet and dry weathers. On the other hand, the $k-C^*$ model has one more model parameter (i.e. $k$) to be calibrated. It has been confirmed that the $k-C^*$ model is very sensitive to both parameters, and that it is not readily transferable from site to site. The Nr model has only one calibration parameter $C^*$. Further work is needed to study whether this parameter can be transferable between sites. However, obtaining an estimate of $C^*$ from collection of field data during a “pilot study” is a much simpler (and less expensive) task than collecting similar data for $k$. $C^*$ can be calibrated by simple grab-sample monitoring during post-storm periods, whilst $k$ can only be measured by continuous (small time step) monitoring during a storm event.

11.2 Recommendations for future work

The research work was undertaken for its fundamental nature, but there remain a number of improvements and knowledge gaps that remain for future investigation. The following recommendations are proposed for the continuation of this study.

1. Further testing of the Nr model

Up to now, the proposed Nr model has been tested against the data collected from the Ruffey’s Creek Wetland. Although it shows promise in practice, more field data from other monitoring wetlands are needed for verification and refinement of the model.

Furthermore, the testing of the proposed Nr model in this study was coupled with the 1-D continuously stirred tank reactors in series to account for the flow hydrodynamics within a
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wetland. Although the CSTRs model can provide reasonable predictions for wetland hydrodynamics, it is too simple to describe the 3-D nature of physical systems. It is expected that the better results may be obtained if the $N_f$ treatment model can be combined with a complex (such as 2-D or 3-D) pollutant transport model, in order to describe more accurately the relationship among flow hydrodynamics, sediment transport and deposition. It is hypothesized that such an approach would increase the reliability of the TSS predictions.

2. Improvements of the $N_f$ model

The reliability of the model regarding small fractions (less than 6 µm) should be improved in the future if possible. This can be achieved through gathering more experimental data using fine input sediment particle size (e.g. $d_{50}$ less than 10 µm) and using a broad measuring range instrument to calibrate samples with high accuracy (such as Mastersizer 2000) for PSD analysis.

On the other hand, the current model uses a lumped parameter - apparent background concentration ($C^*$) to account for the effects of wash-off and re-suspension of fine particles due to environmental factors. The improvement of the proposed model, including a re-suspension model for clearly estimating wind-driven re-suspension would make this method physically sound.

3. Extension of the $N_f$ model

The $N_f$ model was developed in order to predict the trapping efficiency of sediment particles in surface-flow constructed stormwater wetlands. The approach presented here can be extended to predict the removal efficiency of particle-bound pollutants such as total phosphorus and heavy metals.

Furthermore, a similar model, the Aberdeen model (Equation 2.21), has been successfully tested for wet weather stormwater swales and grass filter strips (Deletic & Fletcher, 2006). The $N_f$ model seems to be applicable across a number of surface flow stormwater treatment systems. It may be thus hypothesised that the same methodology can be applied to a wide
Conclusions

range of pollutants under different treatment facilities, by evaluating the main model
coefficients (i.e. $a$ and $b$ in Equation 6.1) for each treatment system and each pollutant.
However, this hypothesis should also be verified using experimental and field data before
the $N_r$ model is recognized as a useful, universal tool for the design and performance
evaluation of a range of surface flow stormwater treatment systems.
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Appendices

Appendix A

Laboratory Study- Mesocosm Wetlands – Wet and Dry Weather Experiments
Appendix A1: The variables for all wet weather experiments (L stands for Low, M for Medium, H for High, F for Flow, C for Concentration).

<table>
<thead>
<tr>
<th>Experiment</th>
<th>Vegetation density</th>
<th>Flow depth</th>
<th>Flow rate</th>
<th>Hydraulic loading</th>
<th>Input sediment concentration</th>
<th>Input median particle size</th>
</tr>
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<tbody>
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<td>Name No.</td>
<td>(culms/m²)</td>
<td>(m)</td>
<td>(m³/s)</td>
<td>(m³/y)</td>
<td>(mg/L)</td>
<td>(µm)</td>
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<td>8842</td>
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<tr>
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<td>7110</td>
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Appendix A2: The variables for all dry weather experiments (L stands for Low, M for Medium, H for High, F for Flow, C for Concentration).

<table>
<thead>
<tr>
<th>Experiment</th>
<th>Vegetation density</th>
<th>Water depth (m)</th>
<th>Flow rate (m³/s)</th>
<th>Hydraulic loading (m/y)</th>
<th>Mean wet-weather sediment concentration in the flume (mg/l)</th>
</tr>
</thead>
<tbody>
<tr>
<td>LF-LC</td>
<td>High=2936</td>
<td>0.04</td>
<td>5.77E-06</td>
<td>359</td>
<td>35</td>
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<tr>
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<td>7.79E-06</td>
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<tr>
<td></td>
<td>Low=590</td>
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<td>1.07E-05</td>
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<tr>
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<td>545</td>
</tr>
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<td>1.74E-05</td>
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<td>9.77E-06</td>
<td>609</td>
<td>480</td>
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<tr>
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<td>1.30E-05</td>
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<td>604</td>
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<td>1.53E-05</td>
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<td>No vegetation=0</td>
<td>0.04</td>
<td>1.12E-05</td>
<td>687</td>
<td>617</td>
</tr>
</tbody>
</table>
Appendix A3: The observed TSS concentrations at three positions (0.5 m, 1.0 m and 2.0 m from the inlet weir) along each mesocosm cell versus time for both wet and dry weather experiments (experiments are grouped in five classes according to wet weather inflow rate and inflow TSS concentration).

Low Flow – Low Concentration experiments:
Low Flow – Medium Concentration experiments:

Experiment 2

Experiment 8
Experiment 9

High Flow – Low Concentration experiments:

Experiment 10
High Flow – Medium Concentration experiments:

Experiment 5

High Flow – High Concentration experiments:

Experiment 3
Appendix A4: Fraction concentrations averaged over each wet weather experiment, versus distance along the mesocosm cell (experiments are grouped in five classes according to wet weather inflow rate and inflow TSS concentration).

Low Flow – Low Concentration experiments:

Experiment 7

Low Flow – Medium Concentration experiments:

Experiment 2
High Flow – Low Concentration experiments:

Experiment 10

High Flow – Medium Concentration experiments:

Experiment 5
High Flow – High Concentration experiments:

Experiment 3

Experiment 4
Experiment 6
Appendix A5: The measured PSD curves at points 0.5 m, 1.0 m and 2.0 m from the inlet weir at different times for wet weather experiments (experiments are grouped in five classes according to inflow rate and inflow TSS concentration).

**Low Flow – Low Concentration experiments:**

<table>
<thead>
<tr>
<th>Experiment 1 (LF-LC) High-vegetated Cell</th>
<th>Experiment 1 (LF-LC) Medium-vegetated Cell</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wet: 0.5m</td>
<td>Wet: 0.5m</td>
</tr>
<tr>
<td>Wet: 2.0m</td>
<td>Wet: 2.0m</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Experiment 1 (LF-LC) Low-vegetated Cell</th>
<th>Experiment 1 (LF-LC) Non-vegetated Cell</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wet: 0.5m</td>
<td>Wet: 0.5m</td>
</tr>
<tr>
<td>Wet: 2.0m</td>
<td>Wet: 2.0m</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Experiment 7 (LF-LC) High-vegetated Cell</th>
<th>Experiment 7 (LF-LC) Medium-vegetated Cell</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wet: 0.5m</td>
<td>Wet: 0.5m</td>
</tr>
<tr>
<td>Wet: 1.0m</td>
<td>Wet: 1.0m</td>
</tr>
<tr>
<td>Wet: 2.0m</td>
<td>Wet: 2.0m</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Experiment 7 (LF-LC) Low-vegetated Cell</th>
<th>Experiment 7 (LF-LC) Non-vegetated Cell</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wet: 0.5m</td>
<td>Wet: 0.5m</td>
</tr>
<tr>
<td>Wet: 1.0m</td>
<td>Wet: 1.0m</td>
</tr>
<tr>
<td>Wet: 2.0m</td>
<td>Wet: 2.0m</td>
</tr>
</tbody>
</table>
Low Flow – Medium Concentration experiments:

Experiment 2

Experiment 8
Experiment 9

High Flow – Low Concentration experiments:

Experiment 10

Experiment 10 (HF-LC) High-vegetated Cell

Experiment 10 (HF-LC) Medium-vegetated Cell

Experiment 10 (HF-LC) Low-vegetated Cell

Experiment 10 (HF-LC) Non-vegetated Cell
High Flow – Medium Concentration experiments:

Experiment 5

High Flow – High Concentration experiments:

Experiment 3
Experiment 4 (HF-HC) High-vegetated Cell

Experiment 4 (HF-HC) Medium-vegetated Cell

Experiment 4 (HF-HC) Low-vegetated Cell

Experiment 4 (HF-HC) Non-vegetated Cell

Experiment 6 (HF-HC) High-vegetated Cell

Experiment 6 (HF-HC) Medium-vegetated Cell

Experiment 6 (HF-HC) Low-vegetated Cell

Experiment 6 (HF-HC) Non-vegetated Cell
Appendix A6: The average measured PSD curves at points 0.5 m, 1.0 m and 2.0 m for wet weather experiments; and the measured PSD curves at 2.0 m at different times for dry weather experiments (experiments are grouped in five classes according to wet weather inflow rate and inflow TSS concentration).

Low Flow – Low Concentration experiments:

![Graphs showing PSD curves for different conditions](image1)

Low Flow – Medium Concentration experiments:

![Graphs showing PSD curves for different conditions](image2)
Experiment 8

Experiment 9
High Flow – Low Concentration experiments:

Experiment 10

High Flow – Medium Concentration experiments:

Experiment 5
High Flow – High Concentration experiments:

Experiment 3 (HF·HC) High-vegetated Cell

Experiment 3 (HF·HC) Medium-vegetated Cell

Experiment 3 (HF·HC) Low-vegetated Cell

Experiment 3 (HF·HC) Non-vegetated Cell

Experiment 4 (HF·HC) High-vegetated Cell

Experiment 4 (HF·HC) Medium-vegetated Cell

Experiment 4 (HF·HC) Low-vegetated Cell

Experiment 4 (HF·HC) Non-vegetated Cell
Experiment 6
Appendix B

Application of the $k-C^*$ Model Using Laboratory Experimental Data
Appendix B1: The predicted versus observed TSS concentrations based on the optimum $k$ and $C^*$ parameter sets for different data categories.

**Vegetation density categories:**

- **High-vegetated cell experimental data**
  - $k=7635 \text{m/y}$
  - $C^*=64 \text{mg/L}$
  - $E=0.87$
  - $R^2=0.88$

- **Low-vegetated cell experimental data**
  - $k=7172 \text{m/y}$
  - $C^*=37 \text{mg/L}$
  - $E=0.96$
  - $R^2=0.96$

**Hydraulic loading categories:**

- **Very high hydraulic loading experimental data**
  - $k=11266 \text{m/y}$
  - $C^*=30 \text{mg/L}$
  - $E=0.86$
  - $R^2=0.86$

- **Medium hydraulic loading experimental data**
  - $k=6811 \text{m/y}$
  - $C^*=42 \text{mg/L}$
  - $E=0.91$
  - $R^2=0.91$

- **Low hydraulic loading experimental data**
  - $k=3346 \text{m/y}$
  - $C^*=22 \text{mg/L}$
  - $E=0.76$
  - $R^2=0.76$
Input sediment concentration categories:

High input sediment concentration data

- $k=20843 m/y$
- $C^*=274 mg/L$
- $E=0.57$
- $R^2=0.57$

Medium input sediment concentration data

- $k=6981 m/y$
- $C^*=72 mg/L$
- $E=0.90$
- $R^2=0.90$

Low input sediment concentration data

- $k=13898 m/y$
- $C^*=18 mg/L$
- $E=0.83$
- $R^2=0.84$

Input sediment particle size categories:

Large input sediment particle size data

- $k=3995 m/y$
- $C^*=0 mg/L$
- $E=0.91$
- $R^2=0.92$

Medium input sediment particle size data

- $k=8924 m/y$
- $C^*=71 mg/L$
- $E=0.83$
- $R^2=0.85$

Small input sediment particle size data

- $k=4566 m/y$
- $C^*=36 mg/L$
- $E=0.95$
- $R^2=0.95$
Appendix B2: The response surfaces for the two parameters $k$ and $C^*$ in the application of the $k$-$C^*$ model for different laboratory experimental data categories. The objective function is the model efficiency $E$ (dashed lines indicate the best 5% fits).

Vegetation density categories:

Hydraulic loading categories:
Input sediment concentration categories:
Input sediment particle size categories:
Appendix B3: Parameter frequency for the $k$ and $C^*$ parameter sets which have a value of objective function $E$ within the top 5% of the maximum for different data categories.

Vegetation density categories:

Hydraulic loading categories:
Input sediment concentration categories:

- High Input sediment concentration data
- Medium input sediment concentration data
- Low input sediment concentration data

Input sediment particle size categories:

- Large input sediment particle size data
- Medium input sediment particle size data
- Small input sediment particle size data
Appendix C

Field Data Collected from the Ruffey’s Creek Wetland
Appendix C1: The observed storm-event flow and TSS data at four locations (Sites 1-4) in the Ruffey’s Creek Wetland.

Event 4

Event 6

Event 7
Event 8

Event 9

Event 10
Event 11

Date: 15/10-16/10/2003
- S1 - Inflow of the inlet pond
- S2 - Bypass flow
- S4 - Outflow of the macrophyte cell

Event 11 TSS Pollutograph
- S1 - Inflow TSS of the inlet pond
- S2 - Inflow TSS of the ephemeral cell
- S3 - Outflow TSS of the ephemeral cell
- S4 - Outflow TSS of the macrophyte cell

Event 12

Date: 28/10-29/10/2003
- S1 - Inflow of the inlet pond
- S2 - Bypass flow
- S4 - Outflow of the macrophyte cell

Event 12 TSS Pollutograph
- S1 - Inflow TSS of the inlet pond
- S2 - Inflow TSS of the ephemeral cell
- S3 - Outflow TSS of the ephemeral cell
- S4 - Outflow TSS of the macrophyte cell
Appendix D

Application of the MUSIC Model Using Field Data Collected at the Ruffey’s Creek Wetland
Appendix D1: The comparisons of the observed and the modelled outflow hydrographs from the MUSIC model for eight storm events. (*Flow In* refers to the measured inflow rate of the inlet pond, *Flow Out* refers to the modelled outflow rate of the macrophyte cell including the weir overflow rate, and *Observed outflow* refers to the measured outflow rate of the macrophyte cell).
Appendix D2: The response surfaces for the two parameters $k$ and $C^*$ in the application of the MUSIC model to the RCW field data based on the predefined number of CSTRs. The objective function is the model efficiency $E$. 

RCW - MUSIC model - 1 CSTR

RCW - MUSIC model - 2 CSTRs

RCW - MUSIC model - 3 CSTRs

RCW - MUSIC model - 4 CSTRs

RCW - MUSIC model - 5 CSTRs
Appendix D3: The comparisons of the observed and the MUSIC modelled outflow TSS concentrations for eight storm events when $N = 1$, $C^* = 20$ mg/L and $k = 3200$ m/y. (TSS Concentration In refers to the input TSS concentration of the inlet pond, TSS Concentration Out refers to the modelled outflow TSS concentrations of the macrophyte cell, and Observed outflow TSS refers to the observed outflow TSS concentration of the macrophyte cell).
Appendix E

Journal Papers Published from the Project
Modelling wet weather sediment removal by stormwater constructed wetlands: Insights from a laboratory study

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Received 1 November 2006; received in revised form 1 March 2007; accepted 5 March 2007

KEYWORDS
Stormwater;
Constructed wetlands;
Total suspended solids;
Particle size distribution;
Modelling

Summary Constructed wetlands are now commonly used to control polluted urban stormwater discharges. A laboratory study was conducted to investigate the treatment of solids in these systems. Three mesocosm stormwater wetlands (vegetated with a well-established canopy of different densities) and one mesocosm non-vegetated pond were used, all sized to achieve particle fall number ($N_f$, a ratio between the times of the particle travel in horizontal and vertical directions) and Particle Shear Velocity Reynolds Number, $Re^+$, which are reflective of full-scale systems. The mesocosm vegetated systems had also similar turbulent Reynolds Numbers ($Re_T$) to those found in full-scale systems. Ten groups of steady-state experiments were carried out, all with different hydraulic loadings and sediment inflow concentrations (also maintained within the ranges found in real systems during wet weather). Samples were taken along the mesocosms and analysed for Total Suspended Solids concentrations (TSS) and Particle Size Distribution (PSD).

It was found that both $Re^+$ and $Re_T$ do not significantly influence the trapping of sediments, and therefore the particle re-suspension induced by water flow is not important for sedimentation in constructed stormwater wetlands. Vegetation density was found not to be an important factor, while particle diameter, and flow characteristics (e.g., flow rate and velocity) do influence trapping efficiency of particles. It was concluded that sediment trapping correlates strongly with particle fall number, $N_f$, and therefore can be explained by this single non-dimensional number. A simple non-linear two-parameter regression model is proposed for prediction of particle trapping efficiency in constructed stormwater wetlands. However, further work is needed before the method can be used in practice. The aim of the ongoing work is to test whether the proposed model could be used across a number of real stormwater constructed wetlands without any further calibration. The...
data collected from a number of stormwater treatment systems in Melbourne, Australia, will be used in this study. © 2007 Elsevier B.V. All rights reserved.

Introduction

Stormwater is a major source of pollution in urbanized areas (Burton and Pitt, 2002). Constructed ponds and wetlands are widely used for control of stormwater. Ponds are deep open water pools, typically with very limited water level fluctuation, while wetlands are shallow pools, with emergent aquatic macrophytes in a marsh-like configuration. Apart from being effective in removal of sediment and pollutants, they can contribute to flood attenuation and landscape amenity (Dalimer, 2002; Wong and Breen, 2002).

Physical sedimentation is the preliminary mechanism for pollution removal in ponds and wetlands (Reinelt and Horner, 1995; Wong et al., 1999; Walker, 2001; WERF, 2005). Removal of particles also removes the large proportion of many pollutants (e.g., heavy metals, and phosphorus) which are particle-bound (Pitt and Amy, 1973; Kadlec and Knight, 1996). Different pollutants are associated with different particle sizes (e.g., heavy metals are predominantly attached to particles less than 63 µm, Sansalone et al., 1998; Deletic and Orr, 2003), suggesting that the particle size distribution (PSD) of TSS has a significant effect on the removal of associated contaminants (Breen and Lawrence, 2006; WERF, 2005). Effective wetland and pond design therefore relies on being able to reliably predict the removal of particles fractions of sediment found in stormwater.

Reinelt and Horner (1995) reported that pollutant removal from stormwater wetlands was influenced by flow conditions, pollutant source and state, residence time and season. Dense vegetation is generally credited for increasing sedimentation by a combination of reducing turbulence and slowing water velocity (Barko et al., 1991; Sand-Jensen, 1998; Schmid et al., 2005; WERF, 2005) and sediment adhesion (Wong et al., 2000).

However, the importance of vegetation in removal of sediment in wetlands (used for management of non-urban runoff) has been widely disputed. Fennessy et al. (1994) demonstrated that vegetation was not important for sedimentation in a freshwater constructed wetland, being instead a function of hydrologic loading near the inlet. Based on field data on flow in a marsh, Leonard et al. (2002) suggested that difference in vegetation cover had no significant effect on flow regime, sediment transport and deposition patterns. Brueske and Barrett (1994) were more specific, stating that vegetation patterns did not influence the deposition of sediments in a high hydraulic loading wetland (26.07 m/y), but had a significant effect in a low-loading wetland (3.13 m/y).

Braskerud (2001) studied four constructed rural runoff wetlands over time and found that re-suspension had decreased approximately 40% and became negligible five years after the system construction. This is consistent with the assertion by Kadlec and Knight (1996) that in the surface flow treatment wetland environment, physical re-suspension is not a dominant process. However, modelling by Tsanis et al. (1998) showed that the critical shear stress for re-suspension had a great impact on sediment behaviour in a natural wetland.

Although constructed wetlands for treatment of runoff from both urban and non-urban areas have been extensively studied in the field, the separate and combined effects of factors such as hydraulic loading, vegetation density, inflow concentration and sediment re-suspension under wet weather conditions are still not well understood.

Many efforts have been made to model wetland sedimentation (Walker, 2001). Some complex models require Computational Fluid Dynamics (CFD) modelling of water flow and particle transport (e.g., Tsanis et al., 1998). Others describe the movement of water or solids only in one spatial direction or without any spatial resolution (Kutzner et al., 2006). However, the most widely used model is the simple first-order kinetic model (known as k−C model), which has been generally developed for prediction of the long-term average performance of wastewater treatment wetlands and ponds (Kadlec and Knight, 1996). Wong and Geiger (1997) suggested that the two-parameter k−C model could be adapted for application in constructed stormwater wetlands. However, the model parameters, k − rate constant and C − background concentration, were found to vary strongly with hydraulic loading and inlet concentration (Kadlec, 2000), and therefore variable even at one site. A recent study by Kutzner and Geiger (2005) concluded that these problems have resulted in a search for alternative approaches in Germany.

Sediment trapping in ponds and wetlands has also been modelled using the well known settling theory of discrete particles (developed by Hazen, 1904), where particle trapping is a simple linear function of the particle fall number Nt, defined as the ratio of settling time to retention time. The original Hazen model has been widely used for combined sewer overflows (CSO) and waste water treatment systems (Kutzner et al., 2006). The model has been recommended for assessment of solids removal in stormwater ponds and wetlands (WERF, 2005). However, this work does not present any testing of the model against either laboratory or field data. Recently, the Hazen model was modified to allow for an observed non-linearity between the particle trapping efficiency and Nt in stormwater grass strips and swales (Deletic, 2001, 2005). This model has been successfully verified (without any further calibration) on two field stormwater grass filters (Deletic and Fletcher, 2006). It is therefore hypothesised that a similar approach (i.e., the non-linear Nt model) could also be suitable for stormwater wetlands.

The first objective of this study was to identify and assess the most important processes and variables that impact upon sediment behaviour in stormwater wetlands. Due to the complexity of the process (Fennessy et al., 1994), it was decided to start by conducting controlled laboratory experiments in mesocosm stormwater wetlands. The second objective was to test whether the modified Hazen model...
Experimental methodology

Hypotheses

During wet weather, sediment behaviour in ponds and wetlands may include particle deposition, re-suspension of particles from the system's bed, and wash-off of particles that are suspended in the pool of water (Kadlec and Knight, 1996). The following hypotheses were tested in this work:

Hypothesis 1. Re-suspension of particles from the system's bed can be neglected. Usually, re-suspension is regarded to be a function of particle size, $d_v$, the shear velocity, or simply a function of Shear Velocity Reynolds Number, $Re^*:\$

$$\text{(1)}$$

where $U^*$ is the shear velocity (m/s), $\nu$ is the kinematic viscosity (m$^2$/s). The shear velocity, $U^*$, could be found by assuming a logarithmic velocity distribution with water depth (Nakayama, 1999; Marsh et al., 2004):

$$\text{(2)}$$

Hypothesis 2. Deposition of particles is the main process that governs sediment behaviour during wet weather, and can be modelled as a simple function of the particle fall number $N_t$, defined as the ratio of the times of particle travel in the horizontal and vertical directions:

$$\text{(3)}$$

where $x$ is the length of the system (m), $h$ the mean depth of flow (m), $V_s$, the particle settling velocity (m/s) and $V$ the mean flow velocity (m/s) determined by

$$\text{(4)}$$

where $q$ is the flow rate (m$^3$/s), $B_o$ the open flow width (m). If the particle falls as a discrete sphere, the particle settling velocity, $V_s$, is given by the Stokes’ law:

$$\text{(5)}$$

where $\rho_p$ is the particle density (kg/m$^3$), $\rho_w$ the water density (kg/m$^3$), $d_v$ the particle diameter (m), $g$ the gravity acceleration (m/s$^2$), and $\mu$ the dynamic viscosity of water (kg/s/m).

Hypothesis 3. Wash-off of fine particles that never settle in the pool of water (i.e., particles that are in constant suspension) does not play an important role during wet weather. However, for the purpose of testing this hypothesis, wash-off is modelled as a function of particle size and RMS Turbulent Reynolds number, $Re_T$, an indicator of the level of turbulence of the flow in vegetated channels, is given as follows (Tollner et al., 1976):

$$\text{(6)}$$

where $R_s$ is the spacing hydraulic radius (m), defined as

$$\text{(7)}$$

where $W_{ss}$ is the distance between two plant stems (m).

In non-vegetated systems, this number becomes Flow Reynolds Numbers, $Re_T$ defined as

$$\text{(8)}$$

where $R$ is the flow hydraulic radius (m), defined as

$$\text{(9)}$$

where $B$ is the flow width of the treatment system (m).

Scaling issues in physical modelling of stormwater wetlands and ponds

To study the processes in controlled laboratory environment, the ranges of the three non-dimensional numbers that govern the processes, shear velocity Reynolds Number, $Re^*$, particle fall number, $N_t$, and Turbulent Reynolds Number $Re_T$, had to be kept the same as in real systems (this is the main principle of physical hydraulic modelling). Therefore, the task was to determine the ranges of these three numbers for real operational conditions, and to then construct a mesocosm (laboratory) system capable of replicating these values.

Field data collected at Hampton Park wetland, Melbourne (Fletcher et al., 2004; Taylor et al., 2006) were used to roughly estimate these ranges in reality (the size of vegetated and non-vegetated ponds of this wetland are shown in Table 1). Hydraulic loadings of 500, 2000 and 7000 m$/y$ were used to cover the possible inflow range over wet weather (Duncan, 1998), while the flow depths were as measured in the system for the chosen inflow rate. The typical range of particle sizes in the stormwater inflow was assumed to be from 1 to 300 µm, while their density from 2300 to 2750 kg/m$^3$ (Deletic and Orr, 2003). The vegetation densities ranged from 500 to 3000 culms/m$^2$, Table 1 shows the three non-dimensional numbers as found in Hampton Park, vegetated and non-vegetated ponds for particle size of $d_v = 40$ µm (assessed to be median for stormwater in Melbourne, adopted from CRCCH, 2005), particle density of 2520 kg/m$^3$, and vegetation density of 590 culms/m$^2$ (average for Hampton Park).

The size of the mesocosms and the experimental inflow rates were then determined, so that the lab system could mimic $Re^*$, $N_t$ and $Re_T$ found in the field. Unfortunately, the mesocosm sizing was constrained to great extent, by the need to utilize an existing laboratory rig (used in a study of nitrogen removal in wetlands, Taylor, 2007). As shown in Table 1, similarity was achieved for $Re^*$ and $N_t$ in both vegetated and non-vegetated systems if the mesocosm was 1.5 m long, 0.25 m wide and 0.05 m deep. The Turbulence Reynolds Number, $Re_T$, was reasonably similar in the mesocosms and real vegetated wetlands (although somewhat lower at the mesocosm, but still in the range of numbers found in real wetlands). For the non-vegetated mesocosm pond, it was impossible to achieve replication of $Re_T$ (that in this case is the Flow Turbulence Number, $Re$, Eq. (8)).
while at the same time also maintaining similarity for $N_t$ and $Re$. For non-vegetated ponds, it is possible to scale for only two of the three numbers at the same time (e.g., if sediment size and density is also scaled, we could achieve similarity for $N_t$ and $Re$ at the same time, but then could not keep similarity for $Re$). It was therefore decided to build the mesocosms as specified in Table 1, and to concentrate on similarity of $N_t$ and $Re$ since the literature review showed that the processes of deposition and re-suspension should are the most important. The wash-off process could also be studied in the scaled vegetated ponds to some extent (since at least $Re$ is in the same order of magnitude), while to study this process in non-vegetated ponds separate experiments are required (this is planned for a future study).

Experimental installation

The experimental rig cross-section and plan view are shown with all dimensions in Fig. 1. It contained a 10 kl tank with regulated discharge rate, and four experimental cells (flumes that represent wetlands). An emergent macrophyte, *Baumea articulata*, capable of staying upright in a range of hydraulic loadings, was established in the cell in a 0.4 m thick sandy loam layer, with daylight lamps to support plant growth. Each cell had a different density of vegetation: (1) no vegetation, (2) low-density = 590 culms/m$^2$, (3) medium-density = 1620 culms/m$^2$, and (4) high-density = 2936 culms/m$^2$. Removable gates at the outlet end of the cells were installed to adjust the water depth of each cell, targeting a depth of $h = 0.05-0.06$ m. V-notch weirs were installed at the inflow into each cell for accurate measurement of flow (Fig. 1b), while weirs were also constructed at the cell outflows for sample collection.

Sediment collected from a nearby stormwater retarding basin, was sieved through 300 µm sieve and added as slurry of known concentrations directly into the inflow. Four cups were installed in front of each V-notch weir, for continuous manual injection of slurry into inflow. Mixing of inflow water and sediment was helped by funnels that directed inflow onto the cell front wall. Although this substantially reduced the turbulence created at the start of the cell by incoming water (i.e., remove un-realistic laboratory conditions), baffles were also placed close to the inflow for further turbulence reduction, as shown in Fig. 1b.

Tubes for siphoning water samples were installed into the cells at 0.5 m and 1 m from the inlet weirs. It was decided to use the point at 0.5 m from the inflow as the starting point of the cell, allowing the first 0.5 m for the flow mixing (establishment of uniform flow and sediment profile).

Experimental procedure and programme

At the start of each experiment, steady flow was established prior to injection of sediment. The mean flow depth and inflow in each cell was measured for each cell. A known mass of slurry of known TSS concentration was added every 2 min over 2 h for low flows, and over 1.2 h for high flows. Water sampling started only after steady sediment concentrations were established along the cells (determined in a pilot study). Samples were then taken every 15 min for the low flows and 10 min for the high flows. Samples were collected by siphoning from 0.5, 1.0 m from the inflow weirs, and at the outflow weirs (2 m) by simple collection of the outflow.

| Table 1 Hampton Park Wetland characteristics and mesocosm system characteristics for particle size, $d_i = 40$ µm and density, $\rho = 2520$ kg/m$^3$ |
|--------------------------------------------------|-----------------|-----------------|-----------------|
| **Hydraulic loading, $q$ (m$^3$/y)**             | 500             | 2000            | 7000            |
| **Vegetated systems: Vegetation density = 590 culms/m$^2$** |
| Hampton Park Wetland                             | Size: length, width and depth (m) | 30, 77, 0.41    | 30, 77, 0.79    | 30, 77, 0.95    |
| Particle shear velocity Reynolds number $Re$     | 4.2E-03         | 7.5E-03         | 1.8E-02         |
| Particle fall number $N_t$                       | 72              | 18              | 5.2             |
| Turbulent Reynolds number $Re$                   | 23              | 51              | 148             |
| Mesocosm system                                  | Flow rate of each cell (ml/s) | 6               | 24              | 82              |
| Size: length, width and depth (m)                | 1.5, 0.25, 0.05 | 1.5, 0.25, 0.05 | 1.5, 0.25, 0.05 |
| Particle shear velocity Reynolds number $Re$     | 3.3E-04         | 9.5E-03         | 2.6E-02         |
| Particle fall number $N_t$                       | 73              | 18              | 5.2             |
| Turbulent Reynolds number $Re$                   | 7.2             | 29              | 101             |
| Non-vegetated systems                            | Flow rate of each cell (ml/s) | 6               | 24              | 83              |
| Hampton Park Wetland                             | Size: length, width and depth (m) | 120, 70, 1.48   | 120, 70, 1.64   | 120, 70, 2.21   |
| Particle shear velocity Reynolds number $Re$     | 4.10E-03        | 1.20E-02        | 2.50E-02        |
| Particle fall number $N_t$                       | 83              | 21              | 5.9             |
| Turbulent Reynolds number $Re$                   | 2092            | 8322            | 28,618          |
| Mesocosm system                                  | Flow rate of each cell (ml/s) | 6               | 24              | 83              |
| Size: length, width and depth (m)                | 1.5, 0.25, 0.05 | 1.5, 0.25, 0.05 | 1.5, 0.25, 0.05 |
| Particle shear velocity Reynolds Number $Re$     | 3.00E-03        | 8.50E-03        | 2.30E-02        |
| Particle fall number $N_t$                       | 84              | 21              | 6               |
| Turbulent Reynolds number $Re$                   | 17              | 68              | 238             |
Samples were analysed for TSS using the standard method (APHA/AWWA/WPCF, 1998), while Particle Size Distribution (PSD) was determined using a Low Angle Laser Light Scattering method (MasterSizer/E). In addition, particle density was measured using a fully-automatic gas displacement AccuPyc 1330 Pyconometer for a known small amount of dry sample.

Table 2 summarises the experimental variables selected for the experimental programme: two different hydraulic loading rates (i.e., flows), four vegetation densities, and three sediment concentrations at the start of the cell (0.5 m from the inflow). It should be noted that the hydraulic loading rates were on the higher end of their ranges found in real conditions (Duncan, 1998). This was mainly due to the limitations in the existing rig (it was impossible to reliably regulated flows lower than those used), but also because we wanted to test the system for the ‘worst’ conditions. One to three experiments were carried out for each similar combination of the experimental variables (however, it was very difficult to replicate sediment concentrations at the 0.5 m from the inflow weirs due to unpredictable sedimentation upstream from this point). In total 40 experiments were carried out, grouped in five series according to hydraulic loading and inflow concentration: LF–LC, LF–MC, HF–LC, HF–MC, HF–HC (where L stands for low, M for medium, H for high, F for flow, C for concentration, as explained in Table 2).

### Table 2: Wet weather experimental variables and number of experiments done for the each combination of the variables

<table>
<thead>
<tr>
<th>Number of experiments</th>
<th>Vegetation density (culms/m²)</th>
<th>Sediment concentration at 0.5 m from the inflow (mg/l)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>LF = 2000</td>
</tr>
<tr>
<td></td>
<td></td>
<td>High = 2936</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Medium = 1620</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Low = 590</td>
</tr>
<tr>
<td></td>
<td></td>
<td>No vegetation = 0</td>
</tr>
<tr>
<td></td>
<td>HF = 7000</td>
<td>High = 2936</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Medium = 1620</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Low = 590</td>
</tr>
<tr>
<td></td>
<td></td>
<td>No vegetation = 0</td>
</tr>
<tr>
<td></td>
<td></td>
<td>LC = 30–110</td>
</tr>
<tr>
<td></td>
<td></td>
<td>MC = 110–500</td>
</tr>
<tr>
<td></td>
<td></td>
<td>HC = 500–900</td>
</tr>
<tr>
<td></td>
<td></td>
<td>2</td>
</tr>
<tr>
<td></td>
<td></td>
<td>3</td>
</tr>
<tr>
<td></td>
<td></td>
<td>2</td>
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<td></td>
<td></td>
<td>3</td>
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<td></td>
<td></td>
<td>2</td>
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<tr>
<td></td>
<td></td>
<td>3</td>
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<td></td>
<td>1</td>
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<td></td>
<td>1</td>
</tr>
<tr>
<td></td>
<td></td>
<td>1</td>
</tr>
<tr>
<td></td>
<td></td>
<td>3</td>
</tr>
<tr>
<td></td>
<td></td>
<td>1</td>
</tr>
<tr>
<td></td>
<td></td>
<td>1</td>
</tr>
<tr>
<td></td>
<td></td>
<td>3</td>
</tr>
</tbody>
</table>

Figure 1 Laboratory experiment rig: (a) plan view and (b) cross-section.
The median particle size, $d_{50}$, of samples collected at the cell starting point (0.5 m from the inflow) varied from 20 to 56 µm, and therefore was within the natural range of stormwater sediment (the PSD curves at 0.5 m were very close to the possible PSD of developing catchments in Melbourne and Brisbane, CRCCH, 2005, and to the PSD measured in stormwater in Scotland, Deletic and Orr, 2003). Similarly, the density of the particles was within 2490–2540 kg/cm$^3$ (similar to the values found in field, Deletic and Orr, 2003).

**Measurement uncertainties**

The accuracy of TSS measurements depended on the accuracy of the analytical method used (the standard method, APHA/AWWA/WPCF, 1998) and the uniformity of the sediment concentrations across a cell profile (since some samples were siphoned from a single point in the cross-section).

Analytical method uncertainties were assessed using standard laboratory protocol (Hines, 2005). It was found that this source of uncertainty was ±3.8% of the measured concentration. However, the uncertainty due to the fact that some samples have been extracted from a single point in the cross-sections (always in the centre of the profile) had to be investigated independently. For five different combinations of the experimental variables (LF–LC, LF–MC, HF–LC, HF–MC, HF–HC), and each cell, samples were siphoned at the same time from five points across each of the profiles at 0.5 m and 1 m from the inflow weirs. They were analysed for TSS, and the mean profile TSS was calculated. For all cases and the point 1 m from the inflow, the difference between the middle point TSS and the mean profile TSS was always below 10%. For 0.5 m from the inflow, this difference was less than 15% except for the cell with the high-vegetation density, where it was around 23% (for all five experiments the middle point had around 23% higher TSS values than the means). This may be because the vegetation upstream from the 0.5 m point was creating a certain flow pattern that consistently produced this effect.

It was concluded that for all but the cell with high vegetation density at 0.5 m from the inflow, the siphoning from the mid point in profiles was adequately accurate. However, it was decided that for the high vegetation cell at 0.5 m point, the measured concentrations had to be adjusted to 77% of the measured.

Particle sizes were measured using MasterSizer/E, based on the Mie theory of laser light scattering due to the presence of particles in water. This theory assumes that particles are spherical, and therefore the method has some built-in inaccuracies (although the manufacturer claims ±2% accuracy on volume median diameter, Malvern, 1990). To cover the size range of all particles present in samples (0–400 µm), the middle measuring range of the instrument was used. This might have caused a significant error in measurements of particle below 2 µm. However, to maximize the accuracy of measurements, at least three readings were taken for one sample, and the mean PSD was calculated.

**Determination of significant variables**

Changes in TSS and PSD along the cells were observed by plotting the experimental data versus time. Mean TSS concentrations and PSDs were calculated for each measurement point over each experiment. Variances around the TSS means were also calculated to evaluate whether steady-state conditions were achieved during experiments.

The measured mean TSS and PSD at each point were used to calculate concentrations of five different particle size fractions: 0–6, 6–21, 21–46, 46–124 and 124–404 µm. The trapping efficiency of fraction $s$, $T_{rs}$, was calculated for three observed sections of the cells: (1) between 0.5 m and 1 m, (2) 0.5 and 2 m, and (3) 1 and 2 m from the inlet weir, using the following equation:

$$T_{rs} = (C_{in,s} - C_{out,s})/C_{in,s}$$

where $C_{in,s}$ is input concentration of fraction $s$ into the section (mg/l), $C_{out,s}$ is output concentration of fraction $s$ into the section (mg/l). In total 465 values of $T_{rs}$ have been calculated.

Relationships between trapping efficiency and factors which may influence it were examined: hydraulic loading, sediment particle size, flow depth, flow velocity, shear velocity, retention time, inflow concentration, plant density, as well as the three non-dimensional numbers: $Re_T$, $N_T$, and $Re_T$. Analysis was undertaken by calculating Spearman correlation coefficients, $r_s$, and corresponding $p$-values (Hinton et al., 2004). The importance of the re-suspension process was also analysed using the Modified Shields Diagram commonly used in the analysis of sediment mobilisation in flow (Govers, 1987).

**Experiment results and discussion**

**General trends**

The change in TSS over the duration of Experiment 9 for LF–MC Experiment (Table 2) is presented in Fig. 2 for two cells: one with highest vegetation density and one with no vegetation. It is clear that steady-state concentrations have been achieved (this was observed in all experiments), as shown by the variances of the mean TSS being within 3–18% of the mean for all experiments. It is also clear that TSS decreased along the cells, showing that deposition is an important process.

An example of the measured cumulative PSD curves at different sampling points and times is presented in Fig. 3, confirming the attainment of steady-state conditions (variations in PSD with time at any point along the cells are small). However, the cumulated PSD curves shifted to the left with the distance, showing a decrease in particle sizes along the cell. As expected, along the cell, the larger particles were removed at higher rate than smaller particles, and the median particle size $d_{50}$ gradually decreased with distance.

Since steady-state conditions have been achieved, it was safe to average measured TSS and PSD over each experiment and then use the mean concentrations (of the particle size fractions and total TSS) in further analyses. For example, the mean fraction concentrations for Experiment 9 and the high-vegetated cell were plotted against the distance in Fig. 4. It is clear that there is an exponential decrease of concentrations along the flume (it follows the first-order decay model), with a decay coefficient being highly dependent on particle size. These findings are very consistent with previous studies (Kadlec and Knight, 1996; Wong et al., 2006).
Influential variables

Table 3 presents the results of the Spearman correlation analyses between the trapping efficiency, $T_r$, and explanatory variables. In this table, the variables are listed according to the descending absolute $r_s$ value, in other words, a decreasing importance of the variable for the particle trapping process. For example, particle fall number (No. 1) has $r_s = 0.92$ and therefore explains 92% of the variation in observed TSS concentrations, while vegetation density (No. 13) does not influence $T_r$ ($r_s = -0.01$). The low $p$-values observed for all correlations but vegetation density is in part a function of the large number of points used in the analysis (465). Results from non-vegetated cell need to be regarded with some caution since typical values of $Re$ were not replicated in the mesocosm. The most important findings from this analysis are summarised below:
Table 3 Spearman correlation statistics between the trapping efficiency, $T_r$, and the tested variables based on 465 measured data points

<table>
<thead>
<tr>
<th>No.</th>
<th>Variable</th>
<th>Correlation coefficient, $r_s$</th>
<th>$p$-Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Particle fall number, $N_f$</td>
<td>0.92</td>
<td>0.000</td>
</tr>
<tr>
<td>2</td>
<td>Particle fall velocity</td>
<td>0.86</td>
<td>0.000</td>
</tr>
<tr>
<td>3</td>
<td>Particle size at the input</td>
<td>0.86</td>
<td>0.000</td>
</tr>
<tr>
<td>4</td>
<td>Particle shear velocity Reynolds number $Re^*$</td>
<td>0.77</td>
<td>0.000</td>
</tr>
<tr>
<td>5</td>
<td>Hydraulic loading</td>
<td>-0.31</td>
<td>0.000</td>
</tr>
<tr>
<td>6</td>
<td>Detention time</td>
<td>0.28</td>
<td>0.000</td>
</tr>
<tr>
<td>7</td>
<td>Input sediment concentration</td>
<td>-0.28</td>
<td>0.000</td>
</tr>
<tr>
<td>8</td>
<td>Flow rate</td>
<td>-0.19</td>
<td>0.000</td>
</tr>
<tr>
<td>9</td>
<td>Water depth</td>
<td>-0.18</td>
<td>0.000</td>
</tr>
<tr>
<td>10</td>
<td>Flow velocity</td>
<td>-0.16</td>
<td>0.001</td>
</tr>
<tr>
<td>11</td>
<td>Shear velocity</td>
<td>-0.15</td>
<td>0.001</td>
</tr>
<tr>
<td>12</td>
<td>Turbulent Reynolds number, $Re_T$</td>
<td>-0.12</td>
<td>0.008</td>
</tr>
<tr>
<td>13</td>
<td>Vegetation density</td>
<td>-0.01</td>
<td>0.835</td>
</tr>
</tbody>
</table>

- Vegetation density does not play any role in the total sediment deposition in wetlands during wet weather. As discussed in the introduction, this has been found previously for freshwater wetlands (Bruske and Barrett, 1994; Fennessy et al., 1994; Leonard et al., 2002), but has been disputed for stormwater wetlands (Wong et al., 2000; WERF, 2005) and constructed wetland ponds (Schmid et al., 2005). Some studies suggested that an increase in vegetation density may reduce the finer particles more efficiently over the long term due to filtration and other physical, chemical and biological processes (Breen, 1990; WERF, 2005). Unfortunately, we were not able to quantify the effect of vegetation on these very fine particles, because they presented very small fraction of the TSS and therefore the experimental error prevented us from assessing accurately their concentrations.

- Flow characteristics, such as flow velocity, depth, and rate have some influence, with lower $T_r$ for higher flows. Finally, hydraulic loading (a combined measure of flow rate and length available for deposition), has a relatively weakly significant influence ($r_s = -0.31$), with a similar influence of detention time (the ratio between pool volume and flow rate).

- The input concentration has a relatively small influence ($r_s = -0.28$) on sediment trapping efficiency, $T_r$, despite the predictions in earlier studies (Duncan, 1998; WERF, 2005).

- Particle size is the most important single variable, explaining 86% of the variance in observed TSS concentration.

- Of the three non-dimensional numbers studied, $Re^*$ had no significant influence while $Re_T$ and $N_f$ were both highly significant.

Tested hypotheses

**Hypothesis 1:** The importance of $Re^*$ for the Trapping Efficiency ($r_s = 0.77$) could be entirely due to the importance of particle size ($r_s = 0.86$), because the shear flow velocity, $U_s$, was not very significant ($r_s = -0.15$). Since particle size is very important for the deposition process, conclusions about the relevance of the re-suspension process could not be easily reached. It was therefore decided to evaluate the importance of this process further, by using the general sediment transport theory and in particular Shield’s diagram (Govers, 1987; Yang, 1996).

To broaden the scope of the lab study, $Re^*$ was calculated for operational conditions for which wetlands are typically designed in Australian practice. According to current Australian design guidelines the design mean flow velocity, $V$, is below 0.05 m/s, while during wet weather it can range between 0.02 m/s and 0.1 m/s (Wong and Breen, 2002). The water depth, $h$, in wetlands is typically between 0.15 m and 0.75 m, while for ponds it is between 1 m and 2 m (Kadlec, 2005). Using these conditions, $Re^*$ values were determined for a range of stormwater sediment particles (the same range as used in the scaling of the mesocosms). These values were then presented against their matching densimetric Froude number $F_r = \frac{U_s^2}{g_d h}$, as shown in Fig. 5. It is clear that almost all calculated points were below the modified Shield’s critical curve, with only points that correspond to the particles of 1 µm and the highest allowed velocity (0.1 m/s) being above the critical curve. This clearly indicates that re-suspension is unlikely to be a governing process in real wetlands.

From both the lab study and the Shield’s diagram it was concluded that re-suspension of particles from the system’s bed can be neglected, even for very high flow rates that could occur in well designed wetlands during wet weather. However, it is possible that re-suspension could important in wetlands and ponds with no bypass for large floods (i.e., where velocities are uncontrolled). In practice a variety of other factors than flow can cause sediment re-suspension in wetlands, such as wind-driven turbulence, water level fluctuations (e.g., drawdown), animals, and gas release from the bed (unfortunately it is very hard to model these activities in a laboratory system).

**Hypothesis 3:** The wash-off process appears not to play a big role during wet weather since flow turbulence does not significantly influence sediment transport (i.e., Turbulent Reynolds Number $Re_T$ explained only 12% of variance in TSS concentrations). This supports our third hypothesis,
Hypothesis 2: Deposition is the main process that governs sediment behaviour in wetlands. The particle fall number, $N_f$, had the far highest $r_i$ of all studied variables (0.92). This is in full agreement with the Hazen settling theory (Hazen, 1904), as well as findings from a similar laboratory study on stormwater sediment behaviour in grass filters (Deletic, 2005).

Modelling methodology

Based on the main conclusions from the experimental programme, as well as on previous work done on modelling of sediment behaviour in grass filters (Deletic, 2001, 2005), the following non-linear relationship between the particle trapping efficiency, $T_{re}$, and the particle fall number, $N_f$, was proposed:

$$T_{re} = \frac{N_f^a}{N_f^b}$$  \hspace{1cm} (11)

where $a$ and $b$ are regression parameters. This regression model was calibrated against all available measured data (465 points) using the Sum of Squared Errors as the objective function. The correlation coefficient between the measured and modelled points was also calculated.

Sensitivity of the model to its parameters was then investigated by calibrating the model independently for a group of experiments. These groups included sets of experiments with identical hydraulic loadings, similar inflow concentrations, as well as groups of six randomly selected experiments. The coefficients derived from the calibration for each of the groups were compared and the mean and variance of the two model parameters ($a$ and $b$) calculated to assess model sensitivity against the experimental variables (i.e., hydraulic loading rates or inflow sediment concentrations).

Finally, the total TSS concentrations were modelled, assuming that particles behave independently from each other (no interaction between the fractions) and that PSD of stormwater can be represented by five fractions. The calibrated two-parameter non-linear $N_f$ model (Eq. (11)) was applied to five fractions of the following median diameter: $d_{50}$ = 3, 13.5, 33.5, 85, and 264 µm, and their concentrations calculated along the cells. The concentrations of the fractions were then summed to determine the total TSS. The reliability of the model was assessed using the coefficient of efficiency, $E$, (Nash and Sutcliffe, 1970). The modelled TSS was also plotted against the measured TSS including 25% and 50% error bands.

Modelling results and discussion

Using all available data the model (Eq. (11)) was calibrated and is presented along with the measured data in Fig. 6. The values of the regression parameters were: $a = 0.43$ and $b = 1.42$. Both the model efficiency $E$ and cross-correlation coefficient $R^2$ between measured and modelled were 0.82 (with $p < 0.01$).

For different flow rates, input sediment concentrations or any six random experiments, the best fits for the model coefficients $a$ and $b$ were listed in Table 4 that presents the model coefficients $a$ and $b$ along with the model efficiencies $E$ and correlation coefficients for all experimental runs, and for groups of runs (with similar flow, similar concentration, or randomly selected). The coefficient of variation was 4.2% and 5.5% of the means for $a$ and $b$, respectively, with $R^2$ always greater than 0.8. Therefore, the regression parameters can be regarded as independent from the key variables (e.g., inflow rates, inflow sediment concentrations, etc.).

It should be noted that the spread of the observed data around the model (Fig. 6) was higher for the smallest particle size fraction (below 6 µm). This is likely due to the proportionally high uncertainty in PSD measurements for small particles. The model slightly underestimated $T_{re}$ of the biggest fraction (124–404 µm), which may also be due to uncertainty associated with measuring the typically very low concentrations of large particles.

![Figure 5](image-url)  
Modelled shear velocity Reynolds number $Re_s$ versus densimetric Froude number $Fr_d$ at stormwater constructed wetlands and ponds for $\rho_s = 2500$ kg/m³.
The predicted TSS concentrations are very consistent with the observed values (Fig. 7). $E$ and $R^2$ coefficients were 0.91, 0.92, respectively, with all modelled concentrations being well within ±50%, and most within ±25% range of the observed values.

It should be noted that the optimum model parameters ($a = 0.43$ and $b = 1.42$) are different from the corresponding values found for grass swales and filter strips ($a = 0.69$ and $b = 4.95$, Deletic, 2005). This indicates that the proposed model (Eq. (11)) may be universally used for different stormwater treatment systems (e.g., wetlands, swales and filter strips), but with the model parameters being dependent on the system type. However, if further research can show that the parameters are constant for any stormwater constructed wetland, as was the case for swales and filter strips (Deletic and Fletcher, 2006), then the model could be confidently used for wetland design.

### Table 4 Model coefficients $a$ and $b$ and the statistics of the model fit

<table>
<thead>
<tr>
<th>Experiment no.</th>
<th>Model coefficients</th>
<th>$E$</th>
<th>$R^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>All: 1,2,3,4,5,6,7,8,9,10</td>
<td>$a$ = 0.43, $b$ = 1.42</td>
<td>$E$ = 0.82</td>
<td>$R^2$ = 0.82</td>
</tr>
<tr>
<td>Low flow: 1,2,7,8,9</td>
<td>$a$ = 0.44, $b$ = 1.32</td>
<td>$E$ = 0.82</td>
<td>$R^2$ = 0.83</td>
</tr>
<tr>
<td>High flow: 3,4,5,6,10</td>
<td>$a$ = 0.41, $b$ = 1.49</td>
<td>$E$ = 0.81</td>
<td>$R^2$ = 0.81</td>
</tr>
<tr>
<td>Low conc.—medium conc.: 1,2,5,7,8,10</td>
<td>$a$ = 0.44, $b$ = 1.45</td>
<td>$E$ = 0.82</td>
<td>$R^2$ = 0.84</td>
</tr>
<tr>
<td>Medium conc.: 2,5,8,9</td>
<td>$a$ = 0.44, $b$ = 1.43</td>
<td>$E$ = 0.83</td>
<td>$R^2$ = 0.83</td>
</tr>
<tr>
<td>Medium conc.—high conc.: 3,4,5,6,8,9</td>
<td>$a$ = 0.43, $b$ = 1.39</td>
<td>$E$ = 0.83</td>
<td>$R^2$ = 0.83</td>
</tr>
<tr>
<td>Random experiments no: 1,2,5,7,9,10</td>
<td>$a$ = 0.43, $b$ = 1.40</td>
<td>$E$ = 0.81</td>
<td>$R^2$ = 0.82</td>
</tr>
<tr>
<td>Random experiments no: 1,3,6,7,8,10</td>
<td>$a$ = 0.42, $b$ = 1.30</td>
<td>$E$ = 0.81</td>
<td>$R^2$ = 0.83</td>
</tr>
<tr>
<td>Random experiments no: 1,2,5,6,7,9</td>
<td>$a$ = 0.46, $b$ = 1.48</td>
<td>$E$ = 0.84</td>
<td>$R^2$ = 0.85</td>
</tr>
<tr>
<td>Random experiments no: 1,3,4,5,7,8</td>
<td>$a$ = 0.45, $b$ = 1.49</td>
<td>$E$ = 0.84</td>
<td>$R^2$ = 0.85</td>
</tr>
<tr>
<td>Random experiments no: 1,3,4,6,7,9</td>
<td>$a$ = 0.45, $b$ = 1.39</td>
<td>$E$ = 0.83</td>
<td>$R^2$ = 0.83</td>
</tr>
<tr>
<td>Random experiments no: 1,2,4,7,8,10</td>
<td>$a$ = 0.42, $b$ = 1.40</td>
<td>$E$ = 0.81</td>
<td>$R^2$ = 0.81</td>
</tr>
<tr>
<td>Random experiments no: 1,3,4,6,7,8</td>
<td>$a$ = 0.45, $b$ = 1.44</td>
<td>$E$ = 0.84</td>
<td>$R^2$ = 0.84</td>
</tr>
<tr>
<td>Random experiments no: 1,4,5,7,9,10</td>
<td>$a$ = 0.42, $b$ = 1.31</td>
<td>$E$ = 0.81</td>
<td>$R^2$ = 0.82</td>
</tr>
</tbody>
</table>

| Coefficient of variation (%) | 4.16 | 5.53 | 1.64 | 1.61 |

Figure 6 The fitting curve based upon Eq. (11) for experimental results at five different ranges of particle sizes.

Figure 7 Predicted versus observed mass of sediment for all experiments.
Conclusions

Four mesocosm wetlands (with different vegetation densities) were used to examine sediment behaviour in wet weather, aiming to replicate typical real-world particle fall number, $N_f$, Particle Shear Velocity Reynolds Number, $Re^*$, and Turbulent Reynolds Number, $Re_T$.

It was confirmed that the sediment concentration decreases exponentially with distance, with the decay coefficient being highly dependent on particle size. Particle trapping efficiency was not dependent on density of vegetation, flow turbulence ($Re_T$) or shear flow velocity. Flow characteristics did have a significant influence (sediment removal decreased with increases in hydraulic loading), as did inflow concentration to some extent (which was not expected). However, the particle size (or settling velocity) was found, not surprisingly, to have the greatest influence. Finally, it was concluded that a single parameter is capable of explaining the process. This particle fall number is strongly correlated with the particle trapping efficiency. Therefore it was concluded that re-suspension of bed sediment, and wash-off of suspended particles can be neglected, with the most important process being deposition.

A simple two-parameter non-linear regression was developed for the prediction of particle trapping efficiency. The model parameters were insensitive to hydraulic loading rate, inflow sediment concentration, and sediment size. However, the model was not very reliable for very fine particles (less than 6 µm). The proposed method for calculation of TSS had relatively high accuracy with majority of concentrations predicted within ±25% of observed values.

However, further work is needed before the method can be used in practice. The aim of the ongoing work is to test whether the proposed model could be used across a number of real stormwater wetlands without any further calibration. The data collected from a number of stormwater treatment systems in Melbourne, Australia, will be used in this study. It should be emphasized that the model presented here attributes our findings for dominant physical removal of settling particles. In practice, the designers should combine it with chemical and biological processes to achieve the maximum ability of removal of pollutants for a constructed stormwater wetland.

References


