AN ABSTRACT OF THE THESIS OF

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Title: Flow Characteristics of Constructed Wetlands: Tracer Studies of the Hydraulic Regime

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Abstract Approved: ______________________

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Treatment efficiency in a constructed wetland is related in part to the amount of time that a wastewater remains in the system. Current design methods idealize the system as a plug flow reactor and use a "residence time" based solely on the volume of the cell and the flow rate. Under this assumption, every element of wastewater entering the wetland cell experiences the same residence time.

It is understood that this idealization ignores the existence of longitudinal dispersion, short circuiting and stagnant regions within the wetland cell. The result of these phenomena is a *distribution* of residence times. In other words, portions of the effluent exit the cell earlier than predicted, resulting in undertreatment, and portions exit late, resulting in excess treatment. The average concentration of treated wastewater at the outlet is a function of this distribution and the reaction kinetics associated with the waste. The overall effect of a distribution of residence times is reflected in a reduction of treatment efficiency at the outlet. Hydraulic regimes of constructed wetland systems were investigated at a pilot project site providing tertiary treatment of a pulp mill wastewater. Two vegetation types, bulrush and cattail, were investigated and compared to nonvegetated and rock-filter cells with identical configurations. Tracer studies used a fluorescent dye and were performed over the course of a year. Dye was input as a pulse at the inlet end of the cell and sampled over time at the outlet end to obtain concentration breakthrough curves. From these curves, time to peak, actual mean detention times, degree of dispersion, and extent of dead space were calculated, as well as predicted treatment efficiency.

Results indicated varying degrees of dispersion, short circuiting, and dead space in the individual cells. Analysis of the residence time distributions provided estimates of the "active" volume of the treatment cell and the degree of short circuiting in the system. Effective volume of the planted cells ranged from 15 to 25% of full volume. Early arrivals

of the peaks of the distibutions, indicative of short circuiting, ranged from 30% to 80% of the theoretical detention times. A first order treatment model and a kinetic coefficient, k, were assumed, and corresponding treatment efficiencies were compared to the theoretical treatment of an ideal plug flow reactor. Reduced treatment efficiencies for the planted systems ranged from 2 to 20 %, by this estimation.

Many references attempt to analyze wastewater treatment systems by refering to two models: dispersed plug flow and an approximation of tank-in-series. These models were investigated as potential descriptions of the hydraulic regime present in constructed wetlands. Residence time distributions of the constructed wetlands in this study indicated flow was not exclusively dispersed plug flow. This simplified model does not account for the exchange of material with "dead" space in the wetland cell. The data suggest a combination model of dispersed plug flow with a transient storage zone component may be more appropriate.

Flow Characteristics of Constructed Wetlands:

Tracer Studies of the Hydraulic Regime

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Introduction

Constructed wetlands (CW) have received considerable attention of late as effective systems for improving water quality. This low-cost, low-maintenance technology has been used predominantly in secondary and tertiary treatments of municipal wastewater, but has also been successfully employed in treating a wide variety of agricultural and industrial wastewaters, ranging from high nutrient livestock wastes to heavy metal leachates.

The wetland system is most simply characterized as a fixed-film biological reactor. Although macrophytes are a key element of the system, the treatment of organic wastes is presently attributed to the microbes that are established on the plant roots, rhizomes and stems, and in the litter layer. The role of the plants is believed twofold: first providing a surface on which microbial populations can attach; and second, transporting oxygen through the stems and to the rhizosphere (root zone) providing an aerobic "micro"-layer in the reduced environment of the wetland.

There are two types of constructed wetland system: Free Water Systems (FWS), and Subsurface Flow Systems (SFS). This paper will focus on the hydraulics of free water systems, although the methodology presented here can, and has been, used on SFS's.

If constructed wetlands are viewed as biological reactors, certain generalities adapted from chemical reactor theory can be applied. In any reactor, there are two primary components of concern, kinetics and contacting pattern. **Kinetics** describe how fast things happen; **contacting pattern** describes how elements move through a system and contact each other. The coupling of the reaction kinetics of a reactor with its contacting pattern comprise the performance equation, relating output to input, or,

Output = *f* (input, kinetics, contacting).

This is an important expression for comparing various reactor designs and conditions. Evaluation of a reactor's performance invariably centers on these two processes, although kinetics often receive the most attention.

The efficiency of biological treatment is partly dependent on the occurrence, and duration, of contact between the pollutants and the microbial populations. This concept is common to any reactor system: the degree of treatment is directly related to the residence time and efficiency of contact in the reactor. To obtain maximum treatment efficiency, it is necessary to maximize contact between the wastewater contaminants and the treatment (in this case the plant surfaces) and minimize short circuiting. (Steiner & Freeman, 1989). Short circuiting is defined as the existence of preferential paths within the hydraulic regime. These "channelizations" allow certain elements of flow to pass through the treatment system ahead of, or faster than, the theoretical detention time of the system. The theoretical detention time is a volumetric calculation, which ignores obstructions, stagnant regions, and velocity gradients. Consequently, if a system is designed based on the theoretical detention time, the elements of the flow that are treated for less time are ignored. By engineering these systems with proper consideration of the hydraulics, the efficiency of the system can be substantially improved.

This all leads to a critical and perhaps insufficiently understood aspect of constructed wetlands design: the hydraulic regime. Currently used hydraulic design criteria in the field of constructed wetlands are largely "rule of thumb". Hydraulics are often idealized as plug flow, resulting in a treatment potential that is limited by the "current incomplete knowledge of hydraulic characteristics" (Fisher, 1990). The highly variable treatment results generated in the field suggest that certain aspects are not being optimized, nor adequately understood (Fisher, 1990). While hydraulic residence time has been widely used as a design parameter, adequate relationships between hydraulic detention time, removal mechanisms, and process performance are unavailable (Weber & Tchobanoglous, 1985). Contemporary research in constructed wetland technology needs to eliminate the "black box" method of measuring success, endeavoring to define the complicated, interrelated processes that exist.

By injecting a conservative tracer into the system, a "visual" assessment of the hydraulic regime can be obtained. Tracer methods have been used extensively in chemical reactor analysis and have been employed frequently in more conventional wastewater treatment technologies, like stabilization ponds. There have been some studies performed on constructed wetlands (Bowmer, 1987; Fisher, 1990), but few have attempted to model the hydraulic regime as an analogous reactor with nonideal flow.

LITERATURE REVIEW

When predicting treatment efficiency for wastewater treatment systems, the most common approach is to assume that the flow pattern closely resembles one of the two ideals: plug flow or complete mixed. Stabilization ponds are generally modeled as complete-mix reactors when the shape is circular or square (Moreno, 1990), while those with a 3:1 or 4:1 length to width ratio more closely resemble plug flow (Reed et al, 1988). Constructed wetlands are usually modeled as plug flow reactors. In reality the flow pattern in these surface flow systems is somewhere in between these two ideals neither plug flow nor completely mixed (Thirumurthi, 1969). The same generally holds true for constructed wetlands.

As the flow patterns of a reactor or treatment system deviate from ideal assumptions, conversion, or treatment efficiency, is often compromised. The extent to which the reactor deviates form ideal can be easily determined by tracing a conservative substance through the system. A "stimulus-response" technique is often used to evaluate hydraulic deficiencies in the system.

Dye Tracers

A tracer is defined as any substance, preexisting or introduced, used to assess some characteristic of a system. In the case of hydraulic studies, there are several attributes of the tracer that Denbigh & Turner (1984) regard as important. First, a tracer should not alter the existing flow of the system and should be conservative (stable). By conservative, it is assumed that the tracer is, for the most part, nonreactive and nonsorptive. Second, analysis of tracer concentration should be feasible above background levels and convenient to perform. Lastly, the tracer should be evenly distributed in the fluid and representative of a comparable unit of that fluid. The density of the tracer should be similar to the density of the fluid being traced, assuring minimal alteration of the hydraulic regime by the tracer itself.

Fluorescent dyes are commonly used in tracer studies of ground- and surface water systems. Fluorescence is the property of a substance that absorbs excitation energy as light at a particular wavelength and then releases it at a lower wavelength. Many naturally occurring substances have this property, so a particular arrangement of light source and filters are used in a *fluorometer* to minimize background effects. A light source is chosen that provides the required excitation energy to facilitate fluorescence. This source is directed through a primary filter that limits the light to the peak excitation wavelength for the particular dye. Light passes through the sample and is reflected at 90° through a secondary filter that narrows the emission to the desired range of fluorescence. Finally the remaining light released is measured on a photomultiplier. There are several sources of detailed information on fluorescent dyes and fluorometric techniques, including Smart & Laidlaw (1977), Wilson (1986) and Gaspar (1987).

Relative cost and ease of detection typically are the reasons for choosing fluorescent tracers. Rhodamine WT has been praised by many sources as one of the more stable (nonreactive) water tracers (Wilson, 1986; Smart, C.C., 1988). Among the advantages expounded by Smart & Laidlaw (1977) were its ease of detection, reasonable stability in a normal water environment, and detectability at very low concentrations. Wilson et al. (1986) added that the dye was only "slightly susceptible to adsorption in most situations", and a general affinity towards organic, but not inorganic, sediments. Numerous studies have cited the conservative nature of rhodamine WT in analysis of tracing experiments. Fox et al. (1991) selected the dye because it was "not adsorbed by plants or sediments". Steinback (1987) contended that rhodamine was "developed to be particularly insensitive to (sorption) by bottom sediments or suspended solids".

Unfortunately there have been equally numerous studies demonstrating that rhodamine WT is susceptible to adsorption and "masking" problems. Most of the assertions to rhodamine's suitability as a water tracer cite the Smart & Laidlaw paper (1977), yet, in a more recent paper, P.L. Smart (1985) recognizes the difficulties of using a fluorescent tracer in contaminated waters. The most significant problem arises when high concentrations of dissolved organic compounds, many of which fluoresce, produce a "masking effect" of the actual dye concentration. Smart further acknowledges that fluorescent dyes, notably rhodamines, adsorb onto surfaces, particularly amorphous oxides and organic solids. In tracer experiments of surface waters (Jones & Jung, 1990; Bencala, 1983), researchers have indicated the appreciable capacity for sorption of rhodamine to streambed sediments. In column experiments comparing rhodamine to bromide, which is generally considered conservative, and lithium, which is generally considered sorptive, rhodamine displayed characteristics similar to the later (Everts et al., 1989), with a reported sorption coefficient, K_{OC}, between 1400 to 3700. Column experiments by Finker & Gilley (1986) investigated the adsorption of rhodamine on upland sediments. Some researchers have even used rhodamine WT to mimic the sorptive characteristics of pesticides in solute transport studies (Sabatini & Austin, 1991).

Steinback (1987) used rhodamine WT in flow calibrations in a pulp mill and also noted the masking effect of the opaque effluent. By using higher doses of the rhodamine dye and diluting the samples with distilled water he reduced the effect of solids blocking light transmission. High concentrations of free chlorine, typical of bleach kraft mill wastewaters, is suspected of bleaching out the fluorescent dyes (Wilson, 1986; Steinback, 1987).

Another method used to overcome masking effects is to make fluorometer standards using water or effluent from the system being traced as a diluent rather than distilled water (Wilson, 1986). This negates the effect of any reactions of the dye with substances in the traced system. The downfall of this technique occurs when the quality of the water being analyzed is not consistent. Standards have to be made often to account for variations in the wastewater and discrepancies in the effect. Regardless, Steinback (1987) chose to use this method to alleviate corrections in his tracer studies in pulp mill effluent.

The above evidence indicates the need for some caution when analyzing residence time data obtained with fluorescent dye tracers. Although **masking and bleaching** effects of contaminated water influence the mass balance (ie. dye recovery) of the system, estimates of detention time and dispersion are still accurate, provided the breakthrough signal is strong enough to be distinguished from the background. To overcome the masking effect, P.L. Smart (1985) also suggests the use of higher doses of tracer to give a stronger breakthrough signal. **Sorption**, however, being a kinetic process and therefore time variant, can potentially distort time-of-travel studies. The resulting long-tailed breakthrough curve produces time parameters that are in error (Levenspeil, 1993). It is for this reason that mass balances on the traced system are crucial to validating time-of-travel studies where sorption is suspected. If there is a substantial quantity of dye unaccounted for, time-of-travel parameters are in question. Furthermore, temporary adsorption can be misinterpreted in tracer studies as dead space (Grobicki & Stuckey, 1992).

Reactor Theory & Tracer Studies

There are two ideal types of reactor systems: the plug flow reactor (PFR) and the continuous stirred tank reactor (CSTR). In a plug flow reactor fluid moves in the chosen spatial direction with no attempt made to induce mixing "between the elements in the direction of flow" (Denbigh & Turner, 1984). In other words, any one position in the system is time invariant in composition- changes in composition are from one location to the next. Plug flow reactors are ideally mixed uniformly in the lateral direction and unmixed in the longitudinal (Tchobanoglous, 1987). In effect, all fluid elements entering the reactor will experience the same residence times . This is often referred to as "piston flow" (Danckwerts, 1953). Constructed wetland treatment systems are most often designed on the premise that they behave as PFRs. It should be stated now that these assumptions and indeed the analyses of the constructed wetland system are based on the supposition that steady state flow prevails.

In a completely mixed reactor, the fluid properties are uniform throughout the reactor, and everywhere identical to the outgoing elements. In this ideal case, it is assumed that the fluid is instantly dispersed throughout the reactor at time $t = 0$.

Consider a conservative tracer injected as an instantaneous pulse into an ideal plug flow reactor. The injection time of the tracer should be short in comparison to the theoretical residence time of the system. If the outlet of the reactor is sampled at periodic intervals beginning at the time of injection, a plot of the tracer concentration vs. time would appear as in Figure 1. This type of stimulus-response curve, called a flow-through curve, depicts the concentration of the dye at the outlet after an interval of time. If there is no longitudinal dispersion (along the flow path), the tracer moves through the system as a pulse. The dye appears as a "spike" with a width of zero at the outlet. The time from injection to when the spike appears is referred to as the **theoretical hydraulic residence time**. This residence time can be calculated by dividing the volume of the system, V, reduced by the porosity fraction, η, by the flow-through rate, Q:

$$\bar{t} = \frac{V\eta}{Q} \tag{1}$$

The porosity fraction, η, is defined in a manner similar to soil systems as the ratio of the volume of voids to the total volume. Insufficient data exists on the void fractions of wetland systems, which in turn makes it difficult to accurately calculate detention times (Kadlec, 1987). Typically, calculated detention times pay no heed to the volume occupied by biomass and detritus (Weber & Tchobanoglous, 1985). Additionally, Kadlec (1981)

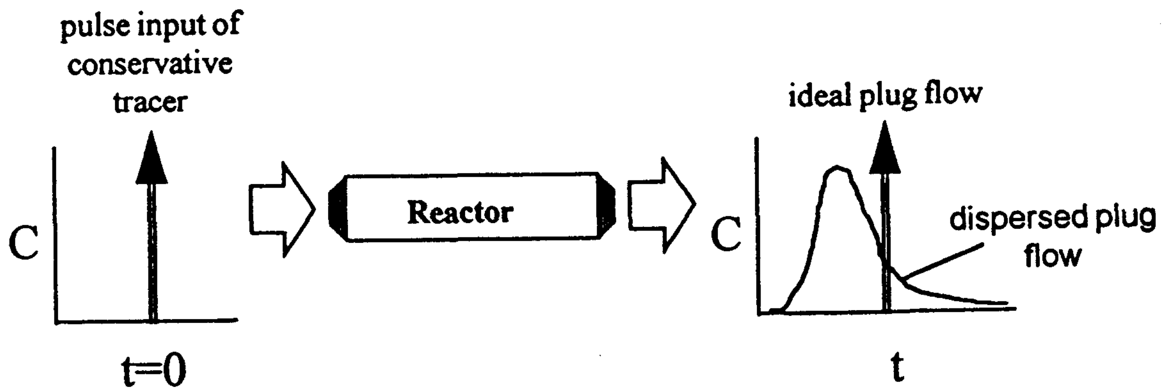

Figure 1. Stimulus- response technique of tracer injection

refers to the "doubly porous" medium of vegetated media: small scale obstructed flow, as *through* plant hummocks; and larger scale channelizations, like flow *around* hummocks and islands. Kadlec suggests the cautious use of a superficial velocity and residence time, which is calculated ignoring the porosity term.

If porosity is to be used in calculations, the literature suggests a wide range of values. Part of the decrepency results from the definition that one uses for porosity. An actual porosity accounts for only the volume occupied by plant matter, whereas an effective porosity accounts for all hydraulic inefficiencies in the system. Watson and Hobson (1989) report porosities of 0.90-0.95 for typha, and 0.86 for bulrush, whereas Reed (1988) uses a porosity for the general case of 0.75. The later value was determined from dye study data in Listowel and Arcata, for a "moderately dense stand" of emergent vegetation. The volume occupied by the vegetation and litter was determined to be about 5%. Kadlec (1990) reported variation in void fraction along the height of the plant stems from 99% at midheight to 90 % in the litter layer.

A correction for the active, or effective, volume of the cell must account for more than just the volume "unavailable" to flow (ie. occupied by some stationary object). Dead space can also result from purely hydraulic phenomena, such as witnessed in eddies and pools of a mountain stream. The result of "hydraulic" dead space is to decrease the active volume of a treatment system. Grobicki and Stuckey (1992) separate dead space into two phenomena: biological and hydraulic dead space. Correcting simply for porosity does not account for other portions of the reactor unused hydraulically (in between plants, in the corners and at the edges of ponds).

Nonideal Flow and Residence Time Distributions

In the "nonideal" world most systems deviate considerably from either of the two ideal reactors previously mentioned (Tchobanoglous & Schroeder, 1885). In some situations, a CFSTR or plugflow reactor approximation may produce negligible error. In poorly designed systems, however, substantial miscalculation may result. Channeling of the fluid, stagnant regions, longitudinal dispersion, and density stratification may result in deviations from assumptions used in design (Levenspeil, 1972). The more a system deviates from the ideal, the wider the distribution of the tracer output curve. To illustrate this concept, Figure 2 shows the flow-through curves for reactors with various magnitudes of longitudinal dispersion. This distribution of residence times indicates an inefficiency in a reactor, requiring a much larger reactor volume than if the plug flow ideal was met (Denbigh & Turner, 1984).

The residence time distributions are experimentally determined by the injection of a pulse input of dye at the inlet end of a reactor and measuring the concentration of dye over time at the outlet end. A plot of the concentration readings vs. time is referred to as the "C-diagram". This graph gives an indication of the nature of fluid movement through the reactor. If the tracer used is considered stable and conservative, the C-diagram provides an accurate representation of the hydrodynamics of the system (Grobicki & Stuckey, 1992).

The area under the residence time distributions should be equivalent to the mass of die input into the system divided by the volumetric flow rate, or,

$$\text{Area} = \int_0^\infty C\,dt = \sum_0^\infty C_i \cdot \Delta t_i = \frac{M_t}{\upsilon} \qquad (2)$$

Dimensionless Residence Time Distributions:

To facilitate the comparison of reactors with varying performances, a normalized plot of the C-diagram, referred to as the E-diagram, or exit age distribution (Levenspeil, 1972), is constructed. To normalize the concentration readings we divide the readings, C_t,

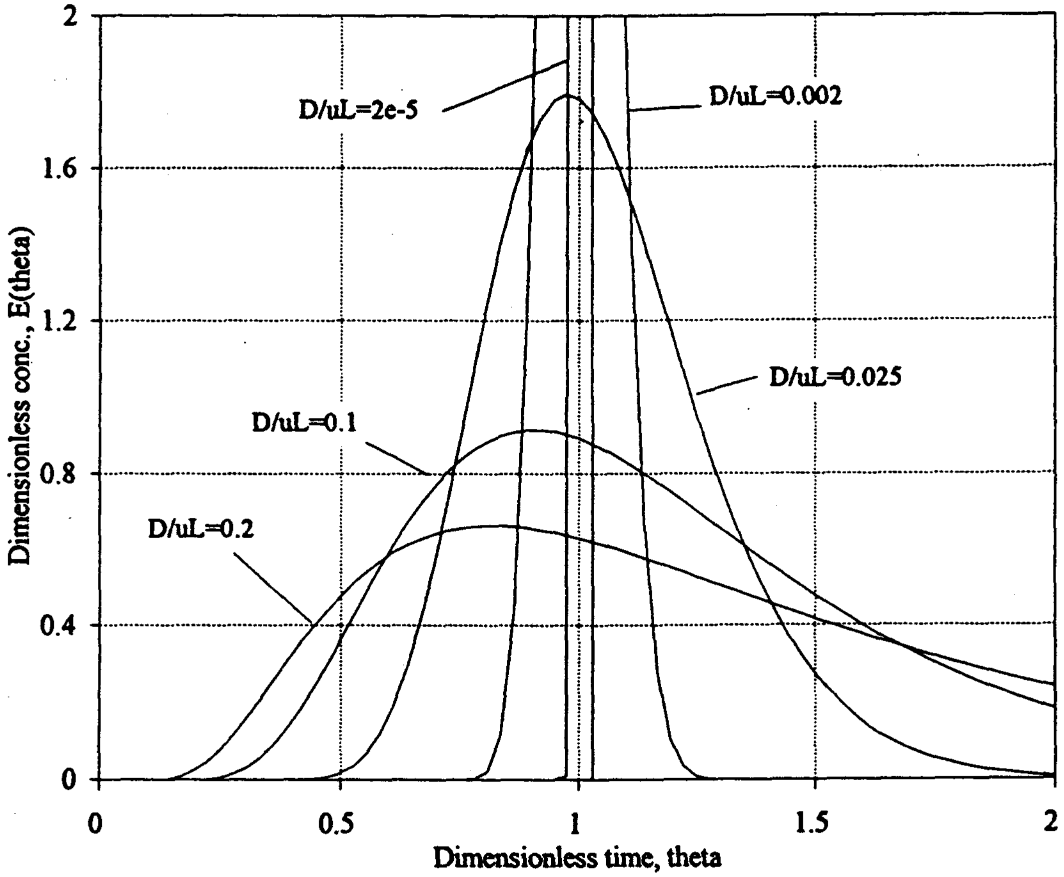

Figure 2. Dispersed plug flow for varying dispersion number

by the initial concentration of dye added, C_o, where,

$$C_o = \frac{M_t}{V} \tag{3}$$

where,

M_t = mass of dye added to reactor

V = volume of reactor.

Thus, the new concentration scale is given as

$$E_\theta = \frac{C}{C_0} = \left(\frac{V}{M_t}\right) \cdot C \tag{4}$$

The normalized time scale, θ, is calculated as the residence time of each fluid element, t_i, divided by the theoretical residence time, $\bar{t}$, as defined above, or,

$$\theta = \frac{Q}{V} \cdot t = \frac{t}{\bar{t}} \tag{5}$$

Note that the area beneath the plot of E_θ vs θ should be equal to unity (Danckwerts, 1953),

$$\frac{V}{Q} \int_0^{+\infty} C_\theta \, d\left(\frac{\upsilon\theta}{V}\right) = 1 \tag{6}$$

The exit age distribution gives the fraction of the flowing stream spending time t + Δt in the reactor.

Mean residence time and variance:

To characterize the residence time distributions, it is convenient to define a few numerical parameters. The **mean residence time,** $\bar{t}$, and the **variance,** σ^2, can be evaluated from the C vs. t curve (Figure 3).

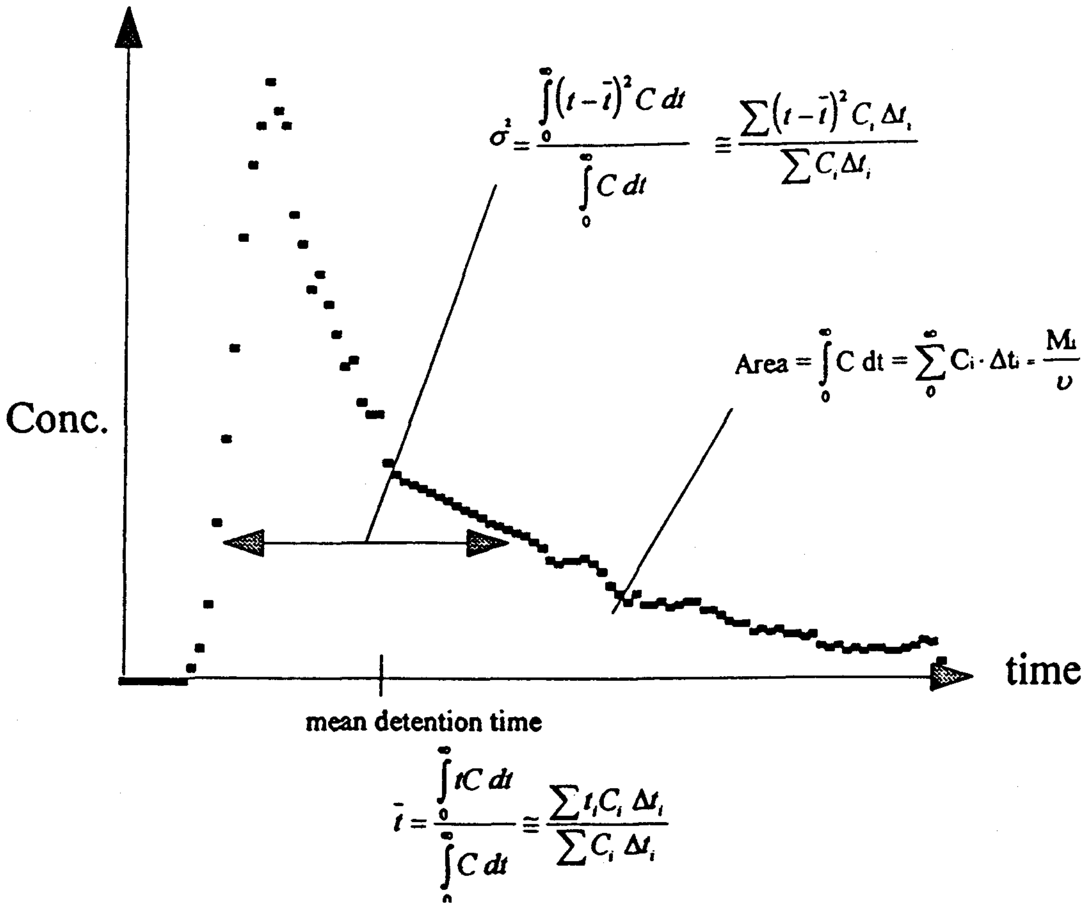

Figure 3. Determining parameters of residence time distribution

The mean is a location parameter describing the centroid of the distribution. For a C vs. t curve the mean is given for discrete time values as (Levenspeil, 1972)

$$\bar{t} = \frac{\int_0^\infty tC\,dt}{\int_0^\infty C\,dt} \cong \frac{\sum t_i C_i \Delta t_i}{\sum C_i \Delta t_i} \tag{7}$$

The variance of the distribution gives an indication of the spread of the residence times and is defined as

$$\sigma^2 = \frac{\int_0^\infty (t-\bar{t})^2 C\,dt}{\int_0^\infty C\,dt} = \frac{\int_0^\infty t^2 C\,dt}{\int_0^\infty C\,dt} - \bar{t}^2 \tag{8}$$

Or in discrete form,

$$\sigma^2 \cong \frac{\sum (t-\bar{t})^2 C_i \Delta t_i}{\sum C_i \Delta t_i} \cong \frac{\sum t_i^2 C_i \Delta t_i}{\sum C_i \Delta t_i} - \bar{t}^2 \tag{9}$$

The variance represents the square of the spread of the distribution and has units of $(\text{time})^2$.

Dead Space:

The existence of stagnant, eddy regions in a reactor reduces the active volume of a treatment system. These "no-flow" regions typically occur in the corners and at the edges of lagoons and stabilization ponds, but in the obstructed flow of a vegetated system like a constructed wetland, they can exist throughout the system. Recall the discussion of the "doubly porous media" proposed by Kadlec (1981), alluding to larger main channels of flow around plant hummocks, and the smaller channels of flow through the plant hummocks between individual plants. Indeed, depending upon the actual nature of the plants used, the variation in plant densities and resulting array of local velocities can discourage any attempt at quantifying the volume of the wetland system that is active, and the volume that is stagnant. To account for the exchange of material between active and dead regions, several authors have attempted to incorporate a mass exchange approximation into flow models (Valentine & Wood, 1979; Bencala 1983; Seo, 1992). Because of the complexity, and uncertainty, in determining mass exchanges in most systems, many other authors have chosen to ignore the exchange, asserting that, in general, the quantity exchanged compared to the bulk flow is usually negligible in hydraulic analysis (Tchobanoglous and Schroeder, 1985). The net effect of such an assumption is to approximate the system with a reactor of reduced volume, sometimes termed the "effective" or "active" volume of the treatment system.

From the analysis of the residence time distributions, the first moment of the area beneath the curve represents the mean detention time, $\bar{t}$, of the system. If the flow rate, Q, was calculated accurately, then the following relationship can be used to approximate the active volume of the system:

$$V_a = \bar{t} \cdot Q \tag{10}$$

This relationship does not include a correction for the volume of the system occupied by plant matter (living and detrital), so in effect it calculates the reduction due to both the volume occupied by plant matter (biological dead space), and the volume attributed to eddies and stagnant flow (hydraulic dead space).

Treatment Efficiency:

Treatment efficiency of a reactor depends on two factors: the kinetics of the reaction, or rate, and the residence time. In the ideal reactors, all fluid elements experience the same residence time. In the nonideal case, there is a *distribution* of residence times; some elements are detained in the reactor longer than expected (stagnant flow),and some are moved through faster than expected (short circuiting). The effect of a distribution of residence times is a decrease in overall treatment efficiency. In almost all cases, deviations from ideal flow patterns results in decreased efficiency (Levenspeil, 1974).

To quantify this decrease in treatment efficiency, we can combine an expression for the kinetics of the reactor with the RTD. If the conversion of the material is assumed to be first order, then the rate of reaction is solely dependent on the time that the material spends in the reactor. The exit age distribution, or E-curve, provides an estimate of the time that each portion of the total flow spends in the reactor. Combining this with the expected conversion of a reactant at each time, t, we have (Levenspeil, 1972),

$$\begin{pmatrix}\text{mean concentration}\\ \text{of reactant}\\ \text{in exit stream}\end{pmatrix} = \sum_{\substack{\text{all elements}\\ \text{of exit}\\ \text{stream}}} \begin{pmatrix}\text{concentration of}\\ \text{reactant remaining}\\ \text{in element of age}\\ \text{between t and}\\ \text{t + dt}\end{pmatrix} \cdot \begin{pmatrix}\text{fraction of}\\ \text{exit stream}\\ \text{of age between}\\ \text{t and t + dt}\end{pmatrix} \qquad (11)$$

or, in symbols,

$$\overline{C_A} = \int_{t=0}^{\infty} C_{A,element}\, E\, dt \qquad (12)$$

in which,

$C_{A,element}$ = concentration of reactant as a function of time
$\overline{C}_A$ = mean concentration of material leaving the reactor unreacted
E = exit age density function

An estimate of the conversion of a substance in the reactor is obtained from a kinetic expression. If the reaction were assumed to be irreversible and first order, for example, the expression is,

$$C_{A,element} = C_{A_O} e^{-kt} \qquad (13)$$

where C_{A_O} = initial concentration

Substituting this rate expression in the above Eqn. (12) we have the unconverted fraction,

$$\frac{\overline{C_A}}{C_{A_O}} = \int_{t=0}^{2} e^{-kt} \, \mathbf{E} \, dt \qquad (14)$$

This expression gives the fraction of material that has not undergone reaction at □t = ∞. The extent of conversion can be related as:

$$\% \text{ treatment efficiency} = 100\left[1 - \frac{\overline{C}_A}{C_{A_O}}\right] \qquad (15)$$

When modeling the treatment kinetics with the first order reaction usually suggested (Kadlec, 1989), a good estimate of the actual residence time of the system is required. The conversion of BOD_5 (Biochemical Oxygen Demand) can be used as an example. A typical value for k of 0.23 day^{-1} (Metcalf & Eddy, 1991) and an initial concentration of BOD_5 can be used in the rate equation to determine treatment efficiency. Using these values together with the exit age distributions, a comparison between reactors with different characteristics can be made in reference to treatment efficiency. This in turn can be compared to the hypothetical BOD conversion based on a plug flow assumption, using the theoretical detention time in (1) and the first order rate equation (13).

Flow Models

Although the data obtained from a tracer experiment can be used directly to obtain some meaningful parameters (ie. mean detention time, % of dead space, and degree of short circuiting), it is often beneficial to employ one or more flow models in the analysis. Theoretical flow models can increase understanding of the phenomena leading to certain attributes of flow. The most often used models are derived from two approaches: a one dimensional simplification of the Fickian convection-dispersion equation and the quasiphysical, compartmental approach. Levenspeil (1972) introduces these two techniques as dispersed plug flow and tank-in-series. Although these models provide some flexibility compared to the ideal assumptions, they are still simplified representations of what is actually occurring in the systems. In, general, the interactions of dispersion and mixing are far too complex to describe mathematically (Gaspar, 1987). Velocity patterns are difficult to quantify because of spatial nonuniformity (Weber & Tchobanolous, 1985). Due to channelizations and preferential flow paths, localized velocities can greatly exceed those predicted. Nevertheless, the dispersed plug flow and the tank-in-series models have been used to provide reasonable approximations of the varying degrees of mixing characteristic of nonideal flow.

The investigation of the flow regime can begin with a description of the flow as either laminar or turbulent. Laminar flow is characterized by smooth flow lines and predominately convective transport (as in pouring honey or oil), whereas turbulent flow denotes mixing between layers and dominance of random diffusional processes . The transition between these two phases of flow depends on the velocity of the fluid, the cross-sectional area of flow, and the viscosity of the fluid. Although predicting the exact point of transition is somewhat elusive, the dimensionless **Reynold's number** is typically used for an estimate of which type of flow is present. The Reynold's number, R_e, is a ratio of the inertial forces to the viscous forces of the fluid. At high Reynold's numbers, the inertial forces prevail, yielding turbulent flow, at lower Reynold's numbers laminar flow exists. For open channel systems this relationship is given as follows:

$$R_e = \frac{Vd}{\nu} \tag{16}$$

where,

V = mean velocity of flow
d = depth of flow
ν = kinematic viscosity of fluid

In open channels, the transition from laminar to turbulent flow was determined experimentally to occur around Re equal to 500 (Roberson & Crowe, 1985). Typical Reynold's numbers in wetlands have been reported in the range of 1-100 (Hammer, DE & Kadlec, 1986). In general, the slope and depth of the system are not large enough to generate the velocities necessary to create the turbulent criteria (Kadlec, 1990).

Convective vs. Diffusive Flow:

The properties of the fluid (viscous properties) and the characteristics of the flow regime (inertia) can provide further qualification of the nature of flow in a system by indicating importance of convective or diffusive flow. Levenspeil (1993) provides an empirically derived graph that specifies which flow regime a particular system is likely to fall into. Based on a few dimensionless parameters, the Schmidt and the Bordenstein numbers, one can determine which forces dominate. If a flow is convective, spreading of material is mainly a result of velocity gradients, whereas if a flow is diffusional, spreading of material results mainly from molecular properties governed by traditional Fickian dispersion equations. Pure convective flow occurs if the flow path is so short as to prohibit molecular diffusion from developing enough to distort the velocity profile. If the flow path is very long, dispersion distorts the velocity profile. It is recognized on river systems that there is some initial length, referred to as the convective period, where the residence time distribution is greatly skewed as convective forces dominate (Fischer, 1968; Nordin & Troutman, 1980). It is assumed after this convective period that concentration distribution approaches a more Gaussian (symmetrical) form, reflective of the theoretical dispersed plug flow model developed below.

It is entirely possible that the flow regime falls somewhere in between these two extremes. This transitional region, like the transition from laminar to turbulent, appears as a broad band between the two zones, as in Levenspeil's graph (1993). In this region, as in many transitional regions, the analytical solution becomes impractically complex to use. Levenspeil (1993) suggests estimating the dispersion caused by pure convection models and the by pure diffusion models and averaging the two as the best solution.

Dispersed Plug Flow:

Ideal plug flow behavior has been discussed previously. A pulse input into a reactor will appear as a spike at the outlet end. As an increasing amount of dispersion is present the spike will begin to flatten. At first symmetrical, the residence time distribution will eventually become heavily skewed with extensive dispersion (deviation from plug flow). To characterize this spreading a diffusion-like process is combined with a plug flow model. Levenspeil (1972) presents these type curves for various inlet and outlet conditions. An example of the curves for "open-open" boundary conditions is depicted in figure 2. A discussion of boundary conditions follows the derivation of the dispersion equation.

The dispersion process is primarily a function of concentration gradient. If dispersion is considered in the x-direction, then,

$$F_{A_x} = -D_x \frac{\partial C_A}{\partial x} \qquad (17)$$

where,

F_{A_x} = mass flux of material A in the x direction , $[M / L^2 \cdot T]$

D_x = coefficient of dispersion in x direction , $[L^2 / T]$

C_A = concentration of material A, $[M / L^3]$

Of course, dispersion typically is three dimensional and the dispersion coefficient varies with direction and degree of turbulence. For simplicity, however, the remaining discussion will address one dimensional dispersion only.

A one dimensional analysis of dispersion depicted in Figure 4 will relate the following mass balance:

Accumulation = inflow - outflow + generation.

Consulting Figure 4, this translates to

$$\frac{\partial C_A}{\partial t} A \Delta x = \left(u A_x C_A - A_x D_x \frac{\partial C_A}{\partial x} \right)\bigg|_x - \left(u A_x C_A - A_x D_x \frac{\partial C_A}{\partial x} \right)\bigg|_{x+\Delta x} + r_A A_x \Delta x \qquad (18)$$

where,

A_x = cross sectional area in x direction, $[L^2]$

u = average velocity in x direction, $[L / T]$

r_A = rate of reaction of material A, $[M / L^3 \cdot T]$

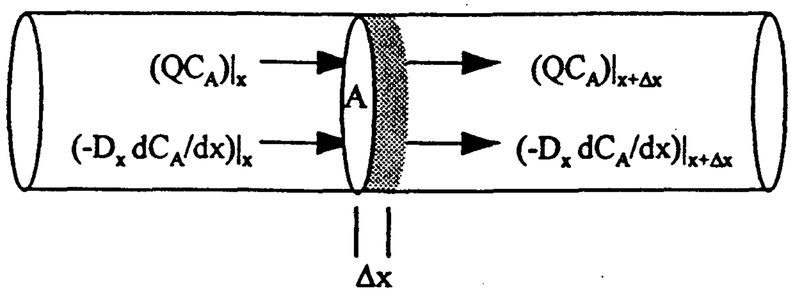

Figure 4. One dimensional mass balance for dispersed plug flow model.

By taking the limit of the above equation as Δx approaches zero, and assuming that material A is nonreactive (ie. a conservative tracer), equation 18 becomes (Tchobanoglous & Schroeder, 1985),

$$\frac{\partial C_A}{\partial t} = D\frac{\partial^2 C_A}{\partial x^2} - \frac{\partial C_A}{\partial \theta_H} \tag{19}$$

In Eqn. (19), the term $\partial\theta_H$, or hydraulic residence time, was substituted for the quantity $\partial x/u$. The subscript 'x' was dropped for convenience, but it should be kept in mind that the remainder of the discussion assumes one dimensional dispersion only.

Boundary Conditions and the Dispersion Model

Tchobanoglous & Schroeder (1985) propose that dispersion can be considered small when the quantity D/uL is less than 0.025, and large when D/uL is greater than 0.2. If the dispersion in the system is small, Levenspeil (1972) provides an analytical solution for the dispersed plug flow model as,

$$E_{t,\infty} = \frac{u}{\sqrt{4\Pi Dt}}\exp\left[-\frac{(L-ut)^2}{4Dt}\right] \tag{20}$$

where,

u = average velocity
D = dispersion coefficient
L = length of travel path

When dispersion in a system is large, outlet concentration curves become increasingly nonsymmetrical and particularly sensitive to boundary conditions (Levenspeil, 1972). This fact has been a topic of discussion in the chemical engineering field since

Danckwerts' paper in 1953. The inlet and outlet boundary conditions can be qualified as either open or closed. A number of mathematical subtleties arise when attempting to provide analytical solutions to the problem when the boundary conditions are considered (Denbigh & Turner, 1984).

If the flow characteristics do not change drastically at a boundary, it is said to be open. In this case it is assumed that backmixing may occur at the boundary, which in turn distorts the residence time distribution. The partial differential dispersion equation above is interpreted to represent the fluid patterns in the reactor, but does not account for the possibility of a particle to cross and recross a boundary through out its life in the reactor (Nauman, 1981). This led Gibilaro (1978) to assert that tracer tests could not be used in systems with open boundaries to find true residence time distributions at a molecular level of scrutiny.

Conversely, if a boundary is considered to be closed, then no back mixing occurs, and the residence time distribution obtained is the true RTD for the reactor. An example of a closed boundary- would be a long, narrow pipe entering a reactor. Unfortunately, there has been no analytical solution for the case of a closed reactor to date. (Nauman, 1981; Levenspeil, 1993).

Determining Dispersion from Tracer Studies:

The most accurate method of determining the dispersion present in a system is by a method analogous to the flood routing procedure used in hydrology (Fischer, 1968). A pulse of dye is released upstream of two sampling points located at some distance apart from one another. As the pulse of dye passes the initial sampling point, it will have a characteristic residence time distribution. As it passes the second sampling point further downstream, the distribution will indicate an attenuated peak and larger variance characteristic of longitudinal dispersion. By comparing the change in the residence time distribution via the variance an estimate of the dispersion in the system can be made. The advantage of this "routing" technique is its insensitivity to the method of tracer injection. For this reason it is commonly used in studies of dispersion in rivers. Fischer (1968) gives the dispersion as:

$$D = \left(\frac{1}{2}\right) \cdot \bar{u}^{2} \frac{\sigma_{t_1}^2 - \sigma_{t_0}^2}{\bar{t}_1 - \bar{t}_0} \tag{21}$$

where,

$\bar{u}$ = mean velocity of the fluid

$\sigma_{t_1}^2, \sigma_{t_2}^2$ = variance of distributions at upstream and downstream locations respectively

$\bar{t}_1, \bar{t}_2$ = mean time of passage of tracer cloud at each station

Tanks-in-series:

An approximation of the dispersed plug flow model is encompassed in the tank-in-series model. By simulating the flow as some number of complete mixed reactors in series, N, a set of curves indicated in Figure 5 results. A truly complete mixed reactor would be modeled with one tank in series (N = 1), and true plug flow would be modeled with an infinite number of complete mixed reactors (N = ∞). By matching the C-curve with one of the type curves shown in Figure 5, an analogous number of tanks in series is obtained.

The derivation of the tank-in-series model results from the expression for one tank, i, in series (Levenspeil, 1993):

$$E_i = \frac{e^{-t/\bar{t}_i}}{\bar{t}_i} \tag{22}$$

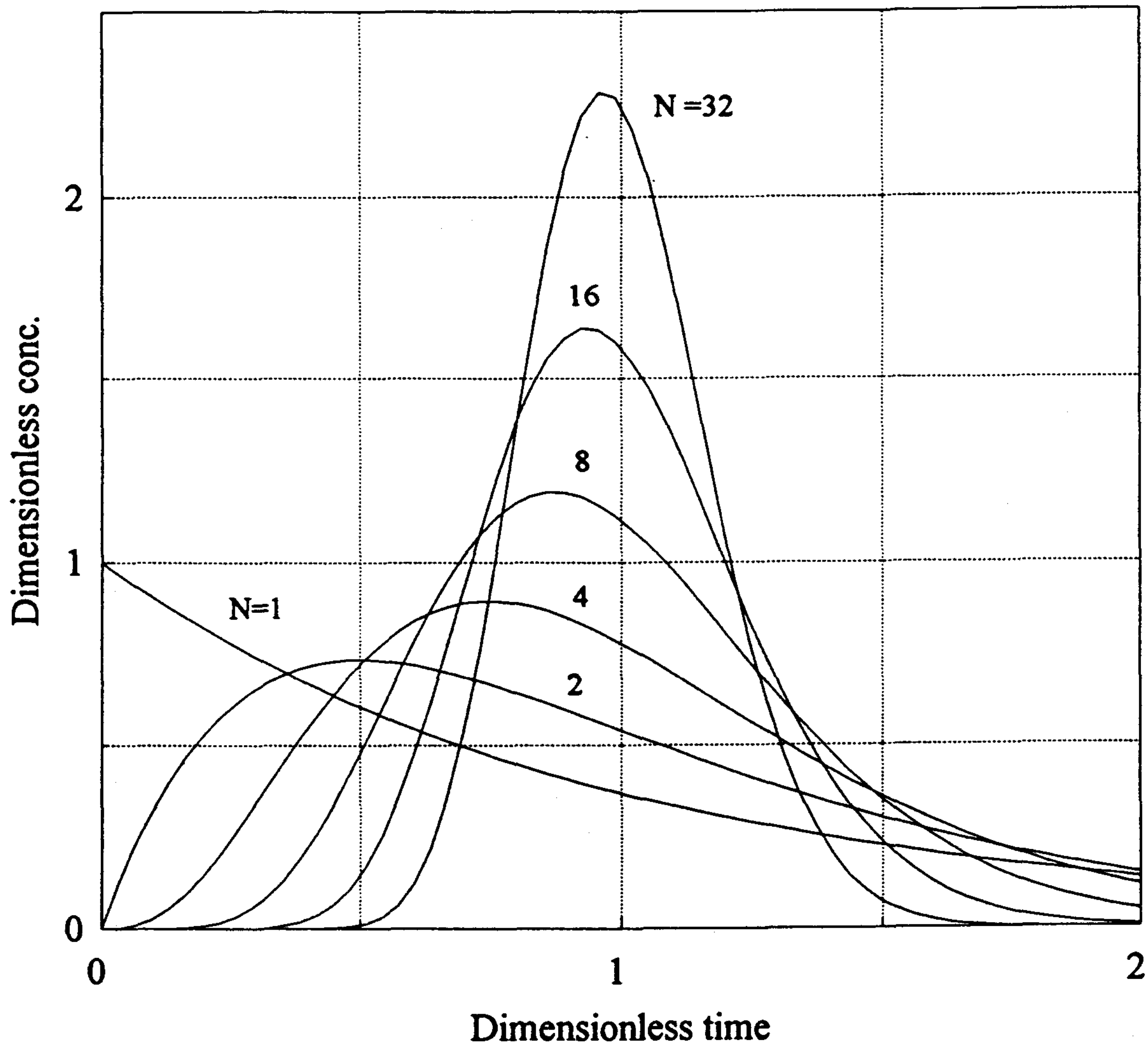

Figure 5. Dimensionless residence time distributions for N reactors in series.

The solution for N reactors in series is given as,

$$E_\theta = (N\overline{t_i})E = \frac{N(N\theta)^{N-1}}{(N-1)!}e^{-N\theta} \tag{23}$$

Where,

$\overline{t_i}$ = mean residence time of one tank

$\bar{t} = N\overline{t_i}$ = mean residence time of N - tank system

$\theta_i = t/\overline{t_i}$ = dimensionless res. time of one tank

$\theta = t/\bar{t}$ = dimensionless res. time of system

The tank-in-series (TIS) model yields identical results as the dispersed plug flow model for small amounts of longitudinal dispersion. The use of one over the other is usually decided as a matter of convenience to the designer, although the TIS model, unlike the dispersed plug flow model, has the advantage of insensitivity to boundary conditions and method of injection and sampling (Levenspeil, 1993).

To determine the number of analogous reactors in series from an experimental residence time distribution, Levenspeil (1993) suggests several approximations, the most reliable being:

$$\sigma_\theta^2 = \frac{1}{N} \tag{24}$$

Wetland Design Criteria

Because this discussion centers on hydraulic efficiency and reactor configuration, it is useful to briefly discuss some of the parameters that affect the hydraulic regime. The general design procedure currently used for constructed wetlands for wastewater treatment is outlined in the EPA Design Manual (EPA, 1988). The design process is based on the assumption that the reactions (BOD reduction) are first order and the hydraulics can be approximated by plug flow. Some of the recommendations presented in the manual are based on empirical results from a few systems. The ultimate design criteria will maximize contact between the wastewater and the plant matrix, while minimizing short circuiting (Hantzche, 1985). There are several factors of importance in the configuration of the wetland system that affect the distribution of the wastewater.

Depth: The depth of the wetland is critical to maintaining the design hydraulic residence time and preventing anoxic conditions from developing (Reed et al., 1988). Often the depth of the wetland is dictated by the selection of the plants to be used. Once the plants have been selected, the depth is set for the conditions dictated by the plant needs.

Slope: The suggested slope of a constructed wetland ranges from 0 to 1% (Watson & Hobson, 1989). Cheri & Malone (1992) have shown that all other features of the cell can be determined from the selected slope with a modified version of the EPA design process (1988). Slope is important in insuring adequate drainage and providing sufficient residence time, particularly in subsurface wetlands.

Location of inlet/outlet control structures: The location of inlet and outlet structures would seem to be an obvious design consideration of importance to efficient flow. However, there are numerous references to faulty designs, placing inlet and outlet structures in close proximity to each other - increasing short circuiting (Moreno, 1990). Inlet and outlet structures should be placed as far from each other as possible, and attempts should be made to inhibit channelizations. Baffles, islands, and dense plant growth tend to eliminate preferential flow patterns.

Flow distribution network: Inlet distributions pipes are often recommended, with the belief that they distribute the flow over the entire width of the wetland and maximize the potential for uniform flow and contact with the entire volume of the cell, and minimize stagnation. Inlet distribution pipes should be gated "manifolds" that extend to at least 90% of the width of the cell (EPA, 1988).

Aspect Ratio: The commonly referenced design guidelines stipulate a 3-5:1 length to width ratio for free water surface wetlands (Hammer, D.A., 1992). Some suggestions range as high as 10:1 (Watson & Hobson, 1989; Tchobanoglous, 1991). The process used to size the wetland usually involves a determination of surface loading quantities and then arbitrarily choosing a l:w ratio. As mentioned before, the variation in treatment efficiencies experienced at different wetland sites suggest that there is a need to optimize the understanding of these geometric relationships.

Studies have shown that a long narrow cell provides maximum treatment efficiency by assuring the assumption of plug flow is met. Listowel data (Wile et al., 1985) documented a long, serpentine channel of l:w ratio 75:1 continually outperformed an equal area cell of 4:1 l:w. The short channel exhibited a high tendency for short circuiting.

The negative side of designing long, narrow cells is the increased construction cost. As the l:w ratio increases, the amount of external berming, and hence cost, increases. So, although a high l:w ratio is lauded as desirable for optimal treatment, there is a trade off in construction cost.

Berms, Dikes: Attempts to simulate long narrow channels have often employed the use of internal baffles, establishing a long, serpentine configuration to the wetland. This design procedure has been shown to effectively control short circuiting (Watson & Hobson, 1989), and can be added to existing systems for corrective action. As with external berms, these features add to the cost and construction difficulty of the wetland, and should be included with some discretion. One variation of the internal berming theme presented by Hammer and Knight (1992) was the incorporation of deep water zones within the wetland. These zones seemed to minimize the potential for short circuiting by providing areas of mixing to cut off preferential flow channels.

OBJECTIVES

The level of performance of a wastewater treatment system is related in part to the residence time of the particular reactor. At present, the design of constructed wetlands for wastewater treatment is based on the premise of plug flow. In this ideal situation, all elements of the waste stream entering the wetland experience the same residence time. In actuality, the hydraulic pattern deviates somewhat from the ideal. Longitudinal dispersion, short circuiting and dead space within the system contribute to this departure from the ideal case. Overall conversion of pollutants (eg. BOD) in the system is directly affected by the resulting distribution of residence times. In almost all cases, a distribution of residence times in a reactor leads to decreased efficiency. Investigations of the hydraulic regime will provide an increased understanding of the important parameters in the design and physical configuration of constructed wetlands.

The objectives of this research are to:

- Investigate the hydraulic regime of a constructed wetland (CW) system via dye tracer techniques.
- Describe the deviation of the CW system from the ideal assumption of plug flow. Specifically, using the residence time distribution, calculate the mean detention time, degree of dispersion and effective volume;
- Demonstrate the decreased treatment efficiency indicated by the residence time distributions incorporated with reaction kinetics;
- Determine suitability of the dispersed plug flow and tanks-in-series models for the nonideal flow of a constructed wetland.

MATERIALS & METHODS

The Wetland System:

The wetland systems used in this study were free water surface (FWS) wetland ponds located at Pope and Talbot, Inc., a pulp mill in Halsey, Oregon. The pilot project wetlands are the result of an on-going five-year cooperative study with a research team from Oregon State University. The intended function of the wetlands being studied is to provide tertiary treatment of the pulp mill effluent.

Configuration: The wetland system consists of ten independent cells each 63 meters long by 21 meters wide, yielding a length to width ratio of 3:1 (Figure 6). The depth is held constant at 46 centimeters. The wetlands have a slope of 0.5%, and the side slopes are approximately 2:1. Each cell was lined with a compacted, relatively impermeable clay layer.

Hydraulics: The effluent to each pond is secondarily treated pulp mill effluent from aeration lagoons. Effluent enters over an adjustable triangular weir and through a gated distribution pipe across the width of the cell. Effluent is collected by a similar perforated pipe at the outlet end. The distribution and collection pipes rest on the cell bottom. Flow is maintained by a constant head tank and by the adjustable weirs. The ponds included in this study had theoretical detention times of two days. These times are based on the effective pond volume divided by the flow rate. Flow is along the length of the ponds, with the inlet and outlet at opposite ends.

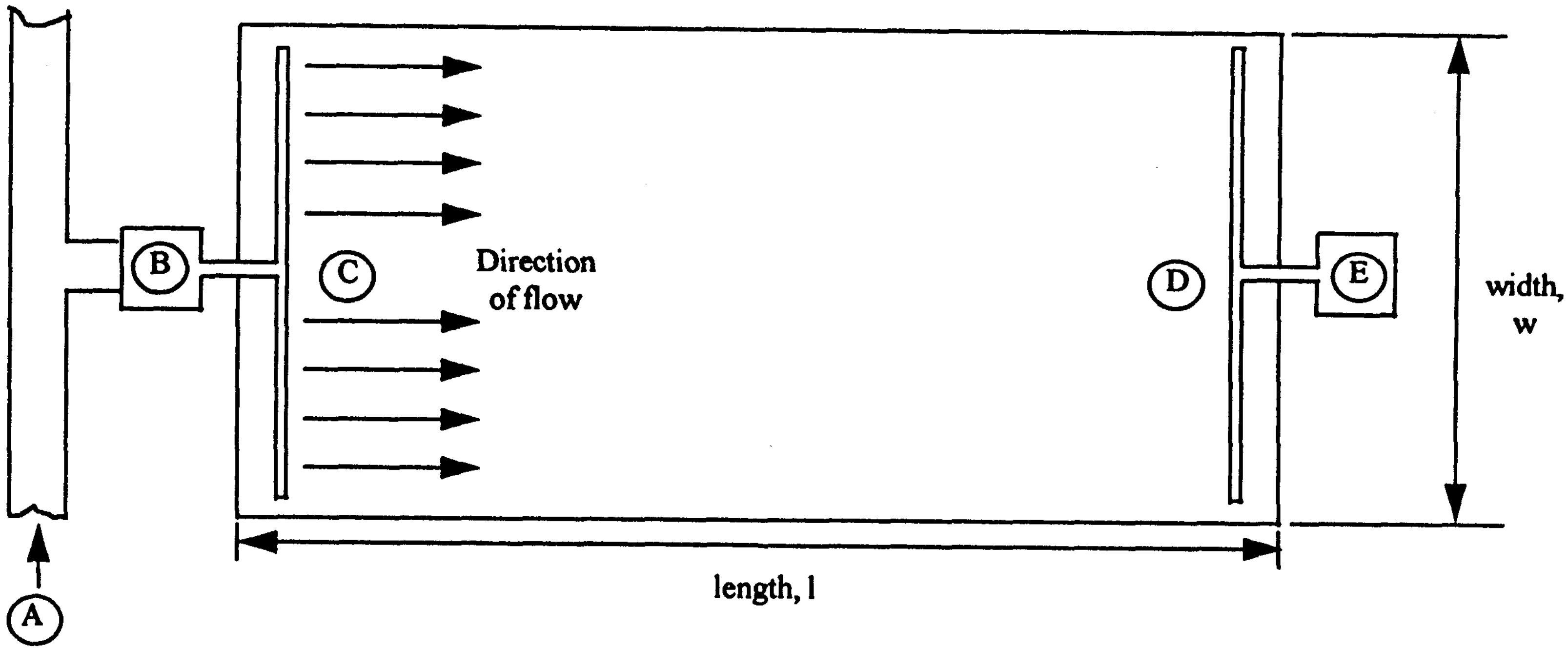

Figure 6. Hydraulic control structures of the constructed wetland cell. (A) influent from constant head tank; (B) flow control box with V-notch weir; (C) inlet distribution pipe; (D) outlet collection pipe; (E) outlet/ depth control box.

Media: The wetlands were planted in late Fall of 1990. Two species are used: hardstem bulrush (*scirpus acutus*) and cattail (*typha latifolia*). Two of the ten cells are not vegetated: one contains a stone bed of 3" diameter or larger rock, and the second is merely an open (empty) cell.

Dye Studies:

The initial step involved selection of a stable dye to be used as a tracer. Several papers are available on this topic. (Smart & Laidlaw, 1977; Gaspar, 1987). The tracer finally chosen was a fluorescent dye, rhodamine WT, in liquid form. Rhodamine WT has a minimum detectability of 0.013 μg/L, and was measured on a Turner 111 Fluorometer. The fluorometer was set up for rhodamine analysis as per Wilson (1986) with a General Electric (GE) G4T4/1 far-UV lamp providing a peak excitation of 546 nm. The primary filter combined a gel Wratten 61 green filter between two glass Corning 1-60 gray filters. The secondary filter arrangement included an orange glass Corning 3-66 and a blue glass Corning 4-97 filter.

Stock dye concentration was 20% by weight as obtained from the manufacturer. Calibration curves were determined using standards prepared according to procedures described by Wilson (1986). Standards were made with rhodamine dye serially diluted from 0.25 μg/L to 10 μg/L in pulp mill effluent. Because rhodamine dye fluorescence is particularly sensitive to temperature, the fluorometer standards and dye study samples were stored in a laboratory cabinet at room temperature prior to analysis.

The stimulus-response technique used in the dye study consisted of a 90ml slug of dye mixed into 5 gallons of effluent, and added as an instantaneous pulse at the inlet. Care was taken to replicate experimental technique exactly in all runs, since reproducibility has been shown highly dependent on experimental technique (Grobicki & Stuckey, 1992). Samples were drawn from the outlet control box at one hour intervals from the time of injection until twice the theoretical detention time (Middlebrooks et al, 1982). The studies were performed in the late spring and fall of 1992, and in the spring of 1993.

RESULTS AND DISCUSSION

Theoretical Detention Times:

The flow rates at the Pope and Talbot site are controlled with 90°, v-notch weirs. The volumetric flow rate, Q (m^3), can be approximated from the head above the weir, H (m), as follows (Cuenca, 1989):

$$Q = 1.3424 H^{2.48} \qquad (25)$$

The flow through rates and corresponding theoretical detention times were calculated as shown in Table 1. For the purposes of this study, a conservative density of 0.90 was assumed for the planted systems, 0.55 for the rock cell (Reed, 1988; Tchobanoglous, 1991), and 1.0 for the non-vegetated cell. A porosity corrected theoretical detention time is also reported.

Table 1. Flow rates and corresponding theoretical detention times.

Pond #	Media	Flow Rate (m^3/d)	Detention Time no porosity correction, (days)	Detention time with porosity correction, (days)
1	rock	340	1.8	0.9
2	cattail	262	2.3	2.1
4	bulrush	256	2.4	2.2
7	empty	288	2.1	2.1
8	cattail	403	1.5	1.4

The flows were checked periodically to verify the the rates indicated above. Occasionally flows were altered due to adjustments being made at the constant head tank. Effort was made to record the fluctuations and account for it in the data analysis.

Dye Characteristics

Standards for the fluorometer calibration were made with distilled water and effluent to compare the potential masking/bleaching effects. Calibration curves with the dye diluted in effluent demonstrated considerably reduced fluorescence from the distilled water samples. To compensate for the affects of the effluent on dye concentration readings, the effluent standards were used for the remaining calibrations.

Residence Time Distributions

The concentration vs. time data for all of the runs is depicted in figures 7-12. Immediately obvious is the deviation of the flow pattern from ideal plug flow. In general, the distributions for all the ponds displayed a sharp smooth rise to an early peak, followed by a gradual, exponential falling limb. The long tailing and attenuated peak are indicative of large values of dispersion and/or the existence of dead zones (Levenspeil, 1993), but may also result from tracer adsorption-desorption or faulty tracer injection technique. A spreadsheet was setup to calculate the statistics for each curve (ie. mean detention time, time to peak, variance), which are summarized in Table 2.

There is some visible fluctuation in the shape of the distributions, possibly attributable to one or more of the following:

- seasonal changes in plant density, due to growth and die-back can alter flow patterns and velocities within the wetland;
- variation in experimental technique; reports in the literature have warned of the sensitivity of tracer study results to the technique of dye injection;

- wind, which is particularly evident on the non-vegetated pond, can play a big role in creating circulation patterns that may lead to short circuiting. An illustration of the effect of wind is apparent in the Apr 92 run of pond #7. A strong wind (up to 12.8 mph) blowing along the length of the pond, from inlet to outlet, produced a very early peak, approximately 4 hours from the time of injection. In vegetated ponds wind is presumed to have less of a effect on circulation patterns. Care should be taken, however, when large open areas exist in vegetated ponds, particularly with species like bulrush that grow in hummocks.

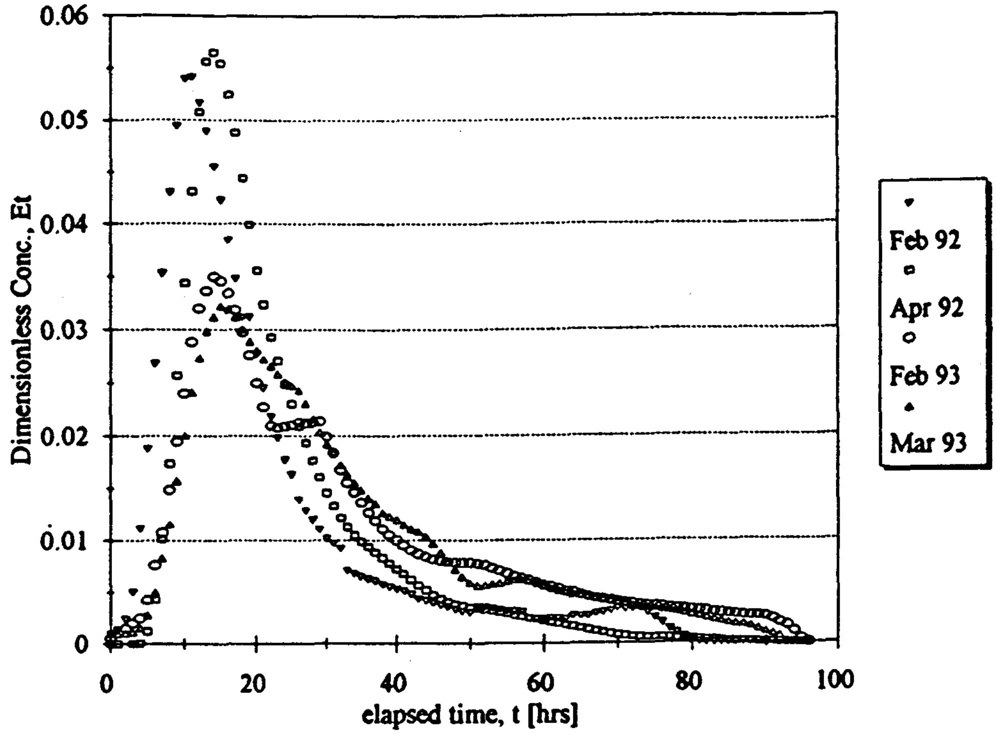

Figure 7. Residence time distributions for Pond 1(rock).

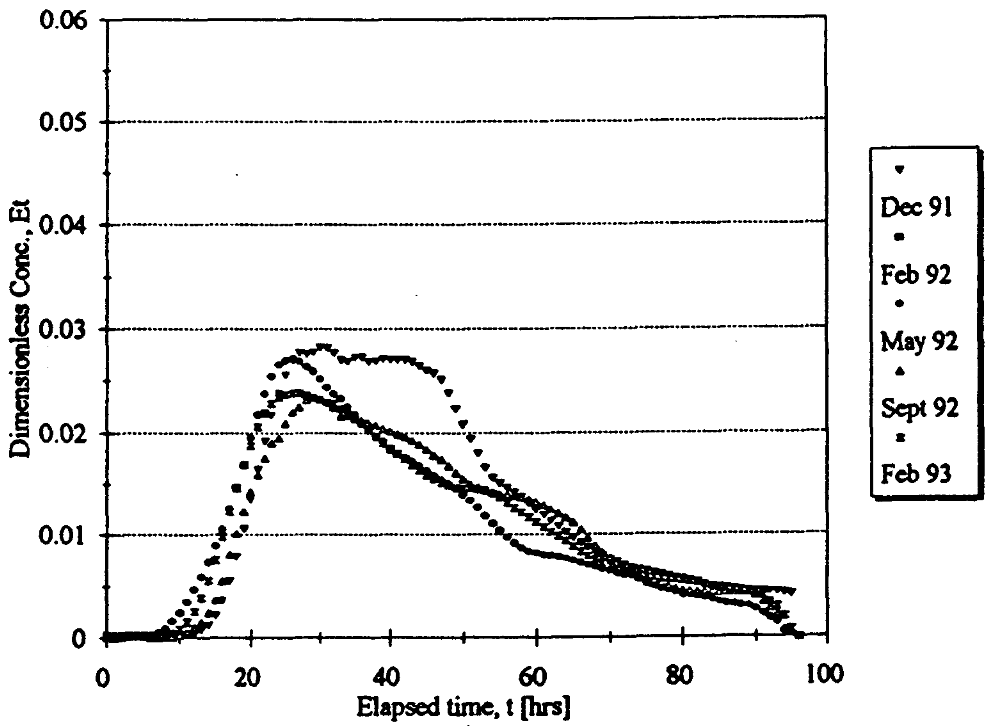

Figure 8. Residence time distributions for Pond 2 (cattail).

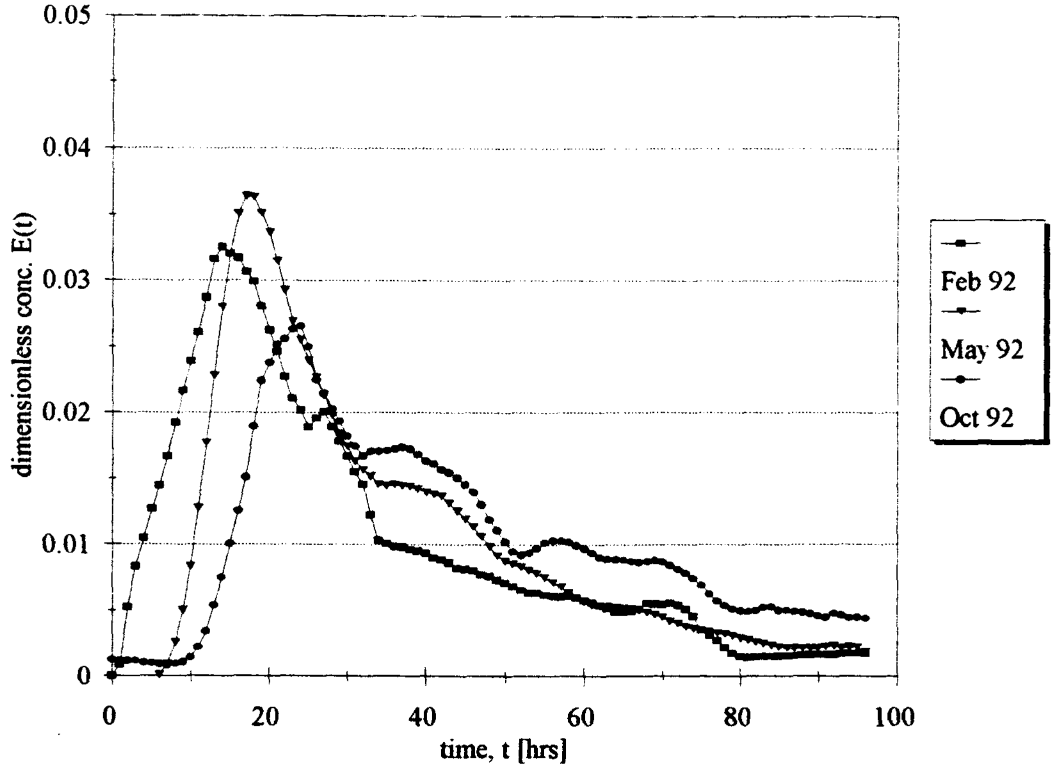

Figure 9. Residence time distributions for Pond 4 (bulrush).

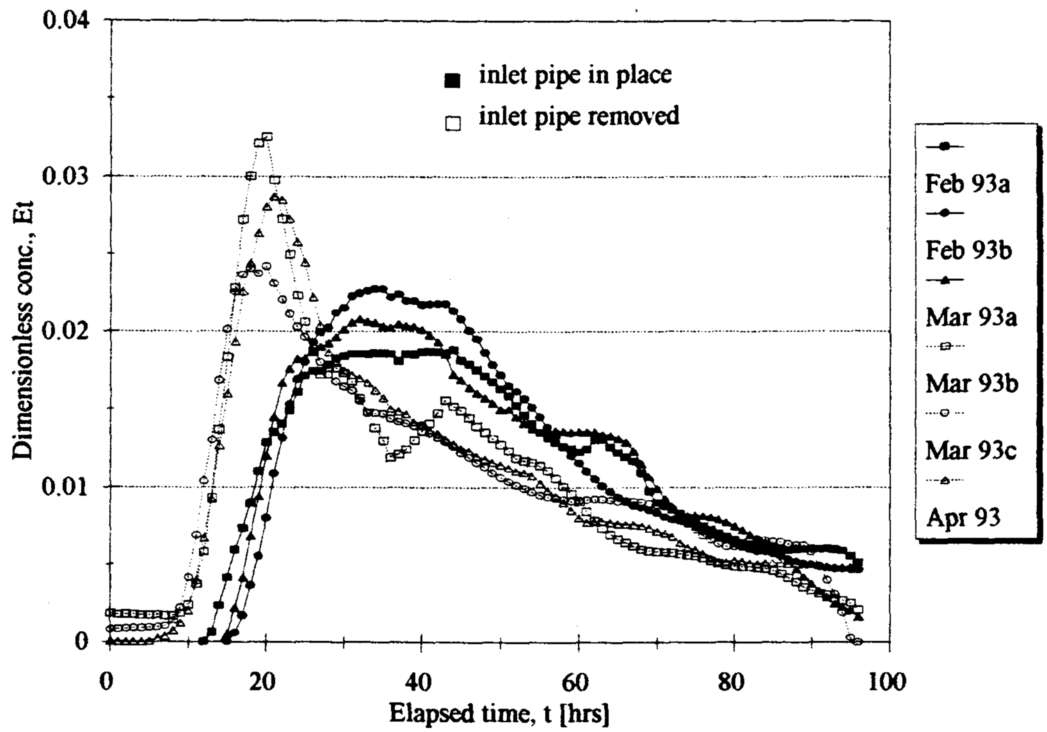

Figure 10. Residence time distributions for Pond 4 (bulrush), with inlet distribution pipe in place and removed.

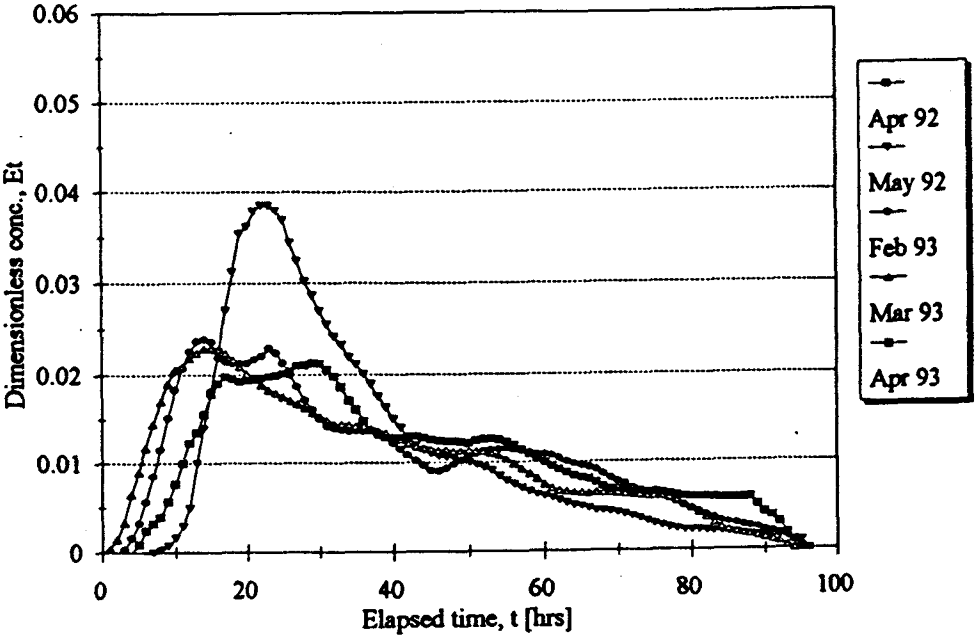

Figure 11. Residence time distributions for Pond 7 (empty).

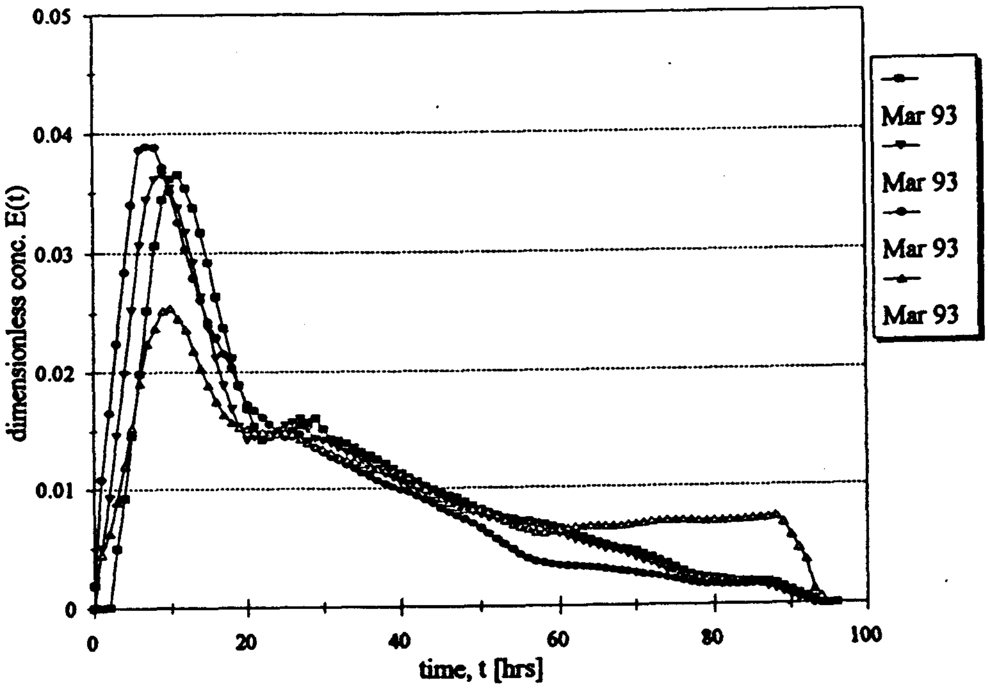

Figure 12. Residence time distributions for Pond 8 (cattail).

The time to the peak of the concentration vs time curve gives some indication of the short circuiting in the cell. If the system behaved as a true plug flow reactor, the peak would appear at a time equal to the theoretical residence time ($t_p = \bar{t}$). Inspection of the breakthrough curves shows time to peaks displayed in Table 2.

One particular comparison of interest appears in Figure 10, a set of runs in the Spring of '93 on the bulrush pond (P4). By only considering these runs spanning 2 consecutive months early in the year, it is hoped that seasonal variability of the system was somewhat avoided. In the first three runs the inlet distribution pipe was left in place, as in all the previous runs. For the second set of runs, the inlet distribution pipe was removed. There is a slight visible difference between the two sets, specifically a sharper peak - indicating a higher degree of short circuiting - with the distribution pipes removed. The average time to peak, t_p, with the distribution pipes in place occurred at 35 hours, while the t_p with the pipes removed was 20 hours.

Dye recovery:

The mass of dye recovered from each run was calculated by integrating the concentration curve over time from $\theta = 0$ to $\theta = 2$, as discussed previously. Dye recoveries are summarized in Table 2. In all cases, less dye was recovered than input into the system. Factors affecting the mass balance could be one or more of the following:

- dye sorption/desorption, to plants or soil (clay) bed
- length of study too short; a long tail attributed to dead space retention in the ponds was truncated.
- variations in background interference, such as process changes in the mill, causing errors in accurate quantitative measurement of dye mass.
- other losses in the system- biological uptake

Table 2. Results of normalized concentration vs. time curves

Run #	Date	Time to Peak	Mean	Variance	Dye recov. (%)
1-1	20 Feb 92	0.47	0.93	0.45	78
1-2	16 Apr 92	0.63	1.00	0.30	46
1-3	9 Feb 93	0.51	1.11	0.50	48
1-4	11 Mar 93	0.58	1.09	0.42	53
2-1	22 Dec 91	0.56	0.75	0.09	80
2-2	20 Feb 92	0.52	0.82	0.12	57
2-3	19 May 92	0.48	0.83	0.14	62
2-4	29 Sept 92	0.60	0.91	0.14	55
2-5	16 Feb 93	0.50	0.89	0.15	55
4-1	20 Feb 92	0.21	0.58	0.16	61
4-2	10 May 92	0.33	0.67	0.14	41
4-3	14 Oct 92	0.43	0.86	0.20	67
4-4	3 Feb 93	0.80	0.96	0.15	47
4-5	16 Feb 93	0.62	0.97	0.15	49
4-6	3 Mar 93	0.62	0.96	0.14	49
4-7*	11 Mar 93	0.38	0.77	0.18	67
4-8*	30 Mar 93	0.33	0.83	0.20	52
4-9*	14 Apr 93	0.43	0.83	0.20	64
7-1	16 Apr 92	0.09	0.65	0.20	46
7-2	5 May 92	0.45	0.77	0.11	48
7-3	9 Feb 93	0.26	0.76	0.18	49
7-4	3 Mar 93	0.30	0.71	0.18	59
7-5	14 Apr 93	0.29	0.77	0.15	69
8-1	6 Mar 93	0.37	0.90	0.39	87
8-2	11 Mar 93	0.28	0.80	0.36	68
8-3	30 Mar 93	0.31	1.08	0.52	62
8-4*	14 Apr 93	0.43	0.99	0.52	53

*inlet distribution pipe removed

It has been discussed previously that failure to close the mass balance of dye recovery can imply incorrect measurement of the residence time distribution. If the cause is any of the reasons mentioned above, then time of travel calculations are in question. Nevertheless, it was assumed for the remainder of the analysis that the concentration vs. time curves are an adequate representation of the general system characteristics. It is evident from the curves in Figures 7-12 that the bulk of the dye had exited the system at a detention time less than $\theta = 2$. The dimensionless E -curves were normalized to yield an area of unity beneath the integrated distribution.

Means, Variances, and Dead Space:

The means and variances of the residence time distributions were calculated from the normalized curves. The results are listed in Table 2 and averages of all the runs with 95 % confidence intervals listed in Table 3. A mean residence time of 1.0 corresponds to $t_{obs} = t_{theoretical}$.

Table 3. Averages of statistics with 95% confidence intervals from normalized residence time distributions.

Pond #	Mean detention time	variance
1	1.03 ± 0.135	0.42 ± 0.128
2	0.84 ± 0.079	0.13 ± 0.030
4	0.83 ± 0.018	0.16 ± 0.024
7	0.73 ± 0.065	0.16 ± 0.044
8	0.94 ± 0.191	0.45 ± 0.316

Dead space was calculated by considering the volume of a reactor that would produce a residence time equal to the mean residence time obtained from analysis of the concentration vs. time diagrams. The resultant "reduced" volume is termed **effective**

volume in Table 4, and the percentage of dead volume was obtained by subtracting this quantity from the actual reactor volume, without porosity reduction, of 609 m^3. By ignoring reactor volume occupied by objects (rock or plants), the "effective porosity" obtained by the exercise includes two phenomena: the reduction attributed to the volume occupied by substrate, and the reduction attributed to hydraulic dead space. This is particularly dramatic in the rock pond where the "effective" porosity ranges from 0.51 to 0.61 (avg = 0.57), while the design porosity (volume occupied by rock) was assumed to be 0.55. This would suggest that there is minimal hydraulic dead space in the rock filter, or that the assumed prosity was wrong. For design of planted systems, the effective volume has been suggested to be 0.75 (Reed et al., 1988). This compares to the following ranges obtained for the planted ponds in this study:

- 0.74 to 0.82 (avg = 0.77) for cattail pond #2
- 0.52 to 0.87 (avg = 0.74) for bulrush pond #4
- 0.72 to 0.97 (avg = 0.85) for cattail pond #8.

Interestingly, the effective reduction in volume of the unplanted pond (pond 7) was in the range of 0.65 to 0.85, with an average value of 0.74.

Middlebrooks et al. (1982) calculated dead space by this method in wastewater stabilization ponds in the range of 45 to 65%, whereas Moreno (1990) found dead space accounted for anywhere from 10 to 21% for facultative stabilization ponds studied. Studies of baffled reactors have yielded values from 1.2 to 21% (Grobicki & Stuckey, 1992). Obviously the extent of dead space depends on the hydraulic features of the particular system being studied. The parameters critical to uniform flow distribution in the system were discussed previously, and included: inlet/outlet location, baffling, and length to width ratio, among others. Given the possibility for variation, the results obtained seem within the reasonable range of 15-25%.

Table 4. Calculation of Active Volume and Dead Space of Wetland Cell

Run	Theoretical detention time, [d]	Mean detention time, [d]	Flow Rate [m^3/d]	Effective Volume [m^3]	Dead Volume [%]	"Effective Porosity" n
1-1	1.79	0.92	340	312	49	0.51
1-2	1.79	0.98	340	334	45	0.55
1-3	1.79	1.35	274	371	39	0.61
1-4	1.79	1.35	274	370	39	0.61
2-1	2.32	1.91	240	459	25	0.75
2-2	2.32	1.71	262	449	26	0.74
2-3	2.32	1.73	262	454	25	0.75
2-4	2.32	1.91	262	501	18	0.82
2-5	2.32	1.85	262	485	20	0.80
4-1	2.38	1.25	256	319	48	0.52
4-2	2.38	1.44	256	368	40	0.60
4-3	2.38	1.85	256	472	22	0.78
4-4	2.38	2.05	256	525	14	0.86
4-5	2.38	2.07	256	530	13	0.87
4-6	2.38	2.05	256	524	14	0.86
4-7*	2.38	1.69	256	432	29	0.71
4-8*	2.38	1.79	256	457	25	0.75
4-9*	2.38	1.78	256	456	25	0.75
7-1	2.09	1.26	312	394	35	0.65
7-2	2.09	1.49	288	429	30	0.70
7-3	2.09	1.61	292	471	23	0.77
7-4	2.09	1.51	292	440	28	0.72
7-5	2.09	1.78	292	519	15	0.85
8-1	1.51	1.23	403	496	19	0.81
8-2	1.51	1.09	403	439	28	0.72
8-3	1.51	1.61	369	593	3	0.97
8-4*	1.51	1.35	403	545	11	0.89

Estimating Reynold's Number:

A conservative estimate of Reynold's number was calculated with equation (16) and the highest flow rate listed in Table 1. Assuming the kinematic viscosity of water (8e-7 m^2/s), depth of 0.46 meters, and velocity of 0.05 cm/s,

$$R_e = \frac{5.0 \times 10^{-4} \mathrm{m/s} \cdot (0.46\mathrm{m})}{8 \times 10^{-7} \mathrm{m}^2 / \mathrm{s}} = 276$$

This conservative Reynold's number is well below the transition value of 500, indicating a laminar flow regime that one would expect in low flow systems like constructed wetlands. The Reynold's number for the rock filter, with a higher estimated velocity of 0.07 cm/s was calculated to be 426, so laminar flow can still be assumed.

To determine whether the flow lies in the convective or dispersive period, Levenspeil's (1993) flow behavior graph was used. The Schmidt number was assumed to be 10^3 for liquids (Levenspeil, 1993). Using a Bordenstein number (Reynold's number multiplied by the Schmidt number) and length to depth ratio of 133, the flow was estimated to be in the region between the purely convective and axial dispersion regions.

Predicting Dispersion from Tracer Studies:

The dye studies in Feb. '92 included sampling ports across the wetland cell at the midpoint. By selecting a representative tracer curve from this point and comparing to the tracer curve at the outlet (Figure 13), an estimate of dispersion is calculated as per Fischer (1968). To make the calculations feasible only the main portions of the distributions were considered, the long tailing was ignored. The mean velocity of the system was calculated by dividing the distance between the two sampling locations by the difference in mean times of passage, resulting in $\bar{u} = 0.06$ cm/sec. The respective mean detention times and variances of the two curves were used with this velocity in Eqn. (21) to arrive at a value for the dispersion coefficient, D, of 2.82×10^{-3} m^2/sec.

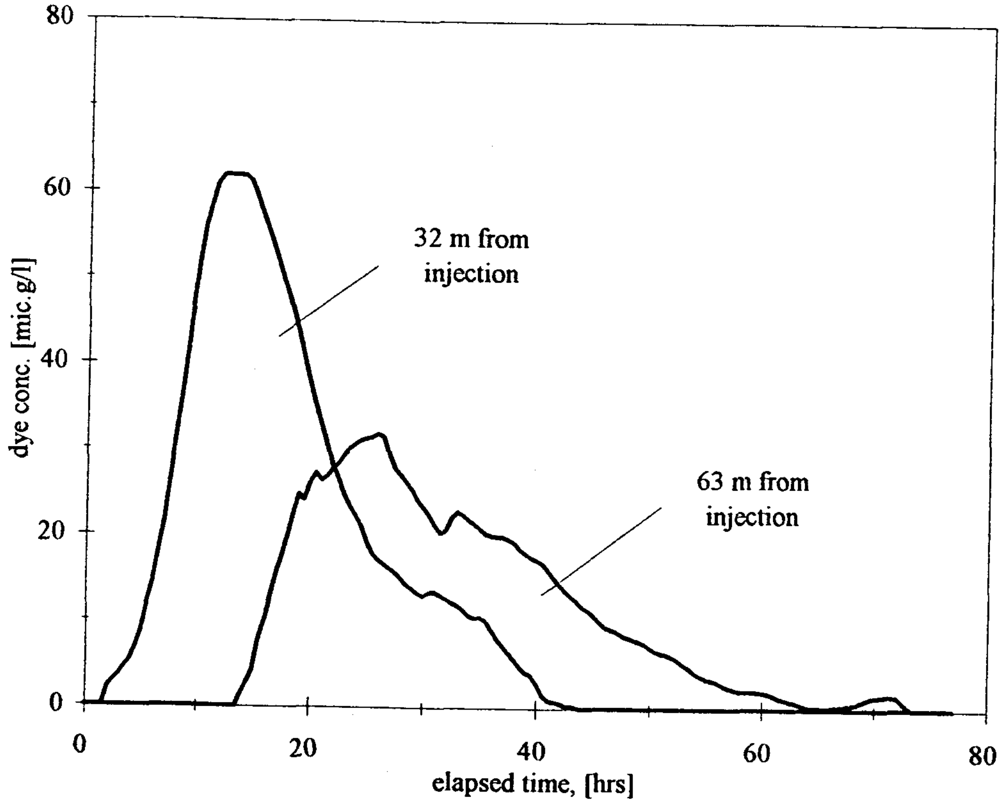

Figure 13. Determining dispersion via "routing" technique. Residence time distributions at midpoint and outlet of pond #2

Treatment Efficiency:

To facilitate a comparison of treatment efficiencies between the ponds, a hypothetical BOD conversion was calculated based on the residence time distributions and the kinetic equation (14). A kinetic coefficient, k, of 0.23 day-1 and an initial BOD value of 45 mg/L was assumed in the calculation. The resulting treatment efficiency was compared to the value that would have been obtained for an ideal plug flow reactor, as per equation 13 and the design detention time. These values are summarized in Tables 5-9.

The average reduction in efficiency was calculated by the residence time distribution method are as follows:

- rock (P1).................... -0.86
- cattail (P2).................. 4.86
- bulrush (P4)................ 3.10
- nonvegetated (P7)........ 4.66
- cattail (P8).................. 0.15

Table 5. Example of decreased BOD conversion for Pond #1 from the residence time distribution (assuming k = 0.073 1/d and BODo = 45 mg/L)

assuming plug flow (PF)	tbar	Ce/Co	% convert
	21.5	0.94	6.25

	1-1	1-2	1-3	1-4
CONVERSION CALCULATED BY INTEGRATION OF RTD				
Ce/Co	0.94	0.94	0.93	0.91
Ce	42.31	42.09	41.71	41.09
% convert	5.98	6.47	7.31	8.68
reduced efficiency* (%)	0.27	-0.22	-1.06	-2.44
			Average reduction =	-0.86

Table 6. Example of decreased BOD conversion for Pond #2 from the residence time distribution (assuming k = 0.073 1/d and BODo = 45 mg/L)

assuming plug flow (PF)	tbar	Ce/Co	% convert
	54.8	0.85	15.16

	2-1	2-2	2-3	2-4	2-5
CONVERSION CALCULATED BY INTEGRATION OF RTD					
Ce/Co	0.89	0.89	0.90	0.89	0.91
Ce	40.19	40.19	40.36	40.08	40.99
% convert	10.69	10.69	10.30	10.93	8.91
reduced efficiency* (%)	4.47	4.47	4.86	4.23	6.25
				Average reduction =	4.86

Table 7. Example of decreased BOD conversion for Pond #4 from the residence time distribution (assuming k = 0.073 1/d and BODo = 45 mg/L)

	tbar	Ce/Co	% convert
plug flow (PF)	52.6	0.85	14.60

	4-1	4-2	4-3	4-4	4-5	4-6	4-7 *no. dist.*	4-8 *no dist.*	4-9 *no dist.*
CONVERSION CALCULATED BY INTEGRATION OF RTD									
Ce/Co	0.92	0.90	0.88	0.87	0.87	0.87	0.89	0.88	0.89
Ce	41.24	40.69	39.73	38.99	39.03	38.99	40.09	39.76	39.88
% convert	8.35	9.59	11.71	13.36	13.26	13.36	10.90	11.66	11.37
reduced efficiency (%)	6.24	5.01	2.88	1.24	1.34	1.24	3.70	2.94	3.22
							Average reduction =		3.09

Table 8. Example of decreased BOD conversion for Pond #7 from the residence time distribution (assuming k = 0.073 1/d and BODo = 45 mg/L)

assuming plug flow (PF)	tbar	Ce/Co	% convert
	50.1	0.86045	13.95502

	7-1	7-2	7-3	7-4	7-5
CONVERSION CALCULATED BY INTEGRATION OF RTD					
Ce/Co	0.89	0.94	0.91	0.90	0.89
Ce	40.01	42.43	41.07	40.58	40.01
% convert	11.10	5.72	8.73	9.83	11.10
reduced efficiency (%)	2.86	8.24	5.22	4.12	2.86
			Average reduction =		4.66

Table 9. Example of decreased BOD conversion for Pond #8 from the residence time distribution (assuming k = 0.073 1/d and BODo = 45 mg/L)

assuming plug flow (PF)	tbar	Ce/Co	% convert
	31.3	0.91	8.96

	8-1	8-2	8-3	8-4
CONVERSION CALCULATED BY INTEGRATION OF RTD				
Ce/Co	0.92	0.93	0.89	0.91
Ce	41.26	41.67	40.26	40.95
% convert	8.32	7.41	10.54	9.00
reduced efficiency (%)	0.65	1.56	-1.57	-0.04
		Average reduction =		0.15

Models:

Dispersed Plug Flow

The residence time distributions were normalized so that the area under the curve was equal to unity and fit with the dispersed plug flow equation (20).

$$E_{t,\infty} = \frac{u}{\sqrt{4\Pi Dt}} \exp\left[-\frac{(L-ut)^2}{4Dt}\right] \qquad (20)$$

where,

u = average velocity
D = dispersion coefficient
L = length of travel path

The least squares estimate of the parameters in the model were determined using a Marquardt's search algorithm supplied by the Statgraphics statistical software package. The three parameters that are determined in the nonlinear search are the length of the travel path, L, the average velocity, u, and the dispersion coefficient,D. Initial estimates for velocity and flow length used in the search algorithm were approximated from the wetland cell configuration. The value of the dispersion coeeficient was approximated from the "routing" procedure described earlier. The estimates used were: mean velocity = 0.05 cm/s, length of flow path = 63 m, and dispersion coefficient = 2.8 x 10^{-3} m^2/s. The algorithm is highly dependent on the initial parameters, so care was taken to provide reasonable initial estimates. The procedure obtains a least squares estimate of the parameters in the model with an iterating search. Results of the nonlinear regression and the percent of variation explained by the model - given as R^2 - are summarized in Table 9. Fits of the equation to the tracer study data were very good, with R^2 values ranging from 0.80 to 0.97.

The average D/uL for each pond was used to construct a model fit for the pooled data, shown in Figures 14-17. For comparison between the ponds, these final models are again summarized in Figure 19.

Tank-in-series

The tank-in-series model was fit by using the mean detention time obtained from the residence time studies, and selecting the number of tanks, N, corresponding to the maximum of the distributions as per Levenspeil (1993). These curves are also displayed in Figures 13-17. The number of tanks required for the model fit are as follows: Pond 1, N=4; Pond 2, N=8; Pond 4 , N=5; Pond 7, N=4; Pond 8, N=3.

All of the tank-in-series (TIS) fits are slightly late compared to the dispersed plug flow fits. This could be a result of the fact that the TIS model is not flexible enough to include the existence of dead space and high degree of dispersion.

On the Appropriateness of the Models

Overall, the models do not seem to adequately compensate for the long tailing of the distributions indicative of exchange with stagnant areas (transient storage) nor the high dispersion in the ponds. Levenspeil (1993) warns against assuming the dispersed plug flow model when dispersion is great or dead space is extensive. Although the model curves can be "fit " to the data, as was done here with a search method for the parameters, the fundamental assumptions in the model do not conform to what is intuitively occurring in the system. A better model for the wetland system suggested by the shape of the distributions and knowledge of the system might be a plug flow reactor with dispersion, interacting with several complete mixed reactors along its length (simulating dead space). Unfortunately, such a model increases complexity over the dispersed plug flow and tank-in-series models, and therefore requires even more "guessing" at appropriate values for the system.

Table 10. Summary of nonlinear regression against dispersed plug flow model.

Run #	Parameters			D/uL	R^2
	u	D	L		
1-1	7	113	87.7	0.18	94
1-2	11.8	231.9	185.9	0.11	95
1-3	4.2	90.3	82.2	0.26	90
1-4	4.9	106.9	101.1	0.22	96
2-1	2	11.8	73.9	0.08	91
2-2	1.9	8.6	60.1	0.08	88
2-3	1.9	15.2	61.4	0.13	97
2-4	2.1	19.8	77.8	0.12	95
2-5	1.9	17.8	65.4	0.14	96
4-1	3.6	82.7	65.9	0.35	89
4-2	5	99.3	115.6	0.17	88
4-3	2.6	39.4	83.2	0.18	82
4-4	2.2	27.9	90.1	0.14	92
4-5	1.9	14.6	76.2	0.10	95
4-6	1.8	15.4	70.6	0.12	92
4-7*	1.5	11.1	59.6	0.12	92
4-8*	1.7	22	51.1	0.25	81
4-9*	1.6	15.7	48	0.20	82
7-1	1.8	40.6	24.1	0.94	93
7-2	2.2	11.8	57.2	0.09	93
7-3	1.5	19	36.9	0.34	84
7-4	1.9	31.8	40.1	0.42	93
7-5	1.6	20.3	49.5	0.26	88
8-1	2.5	40.2	40	0.40	86
8-2	2.5	36.8	34.8	0.42	92
8-3	1.5	24.8	34.9	0.47	80
8-4*	2.7	48.7	46.6	0.39	90

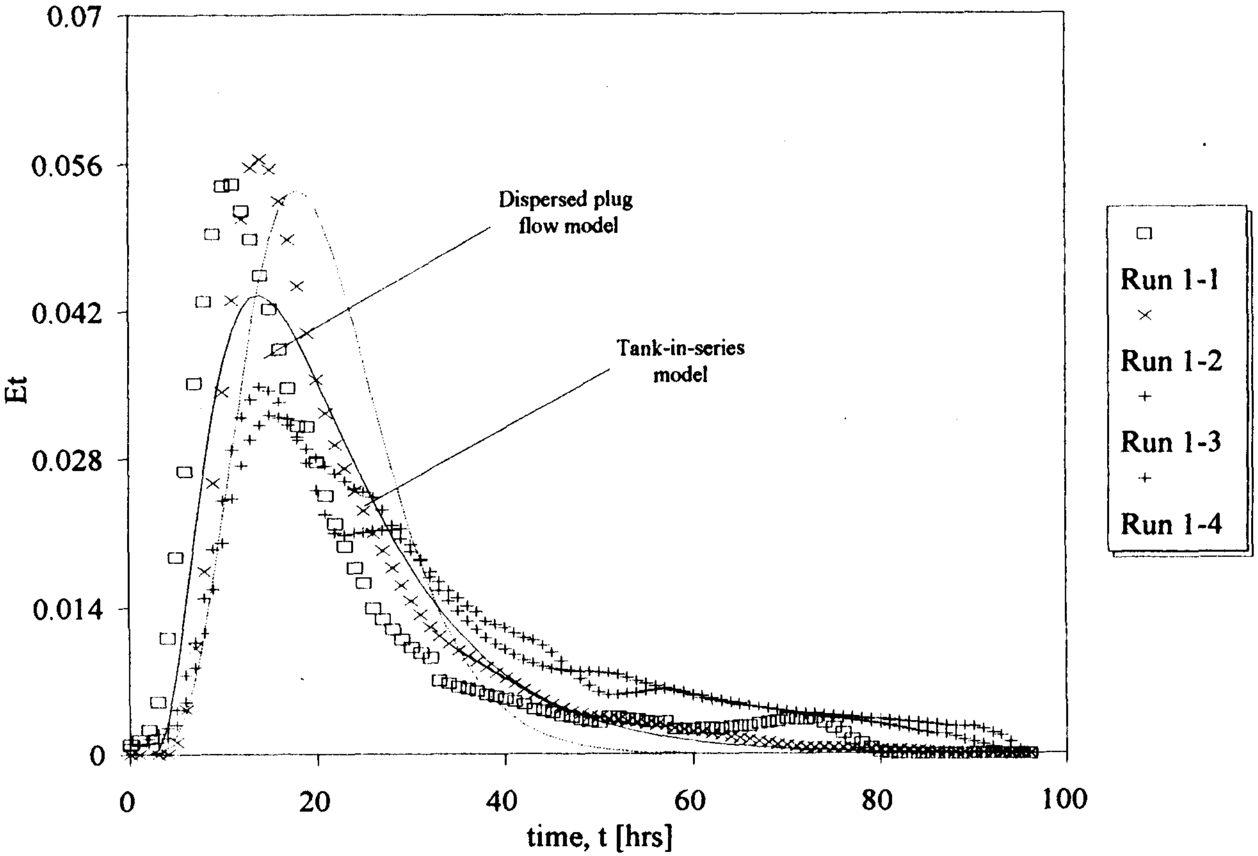

Figure 14. Fit of the dispersed plug flow and tanks-in-series models to tracer study data for pond #1 (roc

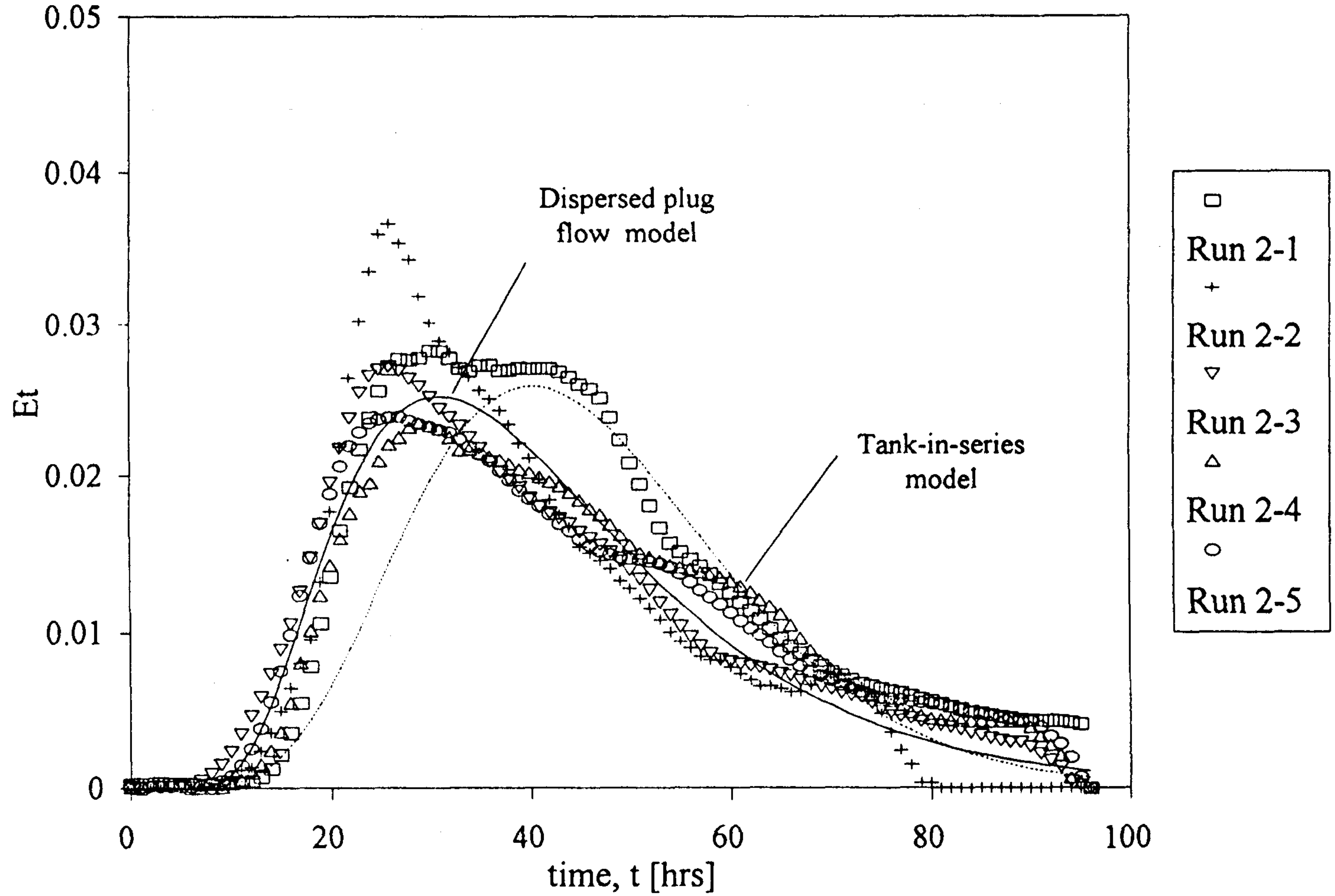

Figure 15. Fit of the dispersed plug flow and tanks-in-series models to tracer study data for pond #2 (cattail)

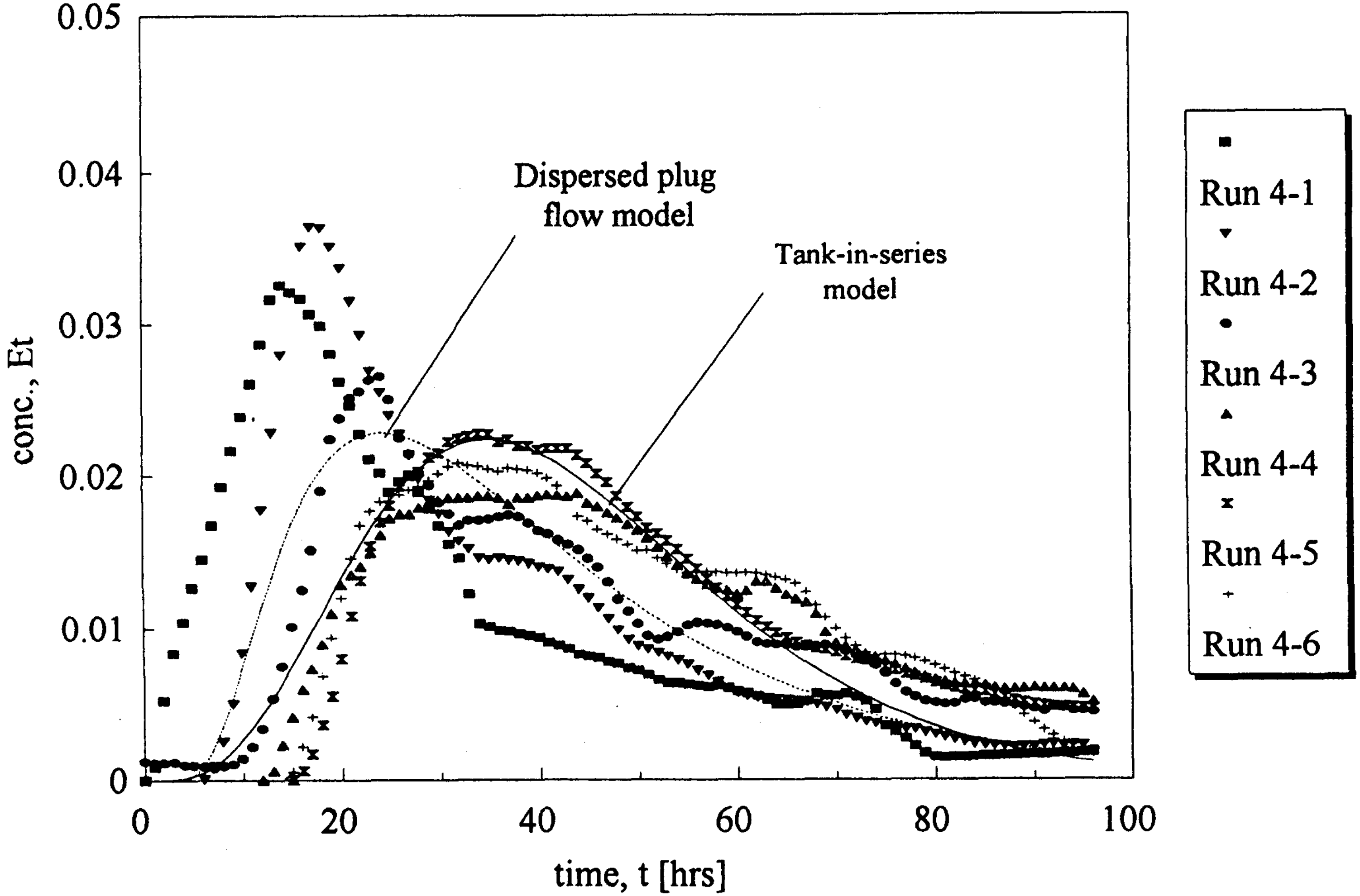

Figure 16. Fit of the dispersed plug flow and tanks-in-series models to tracer study data for pond #4 (bulrush)

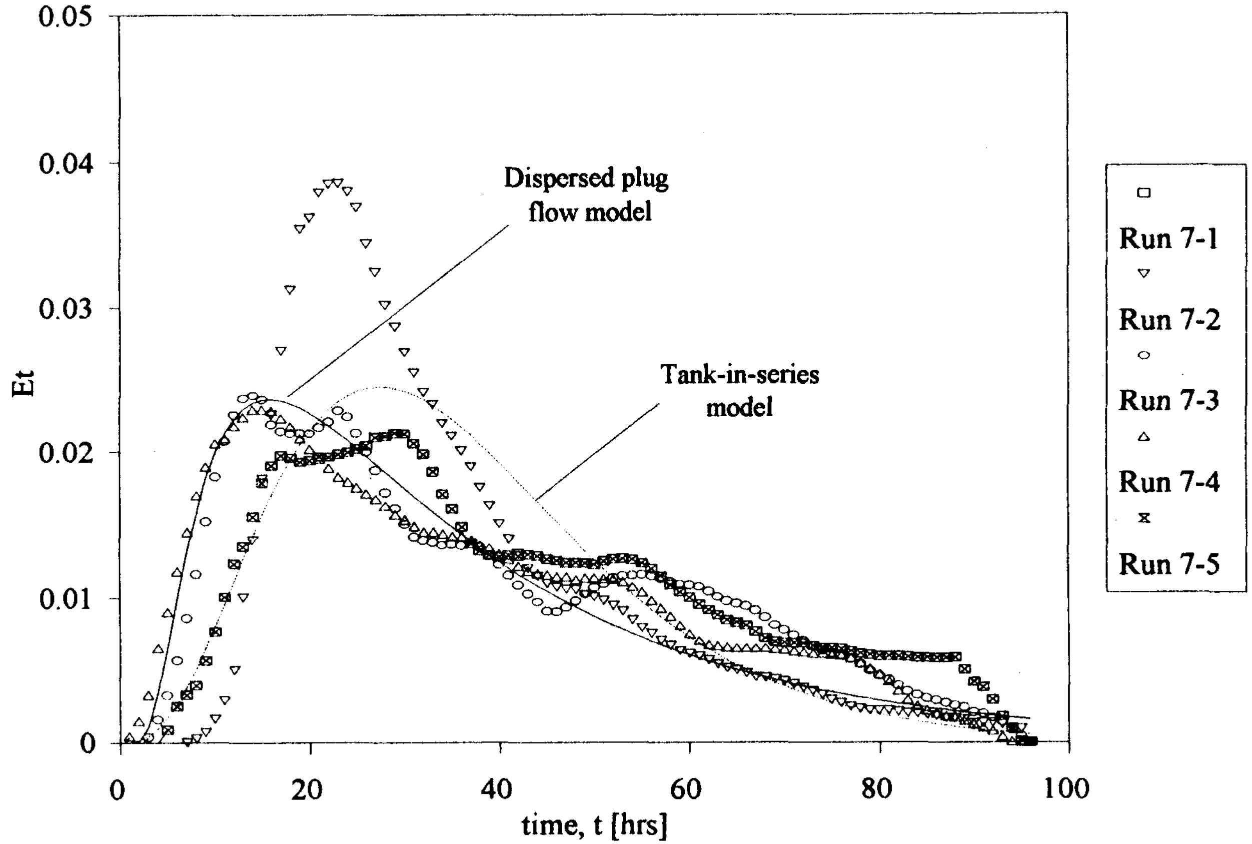

Figure 17. Fit of the dispersed plug flow and tanks-in-series models to tracer study data for pond #7 (unplanted)

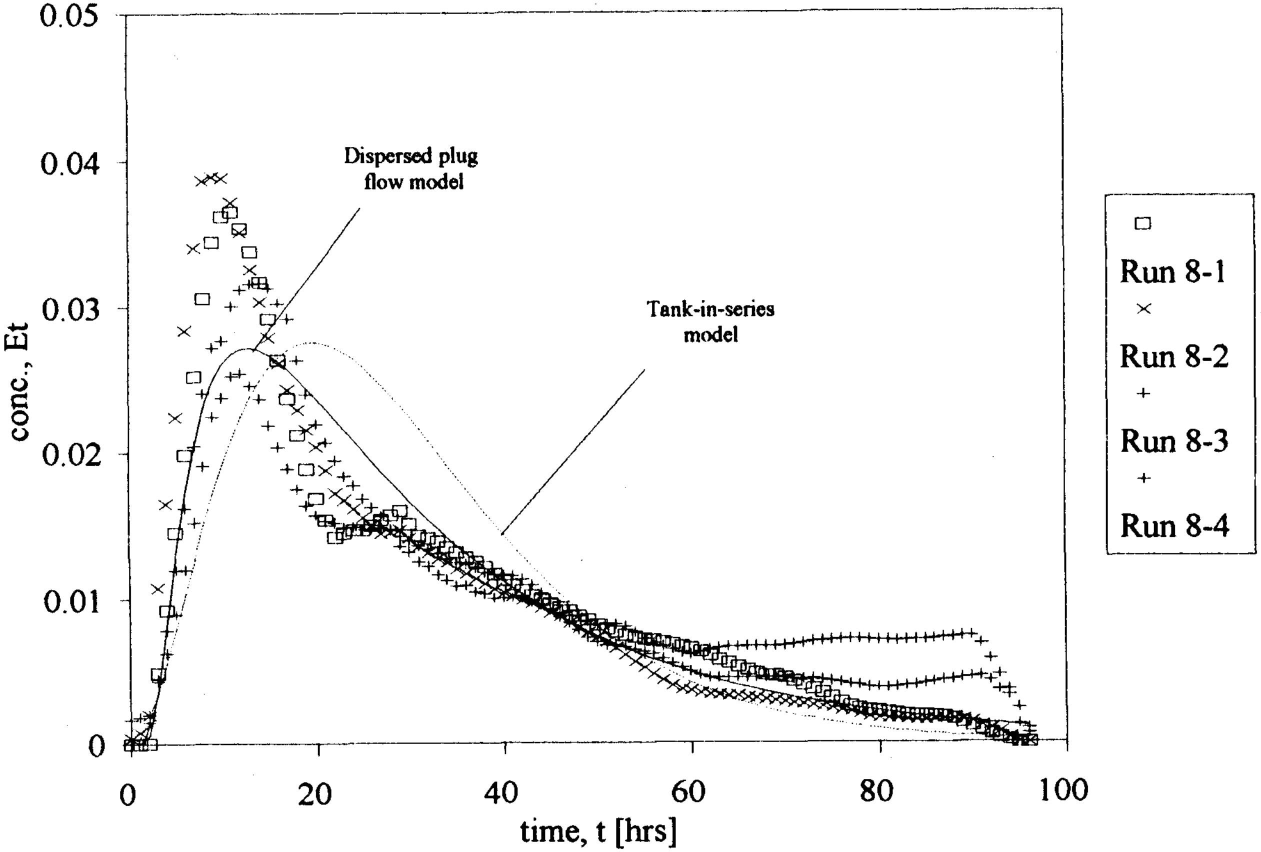

Figure 18. Fit of the dispersed plug flow and tanks-in-series models to tracer study data for pond #8 (cattail)

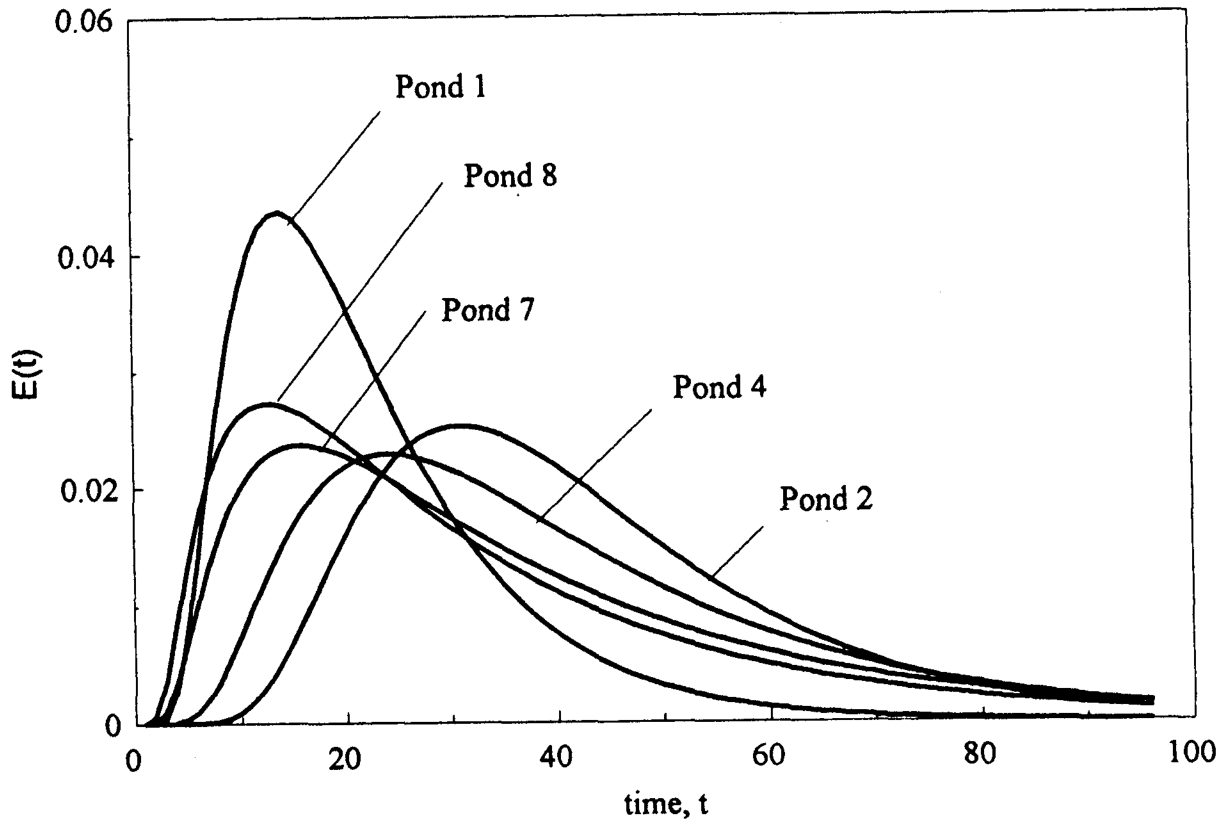

Figure 19. Comparison of dispersed plug flow models for all ponds.

Conclusions

Constructed wetlands for wastewater treatment are currently designed based on an assumption of plug flow. The affects of short circuiting, dead space, and dispersion in these systems result in a deviation from this ideal, a corresponding distribution of residence times. Since treatment efficiency is related in part to the residence time of the wastewater it is of interest to determine the degree to which these "engineered" systems deviate from ideal. Tracer studies of the constructed wetland provide a statistical picture of what the flow patterns of the system are like.

The primary purpose of this study was to apply the dye tracer methodology used in chemical reactor and wastewater treatment system analysis to the constructed wetland system. Rhodamine WT dye was chosen due to its nonsorptive properties, yet mass balances of tracer studies indicated dye losses from 13 to 59%. The dye also was susceptible to masking effects from the dark colored effluent.

The results provided some insight into the variability of these systems. Although all wetland cells investigated were physically the same (ie. length to width ratio, slope, flow rate), the degree of deviation from the plug flow ideal varied between cells, seasons, and plant type. Actual mean residence times of the planted systems were significantly less than would be predicted, yielding an apparent reduction in effective volume ranging from 15 to 25%. This correlates with a typical "porosity" suggested by the literature of 0.75 (Reed, 1988). The significant reduction in effective volume was attributed to two phenomena, that portion of the volume occupied by plant material and the portion unavailable to flow (eg. eddies). Estimating an average velocity of 0.05 cm/sec, the flow in the constructed wetland system was shown to be in the laminar flow region, as demonstrated by an estimate of the Reynold's number of 256. Traditional chemical reactor theory indicates a flow regime that is transitional between the purely convective and purely diffusive regions of flow. Dispersion was estimated via a "routing" procedure in one pond to be about 2.82

x 10^{-3} m^2/sec with a mean velocity of 0.06 cm/sec, verifying earlier estimates. Short circuiting in the system is indicated by very early time to peak concentrations in the dye studies ranging from 30 to 80 % of the theoretical detention time.

By combining the residence time distributions with an assumed a first order kinetic equation for conversion, it was demonstrated that the reduction in treatment due to hydraulic inefficiencies was 0.2 to 4.8% in the planted systems, compared to the ideal assumption of plug flow. For this wetland system the assumption of ideal plug flow would only produce a slight error (maximum 5%) in treatment predictions. This error will increase if either the hydraulic retention time or the kinetic coefficient are greater.

Two chemical engineering models, dispersed plug flow and tank-in-series were fit against the data. Both model fits were generally good with R^2 from 0.81 to 0.97. Unfortunately, these models are not theoretically developed to account for exchange of material in and out of stagnant regions in the system. The general shape of the residence time distributions suggested that fitting to the traditional dispersed plug flow model often used in modeling solute transport would not be altogether appropriate. A combination of the dispersed plug flow model with a component reflecting exchange with dead zones would seem to be intuitively more correct for this system.

RECOMMENDATIONS

The primary aim of this project was to investigate the hydraulic regime of a constructed wetland system via dye tracer studies. A myriad of further research opportunities arise from this introduction to residence time distributions, reactor theory, and flow models. In the quest to eliminate the "black box" approach to designing these systems, the following suggestions may prove beneficial to future applications of the technology.

1.) Although dye tracer studies are relatively effortless to perform, there are a number of anomalies that warrant further consideration. The failure to provide a material balance closure is perhaps the most obvious. Rhodamine WT sorption potential in surface water studies has not been documented. Moreover, insufficient data exists on the masking effects of opaque wastewater (eg. pulp mill wastewater) and the potential for bleaching of dye by chlorine. Perhaps the easiest way to verify the time of travel results obtained with Rhodamine WT are to run cotracer studies with a tracer that is less likely to adsorb, like bromide.

2.) The technique of tracer injection should also be explored, to determine the effect of injecting tracer into the inlet box, the distribution pipe, and directly into the flow path. Possible errors may have resulted in the tracer studies due to extended times required for the tracer to empty from the inlet box, explaining the extremely long tailing of the residence time distributions.

3.) Investigation of the physical parameters of the wetland cell and their affect on residence time is probably the most immediate need in the field of CW design. Treatment efficiency is influenced in part by residence time distribution. Studies of other systems, baffled reactors (Grobicki & Stuckey, 1992), lagoons (Middlebrooks et al.,1982) and wetlands (Wile et al. 1985) have demonstrated the affect of length to width (l:w) ratio on dispersion and consequent treatment efficiency. To eliminate the guess work in l:w ratios and in other parameters, model or prototype studies with dye tracers can compare residence time distributions. Combining these distributions with proper kinetic equations, quantitative values for treatment efficiency can be obtained, and the system optimized for maximum treatment. Other design variables lending to this type of analysis are internal baffling, depth, distribution/collection systems, and vegetation type.

4.) Although the dispersed plug flow and tanks-in-series models fit the data pretty well, there is some question as to there adequacy in describing the system. A more appropriate model would include a compensation for exchange of material between active and dead zones. Unpublished comments of late allude to the behavior of the system as a plug flow reactor interacting with series of complete mixed reactors along the length. A "transient storage" model of this type has been used in stream studies (Sabol & Nordin, 1978; Valentine & Wood, 1979; Bencala & Walters, 1983; Seo & Maxwell, 1992). A compartmental model of this type, although difficult to actually apply, seems intuitively more appropriate in the constructed wetland scenario.

BIBLIOGRAPHY

Agunwamba, J.C. (1992): Field pond performance and design evaluation using physical models. Water Research 26(10), 1403-1407.

Atkinson, B. (1993): The effects of channel modification on characteristics of streams during low flow. MS Thesis, Oregon State University.

Azimi, A; Horan, N (1991): The influence of reactor mixing characteristics on the rate of nitrification in the activated sludge process. Water Research 25(1), 419-423.

Bencala, K.E.; Rathburn, R.E.; Jackman, A.P.; Kennedy, V.C.; Zellweger, G.W.; Avanzino, R.J. (1983): Rhodamine WT dye loss in a mountain stream environment. Water Resouces Bulletin 19(6), 943-950.

Bencala, K.E.; Walters,RA (1983): Simulation of solute transport in a mountain pool and riffle stream: a transient storage model. Water Resources Research 19:3, 718-724.

Bowmer, K.H. (1987): Nutrient removal from effluents by an artificial wetland: influence of rhizosphere aeration and preferential flow studied using bromide and dye tracers. Water Research 21:5, 591-599.

Chen, S; Malone, R; Fall, L (1992): A modified procedure for the design of constructed wetlands. Presented at the International Winter Meeting of the ASAE at Nashville, Tenn. Dec 1992. Paper No. 92-4527.

Chudoba, J; Ottava, V; Madera, V (1973): Control of activated sludge filamentous bulking- I. Effect of the hydraulic regime or degree of mixing in an aeration tank. Water Research 7, 1163-1182.

Cuenca, R.H. (1989): **Irrigation System Design: an Engineering Approach**. Prentice-Hall Inc., Englewood Cliffs, NJ.

Danckwerts, P.V. (1953): Continuous flow systems: distributions of residence times. Chemical Engineering Science 2:1, 1-13.

Denbigh, K.G.; Turner,J.C.R. (1984): **Chemical Reactor Theory**. Cambridge University Press, Cambridge.

EPA (1988): Constructed wetlands and aquatic plant systems for municipal wastewater treatment. (Design Manual: EPA/625/1-88/022)

Everts, C.J. (1989): Comparison of tracer mobilities under laboratory and filed conditions. Journal of Environmetal Quality 18, 491-498.

Feurstein, D.L.; Selleck, R.E. (1963): Fluorescent tracers for dispersion measurements. Journal of the Sanitary Engineering Division of ASCE 89:SA4, 1-21.

Finkner, S.C. & Gilley, J.E. (1986): Sediment and dye concnetration effects on fluorescence. Applied Engineering in Agriculture 2:2, 104-107.

Fischer, H.B. (1968): Dispersion measurements in natural streams. J. of Sanitary Engineering Div. of ASCE 94:SA5, 927-943.

Fisher, P.J. (1990): Hydraulic characteristics of constructed wetlands at Richmond, NSW, Australia. In: **Constructed Wetlands in Water Pollution Control**. (Eds: Cooper,PF; Findlater,BC) (IWAPRC.) Permago Press, Oxford, 21-31.

Fox, A.M.; Heller, W.T.; Getsinger, K.D. (1991): Factors that influence water exchenge in spring fed tidal canals. Estuaries 14:4, 404-413.

Gaspar,E (Ed.) (1987): **Modern Trends in Tracer Hydrology. Vol. I & II**. CRC Press, Inc., Boca Raton, Fla.

Gibilaro, L.G. (1978): On the residence time distributions for systems with open boundaries. Chemical Engineering Science 33, 487-492.

Grobicki, A; Stuckey, D.C. (1992): Hydrodynamic Characteristics of the Anearobic Baffled Reactor. Water Research 26(3), 371-378.

Hammer, D.A. (1992): Designing Constructed Wetland Systems to Treat Agricultural Nonpoint Source Pollution. Ecological Engineering 1:1/2(March), 49-82.

Hammer, D.E.; Kadlec, R.H. (1986): A model for wetland syrface water dynamics. Water Resources Research 22:13, 1951-1958.

Hantzsche, N.N. (1985): Wetland systems for wastewater treatment: engineering applications. In: **Ecological Considerations in Wetlands Treatment of Municipal Wastewaters**. (Eds: Godfrey, P.T.; Kaynor, E.R.; Pelczarski, S.; Benforado, J.). Van Nostrand Reinhold Co., New York.

Jones, D.R.; Jung, R.F. (1990): Analytical problem arising from the use of bromide and rhodamine Wt as co-tracers. Water Research 24:1, 125-128.

Kadlec, R.H. (1987): The Hydrodynamics of Wetland Water Treatment Systems. In: **Aquatic Plants for Water Treatment and Resource Recovery**. (Eds: Reddy, K.R.; Smith, W.H.) Magnolia Publ., Orlando,Fla, 373-392.

Kadlec, R.H. (1989): Hydrologic Factors in Wetland Water Treatment. In: **Constructed Wetlands for Wastewater Treatment--Municipal, Industrial, and Agricultural**. (Ed: Hammer, D.A.) Lewis Publ., Chelsea, Michigan, .

Kadlec, R.H. (1990): Overland Flow in Wetlands: Vegetation Resistence. Journal of Hydraulic Engineering 116:5(May), 691-706.

Kadlec, R.H.; Hammer, D.E.; Nam, I.; Wilkes, O. (1981): The Hydrology of Overland Flow in Wetlands. Chemical Eng. Comm. 9, 331-344.

Kreft, A; Zuber, A (1978): On the physical meaning of the dispersion equation and its solution for different initial and boundary conditions. Chemical Engineering Science 33, 1471-1480.

Levenspeil, O (1972): **Chemical Reaction Engineering**. John Wiley & Sons, Inc., New York.

Levenspeil, O (1993): **Chemical Reactor Omnibook**. OSU Bookstores, Inc, Corvallis, OR.

Martin, E.J.; Martin, E.J.(1991): **Technologies for Small Water and Wastewater Systems**. Van Nostrand Reinhold, New York.

Middlebrooks, E.J.; Middlebrooks, C.H.; Reynolds, J.H.; Watters, G.Z.; Reed, S.C.; George, D.B. (Eds.) (1982): **Wastewater Stabilization Lagoon Design, Performance, and Upgrading**. MacMillan Publ. Co. Inc., New York.

Moreno, M.D. (1990): A tracer study of the hydraulics of facultative stabilization ponds. Water Research 24(Aug), 1025-30.

Nauman, E.B. (1981): Residence time distributions in systems governed by the dispersion equation. Chemical Engineering Science 36, 957-966.

Netter, R; Biscofsberger, W (1990): Hydraulic investigations on planted soil filters. In: **Constructed Wetlands in Water Pollution Control.** (Eds: Cooper,PF; Findlater,BC) (IWAPRC.) Permagon Press, Oxford, 11-20.

Nordin, C.F.; Troutman, B.M. (1980): Longitudinal dispersion in rivers: the persistence of skewness in observed data. Water Resources Research 16:1, 123-128.

Reed, S.C.; Middlebrooks, E.J.; Crites,R.W. (1988): **Natural Systems for Waste Management and Treatment**. McGraw-Hill, New York.

Reichert, P; Wanner, O (1991): Enhanced one-dimensional modeling of transport in rivers. Journal of Hydraulic Engineering 117, 1165-1183.

Roberson, J.A.; Crowe, C.T. (1985): **Engineering Fluid Mechanics**. Houghton Mifflin Co., Boston, MA.

Sabatini, D.A.; Austin, T.A. (1991): Characteristics of Rhodamine WT and Fluorescein as adsorbing ground water tracers. Ground Water 29:3, 341-349.

Sabol, G.V.; Nordin, C.F. (1978): Dispersion in rivers as related to storage zones. J.Hydraulics Div. ASCE 104:HY5, 695-708.

Seo, I.W.; Maxwell, W.H. (1992): Modeling low-flow mixing through pools and riffles. Journal of Hydraulic Engineering 118:10, 1406-1423.

Smart, C.C. (1988): Artificial tracer techniques for the determination of the structure of conduit aquifers. Ground Water 26:4, 445-453.

Smart, P.L. (1985): Applications of fluorescent dye tracers in the planning and hydrological appraisal of sanitary landfills. Quarterly J. of Engineering Geology 18, 275-286.

Smart, P.L.; Laidlaw, I.M.S. (1977): An evaluation of some fluorescent dyes for water tracing. Water Resources Research 13(1, Feb), 15-33.

Steinback, B.J (1987): Effluent flowmeter calibration using fluorescent dye tracers. Pulp & Paper Canada 88:12, 227-230.

Steiner, G.R.; Freman, R (1989): Configuration and substrate design for constructed wetlands in wastewater treatment. In: **Constructed Wetlands for Wastewater Treatment--Municipal, Industrial and Agricultural**. (Ed: Hammer,Donald A) Lewis Publ., Chelsea, MI., 1989.

Tchobanoglous, G; Burton, F.L. (1991): **Wastewater Engineering: Treatment, Disposal, and Reuse**. 3rd ed. (Metcalf & Eddy) McGraw-Hill, Inc., New York.

Tchobanoglous, G; Schroeder, E (1985): **Water Quality**. Addison-Wesley Publishing Co., Reading, Mass.

Thirumurthi, D. (1969): A break through in tracer studies of sedimentation tanks. Journal of the Water Pollution Control Federation 41(11, part 2, Nov), R405-R418.

Valentine, E.M.; Wood, I.R. (1979): Experiments in longitudinal dispersion with dead zones. J.Hydraulics Div. ASCE 105:HY8, 999-1016.

Watson, J.T.; Hobson, J.A. (1989): Hydraulic design considerations and control structures for constructed wetlands for wastewater treatment. In: **Constructed Wetlands for Wastwater Treatment--Municipal, Industrial, and Agricultural.** (Ed: Hammer,Donald A) Lewis Publ., Chelsea, MI., .

Weber, A.S.; Tchobanoglous, G (1985): Rational design parameters for ammonia conversion in water hyacinth treatment systems. Journal WPCF 57:4, 316-323.

Wile, I; Miller, G; Black, S (1985): Design & use of artificial wetlands. In: **Ecological Considerations in Wetland Treatment of Municipal Wastewaters**. (Eds: Godfrey,Paul et al.) Van Nostrand Reinhold Co., New York, 26-35.