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Simulating short-circuiting flow in a constructed wetland: the implications of bathymetry and vegetation effects

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Abstract:

Short-circuiting flow, commonly experienced in many constructed wetlands, reduces hydraulic retention times in unit wetland cells and decreases the treatment efficiency. A two-dimensional (2-D), physically based, distributed modelling approach was used to systematically address the effects of bathymetry and vegetation on short-circuiting flow, which previously have been neglected or lumped in one-dimensional wetland flow models. In this study, a 2-D transient hydrodynamics with advection-dispersion model was developed using MIKE 21 and calibrated with bromide tracer data collected at the Orlando Easterly Wetland Cell 7. The estimated topographic difference between short-circuiting flow zone and adjacent area ranged from 0·3 to 0·8 m. A range of the Manning roughness coefficient at the short-circuiting flow zone was estimated (0·022–0·045 s $m^{-1/3}$). Sensitivity analysis of topographical and vegetative heterogeneity deduced during model calibration shows that relic ditches or other ditch-shaped landforms and the associated sparse vegetation along the main flow direction intensify the short-circuiting pattern, considerably affecting 2-D solute transport simulation. In terms of hydraulic efficiency, this study indicates that the bathymetry effect on short-circuiting flow is more important than the vegetation effect. Copyright © 2009 John Wiley & Sons, Ltd.

KEY WORDS constructed wetland; short-circuiting; hydraulic efficiency; hydrodynamic modeling; bathymetry; vegetation

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INTRODUCTION

Constructed wetlands are increasingly being used to facilitate nutrient removal from conventional wastewater treatment plant effluents and agricultural runoffs (Hammer, 1989; Brix, 1994; Kadlec and Knight, 1996; Chimney and Goforth, 2001; Dierberg *et al.*, 2002). This is a result of the acknowledgment, development, and application of the natural ability of wetlands to transform and store organic matter and nutrients among the newly-recognized multiple functions and values of wetlands, which once were regarded as wastelands (Mitsch and Gosselink, 1993; Brix, 1994). In addition, these extensive applications of the wetland function of water quality improvement are attributed to the fact that using constructed wetlands may be one of the most ecologically friendly and cost-effective technologies for removing nutrients, primarily nitrogen and phosphorus (Goforth, 2001).

Solutes, including tracers, nutrients, and pollutants, are transported through primary flow paths in the main channel of a wetland and have limited contact with stagnant water areas along the banks, in dense emergent vegetation, or in the subsurface hyporheic zone (Keefe *et al.*, 2004). To enhance the efficiency of a treatment wetland, a systematic understanding of various factors affecting the solute transport or retention is required. Wetland hydraulics is one of the most fundamental and significant factors because the hydraulic efficiency is directly related to the treatment efficiency of constructed wetlands (Martinez and Wise, 2003b).

Flow is the most critical factor sustaining the structure and function of various treatment wetland systems (Hammer, 1989; Arnold *et al.*, 2001). In addition, to effectively simulate the removal or retention of a variety of pollutants in constructed wetlands, an understanding of the hydraulic characteristics controlled by the physical settings, such as shape, bathymetry, slope, the existence of hydraulic structures and their types, and vegetation, is essential because the flow basically determines the availability of pollutants for assimilation by biota and sorption by soils. The residence time distribution (RTD) determined by tracer experiment in a wetland has been used as an important tool to characterize the wetland hydraulics (Kadlec and Knight, 1996; Persson, 2000; Werner and Kadlec, 2000; Holland *et al.*, 2004). Specifically, through moment analysis of an RTD, several parameters for measuring and comparing the hydraulic performances of various constructed wetlands may be determined (Persson *et al.*, 1999; Martinez and Wise, 2003b).

In general, naturally formed or man-made low elevation areas exist in a wetland, even within artificially constructed wetlands. Water flow interconnects and canalizes the low elevation zones. Those areas show greater water depth and less vegetation density (less hydraulic resistance) than the adjacent shallow-depth,

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flat-bottomed, vegetative areas, causing hydraulic short-circuiting (Dierberg *et al.*, 2005). This reduces the hydraulic retention time in a treatment cell and leaves large isolated stagnant areas only tangentially available to the incoming water, thus decreasing the treatment efficiency. Hence, short-circuiting has been recognized as one of the biggest obstacles to successful treatment wetland design and management (Persson, 2000). Several researchers have reported factors affecting the short-circuiting flow in constructed wetlands (Thackston *et al.*, 1987; Kadlec and Knight, 1996; Persson *et al.*, 1999; Persson, 2000; Koskiaho, 2003; Persson and Wittgren, 2003; Holland *et al.*, 2004; Dierberg *et al.*, 2005; Jenkins and Greenway, 2005; Wörman and Kronnäs, 2005; Bracho *et al.*, 2006) as follows:

- Wetland shape or the length-to-width ratio
- Wetland depth or water level
- Wetland flow rate
- Wetland bathymetry or basin morphology
- Wetland vegetation: type, density, and spatial distribution
- Wetland hydraulic structures: type and location
- Wetland internal structures such as subsurface berms and islands
- Wind.

In this study, we focus on the impact of topographic and vegetative heterogeneity on the simulation of short-circuiting flow zone through the modification of those factors. One of the reasons that these two factors were chosen is that it is usually difficult to obtain these data in the field directly; as a result, few studies on these, compared with the other factors, have been reported based on actual wetland systems, rather than hypothetical ones. In addition, they are the most critical components for the application of multi-dimensional flow dynamics models in a constructed wetland with a given shape and hydraulic structure.

Two-dimensional (2-D), physically based, distributed hydrodynamics and solute transport models are often used to obtain valuable insights into the hydrodynamic behaviour of water systems (Persson *et al.*, 1999). This is due to the fact that they allow researchers to estimate the effect of each factor separately based on realistic flow conditions. In the present work, MIKE 21 is used for the simulation model. Each impact was estimated and compared in terms of hydraulic efficiency as defined by Persson *et al.* (1999), which is based on breakthrough curves (BTCs) generated by each simulation.

Therefore, the main objective of this study is to estimate bathymetry and the associated hydraulic resistance favourable to short-circuiting flow; in other words, to examine the quantitative difference between the short-circuiting flow zone and adjacent areas through manipulations in the 2-D, spatially distributed modelling framework (MIKE 21). The primary interest of this paper is not on the development of a 2-D, physically based, fully distributed model, which is precisely calibrated and validated with field measurement, but on the derivation of useful implications obtained through model calibration and sensitivity analysis of the effects of bathymetry and vegetation density (Manning roughness coefficient) on short-circuiting flow in constructed wetlands. The results of this modelling work will also help identify areas where further study is needed in terms of the resolution and scale of spatial data to be obtained in developing a more robust 2-D, spatially distributed hydrodynamics and transport model in constructed wetlands or applying it to other wetland ecosystems, in particular, ridge and slough landscape in the Florida Everglades.

METHODS

Moment analysis of RTD and hydraulic performance

Key parameters used to characterize the hydraulic performance of a wetland are derived from the moment analysis of an RTD (Kadlec and Knight, 1996; Martinez and Wise, 2003b; Holland *et al.*, 2004). The zeroth absolute moment (M_o) of an RTD is equivalent to the total mass of tracer recovered and the first normalized absolute moment (μ_1) yields the centroid of the RTD, which is the mean residence time, $\bar{t}$ (days). The second normalized central moment (μ_2) is the variance, σ^2 (days^2), of the RTD, which accounts for the spreading. These moments can be used to describe the outlet response of a wetland system to a Dirac input function for a tracer (Sardin *et al.*, 1991). However, if the input boundary condition is a continually changing time series, the moments can be formulated under the assumption that the system behaves linearly (Das and Kluitenberg, 1996) as below:

$$\mathrm{MRF} = \frac{\int_0^\infty Q_{\mathrm{eff}}(t)C_{\mathrm{eff}}(t)dt}{\int_0^\infty Q_{\mathrm{inf}}(t)C_{\mathrm{inf}}(t)dt} \quad (1)$$

$$\mu_n = \mu_n^{\mathrm{eff}} - \mu_n^{\mathrm{inf}} \ (n = 1 \text{ and } 2) \quad (2)$$

where $Q_{\mathrm{eff}}(t)$ and $Q_{\mathrm{inf}}(t)$ are the volumetric flow rates (m^3 s^{-1}) of effluent and influent, respectively; $C_{\mathrm{eff}}(t)$ and $C_{\mathrm{inf}}(t)$ are the tracer concentration (mg L^{-1}) of effluent and influent, respectively; and μ_n^{eff} and μ_n^{inf} are the first ($n = 1$) or second ($n = 2$) normalized moment of effluent and influent, respectively. Mass recovery fraction (MRF) is defined in Equation (1) as the ratio of tracer mass recovered at the outlet point to the total mass of tracer injected. Equation (2) serves as a general formula for calculating μ_1 and μ_2 for wetland systems with arbitrary input boundary conditions. In this case, the mean residence time is considered as a time interval between each centroid of inlet and outlet RTD curves. Equations (1) and (2) were used for the moment analysis in this study.

In general, the nominal residence time, τ (days), is defined as

$$\tau = \frac{V_{AVG}}{Q_{AVG}} \quad (3)$$

where V_{AVG} is the volume of a wetland cell (m^3) and Q_{AVG} is the average cell flow (m^3 day^{-1}). In Martinez and Wise (2003b), hydraulic efficiency was defined as a ratio of experimentally determined mean residence time ($\bar{t}$) to nominal residence time (τ), which was suggested as the effective volume ratio (e) by Thackston *et al.* (1987).

$$e = \frac{\bar{t}}{\tau} \quad (4)$$

Persson *et al.* (1999) proposed an alternative metric for hydraulic efficiency (λ) that combined existing metrics of flow uniformity ($1-1/N$) and effective volume (e):

$$\lambda = e\left(1 - \frac{1}{N}\right) = \left(\frac{\bar{t}}{\tau}\right)\left(1 - \frac{(\bar{t} - t_p)}{\bar{t}}\right) = \frac{t_p}{\tau} \quad (5)$$

where N is the number of continuous stirred tank reactors (CSTRs) in series required to simulate flow conditions and t_p is the peak time for a BTC. The number of CSTRs in series (N) is calculated using the difference in the time-to-peak concentration and the mean residence time as follows: $N = \bar{t}/(\bar{t} - t_p)$ (Kadlec and Knight, 1996).

For a detailed description of the RTD moment analysis or the parameters for measuring hydraulic performance, the reader is referred to Levenspiel (1972), Kadlec and Knight (1996), Werner and Kadlec (1996), and Persson *et al.* (1999).

Site description

The study area, the Orlando easterly wetland (OEW), is located east of the City of Orlando and about 3 km west of the St Johns River in central Florida (Figure 1). It is one of the older and larger constructed treatment wetlands in the USA (Black and Wise, 2003; Martinez and Wise, 2003b). Since 1987, it has been used to reduce nutrient loading (nitrogen and phosphorus) of the treated wastewater discharged to the St Johns River (Kadlec and Newman, 1992). Figure 1 shows the plan view of the northern part of the entire wetland system, including seven compartmentalized cells (and a downstream lake) divided by earthen berms. The inflow is typically distributed into three flow paths of sequential cells at influent structure (closed circles) and the flow control in each cell is achieved through an adjustable, sharp-crested rectangular weir (closed rectangles).

For a detailed 2-D hydrodynamic and solute transport modeling, Cell 7, located in the middle of the northern flow train was selected. The cell has a topographic slope of approximately 0·2% along the main flow direction from influent weir (3-X) to the effluent weir (7-X) and the average drop in bed elevation across the cell is about 0·6 m. Martinez and Wise (2003b) reported that site grading was not conducted in the interior of the wetland cells during the conversion of the site from pasture land for cattle operations to treatment wetland.

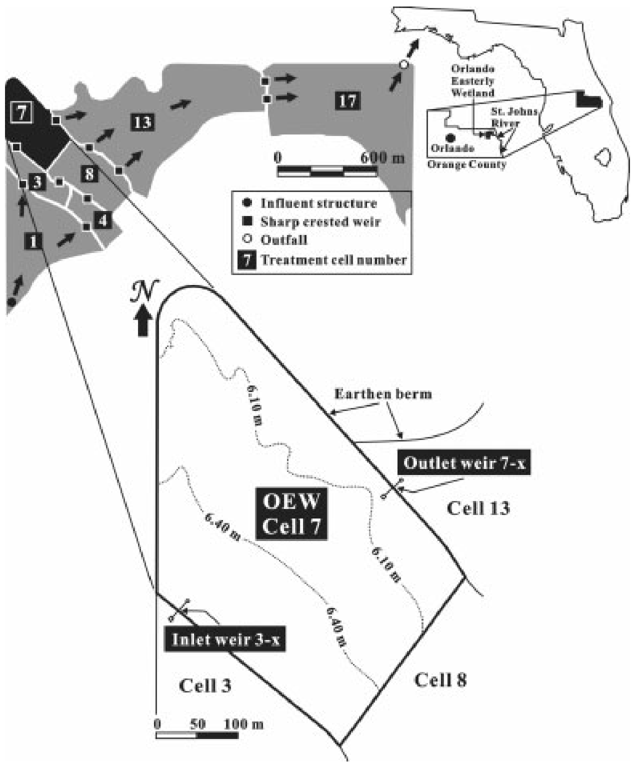

Figure 1. Location and plan view of study area, the Orlando Easterly Wetland (OEW) Cell 7

Artificial landforms, such as ditches and roads, still exist within the cells. Cell 7 is a wet prairie of which the dominant vegetation is cattail (*Typha* spp.) and bulrush (*Scirpus* spp.) (Martinez and Wise, 2003b).

Previous research at the OEW

Bromide tracer test and hydraulic analysis. Martinez and Wise (2003b) conducted bromide tracer experiments on the OEW. From RTD-moment analyses of the treatment cells, the hydraulic efficiencies of the individual treatment cells were estimated; the entire wetland system was operating at near 50% efficiency during the test period. The existence of preferential flow paths resulting in short-circuiting flow and the formation of dead zones was proposed as the main factor causing these low hydraulic efficiencies within the OEW treatment cells. For a detailed description of the bromide tracer tests, such as tracer selection, injection, sampling, and analysis, as well as the results, see Martinez and Wise (2003b).

1-D transient storage modelling study. Martinez and Wise (2003a) reported the results of a one-dimensional (1-D) transport model with inflow and storage (OTIS) that was applied to simulate the results of the OEW bromide tracer experiment. The transient storage model consists of a main channel described by an advection–dispersion equation with completely mixed storage zones and has been applied mainly to river or stream systems where a 1-D approach is reasonable (Bencala and Walters, 1983; Harvey *et al.*, 1996). Despite its usefulness and conceptual simplicity, the model has several limitations. As

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the authors wrote, it cannot account for the spatial distribution of multiple parallel flow paths and can only provide lumped parameters, such as main channel and storage zone cross-sectional areas and an exchange coefficient between the main channel and storage zone, of which the physical meaning is not clear. This shows that a multi-dimensional, physically based, distributed modelling approach is desirable to overcome the limitations of 1-D wetland flow models.

Model configurations

MIKE 21, a 2-D, physically based, distributed model consists of three interconnected fundamental submodules for environmental hydraulic simulations: (1) hydrodynamics (HD), (2) advection–dispersion (AD), and (3) ecological process/water quality (ECO Lab) module (DHI, 2004). It was used for the 2-D transient HD and AD simulation in this study. A 2-D model was preferred to a three-dimensional (3-D) model because water depth in most constructed wetlands is shallow, so that stratification is not dominant and a 3-D model would be much more difficult to calibrate and implement. The AD module was used to simulate the spreading of solutes subject to advection–dispersion process. Although the MIKE 21 HD and AD model was developed for 2-D free surface flows occurring in such areas as coastal hydraulics and oceanography, it was deemed appropriate for the simulation of hydraulic and environmental phenomena in wetlands because the basic hydraulic principles are almost identical and the model can be adjusted easily to specific wetland conditions through relatively simple conversions of a few model options.

HD model setup. The 2-D HD model was used to simulate the dynamic flow conditions observed at OEW Cell 7 (Figure 2). The HD module is based on the depth-averaged Saint-Venant equations, which describe the evolution of the water level and two Cartesian velocity components u and v, of which solutions are numerically obtained from a finite difference form of the equations (Rungø and Olesen, 2003; DHI, 2004).

The bathymetry of the modeled area was formed on a 30×30 m grid, initially based on the constructed wetland blueprint for OEW reclamation project (Post, Buckley, Schuh & Jernigan, Inc., 1985) and finally determined after calibration. The simulation period was set from 17 November 2000, 2 : 00 PM to 10 December 2000, 8 : 00 AM (22·75 days), which corresponds to the period in which flow data were measured and the samples for bromide concentration analysis were collected. Each simulation time step for both flow and conservative transport modelling was 5 s. The wetland boundary was surrounded by the cells where land values were assigned, resulting in no flow boundary conditions along the wetland boundary cells to reflect the surrounding berm. Inflow and outflow that occurred through inlet and outlet weirs were simulated using the source and sink options. Two cells corresponding to the location of each weir were selected and time series inflow/outflow data calculated at each weir were assigned to the cells. In this study, if the water depth at a cell was less than 0·02 m, the cell was considered to have become dry; however, once the depth was greater than 0·03 m, the cell was regarded as in a flooded condition subject to flow.

Because a typical Florida wetland has a low slope, shallow water depth, dense herbaceous vegetation, and very slow flow velocity, the flow regime is generally regarded as laminar to sublaminar (Lee *et al*., 2004). Therefore, turbulence generated by flow velocity or vegetation is not often considered as a dominant controlling factor in surface water flow and solute transport in these wetland systems (Harvey *et al*., 2005; Leonard *et al*., 2006). Likewise, it was assumed that the impact of eddy viscosity on flow dynamics and solute transport in the OEW is negligible, so the viscosity was set to zero. The initial surface elevation was set as a constant value of 6·85 m corresponding to the initial water level over the top elevation of the outlet weir. For flow resistance, which is expressed as a form of Manning roughness coefficient, a constant value of $n = 0{\cdot}11$ was initially set and later adjusted for model calibration. The time series results of

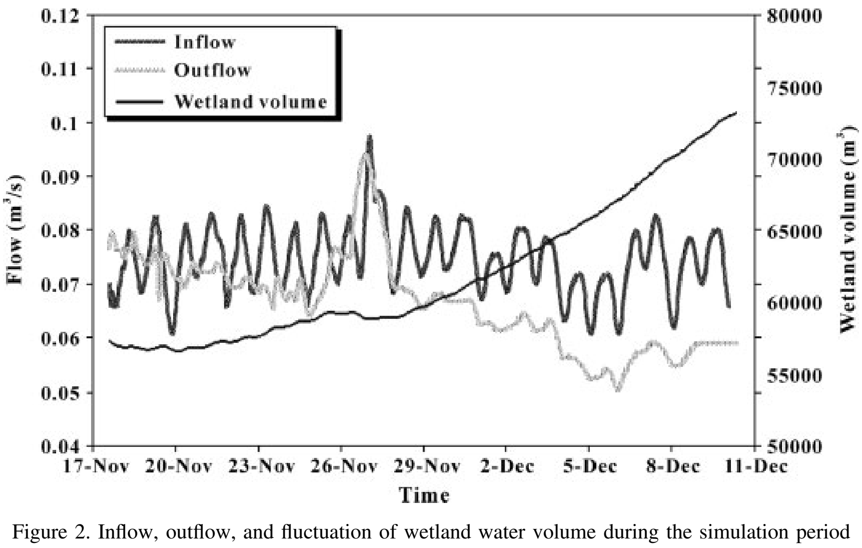

Figure 2. Inflow, outflow, and fluctuation of wetland water volume during the simulation period

the HD simulation, such as flow velocity and water depth in every grid cell, were used as input data for the bromide transport simulation in the AD module.

Rainfall was minimal to non-existent during the period of field experiment (Martinez and Wise, 2003a). As precipitation and evaporation during the simulation period were negligible compared with the flow through the Cell 7, they were not included in this simulation. Wind (or wave) action in shallow aquatic bodies plays an important role in solute or sediment dispersion (Chen, 1994); however, this effect can be reduced in densely vegetated constructed wetland systems like OEW Cell 7 because emergent vegetation dampens wave energy and shelters the water surface from wind stress (Nepf, 1999; Braskerud, 2001). According to the hourly averaged wind speed data (NOAA National Climatic Data Center) monitored at the Orlando Executive Airport located approximately 30 km west of the OEW, there were no major/strong wind events during the sampling period (Mean $\pm$ SD $= 2{\cdot}75 \pm 1{\cdot}67$ m s^{-1}; light breeze level categorized by Beaufort scale); therefore, wind effects were not modelled in this study.

AD model setup. The OEW Cell 7 bromide tracer experiment carried out during November–December 2000 was simulated using the conservative solute transport option of the MIKE 21 AD module. For the AD simulation, the initial concentration of bromide was set to zero and the time series bromide concentration profile measured at the inlet was assigned at the source cell. It is generally known that it is difficult to select 2-D dispersion coefficients, one of the most important parameters in advection–dispersion simulation (DHI, 2004). Of several empirical estimates, the following formulation was used to estimate the coefficients (DHI, 2004):

$$D_x = a_x \cdot \Delta x \cdot v_x \text{ and } D_y = a_y \cdot \Delta y \cdot v_y \qquad (6)$$

where D_x and D_y are the x and y direction dispersion coefficients (m^2 s^{-1}), $\Delta x = \Delta y$ is the constant grid spacing (30 m), v_x and v_y are the local current velocity components (m s^{-1}), and a_x and a_y are the proportionality factors. An isotropic, non-dimensional proportionality factor of 0·6 was used in this study. The value was not determined from the model calibration, but simply selected through modelling efforts of avoiding some simulation instabilities of input and output bromide concentration profiles. To the authors' knowledge, reliable scientific information related to the determination of this parameter in vegetated, free surface water constructed wetland systems has rarely been reported. As the local current velocity is not constant with respect to time, the water velocity on a time step or average water velocity of each cell was required in both x and y directions to determine v_x and v_y. In this simulation, the time step selected was when the peak concentration of bromide measured at the inlet weir was observed. The dispersion coefficient values determined using Equation (6), assigned to each grid cell, varied according to the differences in local current velocities on the time step ($D_x = 0{\cdot}0001$ to $0{\cdot}078$ m^2 s^{-1} and $D_y = 0{\cdot}0001$ to $0{\cdot}081$ m^2 s^{-1}). These values are within the ranges reported by other researchers in constructed wetlands (Table I).

RESULTS AND DISCUSSION

Moment analysis and hydraulic performance

Table II shows the moment analysis results of the bromide BTC and metrics from the previous and this study. The discrepancies between the previous research and this study are largely attributed to the difference of simulation period and flow conditions simulated. Although the previous research used a slightly longer periods of flow and bromide measurement, it was carried out assuming steady-state flow conditions for the hydraulic analysis and 1-D transient storage model application for transport (Martinez and Wise, 2003a, 2003b); in contrast, 2-D transient flow conditions were reflected in this study. Figure 2 shows the inflow, outflow, and fluctuation of wetland water volume during the simulation period. The volume of each simulation time step was estimated from the water depth–volume relationship of the cell, which is based on the time series water depth simulation results; the average cell volume of 61 877 m^3 was used to calculate the nominal residence time.

Minimum travel time, t_m, defined as the shortest travel time from inlet to outlet, can be considered a characteristic of the RTD, which identifies short-circuiting of flow in wetlands because it indicates the fastest flowpath through a wetland (Holland *et al*., 2004). In this study, a minimum travel time was defined as the time of period between two points of the last measurement before the measured concentration (about 0·3 mg L^{-1}) reached 5% of the peak concentration in both inlet and outlet RTD curve. The time of each point

Table I. Values (or range) of dispersion coefficients in constructed wetlands

Source	Dispersion coefficients (m^2 s^{-1})	Comment
This study	0·0001–0·078/0·081*	OEW Cell 7, Florida
Kadlec (1994)	0·0169 ± 0·00976	Des Plaines River
Martinez and Wise (2003a)	0·157 (0·010–0·512)	OEW(Cell 1–15), Florida
Keefe *et al*. (2004)	0·00253 ± 0·000775	Tres Rios Wetlands,
	0·00997 ± 0·000235	Phoenix, Arizona
	0·0213 ± 0·000401	

* D_x/D_y.

Table II. Moment analysis results and metrics showing hydraulic performance of the OEW Cell 7

	Previous research[a]	This study
Cell area (m^2)	117,000	117,900
Cell volume (V_{AVG}, m^3)	67,000	56,600–73,200 (61,877[b])
Average cell flow (Q_{AVG}, $m^3\ s^{-1}$)	0·060	0·072[c]
Nominal residence time (τ, days)	13·0	10·0
Mass recovery fraction (MRF, %)	97	96
Mean residence time ($\bar{t}$, days)	2·63	2·38
Variance (σ^2, $days^2$)	18·3	8·8
Effective volume (e, %)	20 (31[d])	24
Time to peak (t_p, days)		1·47
The number of CSTRs in series (N)		2·6
Hydraulic efficiency (λ, %)		15

[a] Martinez and Wise (2003b).
[b] Mean value.
[c] Mean value of average inflow and outflow during the simulation period.
[d] Expressed as the sum of the volume of the main channel and storage zone, as determined by OTIS (Martinez and Wise, 2003a).

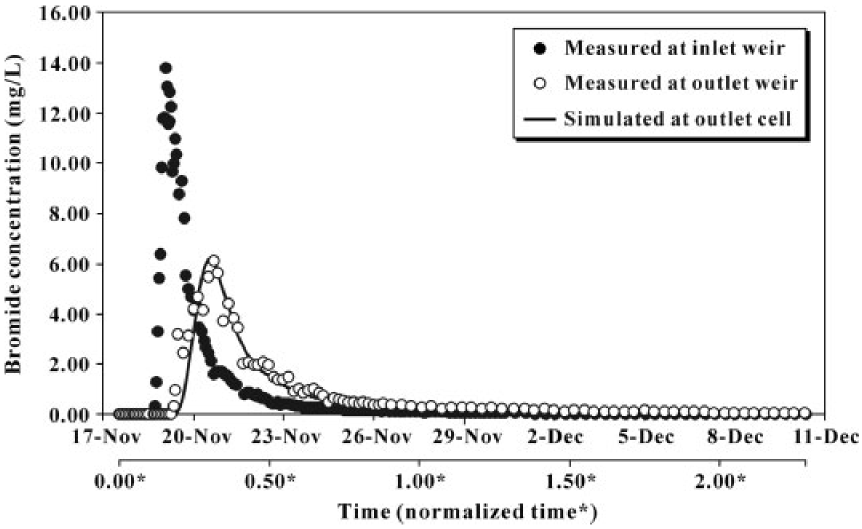

Figure 3. Bromide tracer experiment simulation: MIKE 21 model fit on the measured BTC

calculated using this method matched the first rise of each RTD curve. The minimum travel time was determined to be approximately 14 h. Several other parameters to measure the hydraulic performance of a wetland cell are also summarized in Table II. Many are indicators of short-circuiting flow, including a relatively short mean residence time (2·38 days), a small effective volume (24%), and very low hydraulic efficiency (15%).

Model calibration: bromide tracer BTC and transient 2-D plume

Figure 3 shows the calibrated MIKE 21 model fit for the bromide BTC measured at the outlet weir during the entire simulation period. The bromide concentration profile (closed circles in Figure 3) measured at the inlet weir (weir number 3-x in Figure 1) was assigned as the input boundary condition and the simulated BTC (solid line in Figure 3) was compared with the profile (open circles in Figure 3) measured at the outlet weir (weir number 7-x in Figure 1). Model calibrations on the bathymetry and hydraulic resistance were started with the blueprint bathymetry and a constant value of $n = 0{\cdot}11$. Unexpectedly, a considerable discrepancy was found between observation and simulation at the initial trial. To adequately fit the simulated BTC to the measured bromide concentration profile, it was required to find the bathymetry condition and the associated hydraulic resistances that contributed to decrease the time to peak or mean residence time and increase the peak concentration. Although the initial trial was far from fitting the observed outlet BTC, it provided a 2-D flow velocity vector map that allowed the delineation of a short-circuiting flow zone. In this study, it was assumed that the striking change of bromide BTC at the outlet cell was caused primarily by the change of bathymetry and the associated hydraulic resistance in the cells located along the main flow direction (short-circuiting flow zone) identified by the 2-D flow velocity profile.

To match the short-circuiting pattern of observed outlet BTC, the bathymetry of the short-circuiting flow zone was manipulated first, followed by hydraulic resistance. This calibration sequence was repeated through trial and error until the root mean square error (RMSE) of simulation was minimized. Noticing the fact that

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a ditch-shaped bathymetry is most favourable to the model fit, three different Manning roughness coefficients were assigned to each hydraulically divided conceptual flow zone. Model calibration on the bathymetry and a set of three different Manning coefficients was not finalized until the simulation approached the best model fit (RMSE $= 0{\cdot}45$ mg L^{-1}; $R^2 = 0{\cdot}90$). The estimated topographic difference between the short-circuiting flow zone and the adjacent area ranged from 0·3 to 0·8 m and Manning roughness coefficient of 0·035 s $m^{-1/3}$ was estimated at the short-circuiting zone. Figure 4 illustrates six transient 2-D bromide concentration plumes of the calibrated simulation at the OEW Cell 7 every 12 h after 1·5 days from the start of simulation. It shows that approximately two-thirds of the entire wetland cell area did not contribute to flow and the flow and solute transport were mainly controlled by the short-circuiting zone developed narrowly along the central flow path. The layout of the calibrated bathymetry and the associated flow zones is presented as a baseline case in Figure 5e. The areal portion of the main flow channel and storage zones is 26·4% of the model domain, which is close to the relative portion (31%) of volume of these two zones as calculated by Martinez and Wise (2003a) (Table II).

In the following sections, the computed impacts of change in bathymetry and vegetation on short-circuiting flow within a treatment wetland cell will be presented by reviewing the results of sensitivity analysis on the two factors, based on their spatial distribution delineated by the 2-D bromide plume distribution. Several implications of the analysis results and the modeling approach used in this study are subsequently discussed. These sensitivities were deduced during model calibration.

The effect of bathymetry on short-circuiting flow

To determine the most favourable 2-D bathymetry condition to create a short-circuiting flow in Cell 7, five simulation cases with different bathymetry manipulations, including model calibration (base case), were tested. Each layout of modified bathymetry and the simulated BTC are shown in Figures 5 and 6, respectively. The hydraulic efficiency calculated from the each BTC simulation is denoted for comparison in Figure 6.

An initial bathymetry map of Cell 7 for 2-D modelling was made with measured topographic elevation data of 35 points displayed on the construction blueprint for the OEW reclamation project (Post, Buckley, Schuh & Jernigan, Inc., 1985). An inverse distance weighted interpolation scheme of MIKE ZERO Grid Series Editor

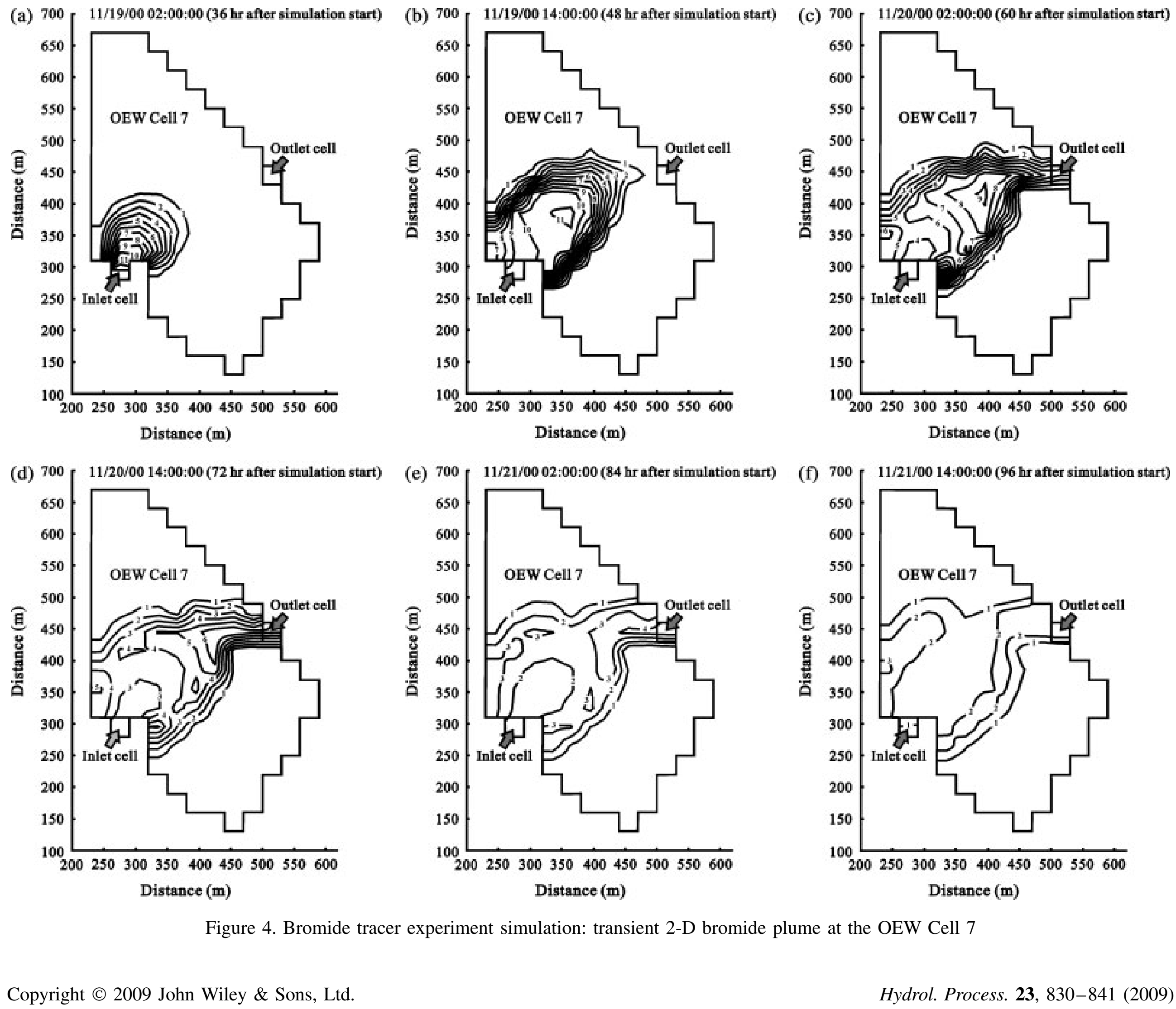

Figure 4. Bromide tracer experiment simulation: transient 2-D bromide plume at the OEW Cell 7

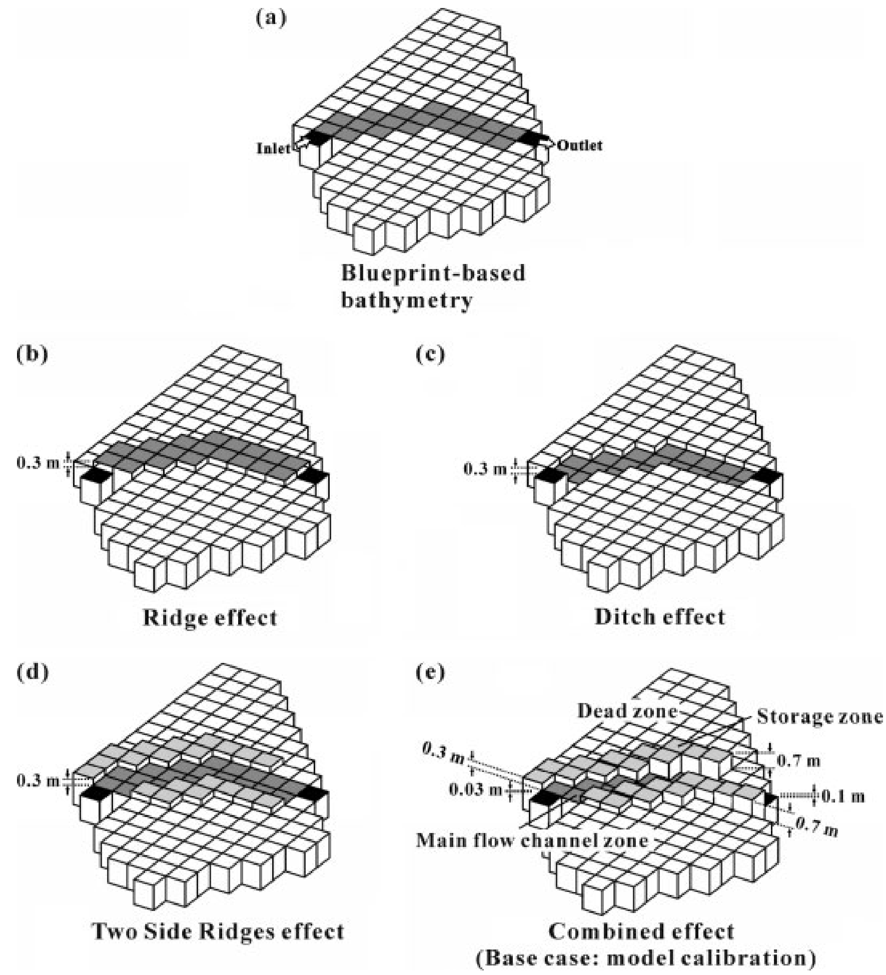

Figure 5. Layouts of bathymetry manipulations

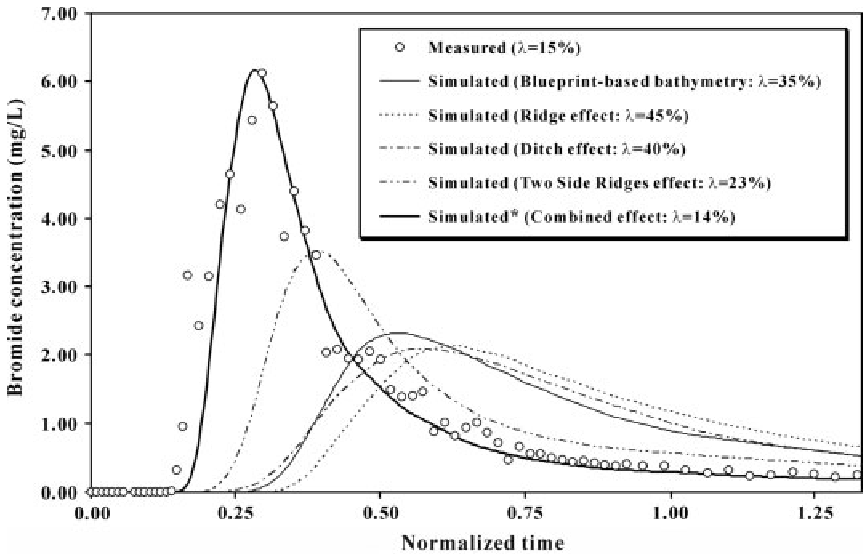

Figure 6. The effect of bathymetry on short-circuiting flow

was used to make the spatially continuous contour map (Figures 1 and 5a). Although this bathymetry map was based on field observations, the simulated outlet BTC was far from the measured one or the base case (Figure 6; solid line). Secondly, bathymetry levels of 0·3 m were increased in the 19 cells located within this main flow channel (Figure 5b). This corresponds to a 0·3 m high middle ridge dividing the entire cell into two areas. Even compared to the blueprint-based case, the simulation result for this case was worse in that the peak

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concentration decreased and the mean residence time and time to peak increased adversely, as shown by the dotted line in Figure 6. Third, bathymetry levels of 0·3 m were reduced in this time under the same conditions as before to create a 0·3 m deep middle ditch (Figure 5c). This simple 'ditch effect' was shown to be favourable to reducing the minimum travel time but still lacking compared with the base case (Figure 6). Next, bathymetry levels of 0·3 m for both side outer cells adjacent to the cells of the main flow channel area were increased without changing any bathymetry within the channel, causing effects similar to immersed levees on both sides of a channel (Figure 5d). In this case, a better simulation result was obtained compared with three previous trials, but still not enough to generate the short-circuiting flow pattern. Finally, Figure 5e shows the layout of the calibrated bathymetry. To generate this base case bathymetry, the 'two side ridges' effect (Figure 5d) was combined with the 'ditch' effect (Figure 5c) to enhance the impact of one large ditch, like a 1-D channel with levees that formed along the main flow direction. During the model calibration, numerous cases of these combined effects were tested by trial and error. The simulated BTC, represented by the bold solid line in Figure 6, was finally generated from the combined manipulations of shortening the number of short-circuiting grid cells between two ridges (19 to 12), increasing the height of each ridge to between 0·3 and 0·7 m, and decreasing the bathymetry levels of the main flow channel area to between 0·03 and 0·1 m (Figure 5e). The hydraulic efficiency of this base case (14%) is closest to one of the measured data (15%).

Bathymetric description is known to be one of the most important tasks in the hydrodynamic modelling process since it plays a key role in determining water depths in a model area (DHI, 2004). This result shows that a bathymetry condition resulting in the water depth increasing by 0·3 to 0·8 m at the narrow main flow zone compared to the adjacent areas in study area is required to generate the highly short-circuiting flow (low hydraulic efficiency) experienced in the treatment cell. This effect of water depth on RTD characteristics was also reported by Holland *et al*. (2004) from the results of dye tracer experiments conducted at a stormwater treatment wetland and by Persson (2005). Martinez and Wise (2003b) suggested that the presence of relic ditches running parallel to flow direction and other landforms in the cell bottom was one of the major factors creating short-circuiting flows experienced at the OEW. These sensitivity test results of different bathymetries reinforce the fact that the relic ditches and other ditch-shaped landforms developed along the main flow direction enhance the wetland preferential flow characteristic, reducing the residence time.

The effect of vegetation on short-circuiting flow

The variation of flow resistance caused by the spatial or temporal distribution of wetland vegetation can affect the treatment performance of constructed wetlands (Poiani and Johnson, 1993; Lee *et al*., 2004; Wörman and Kronnäs, 2005). In many cases, however, it is difficult to determine the effect accurately due to a lack of available data. In this section, the impacts of the spatial distribution of wetland vegetation (hydraulic resistance) on short-circuiting flow are shown through a sensitivity analysis of Manning roughness coefficients assigned along the hydraulically divided, conceptual flow zones (main flow channel, storage zone, and dead zone in Figure 5e).

Although hydraulic resistance parameters originated from the theory of open channel hydraulics are known not to be applicable directly to wetland systems (Kadlec and Knight, 1996), the Manning roughness coefficient has been most widely used to describe the lumped flow resistance of drag due to the stems or litter of wetland vegetation as well as the channel bottom (Kadlec and Knight, 1996; Loftin *et al*., 2001; Lee and Shih, 2004), particularly in flow dynamics modelling approaches. Also, hydraulic resistance is generally known to be a function of water depth as well as vegetation type and density. In this study, the temporal variation effect of ponding depth simulated in each grid cell was not implemented due to the limitation of the modelling platform (MIKE 21) and, considering the relatively short simulation period (22·75 days), no temporal change on assigned vegetation type/density was assumed. Spatial variation of the roughness coefficient within each flow zone was not considered due to the lack of field data on the spatial distribution of each vegetation type. Although the coefficients were conceptually assigned in the fully distributed modelling platform, this semi-distributed modelling approach may be useful in constructed wetland systems in which vegetation types, distribution, and density are so complex that it is difficult to verify all the sources of hydraulic resistance.

Figure 7 shows BTCs generated from the sensitivity test and each hydraulic efficiency. Half of those were tested using constant Manning coefficients ($n = 0{\cdot}1$, 0·05, and 0·025) through the entire model domain, assuming no spatial heterogeneity of vegetation. A significant difference among the simulation results of those three cases was not observed and none of the results was sufficient to fully simulate the short-circuiting flow at the Cell 7 (Figure 7). This result indicates that assigning only one roughness coefficient to a wetland system, which has been generally used in most wetland studies, may not fully explain the flow characteristics.

Next, Manning coefficients were assigned along the hydraulically divided conceptual flow zones: (1) a hydraulic 'dead zone' consisting of background area; (2) a transient 'storage zone' located at the transitional area between dead zone and main flow channel; and (3) a 'main flow channel zone' resulting in short-circuiting flow. First, to determine the roughness coefficient in the dead zone, the maximum roughness coefficient ($n = 0{\cdot}11$ s m$^{-1/3}$) for 'medium to dense brush floodplains in winter ($n = 0{\cdot}045$–$0{\cdot}11$)' was selected from Chow (1959). This value was compared with other data on the roughness coefficient of wetland types similar to that in

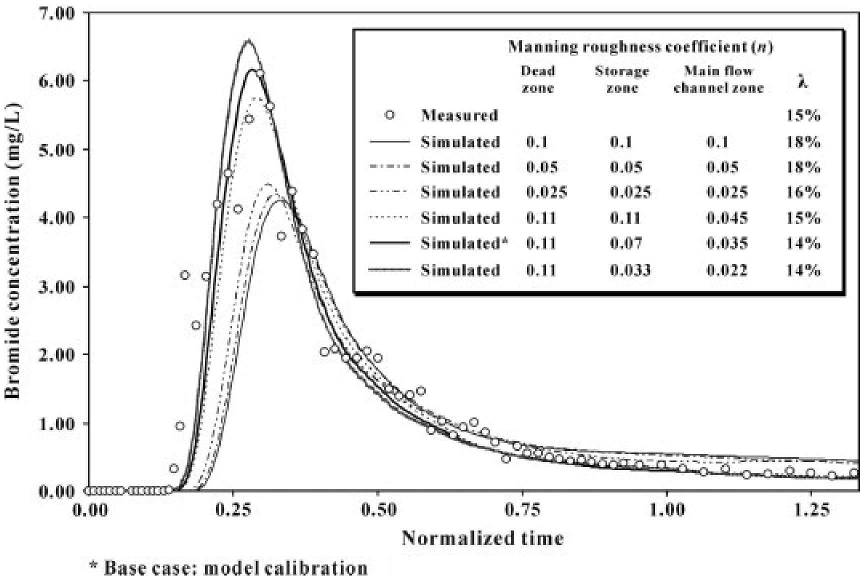

Figure 7. The effect of vegetation (hydraulic resistance) on short-circuiting flow

the Cell 7. According to SFWMD (2005), hydraulic resistance can be computed as a function of ponding depth and vegetation type as follows:

$$n = a \times {h_m}^b \tag{7}$$

where a and b are constants that are a function of wetland type or vegetation, and h_m is water depth. The wetland type of the OEW Cell 7 is wet prairie ($a = 0{\cdot}150$ and $b = -0{\cdot}77$). As the average water depth was regarded as approximately 0·6 m in Cell 7 during the entire simulation period, the roughness coefficient of 0·13 s $m^{-1/3}$ was calculated using Equation (7). In addition, the value of the Manning coefficient in wet prairie was reported to be 0·10 by Loftin *et al.* (2001). Therefore, choosing the value of $n = 0{\cdot}11$ as the background area roughness coefficient in this study seemed to be reasonable.

With an estimated value of 0·11 for the dead zone, the coefficients for the main flow channel and the storage zones were changed. For the base case, denoted by an asterisk in Figure 7, both end values (0·035 s $m^{-1/3}$ for the main flow channel and 0·07 s $m^{-1/3}$ for the storage zone) of the roughness coefficient range for 'scattered brush floodplains with heavy weeds ($n = 0{\cdot}035$–$0{\cdot}07$)' (Chow, 1959) were used. First, the minimum roughness coefficient value ($n = 0{\cdot}045$; Chow, 1959) for 'medium to dense brush floodplains in winter' was assigned for the main flow channel zone and the value for the storage zone was assumed to be same as the one for the dead zone ($n = 0{\cdot}11/0{\cdot}11/0{\cdot}045$ in Figure 7). Second, the minimum (0·022 s $m^{-1/3}$) and maximum roughness coefficients (0·033 s $m^{-1/3}$) for 'straight and uniform earth dredged area with short grass and few weeds ($n = 0{\cdot}022$–$0{\cdot}033$)' (Chow, 1959) were assigned for the main flow channel and storage zones, respectively ($n = 0{\cdot}11/0{\cdot}033/0{\cdot}022$ in Figure 7). These simulated BTCs look slightly underestimated or overestimated around the peak time compared to the base case (Figure 7).

The heterogeneous distribution of vegetation causing spatially non-uniform resistance to flow is a main factor creating short-circuiting flows, resulting in low efficiency of treatment wetlands (Kadlec and Knight, 1996; Martinez and Wise, 2003b). In the previous study (Martinez and Wise, 2003a), a 1-D solute transport model was used to estimate the ratio of the hydraulically divided conceptual flow zones. However, the approach did not provide any information on the hydraulic resistance of each zone, even though separation of the zones is, to some extent, attributed to the differences of flow resistance by wetland vegetation types and density. In this study, the values of 0·035, 0·07, and 0·11 s $m^{-1/3}$ were estimated as the roughness coefficients in the faster-moving flow area, the transition zone, and the dead zone covering the other area of the model domain, respectively. According to the model used by Jenkins and Greenway (2005), the non-vegetated part of the wetland was assumed to have a Manning coefficient of 0·035. However, it may be more reasonable to estimate the roughness coefficient value as a certain range rather than a constant value, considering possible deviations, including analytical measurement error. In addition, hydraulic efficiencies of the last two simulation trials were almost identical to those of the base case or measurement (14–15%). Therefore, a range between 0·022 and 0·045 s $m^{-1/3}$ was suggested as the Manning roughness coefficient values for short-circuiting flow zones in constructed wetlands similar to the OEW Cell 7.

Implications of this modelling study

In this study, a 2-D, physically based, distributed hydraulic and solute transport model was developed using MIKE 21 based on the hydraulic measurement and bromide tracer experiment carried out at the OEW Cell 7. With the transient flow conditions, it simulated the measured bromide BTC successfully ($R^2 = 0{\cdot}90$) and the hydraulic performance was compared with the results of

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previous studies. In this section, some implications of this modelling study are reviewed.

First, the application of hydraulic efficiency (λ), proposed by Persson *et al*. (1999) will lead to even less hydraulic efficiency of the entire OEW than the values reported in the research of Martinez and Wise (2003a, b), because the efficiency in those studies was defined as the metric of effective volume (e). When the alternative parameter, λ, was applied, the hydraulic efficiency of 24% was reduced to 15% at the OEW Cell 7. Therefore, this shows the necessity of more consistently comparing the hydraulic performance of constructed wetland systems, such as short-circuiting and hydraulic efficiency, avoiding a confusion of terminology.

Second, the impacts of changes in bathymetry and vegetation distribution on short-circuiting flow can be compared in terms of hydraulic efficiency (λ). The deviation from the hydraulic efficiency of measured data (15%) was much higher during the simulations for the bathymetry effect (14–45%) than was the vegetation effect (14–18%). It shows that the bathymetry effect on hydraulic efficiency is more critical than the vegetation effect. Recent research also highlighted the impact of water depth among all factors affecting hydraulic efficiency. Holland *et al*. (2004) reported that water depth, rather than flow rate, had a direct impact on the RTD of a wetland and Kadlec (2005) also suggested that lower pond, compared with wetland, efficiencies were primarily due to a depth effect, not the absence of vegetation. This modelling study suggests that the impact of water depth change induced by bathymetry change on hydraulic efficiency may be greater than that of vegetation density change, particularly showing that increasing the water depth due to the ditch-shaped bathymetry results in a decrease in hydraulic efficiency.

Finally, water flow is a basic component of most solute transport/retention models because water is the medium in which solutes, including various pollutants, are transported or transformed. In general, most traditional wetland flow and transport models have relied primarily on 1-D, conceptual, parameter-lumped modelling approaches under such unrealistic hydraulic conditions as plug-flow and steady-state. In other words, unrestricted chances for contact between the incoming water and the organisms responsible for treatment were assumed (Persson, 2000). In addition, it is difficult to find the physical meanings of their key parameters and not possible to independently estimate the effects of various physical and ecological factors affecting wetland hydraulics. In terms of treatment efficiency, manipulating the hydraulic regime to increase efficiency is a desirable strategy for constructed wetlands (Wang and Mitsch, 2000). For this, understanding the characteristics of solute transport/retention on the basis of transient wetland flow conditions is essential. This paper reviewed the simulated impacts of two physical settings of a constructed wetland on its hydraulics, resulting in short-circuiting flow. The MIKE 21 transient flow-based transport model used in this study described the effects successfully. This shows that 2-D, spatially distributed, HD- and AD-linked models can overcome most limitations that traditional wetland models have, provided suitable spatial data are available. Even though field verification of the bathymetry and vegetation implied by the modelling of tracer data was not carried out in this study due to rejuvenating management activities initiated in 2002, which include muck/plant removal, site grading, construction of baffles and islands, and re-vegetation (Wang *et al*., 2006), the quantitative understanding of topographic and vegetative heterogeneities on short-circuiting flow described in this paper may provide future modellers with some useful insights in terms of scale and resolution of these two key spatial data when a multi-dimensional, physically based, spatially distributed modelling approach is considered as a flow and transport modelling tool for constructed wetlands.

CONCLUSIONS

Understanding flow behaviour in constructed wetlands is necessary primarily because it provides the framework for a solute transport/retention model and directly affects the treatment efficiency. The OEW Cell 7 experienced extreme short-circuiting flow, proved by the metrics showing the hydraulic performance, including an efficiency of 15%. In this study, a 2-D hydrodynamics and solute transport model, MIKE 21, was used to simulate the effects of modifications to wetland bathymetry and vegetation distribution on short-circuiting flow. The simulation results reveal that the RTD of a treatment wetland cell is highly sensitive to changes in bathymetry and vegetation distribution. It also clearly shows that a ditch-shaped bathymetry and the associated sparse vegetation distribution (less hydraulic resistance) in the central narrow region of the OEW Cell 7 have significant impacts on the short-circuiting flow. Topographic depression (0·3–0·8 m) and a range of relatively low Manning roughness coefficients (0·022–0·045 s m$^{-1/3}$) were estimated at the short-circuiting flow zone. In this study, the MIKE 21 simulation results of the OEW Cell 7 bromide tracer test show its capability as a robust modelling tool for more advanced flow and transport models in constructed wetlands, providing future modellers with some useful insights in terms of scale and resolution of these two key spatial data to be obtained. In this respect, it may be broadly applied in constructed wetlands to develop a useful modelling tool for quantitatively understanding the internal processes of pollutant removal/retention under dynamic flow conditions and testing various hypotheses according to the changes in the hydraulic regime.

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