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Tracer behaviour and analysis of hydraulics in experimental free water surface wetlands

Hristina Bodin ^{a,*}, Anna Mietto ^b, Per Magnus Ehde ^c, Jesper Persson ^d and Stefan E.B. Weisner ^c

^a IFM, Division of Ecology, Linköping University, 581 83 Linköping, Sweden

^b Department of Environmental Agronomy and Crop Science, University of Padova, Agripolis, Legnaro (PD), Italy

^c Wetland Research Centre, School of Business and Engineering, Halmstad University, Halmstad, Sweden

^d Department of Landscape Management, Design and Construction, Swedish University of Agricultural Sciences, Alnarp, Sweden

* *Corresponding author.* Tel.: +46-(0)13 281296; fax: +46-(0)13 281399.

E-mail addresses: inuita@ifm.liu.se (H. Bodin), stefan.weisner@hh.se (S.E.B. Weisner)

Abstract

Effects of inlet design and vegetation type on tracer dynamics and hydraulic performance were investigated using lithium chloride in eighteen experimental free water surface wetlands. The wetlands received similar water flow but had different vegetation types: 6 emergent vegetation wetlands (EVWs), 6 submerged vegetation wetlands (SVWs) and 6 free development wetlands (FDWs). Two types of inlet designs were applied: half of each wetland vegetation type had a barrier near the inlet to help distribute incoming tracer solution, while the rest had no barrier. Residence time distribution (RTD) functions were calculated from tracer data using two techniques: method of moments and a novel Gauss modelling approach. RTD functions were used to quantify hydraulic parameters: active wetland volume (e -value), water dispersion (N -value) and hydraulic efficiency (λ -value).

For wetlands without barrier, significantly lower tracer mass recoveries were found from EVWs compared to FDWs and SVWs, signifying a risk of tracer methodological problems in small densely vegetated wetlands. These problems were minimized in wetlands with an inflow construction promoting distribution of incoming tracer solution.

Compared to the method of moments, Gauss modelling seemed to produce more reliable λ -values but less reliable N -values. Data for precise hydraulic quantification were lost by Gauss modelling, as indicated by overall lower variance in these data sets and lower mass recoveries. However, Gauss modelling may minimize uncertainties associated with lithium immobilization/mobilization. Parameters were significantly affected by the RTD data analysis method, showing that the choice of method could affect evaluation of wetland hydraulics.

The experimental wetlands in this study exhibited relatively high e -values and low N -values. This was probably caused by the small size of the wetlands and low water flow velocities, emphasizing that hydraulic parameter values obtained in small experimental wetlands may not be applicable to hydraulics in larger wetlands.

The method of moments revealed lower e -values from EVWs compared to SVWs and FDWs. It was indicated that lower e -values were mainly caused by vegetation volumes. This highlighted a need for regular maintenance to secure efficient treatment volume in wetlands with dense vegetation.

Keywords: lithium tracer, hydraulic performance, free water surface wetlands, vegetation, inlet, Gauss modelling, data analysis

1. Introduction

Free water surface (FWS) wetlands have been used in the past 30 years for removal of pollutants from different water sources, such as rivers, municipal and industrial wastewater, and also urban and agricultural runoff (Nichols, 1983; Gerke et al., 2001;

Kadlec and Wallace, 2009). It has been recognized that water flow distribution, i.e. hydraulic conditions within wetlands, can affect their pollutant removal efficiency (Kadlec, 1994; Persson et al., 1999; Dal Cin and Persson, 2000; Serra et al., 2004; Dierberg et al., 2005; Wang and Jawitz, 2006). Therefore, wetlands may fail to meet water quality standards due to hydraulic problems associated with poor wetland design (Thackston et al., 1987; Persson et al., 1999; Kadlec, 2000; Goulet et al., 2001; Persson, 2005).

Several features, such as wetland shape (Persson, 2000; Wörmann and Kronnäs, 2005), inlet and outlet location (Persson et al., 1999; Suliman et al., 2006), wetland bottom topography (Kjellin et al., 2007; Lightbody et al., 2007) and vegetation distribution (Persson et al., 1999; Dal Cin and Persson, 2000; Serra et al., 2004; Kjellin et al., 2007; Keefe et al., 2010), can affect hydraulic characteristics of FWS wetlands.

Persson et al. (1999) and Persson (2000) showed that an island located near the wetland inlet enhanced the distribution of incoming water and thus the hydraulics. Also, Persson (2005) showed that the position of islands inside wetlands was important for the hydraulic performance. Since the cited studies were primarily based on computer simulations, there is need for more empirical data to elucidate the significance of internal physical structures for wetland hydraulics.

Typically, wetlands include a variety of vegetation types such as emergent, floating-leaved attached, free-floating and submerged macrophytes (Kadlec and Wallace, 2009). In fact, macrophytes are often the major component of wetlands and several studies have shown their importance for pollution removal (Nichols, 1983; Weisner et al., 1994; Brix, 1997; Karathanasis et al., 2003). Macrophytes play an important role in natural and physical processes such as filtration and stabilization of

sediments, and also provide increased surface for biofilm growth (Braskerud, 2001; Bastviken et al., 2003; Kadlec and Wallace, 2009). In addition, by dominating the resistance to water flow, macrophytes can influence wetland hydraulics (Kadlec, 1990). Only a few studies have investigated effects of different vegetation types on wetland hydraulics (Stern et al., 2001; Bodin and Persson, in press). Instead, the focus of most published wetland hydraulic studies has been comparing effects on wetland hydraulics caused by different vegetation layouts (Persson et al., 1999; Dal Cin and Persson 2000; Jenkins and Greenway, 2005) or vegetation heterogeneity levels within the same vegetation type (Wörman and Kronnäs, 2005; Kjellin et al., 2007; Min and Wise, 2009). Thus, there is a need to further investigate the significance of highly different vegetation types on wetland hydraulics.

Generally, the hydraulic performance of wetlands has been analyzed using hydraulic tracers (Graham, 1984; Martinez and Wise, 2003; Wang and Jawitz, 2006; Speer et al., 2009; Keefe et al., 2010). The basic assumption of hydraulic tracer studies is that the chosen tracer is inert and thus should represent the water flow through a wetland. However, salt tracer solutions normally have higher density than wetland water, so measures to prevent tracer settling at the wetland bottom should be taken. If the salt tracer is instantaneously injected into calm wetland areas, density effects may cause low tracer mass recoveries and seriously affect the quality of the tracer study. Hence, in wetlands receiving relatively low influent water flows, it may be important to ensure that the initial salt tracer injection method results in properly distributed transport of tracer through the wetland. Still, many published wetland salt tracer studies do not give enough details about tracer injection methods (Dal Cin and Bendoricchio, 2002; Persson, 2005; Dierberg and DeBusk, 2005; Ronkanen and Kløve, 2007; Speer et

al., 2009; Wahl et al., 2010). Thus, in some salt tracer studies, insufficient tracer mass recoveries could be associated with flaws in tracer injection methods. Therefore, there is a need to investigate effects of salt tracer injection methods on tracer behaviour and subsequent tracer mass recovery.

By studying the movement of an inert substance through the wetland, a residence time distribution (RTD) can be determined. Wetland hydraulic parameters can be derived from the RTD by using the method of moments, which is based on numerical integration (Kadlec, 1994; Whitmer et al., 2000; Martinez and Wise, 2003; Wang and Jawitz, 2006; Muñoz et al., 2006), or by modelling of the RTD using probability distribution functions, such as the Gauss function (Levenspiel, 1972; Serra et al., 2004; Fogler, 2006). The main advantage of the method of moments analysis of the RTD is that the hydraulic parameter values are attained directly from the measured data. However, a problem with the method of moments is that it is sensitive to non-continuous data sets and minor concentration variances in the tail region of the RTD, all of which may result in biased hydraulic parameter values (Kadlec and Wallace, 2009). In such cases it may be more advantageous to fit the measured RTD data to a suitable model (Headley and Kadlec, 2007; Kadlec and Wallace, 2009). Yet the chosen fitting procedure may be biased to certain parts of the RTD, resulting in poor fit between the measured data and the model (Wang and Jawitz, 2006; Kadlec and Wallace, 2009). Hence, at present a cautionary principle is justified, using more than one type of tracer data analysis. Also, comparing the outcomes of different types of analysis of tracer data might help future methodological refinement.

1.2. Aim

The present study aimed to evaluate effects on tracer behaviour initiated by inlet design, and to determine how vegetation affected hydraulics in small experimental FWS wetlands. Another purpose was to investigate the potential of a new Gauss modelling approach, as a more stable method for analysis of tracer RTD data than the conventional method of moments analysis.

2. Methods

2.1. Study site

This study was performed in an experimental FWS wetland system (Fig. 1a), near Halmstad, Sweden (56°43'45"N, 12°43'33"E). The system consisted of 18 rectangular wetland basins, each with a flat bottom area of 12 m² (1.6 m x 7.6 m) and ground surface area of 40 m² (4 m x 10 m). Each wetland basin was 1.2 m deep and had side slopes of 45° (Fig. 1b). Incoming water to the wetlands was distributed through three tanks that continuously received groundwater from a common source. Water from each tank was distributed through underground pipes to six wetlands (Fig. 1a). The tank water entered each wetland at one short side through an inlet pipe, positioned approximately 0.1 m above the wetland water level (Fig. 1b). The outlet pipe of each wetland could be adjusted to control water depth. In this study, the mean and maximum water depths were set to 0.55 m and 0.8 m, respectively. This corresponded to a water surface area of 29 m² (9.2 m x 3.2 m) in each wetland. During the study period, the inlet water had a mean temperature of 8.8°C and the mean water temperature measured at the wetland outlets was 10.7 °C.

The experimental design during this study consisted of six wetlands with free vegetation development (FDW), six with planted emergent vegetation (EVW) and six

with planted submerged vegetation (SVW). In nine wetlands (three from each vegetation type), vegetation was removed in the first part of the wetland (0.5 m x wetland width) and a barrier (a board with a width of 0.24 m) was placed at a distance of 0.1 m in front of the inlet pipe (Table 1; Fig. 1a,c). The two different types of inlet designs will subsequently be entitled *b* (barrier) and *nb* (no barrier). The main purpose of the barrier was to investigate effects on tracer mass recovery.

Figure 1. (a) Schematic diagram of the free water surface wetland system near Halmstad, Sweden. The system consisted of six wetlands with free vegetation development (white), six with planted submerged vegetation (interrupted lines) and six with planted emergent vegetation (grey). The incoming water to each wetland came from one of three tanks through an underground inlet pipe (not shown) and entered at one short side of the wetland through an inlet pipe (white cylinder). Black rectangles show inlet design (*b*) and circles show outlet pipe.

Figure 1. (b) Longitudinal three-dimensional cross-section of a wetland with inlet design *nb*.

Figure 1. (c) Schematic diagram of inlet design *b*. Diagrams are not to scale and wetland vegetation is not shown. Abbreviations: *nb* = no barrier; *b* = barrier.

Twelve days before the start of the hydraulic tracer experiment, the inlet water flows were adjusted, using gate valves on the inlet pipes of each wetland, to correspond to a theoretical hydraulic residence time (t_n) of around 4 days. Regular manual measurements of inlets and outlets were made, using a stopwatch and a measuring cylinder, to ensure that the t_n -value did not deviate from 4 days. Infiltration was assumed to be negligible due to the low permeability of the heavy clay soil on site. Table 1 contains physical and hydraulic information about the studied wetland system.

Table 1. Physical and hydraulic data about the experimental free water surface wetland system near Halmstad, Sweden during the study period 24 September – 3 October 2008 ($n = 18$ if not stated otherwise). Standard deviation is shown in brackets.

Volume (m ³) ^a	Area (m ²) ^b	Mean & max water depth (m)	t_n (day) ^c	Height of inlet pipes (m) ^d	Barrier width (m) ^e	Distance: barrier to inlet pipe (m) ^e	Distance: water level to barrier base (m) ^e
16.3	29	0.55	4.4	0.1	0.24 (±0.03)	0.1	0.5
		0.80	(±0.5)	(±0.06)		(±0.06)	(±0.10)

^a volume of one wetland; ^b area of one wetland; ^c based on outflows ($n = 54$); ^d above water level; ^e $n = 9$

The cover of vegetation in the wetlands was estimated on 30 September 2008. The types of vegetation targeted were filamentous green algae (FGA), emergent, floating-leaved and submerged vegetation. The vegetation survey followed the methodology outlined by Weisner and Thiere (2010). Results from the vegetation survey are shown in Table 2.

Table 2. Results from the vegetation survey performed 30 September 2008 in the experimental free water surface wetland system near Halmstad, Sweden. The wetland system consisted of 6 free development wetlands (FDWs), 6 emergent vegetation wetlands (EVWs) and 6 submerged vegetation wetlands (SVWs). Each value represents the mean for each wetland vegetation type. Standard deviation is shown within brackets ($n = 6$).

	Vegetation cover (%)			
	Filamentous green algae	Emergent reeds	Floating-leaved vegetation	Submerged vegetation
FDWs	50(±24)	29(±10)	40(±18)	3 (±8)
EVWs	0	94(±3.2)	0	0
SVWs	88(±7.8)	19(±6.2)	18(±13)	70(±25)

Vegetation species compositions were similar to those found in 2006 (Weisner and Thiery, 2010). The EVWs were dominated by emergent macrophytes, mainly *Phragmites australis* and *Glyceria maxima*. The FDWs contained a mixture of FGA, floating-leaved (mainly *Potamogeton natans*) and emergent macrophytes (mainly *Typha latifolia*). The SVWs contained similar vegetation as the FDWs wetlands but had a higher cover of submerged vegetation (mainly *Myriophyllum alterniflorum* and *Elodea canadensis*).

2.2. Hydraulic tracer experiments

The hydraulic tracer experiment was carried out in all the 18 wetlands during 24 September – 3 October 2008. Lithium chloride (LiCl) was chosen as a hydraulic tracer because it behaves more conservatively than other commonly used tracers (Dierberg and DeBusk, 2005). The tracer solution was prepared in a bucket by adding 300 g of LiCl in 9 litres of water and mixed thoroughly so that the tracer salt was completely dissolved. The tracer solution was then pulse-injected into one of the three wells (Fig. 1a) with a volume approximately of 500 litres, from where water flowed into 6 of the 18 wetlands. During the tracer injection procedure, the well water was thoroughly mixed so that a homogeneous distribution of the tracer solution was ensured. The tracer injection procedure was done at the same time for the other 2 wells and corresponding 12 wetlands. Wetland inlet water flows measured directly after the addition of LiCl were used to calculate the amount of Li^+ entering each wetland.

At each wetland outlet, water samples for Li^+ analysis were collected in 50 ml polyethylene bottles, and water flows were measured at each Li^+ sampling occasion. The first sample was taken a few minutes after the addition of LiCl (time 0) and the

second one after 20 hours had passed. Thereafter sampling was done at intervals of 6 hours up to the 56th hour after tracer addition. Subsequent sampling was done after 68, 96, 144 and 216 hours had passed since the tracer addition.

Water samples were acidified with concentrated HNO₃ *in situ* until pH < 2 was reached and 1 ml KCl solution was added to each sample. Unfiltered water samples were analyzed for Li⁺ concentrations with an atomic spectrophotometer (Varian Spectra 100), by direct intensity measurements at a wavelength of 670.8 nm.

2.3. Background on wetland hydraulic theory and modelling

Generally, mass transport through wetlands has been conceptualized either to be plug flow (PF) or to be represented by one continuously stirred tank reactor (CSTR). In theory, under plug-flow conditions all of the water parcels move through the wetland at the same velocity, and thus reach the wetland exit at exactly the same time. This exit time is referred to as the wetland nominal hydraulic residence time, t_n and can be calculated as in Eq. 1:

$$t_n = \frac{V}{Q} \quad (1)$$

where t_n = nominal hydraulic residence time (hours); V = wetland volume (m³); Q = water flow rate through wetland (m³ hour⁻¹).

In contrast to the PF model, water in the CSTR model is uniformly and instantly mixed throughout the wetland. Thus, a CSTR wetland experiences a distribution of hydraulic residence times characterized by an exponential decay curve. However, numerous hydraulic tracer studies have confirmed that neither of these water flow patterns dominates in wetlands (Bowmer, 1987; Kadlec, 1994; Kadlec, 2000; Werner

and Kadlec, 2000; Dierberg et al., 2005; Lightbody et al., 2008). Thus, in an effort to describe the water flow in existing wetlands, it has been important to consider the degree of deviation from the PF and CSTR models (Kadlec et al., 1993; Kadlec and Wallace, 2009).

The tanks-in-series model (TIS) bridges the gap between PF and CSTR models, and has received much attention in treatment wetland science during the last decade due to its reasonably good capacity to describe non-ideal flow characteristics (Kadlec and Wallace, 2009). In the TIS model, the wetland is partitioned into a number of equally sized (N) CSTR tanks (Levenspiel, 1972; Kadlec and Wallace, 2009).

Generally, wetlands can experience deviations from these ideal flow conditions due to heterogeneous mixing (normally caused by wind and water flow) and velocity profiles (commonly caused by spatial variation of vegetation and topography). The resulting water dispersion causes some water parcels to exit earlier than the t_n -value and some to exit later, behaviour which indicates short-circuiting and dead zones, respectively (Thackston et al., 1987; Lightbody et al., 2008; Headley and Kadlec, 2007). The term ‘short-circuiting’ describes preferential flow paths as a result of wetland areas with low resistance, which allow some portion of incoming water to travel directly to the outlet in much less time than the t_n -value (Dal Cin and Persson, 2000; Dierberg et al., 2005; Lightbody et al., 2007). In contrast are dead zones, i.e. volumes of water which are not a part of the flowing water volume through a wetland.

2.4. Normalization of the residence time distribution data

To facilitate the comparison of RTD data between different wetlands, the outlet tracer concentration was normalized (Eq. 2) and expressed as a function of flow-weighted time (Eq. 3) (Headley and Kadlec, 2007):

$$C(\phi) = \frac{C_{out}(t) V_{sys}}{M_{out}} \quad (2)$$

$$\phi = \frac{V_{out}}{V_{sys}} \quad (3)$$

where $C(\phi)$ = normalized outlet tracer concentration (-); $C_{out}(t)$ = time-specific outlet tracer concentration (mg L^{-1}); V_{sys} = theoretical (empirical) wetland volume (m^3); M_{out} = total mass of recovered tracer (g); ϕ = normalized flow-weighted time (-) and V_{out} = water volume that has exited the wetland since tracer addition (m^3).

2.5. Residence time distribution (RTD) data analysis

2.5.1. The method of moments analysis of residence time distribution

In order to quantify wetland hydraulics, the RTD data from the hydraulic tracer experiments were analyzed by using the method of moments as outlined by Fogler (2006) with the rectangle method as the interpolating function. This method has been the most common analysis procedure for assessing wetland RTD data in the last three decades (Thackston et al., 1987; Whitmer et al., 2000; Wang and Jawitz, 2006; Keefe et al., 2010). In accordance with this method, the outlet pulse of the tracer was given by the RTD function $f(t)$ as

$$f(t) = \frac{Q_{out}(t) C_{out}(t)}{\int_0^{\infty} Q_{out}(t) C_{out}(t) dt} \quad (4)$$

where $f(t)$ = RTD function (hour^{-1}); $Q_{out}(t)$ = volumetric outflow rate of water at time t ($\text{m}^3 \text{ hour}^{-1}$); $C_{out}(t)$ = outlet tracer concentration (mg L^{-1}); t = time of sampling (hour); dt = difference in time between samplings (hour).

Also, to attain as much RTD data as possible, the tail of the RTD was extended to $3 t_n$ using an exponentially decreasing function (Kadlec and Knight, 1996). To achieve the highest possible R^2 -values in the exponential curves, and thus to ensure reliable tracer concentrations in the RTD tail region, between 3 and 9 data points were used to plot these curves, depending upon the data set from each individual wetland. This resulted in R^2 -values of between 0.89 and 1.

Further, the total mass of recovered tracer at the outlet of each wetland at the end of the tracer study (Whitmer et al., 2000) was defined as

$$M_{out} = \int_0^{\infty} Q_{out}(t) C_{out}(t) dt \quad (5a)$$

where M_{out} = total mass of recovered tracer at outlet (g); $Q_{out}(t)$ = outflow rate of water at time t ($\text{m}^3 \text{ hour}^{-1}$); $C_{out}(t)$ = outlet tracer concentration at time t (mg L^{-1}); t = time of sampling (hour); dt = difference in time between samplings (hour). The relative tracer mass recovery, *mass recovery* (%), was defined as

$$\text{mass recovery} = \frac{M_{out}}{M_{added}} 100 \quad (5b)$$

where M_{added} = total mass of added tracer to wetland inlet (g).

The mean residence time, t_m (hour), i.e. the average time that a tracer particle had resided in the wetland, was represented by the first moment of the RTD (Eq. 6),

$$t_m = \int_0^{\infty} t f(t) dt \quad (6)$$

and the variance (hour²), i.e. the second moment, was calculated as in Eq. 7:

$$\sigma^2 = \int_0^{\infty} (t_m - t)^2 f(t) dt \quad (7)$$

The effective volume ratio of each wetland was estimated using the relationship between t_m and t_n (Thackston et al., 1987) as in Eq. 8:

$$e = \frac{t_m}{t_n} = \frac{V_{effective}}{V_{sys}} \quad (8)$$

where e = effective volume ratio (-); V_{sys} = theoretical (empirical) wetland volume, synonymous with total wetland volume (m³); $V_{effective}$ = total effective wetland volume (m³).

The number of cells (N) in the TIS model was estimated by using Eq. 9 (Kadlec and Wallace, 2009):

$$N = \frac{t_m^2}{\sigma^2} \quad (9)$$

where N = number of tanks in TIS model (-), t_m^2 = squared mean hydraulic residence time (hour²), and σ^2 = variance of RTD (hour²).

Also, to enable a more general and comprehensive measure of the hydrodynamic conditions in the studied wetlands, the hydraulic efficiency parameter was calculated as in Eq. 10 (Persson et al., 1999):

$$\lambda = e^{\left(1 - \frac{1}{N}\right)} = \frac{t_p}{t_n} \quad (10)$$

where λ = hydraulic efficiency (-), e = effective volume ratio (-), and t_p = time at which the maximum tracer concentration occurred (hour). One main advantage of the λ -parameter is that it is not associated with problems related to uncertainties in determination of t_m from RTDs with long residing tails (Persson et al. 1999).

2.5.2. Modelling the residence time distribution

The hydraulic parameters derived from the method of moments provided a starting point to assess the hydraulic performance of the studied wetlands. Upon reviewing the RTDs obtained in this study, it was concluded that the data sets frequently contained multiple peaks, which can be viewed as a discontinuity in the RTD data. Thus, in an attempt to estimate more reliable hydraulic parameters, a modified Gauss model (Eq. 11) was used to estimate the hydraulic parameters (Levenspiel, 1972; Fogler, 2006):

$$y(t) = \frac{k_1}{\sqrt{t}} \exp\left(\frac{k_2}{t} + k_3 + k_4 t\right) \quad (11)$$

where $y(t)$ = modelled outlet tracer concentration values (mg L^{-1}); t = elapsed time since start tracer addition (hour); k_1, k_2, k_3, k_4 = model-fitting parameters; exp = the natural logarithm (-).

The process of data modelling is basically a curve-fitting method which aims at producing a model that best describes the RTD curve and thus yields least biased hydraulic parameters. There are many objective curve-fitting methods but they all have the same aim which is to minimize the residuals between the data and the model. In each case, the Gauss model was fitted to the measured RTD data using MathCad 14.0 and the curve-fitting function *genfit* which is based on the Levenberg-Marquardt optimization algorithm. To the authors' knowledge, the proposed data modelling efforts

using MathCad 14.0 and *genfit* have never before been made for analysis of wetland RTD from hydraulic tracer studies. However, the Levenberg-Marquardt algorithm has been used for calibrations of a variety of models where it considerably improved model fits to measured data (Olyphant, 2003; Lamers et al., 2007; Stein et al., 2007). Using the *genfit* function, an expanded Gauss data set was created by interpolating between measured data points at 1-hour intervals ($dt = 1$ hour). To obtain mass tracer recoveries and hydraulic parameters (e ; N and λ), the Gauss data were numerically integrated using the rectangle rule.

Also, the goodness-of-fit between the measured data and the Gauss model was assessed for each wetland separately by calculating the sum of squared errors (Eq. 12)

$$SSQE = \sum (d - m)^2 \quad (12)$$

where $SSQE$ = sum of squared errors; d = measured outlet tracer concentration; m = Gauss-modelled outlet tracer concentration. Within this context, a low $SSQE$ value was indicative of good association between measured and modelled data, and thus signified reliable parameter estimations.

2.6. Statistical analysis

Statistical analysis was computed using SPSS 17.0 for Windows (SPSS Inc., Chicago, IL, USA). Levene's and Kolmogorov-Smirnov tests were used to examine the data for homogeneity of variance and normal distribution. Data were log-transformed to equalize variances. Effects of the fixed factors (vegetation type and inlet design) and their interaction on hydraulic parameters were evaluated using two-way ANOVA. Because the wetlands were divided and randomized within three blocks (Fig. 1a) a

block factor was also included. The random block factor had no statistical significance and was not used in the further statistical analyses. Since interaction effects between the two fixed factors were found in the measured data set, separate one-way ANOVAS were performed for factor vegetation (within same inlet design) for all hydraulic parameters within this data set. The one-way ANOVAS were followed by Tukey's HSD post-hoc test if effects of vegetation were significant. To test for significant differences between the two data analysis methods (the method of moments and Gauss modelling), a paired t-test was employed if variance was equal, or Wilcoxon signed ranks test if variance was not equal. Also, to test for significant differences in goodness-of-fit of the Gauss model to measured data, $SSQE$ values (Eq. 12) were evaluated using one-way ANOVA with vegetation as a fixed factor. $P < 0.05$ was chosen as the level for significant differences in all cases.

3. Results

3.1. RTD characteristics and model fit to measured data

The RTDs obtained from the hydraulic tracer study in the 18 wetlands, grouped by different vegetation types, are presented in Figure 2. The

RTDs, both measured data and Gauss-modelled data, were all non-symmetrical due to the long tails and right-skewed characteristics.

An overview of the hydraulic parameter data for each wetland during the study period is given in Table A of the Appendix.

Figure 2. Normalized residence time distributions (RTDs) for free development wetlands (FDW), emergent vegetation wetlands (EVW) and submerged vegetation wetlands (SVW), with (*b*) and without (*nb*) barrier at inlet. The x-axis shows time normalized by flow-weighted time (Eq. 3). The y-axis shows normalized Li^+ concentrations (Eq. 2). RTDs were derived from measured Li^+ data (—•—) and Gauss modelling (—) (Eq. 11).

It is possible to distinguish a similar shape in the RTDs within the six replicates of each vegetation type (Fig. 2a-f, Fig. 2g-l and Fig. 2m-r, respectively). Also, similar shapes of the RTDs among FDWs and SVWs were observed, with curves characterized by elongated tails to the right. In contrast, the shapes of the RTDs from the EVWs were sharper compared to the others. Moreover, measured data displayed multiple peaks, especially for the EVWs (Fig. 2).

There was a difference between the measured and Gauss-modelled RTDs. Generally, there was a dissimilarity in the tail region between the two RTDs (Fig. 2 d,f,g,i,j,k,l,r). In contrast, the t_p -value was relatively similar between the two methods, since the Gauss model commonly placed the t_p -value between peaks in the measured RTDs. However, there was an obvious difference in the t_p -value between the two RTDs from EVW 18 (Fig. 2 l). Results from the goodness-of-fit test between the measured data and the Gauss model (Eq. 12) showed statistically significantly higher SSQE values (SSQE = 0.052) for the EVWs compared to the FDWs and SVWs ($p < 0.01$; SSQE = 0.0036 and $p < 0.01$; 0.0052, respectively). There was no statistically significant difference in SSQE values between the FDWs and SVWs.

3.2. Effects of vegetation type and inlet design on estimated hydraulic parameters

3.2.1. Measured data analyzed with method of moments

The statistical analysis, based on measured data, showed that inlet design had a significant effect on e -values ($p < 0.05$; Table 3) and that significant interactions between vegetation type and inlet design existed for e -values and λ -values (both $p < 0.01$; Table 3).

Comparisons between vegetation types, including only wetlands with barrier, revealed statistically lower e -values for EVWs compared to the other wetland types ($p < 0.05$; Table 4; Fig. 3a, measured data). For wetlands without barrier, significantly higher λ -values were found for EVWs compared to SVWs ($p < 0.05$; Table 4; Fig. 3b, measured data), and mass recoveries were

significantly lower from the EVWs compared to the other two wetland vegetation types ($p<0.01$; Table 4; Fig. 3d, measured data).

3.2.2. Gauss model data

Statistical analyses of the Gauss model RTD data showed that the EVWs had significantly lower e -values compared to the other two wetland vegetation types ($p<0.001$; Table 3; Fig. 3a, Gauss model data). Also, higher N -values for EVWs compared to SVWs were confirmed ($p<0.05$; Table 3; Fig. 3c, Gauss model data). Furthermore, a statistically significant effect of vegetation type was obtained for *mass recovery*, and the Tukey HSD post-hoc test revealed a p -value close to 0.05 for EVWs compared to SVWs (Table 3; Fig. 3d; Gauss model data).

Figure 3. Mean values ($n = 3$) of a) effective volume ratio (e); b) hydraulic efficiency (λ); c) number of tanks (N) in the tanks-in-series model; and d) relative tracer mass recovery (*mass recovery*), in free development wetlands (FDWs), emergent vegetation wetlands (EVWs) and submerged vegetation wetlands (SVWs) with (white) or without (grey) barrier at inlet, calculated using method of moments (M) and Gauss modelling (G). Standard deviation is shown in error bars.

Table 3. Effects of vegetation type and inlet design on hydraulic parameters, and their interaction according to ANOVA. Parameter values were calculated using measured data and Gauss modelling. Parameters were: effective volume ratio (e), hydraulic efficiency (λ), number of tanks in tanks-in-series model (N) and relative tracer mass recovery (*mass recovery*). For each parameter, p -values from Levene's test and two-way ANOVA tests are shown. DF = 2 (factor vegetation type); DF = 1 (factor inlet design) and residual = 12. Abbreviations: n.s. = not significant; FDW = free development wetlands, EVW = emergent vegetation wetlands; SVW = submerged vegetation wetlands; DF = degrees of freedom.

Parameter	Factor	Levene's test	Measured data		Levene's test	Gauss model data	
		p -value	p -value	Details ^a	p -value	p -value	Details ^a
e	Vegetation type		n.s.	-		$p<0.001$	$p=0.001$ (FDW>EVW)
							$p=0.001$ (SVW>EVW)
	Inlet design	$p<0.05$	$p<0.05$	-	$p>0.05$	n.s.	-
	Vegetation*Inlet		$p<0.01$	-		n.s.	-
λ	Vegetation type		n.s.	-		n.s.	-
	Inlet design	$p<0.05$	n.s.	-	$p>0.05$	n.s.	-
	Vegetation*Inlet		$p<0.01$	-		n.s.	-
N	Vegetation type		n.s.	-		$p<0.05$	$p=0.044$ (EVW>SVW)
	Inlet design	$p<0.05$	n.s.	-	$p>0.05$	n.s.	-
	Vegetation*Inlet		n.s.	-		n.s.	-
<i>mass</i>	Vegetation type		n.s.	-		$p<0.05$	$p=0.052$ (SVW>EVW)
<i>recovery</i>	Inlet design	$p>0.05$	n.s.	-	$p>0.05$	n.s.	-
	Vegetation*Inlet		n.s.	-		n.s.	-

^a Tukey HSD post-hoc test

Table 4. Effects of vegetation type on hydraulic parameters for each inlet type separately: effective volume ratio (e), hydraulic efficiency (λ), number of tanks in tanks-in-series model (N) and relative tracer mass recovery (*mass recovery*). For each parameter, p -values from Levene's test and one-way ANOVA are reported. Parameters were calculated using measured data and method of moments. In all cases, DF was 2. Abbreviations: n.s. = not significant; FDW = free development wetlands, EVW = emergent vegetation wetlands; SVW = submerged vegetation wetlands; b = barrier at inlet; nb = no barrier; DF = degrees of freedom. $N = 3$ in all cases.

Wetland type	Parameter	Levene's test	Measured data	
		p -value	p -value	Details ^a
With barrier (b)	e	$p>0.05$	$p<0.05$	$p=0.027$ (FDW _{b} >EVW _{b}) $p=0.047$ (SVW _{b} >EVW _{b})
	λ	$p<0.05$	n.s.	-
	N	$p>0.05$	n.s.	-
	<i>mass recovery</i>	$p>0.05$	n.s.	-
Without barrier (nb)	e	$p<0.05$	n.s.	-
	λ	$p>0.05$	$p<0.05$	$p=0.03$ (EVW _{nb} >SVW _{nb})
	N	$p>0.05$	n.s.	-
	<i>mass recovery</i>	$p<0.05$	$p<0.01$	$p=0.008$ (FDW _{nb} >EVW _{nb}) $p=0.022$ (SVW _{nb} >EVW _{nb})

^a Tukey HSD post-hoc test

3.3. Effects of RTD data analysis method on estimated hydraulic parameter values

Hydraulic parameter values for FDWs and EVWs without barrier were statistically affected by RTD data analysis method (Table 5; Fig. 3). For FDWs, *mass recovery* and *e*-values decreased with Gauss modelling, while the λ -values increased when modelling was applied (Table 5; Fig. 3). There were no statistically significant effects of RTD data analysis method on estimated hydraulic parameter values for the EVWs with barrier and SVWs (Table 5; Fig. 3).

Table 5. Effects of RTD data analysis method on estimated hydraulic parameter values: relative tracer mass recovery (*mass recovery*), effective volume ratio (*e*), hydraulic efficiency (λ) and number of tanks in tanks-in-series model (*N*). Parameter values were calculated using two data analysis methods: method of moments with measured data (M) and Gauss-modelled data (G) fitted with the Levenberg-Marquardt algorithm. *p*-values between the two data analysis methods and associated details are shown. Abbreviations: RTD = residence time distribution; n.s. = not significant; FDW = free development wetlands, EVW = emergent vegetation wetlands; SVW = submerged vegetation wetlands, *b* = barrier at inlet; *nb* = no barrier. *n* = 3 for EVW; *n* = 6 for FDW and SVW.

Wetland type	<i>mass recovery</i>	<i>e</i>	λ	<i>N</i>
FDW	$p < 0.01^a$ M > G	$p < 0.05^a$ M > G	$p < 0.01^a$ M < G	n.s. ^a
EVW <i>b</i> ^c	n.s. ^a	n.s. ^a	n.s. ^a	n.s. ^a
EVW <i>nb</i> ^c	n.s. ^{a,d}	$p < 0.05^a$ M > G	$p < 0.01^a$ M > G	n.s. ^a
SVW	n.s. ^a	n.s. ^b	n.s. ^a	n.s. ^a

^a paired t-test; ^b Wilcoxon signed ranks test; ^c values for effective volume ratio (*e*) were divided into two groups due to statistically significant lower relative tracer mass recoveries (*mass recovery*) in EVWs *nb* (see Table 4). ^d Levene's test *p*-value < 0.05.

4. Discussion

4.1. Tracer mass recovery

Kadlec and Wallace (2009) refer to mass recoveries of 80-120% as indicators of successful wetland hydraulic tracer studies. Thus, the majority of mass recoveries obtained in the present study indicated that the present method with lithium chloride was adequate for describing hydraulics in most of the studied wetlands (Table A in Appendix; Fig. 3d). Mass recoveries greater than 100% could be related to imperfections in both water flow measurements and frequency of Li^+ sampling, but are generally not addressed as problematic in hydraulic tracer studies (Martinez and Wise, 2003; Wang and Jawitz, 2006; Kadlec and Wallace, 2009).

However, all tracer mass recovery values from EVWs without barrier were below 80% (Table A in Appendix; Fig. 3d), which indicated problematic tracer behaviour and that hydraulic parameter values derived from these wetlands were associated with more uncertainty compared to the other wetland types. This was further supported by statistically significant differences between wetland types in tracer mass recoveries (Table 4; Fig. 3d).

The physical attributes such as height of inlet pipe above water level, wetland bottom slope, differences between inflow and residing wetland water temperature and concentration of injected tracer solution were similar for all wetlands. Thus, most likely, the low tracer mass recoveries in the EVWs without barrier were linked to the combination of emergent vegetation and the lack of an inlet barrier. Some of the physical wetland features may have aided the low tracer recoveries from EVWs without barrier. The height of the inlet pipe above the water level meant that inlet water fell with some force down towards the wetland bottom, and thereafter, in wetlands without barrier, may have followed the relatively steep slope (1:1) down to the maximum depth located just after the inlet pipe. This flow pattern may have been strengthened by the fact that incoming water was colder (and thus more dense) than residing water in the wetlands. In EVWs without barrier, immobilization of inlet water containing tracer at the bottom may have been caused by dense vegetation and litter

layers in combination with inlet water containing tracer having an increased density. Inlet barriers may have prevented this type of scenario as supported by higher mass recoveries in EVWs with barrier.

Thus, immobilization of the tracer in EVWs without barrier may have been aided by both stratification induced by top-to-bottom temperature differences and tracer solution densities as observed by Schmid et al. (2004). In addition, it has been reported that density-induced stratification effects in wetlands may be aided by low flow velocities (Schmid et al., 2004; Headley and Kadlec, 2007). Kadlec and Wallace (2009) consider differences in water temperature as a minor factor for creating stratification in FWS wetlands, because the top-to-bottom temperature variation is generally less than 1°C. However, in the present study, the difference in temperature between wetland water (10.7°C) and incoming water (8.8°C) was 1.9°C, which could have contributed to stratification.

Dierberg and DeBusk (2005) performed tracer studies in small (1L) batch vessels containing high organic sediments collected from cattail-dominated wetlands. The vessels were incubated for 7 days with agricultural drainage water and 106 µg Li⁺/L. In the cited study, 10-19% of added lithium was immobilized (in the cited study, referred to as adsorbed) after the incubation period and 19-21% of the initially immobilized was mobilized (in the cited study, referred to as desorbed) 1 day after the incubation period ended. This type of lithium behaviour could explain low tracer recovery and also the high y-values in the measured RTD tails from EVWs without barrier (Fig. 3d; Fig. 2j,k,l). Further, tracer at the bottom of the water column may have been pulled into the soil medium through the phenomenon known as “transpiration flux” of FWS wetlands as described by Kadlec and Wallace (2009).

Mass recovery results from the current study indicate that density-induced stratification may occur at low concentrations of injected tracer solution (600 g LiCl/m³ for the present study). In conclusion, high coverage of emergent vegetation and/or vegetation litter, in combination with

physical factors (height of inlet pipe, bottom slope, temperature- and density-induced stratification, and low water flow velocities), created conditions that were favourable for immobilizing and stratifying tracer fractions at the wetland bottom. These results revealed a potential methodological problem when implementing hydraulic tracer studies in wetlands with high cover of emergent vegetation. To avoid tracer mass recovery problems in small experimental wetlands dominated by emergent vegetation, proper spreading of incoming tracer water (e.g. by using a barrier) seems to be important.

Concerning EVWs without barrier, with the current tracer data sets, it was not possible to clearly distinguish hydraulic effects caused by vegetation from those produced by tracer methodological factors related to inlet design which affected tracer recovery. Thus, results related to hydraulic parameters (e , λ and N) specifically from EVWs without barrier are not discussed further in this paper.

4.2. RTD characteristics and data analysis method

Both measured and Gauss-based RTDs showed asymmetry, which can be expected for water flow through vegetated systems, indicative of both dispersion and “dead zones” that are inaccessible to the main flow (Fig. 2). These results were in accordance with observations made by Kadlec (1994) who stated that RTDs from FWS wetlands are normally characterized by a large amount of dispersion. Both measured and modelled RTDs of EVWs had an earlier tail formation compared to RTDs from the other two wetland types. This was an indication that EVWs contained more dead zones compared to the other wetland types (Thackston et al., 1987).

Many researchers have reported that vegetation causes water velocity heterogeneity in wetlands (Persson, 2005; Lightbody et al., 2008; Keefe et al., 2010). Multiple peaks observed in the RTDs based on measured data (Fig. 2) revealed that the wetlands experienced short-circuiting as described by Thackston et al. (1987), suggesting presence of water velocity heterogeneity. Visual

inspection of the RTDs suggested that the FDWs contained less marked multiple peaks compared to the other wetland types (Fig. 2a-f, measured data). This indicated that the multiple peaks in the RTDs of the EVWs and SVWs (Fig. 2g-r, measured data) were most likely related to the high coverage of emergent and submerged vegetation, respectively, in these wetlands (Table 2).

On the other hand, Gauss modelling removed all multiple peaks in the RTDs (Fig. 2), and resulted in RTDs that were generally characterized by lower variance compared to the measured ones (Table A in Appendix). However, as a consequence, some portion of data needed for accurate hydraulic quantification was lost due to the Gauss modelling. For example, the overall lower variance of the Gauss-modelled RTDs resulted in lower tracer mass recoveries compared to corresponding recoveries based on measured data (Table A in Appendix). Thus, in addition to using SSQE values (Eq. 12) to validate model quality and derived hydraulic parameters, comparisons between corresponding measured and modelled tracer mass recoveries should also be made. However, published literature reveals many studies where the latter comparisons are not considered (Wörman and Kronnäs, 2005; Wang and Jawitz, 2006; Min & Wise, 2009; Keefe et al., 2010). For the present study, the consequence of such comparisons showed that Gauss-based parameter values, especially from EVWs, were generally not reliable (Table A in Appendix). A Spearman rank order-correlation test conducted on Gauss-based N -values versus Gauss-based mass recoveries revealed a statistically significant negative correlation between the two parameters ($r_s = -0.652$; $p < 0.01$). Consequently, these N -values were considered as unreliable compared to the ones based on measured data and also compared to the other modelled N -values.

When measured data are well represented by a selected model, the objective model-fitting function should yield reliable modelled parameter values (Wang and Jawitz, 2006). Results from the present study generally indicated that measured data, especially from the EVWs, were imperfectly described by the Gauss model as shown by the high mean SSQE value (0.052). Nevertheless, the Gauss model produced more acceptable RTDs for the FDWs and SVWs (Fig. 2a-

f; m-r) as confirmed by lower mean SSQE values compared to those acquired from the EVWs. This result further supported the inference that, generally, Gauss-based hydraulic parameters from EVWs were less reliable than those from the other two wetland types.

Wang and Jawitz (2006) have reported that the SSQE function is good for capturing the mean behaviour of the RTD but not the tail region, and thus also not the variance. Hence, similarly, the most likely explanation for the diverging goodness-of-fit (i.e. different SSQE values) was that the Gauss model was less sensitive to measured RTD data with low variance compared to data with high variance and multiple peaks. Also, the Gauss model produced N -values with large standard deviation for the EVWs (Fig. 3c), indicating that this model was not stable for these measured data sets. In summary, results indicated that the Gauss model should be used with caution when applied to measured RTD data with high variance. However, the question remains at what level of RTD variance caution about the Gauss model should be implemented.

Indications of immobilization/mobilization behaviour of lithium added uncertainties to the RTDs based on measured data and subsequently on these hydraulic parameters. Gauss modelling offered a possibility to overcome lithium immobilization/mobilization problems reflected in many RTD tail regions based on measured data (Fig. 2). Also, an opportunity to stop field tracer sampling at an earlier stage was presented by Gauss modelling, although at the expense of lower tracer mass recoveries (Fig. 3d; Table A in Appendix).

Wang and Jawitz (2006) criticized the use of λ -values derived from RTDs with multiple peaks, since determining accurate t_p -values from such RTDs and few measured data points in the peak region is associated with uncertainty. Gauss modelling may provide an alternative for obtaining more confident λ -values from RTDs with multiple peaks. Within this context, Gauss-based λ -values in the present study were more plausible than the ones based on measured data (Table A in Appendix). In addition, this study showed that the use of RTDs derived from Gauss modelling instead of measured data may have an indirect effect on quantification of wetland

treatment by significantly affecting the magnitude of λ - and e -values as shown for FDWs (Table 5). Persson and Wittgren (2003) showed that the e -parameter was important for accurate modelling of nitrogen removal in wetlands using the TIS model. Thus, if the TIS is used to model FDW treatment performance, the final treatment assessment would be highly dependent upon which RTD data analysis method was used to calculate the e -value.

4.3. General hydraulic conditions

The e -values from the present study (Fig. 3a; Table A in Appendix) were on the same order of magnitude as those reported from other published hydraulic tracer studies in FWS wetlands (Dierberg et al., 2005; Kadlec and Wallace, 2009; Speer et al., 2009; Keefe et al., 2010). According to Thackston et al. (1987), wetlands with $0.5 \leq e \leq 0.75$ have a moderate amount of dead zones, whereas those with values above 0.75 have a small amount. This would mean that most wetlands in the present study, based on measured data, contained small amounts of dead zones. However, the mean Gauss-based e -value characterized wetlands with a moderate amount of dead zones.

N -values for FWS wetlands are generally in the range $0.3 < N < 10.7$ with a mean of $N = 4.1 \pm 0.4$ (Kadlec and Wallace, 2009). Thus, the range of N -values reported in this study (Fig. 3c; Table A in Appendix) was representative for FWS wetlands and not unusual for wetlands with open water areas due to wind-induced mixing (Kadlec and Knight, 1996). Still, since N -values generally were in the range of 1-2, it was indicated that relatively high mixing conditions were present in the experimental wetlands in this study. In fact, only the Gauss-based N -values from the EWVs were above 2.3, and as previously mentioned, the Gauss-based N -values from the EWVs were considered as unreliable.

The mean λ -value in the current study was similar to that reported by Dierberg et al. (2005) of 0.24 for a FWS wetland dominated by submerged vegetation, but lower than the λ -values from a FWS wetland dominated by emergent vegetation reported by Speer et al. (2009) of 0.74. Generally,

λ -values from the present study (Fig. 3b; Table A in Appendix) were indicative of wetlands with ineffective hydraulic conditions, i.e. $\lambda \leq 0.5$ (Persson et al., 1999).

The wetlands in the present study were small, with rectangular shape, and received low water velocities of 1.4 m/day (calculated as mean wetland water length divided by t_n). This would suggest that spreading of incoming water and mixing was aided both by the regular shape of the wetlands and by the low water velocities. Therefore, obtaining high e - and low N - values from such wetlands may not be surprising. LiCl tracer studies in large FWS wetlands (3-6 hectare) have also yielded high e -values in the range of 0.66-1.7 (Polychronopoulos and Chapman, 2001; Dal Cin and Bendoricchio, 2002). However, the N -values from the two cited studies were somewhat larger (2.7-5.9) than those from the current study. Thus, the hydraulic parameter values and results from the experimental wetlands in this study may not be directly applicable to hydraulics in larger wetlands with irregular shape and higher water velocities.

4.4. Effects of vegetation type and inlet design on estimated hydraulic parameters

Researchers have reported decreases in effective volume as an effect of vegetation density and vegetation layout. Computer simulations have shown that fringing wetland vegetation decreased e -values compared to full (Persson et al., 1999) or banded vegetation layout (Jenkins and Greenway, 2005). Also, empirical results have confirmed that an increase in fringing vegetation or increase in stem density of emergent vegetation decreased e -values in wetlands (Dal Cin and Persson, 2000; Bodin and Persson, accepted). Similarly, in the current study, emergent vegetation may have caused inefficient utilization of the wetland volume, as indicated by significantly lower e -values for emergent, compared to submerged and free development vegetation wetlands (Table 4). Bastviken et al. (2009) performed a study in the present wetland system and showed that EVWs had significantly higher nitrate-N area-specific mass removal compared to FDWs and SVWs. However, nitrate-N data analyzed in the mentioned study was from the years 2003-2006. The present study

indicated that emergent vegetation had a negative impact on wetland hydraulics. Still, it is well acknowledged that vegetation is positive for denitrification and thus overall nitrogen removal in wetlands (Kadlec and Wallace, 2009). Removing emergent vegetation to improve wetland hydraulics could negatively affect nitrogen removal. Thus, the question that remains to be answered is how the balance between effective wetland hydraulics and effective nitrogen removal is achieved.

Yet low e -values in wetlands may be a consequence of overestimated t_n -values (Eq. 1). Ideally, from a hydraulic point of view, vegetation volume should not be included when calculating wetland t_n -values (Kadlec, 1994; Whitmer et al., 2000; Smith et al., 2005; Headley and Kadlec, 2007; Ronkanen and Kløve, 2007). More specifically, because the tracer only flows in the wetland water, the t_m -value, logically, only reflects the wetland water volume. Consequently, when the tracer-based t_m -value is compared against the t_n -value, wetlands with lower vegetation volumes may yield higher e -values, especially when compared to wetlands with similar total volumes, but containing higher vegetation volumes (i.e. less water volume). Similarly, if the t_n -value is overestimated, because vegetation volume is not removed, the λ -value may be underestimated. However, from a practical point of view, it can also be argued that the vegetation volume should be included since it represents a part of the wetland volume, and that the influence of the vegetation volume on hydraulics thus should be reflected in e - and λ -values.

To account for emergent vegetation volumes in EVWs and place their e -values in the same magnitude as those from the other two wetland vegetation types, porosity fractions would have needed to be in the order of 0.73 and 0.95. These porosity fractions agree well with values recommended in literature to account for vegetation volume in wetlands (Whitmer et al., 2000; Kadlec and Knight, 1996). Thus, most likely emergent vegetation (both dead plant parts and living plants) caused lower e -values in the EVWs compared to the other wetland types. This pinpointed the significance of regular maintenance of wetlands to secure efficient utilization of the wetland volume, especially those containing dense emergent vegetation.

5. Conclusions

The present study demonstrated that:

- Hydraulic tracer studies performed in small experimental wetlands with large cover of emergent vegetation may exhibit tracer methodological problems that negatively affect tracer mass recoveries. It was suggested that such problems may be avoided by apt spreading of incoming tracer solution, e.g. by using an inlet barrier.
- Gauss modelling produced RTDs with lower variance compared to the method of moments, illustrating that portions of data needed for accurate hydraulic quantification were lost with the former method. Also, the Gauss model was unreliable for describing mixing conditions (N -values) for RTDs with high variance. Still, more reliable λ -values were demonstrated by applying the Gauss model to RTDs with multiple peaks. In addition, Gauss modelling illustrated a possibility to minimize uncertainties in measured hydraulic parameters caused by lithium immobilization/mobilization.
- The experimental wetlands in the present study were small, regularly shaped, and had low water flow velocities. This resulted in comparatively high e - and low N -values, showing that hydraulic parameter values from small experimental wetlands may not be directly relevant to hydraulics in larger wetlands.
- The RTD data analysis method significantly affected hydraulic parameter values. This indicated that the method of analyzing RTD data can be important for evaluation of treatment wetlands.
- Lower effective treatment volumes in emergent vegetation wetlands compared to other wetlands were most likely caused by dead plant parts and living plants. This indicated the need for

systematic maintenance of wetlands with dense emergent vegetation to secure efficient volume utilization for treatment.

Appendix

Supplementary material associated to this article can be found in Table A, Appendix

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Table A. Results from the hydraulic tracer study during 24 September – 3 October 2008 in the experimental free water surface wetland system near Halmstad, Sweden. The system consisted of 6 free development wetlands (FDWs), 6 emergent vegetation wetlands (EVWs) and 6 submerged vegetation wetlands (SVWs), with (*b*) or without barrier (*nb*) at wetland inlet. Shown are effective volume ratio (*e*), hydraulic efficiency (λ), number of tanks in TIS model (*N*), mean hydraulic residence time (t_m), time for peak tracer concentration in the RTD distribution (t_p), variance of RTD distribution (σ^2) and relative tracer mass recovery (*mass recovery*). Values are calculated with measured data and the method of moments (M) or with Gauss modelling (G) using equation Eq. 11. Abbreviations: TIS = tanks in series; RTD = residence time distribution.

Treatment	Wetland no	Inlet design	<i>e</i> (-)		λ (-)		<i>N</i> (-)		t_m (hour)		t_p (hour)		σ^2 (hour ²)		<i>mass recovery</i> (% of added)	
			M	G	M	G	M	G	M	G	M	G	M	G	M	G
FDW	3	<i>b</i>	0.79	0.67	0.26	0.31	2.0	2.1	78	67	26	31	2969	2154	92	79
	8	<i>b</i>	0.72	0.65	0.23	0.26	1.9	1.8	79	72	26	29	3372	2850	112	100
	13	<i>b</i>	0.86	0.85	0.28	0.37	2.1	2.1	92	91	30	39	4102	3965	85	80
	2	<i>nb</i>	0.79	0.74	0.25	0.33	2.3	2.0	82	75	26	33	2967	2828	97	86
	11	<i>nb</i>	0.82	0.74	0.26	0.35	2.1	2.1	79	74	26	35	2985	2538	97	87
	16	<i>nb</i>	0.77	0.76	0.22	0.27	1.9	1.7	88	87	26	31	4153	4335	92	87
EVW	6	<i>b</i>	0.66	0.43	0.30	0.35	1.7	7.6	67	42	30	34	2605	234	101	68
	12	<i>b</i>	0.63	0.54	0.27	0.27	1.6	2.0	59	51	26	26	2226	1284	110	101
	15	<i>b</i>	0.69	0.47	0.27	0.33	1.9	4.0	77	54	30	38	3213	716	82	61
	1	<i>nb</i>	0.82	0.49	0.52	0.41	2.0	7.4	78	47	50	39	3017	303	77	51
	9	<i>nb</i>	1.07	0.63	0.52	0.43	2.1	3.5	115	70	56	47	6359	1397	78	54
	18	<i>nb</i>	0.83	0.55	0.27	0.16	1.3	1.4	78	52	26	15	4618	1986	77	72
SVW	4	<i>b</i>	0.78	0.78	0.42	0.27	1.8	1.7	81	82	44	28	3755	3874	102	99
	7	<i>b</i>	0.76	0.73	0.26	0.27	1.8	1.7	75	71	26	26	3037	2928	92	87
	14	<i>b</i>	0.78	0.70	0.46	0.34	1.9	2.2	73	68	44	33	2805	2142	84	76
	5	<i>nb</i>	0.69	0.73	0.16	0.13	1.6	1.3	82	87	20	16	4134	5637	83	87
	10	<i>nb</i>	0.79	0.82	0.20	0.20	1.9	1.6	75	79	20	19	3015	3957	98	101
	17	<i>nb</i>	0.80	0.63	0.25	0.31	1.8	2.1	80	64	26	31	3492	1939	94	76