Temperature and plant effects on secondary wastewater treatment in model constructed wetlands
by Joel Aaron Biederman

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in Environmental Engineering
Montana State University

Abstract:
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Eight bench-scale horizontal subsurface-flow wetlands were constructed in late 1995 and placed in a climate-controlled greenhouse. Three each were planted with typha (cattail) and scirpus (bulrush) while two remained unplanted. Application of a synthetic secondary wastewater was maintained to these flow cells for nearly three years, beginning in April 1996. Samples of influent and effluent were analyzed for parameters of concern, including chemical oxygen demand (COD), nitrogen, phosphorus, and sulfur. A second research system consisted of 16 batch-mode CW columns, planted with typha, scirpus, or carex (sedge), or left unplanted as controls. These were placed in the same greenhouse as the subsurface flow wetlands in April 1997 and received an identical wastewater. Samples were extracted during each 20-day batch application and analyzed for COD. Greenhouse temperature was modulated between 4 and 24 °C on an annual cycle in increments of 4 °C lasting approximately one month each. COD data from the columns were fitted to a first-order model, modified to include a nonzero asymptote. Assessments of other data were performed using effluent-to-influent concentration ratios and moving average plots of influent and effluent concentrations.

The modified model fit the data well. Plant effects on COD kinetics were noted in many cases, with carex and scirpus generally producing better degradation than typha or control treatments. Effects of temperature on kinetics were substantial, although the carex and scirpus treatments seemed less sensitive. Plant effects on COD removal were less noticeable in the subsurface-flow wetlands, but temperature effects were apparent. The subsurface-flow wetlands showed excellent removal of COD and sulfate, Phosphorus removal was of short duration, and nitrogen removal results varied. The behavior of both systems continued to change after several years of operation.

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APPROVAL

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Joel Aaron Biederman

This thesis has been read by each member of the thesis committee and has been found to be satisfactory regarding content, English usage, format, citations, bibliographic style, and consistency, and is ready for submission to the College of Graduate Studies.

Otto R. Stein, Ph. D. — Otto R Stein (Signature) — 9/7/99 Date

Approved for the Department of Civil Engineering

Donald A. Rabern, Ph. D. — Donald A. Rabern (Signature) — 9/7/99 Date

Approved for the College of Graduate Studies

Bruce McLeod, Ph. D. — Bruce R. McLeod (Signature) — 9-8-99 Date

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ABSTRACT

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INTRODUCTION

The use of artificial wetlands to treat wastewater is gaining attention and acceptance in many parts of the United States. This relatively new technology is usually referred to as treatment wetlands or constructed wetlands (CW). Compared to conventional treatment methods, constructed wetlands can offer simple design, low capital costs, minimal operation and maintenance concerns, and a variety of ancillary benefits, such as the provision of habitat for birds and other wildlife. These attributes have made CW an attractive alternative for small and onsite wastewater treatment systems. In particular, CW may be appropriate for many farm wastewaters, which contribute over 65% of the total pollution to U.S. surface waters (USEPA, 1989).

Although CW systems have been used successfully in various applications for several decades, many projects have reported mixed results (Brix, 1994). Historically, CW design has relied upon rules of thumb and other empirical methods. Widespread application of the technology will require the development of more rational, process-based design criteria. The first step in this direction has involved the adaptation of performance criteria for conventional organic wastewater systems to the design of CW. Such criteria are generally based upon assumed first-order degradation of organic matter and nitrogen compounds via microbial pathways. These microbial processes are known to be strongly temperature-dependent, which has limited the use of CW systems outside of warm and temperate regions. The effective use of CW in colder climates is hindered by a lack of operational data, and of information about the physical and biological controls that regulate constructed wetlands behavior.

Goals and Objectives

The goal of the present research is to contribute to the development of rational design approaches for constructed wetlands in cold climates. This contribution consists of quantification and modeling of the effects of a cold climate on the performance of constructed wetlands for the treatment of water pollutants associated with rural environments. Specific objectives of this project are: (a) to provide additional information on the feasibility of using subsurface-flow wetlands to treat wastewater in cold climates, (b) to develop objective design and operational criteria for subsurface-flow constructed wetlands based on evaluation and modeling of seasonal and temperature effects, (c) document the water quality of effluent from model CW wetlands. This research did not address mechanical operation in cold climates. This issue has been addressed elsewhere, with generally good results, as shall be discussed below.

The aim of this thesis is to report the results from two subsurface CW test systems operated for several years at Montana State University. Bench scale wetlands and wetland columns were constructed and placed in a climate-controlled greenhouse. Native wetland plants were tested for their possible effects on treatment rate and extent. Temperature and daylight conditions were varied on an annual cycle to approximate the environment of a field-scale subsurface CW system. A synthetic wastewater was designed to mimic critical attributes of secondary wastewater. Concentrations of several important parameters were measured, including chemical oxygen demand (COD), total organic carbon (TOC), several nitrogen species, phosphate, total phosphorus (TP) and sulfate. Results were analyzed to consider the effects of temperature, season, and plant type on wastewater treatment efficacy.

Approach

COD data from the wetland columns were fitted to a first-order model, modified to contain a non-zero asymptote to account for the background COD in the CW. A nonlinear mixed-effects regression technique was used to generate estimates of this background COD and the first-order rate constant. A separate modeling run was conducted for each of sixteen 20-day cycles, spanning a temperature range of 4 to 24 °C over fourteen months termed the kinetic study period. In addition to the modeling results, COD performance data from both flow cells and columns are presented and evaluated for seasonal, temperature, and plant treatment effects. TOC data from the flow cells are presented and compared to COD as a measure of organic matter content.

Although the emphasis here is on the kinetic study period, results are presented for the wastewater monitoring performed during the first three years of this ongoing CW project. These include hydrodynamics, COD, ammonium and total N, phosphate and total P, and sulfate. Data for each of the wastewater constituents are evaluated for the effects of season, temperature, plant treatment and influent wastewater strength.

BACKGROUND

Definition of Natural, Created, and Constructed Wetlands

A wetland is generally defined as an ecosystem or land area that is partially or entirely flooded at least some of the time. A wetland may range from those areas that have saturated conditions existing below the soil surface to those that are deeply and permanently flooded. A wetland ecosystem gives way to an aquatic ecosystem where the depth or duration of flooding is such that the growth of emergent or submerged vegetation is prevented (Kadlec and Knight, 1996). The great majority of wetlands are natural, though their common proximity to bodies of water makes them prime targets for drainage and development.

A wetland that is man-made for non-wastewater purposes is generally termed a created wetland. Reasons for creation of wetlands include habitat improvement, aesthetic benefits, flood control, and wetland loss mitigation. Under the policy of "no net wetland loss" proclaimed by former U.S. President George Bush, many development concerns have been required to create wetlands in mitigation of those destroyed in development.

Constructed wetland is the term usually applied to systems specifically designed and built for the treatment of polluted water. These may also be referred to as treatment wetlands or artificial wetlands.

Historical Development of Constructed Wetlands

The association of wastewater and wetlands is not a new one. For centuries, humans have disposed of their wastewaters into bodies of water or depressions in the landscape. In the latter case, a wetland was often present or else resulted from the disposal (Cooper and Boon, 1987). This practice continues today in developing nations and many rural portions of developed nations, with wastewater disposed into ditches, trenches, or shallow infiltration systems, often giving rise to wetland conditions. Though the recognition and classification of this practice as a treatment technology is relatively recent, there exists evidence that ancient Egyptians and Chinese cultures used wetlands for the disposal of municipal wastes (Brix, 1994).

North American experience includes various examples of wetland use for wastewater disposal, including the Great Meadows in Lexington MA, beginning in 1912; the Brillion Marsh in Wisconsin which has received municipal discharge since 1923; the Cootes Paradise natural wetland in Hamilton, Ontario, receiving treated effluent since 1919, and a discharge to a natural cypress swamp in Waldo, FL since 1939. In 1983, the USEPA reported finding about 324 unmonitored and unregulated "swamp discharges" in the 14 states comprising EPA regions 4 and 5. When monitoring was initiated at some of these sites, the treatment potential of wetlands came to light (Kadlec and Knight, 1996).

The first scientific consideration of wetlands for treatment use is credited to Dr. Kathe Seidel of the Max Planck institute. Throughout the 1950s, '60s, and '70s, Dr. Seidel conducted experiments with CW for treatment of domestic wastes, industrial wastes, road runoff and heavy metals, using bulrushes and other higher plants (Brix 1994). CW research expanded in Germany and other parts of Western Europe in the 1960's and 1970's, leading most notably to the development of the "Root Zone Method" of wastewater treatment by Dr. R. Kickuth of Germany. This term referred to CW systems planted in heavy soil or light clay, sometimes with amendments to improve the sorptive capacity. The common failure in such systems was due to the incorrect assumption that plant roots would increase the permeability over time, which often resulted in ponding and overland flow (Brix, 1994).

Usage of both natural and constructed wetlands for water treatment gained momentum in the 1970s and '80s in North America. Work by Spangler et al. (1976), Nichols (1983) Gersberg et al. (1984) and others led to the development of full-scale demonstration projects, such as the wetlands system at Arcata, CA (Gearheart et al., 1989). Constructed wetlands have been used with modest to substantial success for wastewaters with various pollutants, including animal wastes (Szogi et al., 1997; Zimmerman et al., 1994; Hill, 1998), phenanthrene (Machate et al., 1997), human waste (Pride, 1990) and metals (Tarutis and Unz, 1995).

Removal of Organic Matter and Nitrogen Compounds in CW

Performance Expectations

CW are designed to target the removal of organic matter (OM) more often than any other parameter, and good results may generally be expected (WPCF, 1990). It is thought that degradation of OM in CW occurs by primarily aerobic pathways, although concentrations of dissolved oxygen are usually less than 1 mg/l (WPCF, 1990), and this assumption has been challenged (Burgoon, 1995). Some degree of OM production or cycling occurs in all wetlands, and this may result in minimum attainable concentrations in the effluent. As a consequence, OM removal efficiency is often decreased at low influent concentrations (WPCF, 1990; Kuehn and Moore, 1995). Emergent plants affect metabolic pathways and overall removal efficiencies in some cases (Burgoon, 1991a) but not in others (Kuehn and Moore, 1995). Documentation of successful removal of OM using CW abounds (i.e. Wolverton, 1983; Tanner, 1995a). Excellent reviews may be found in Brix (1997) and Kadlec and Knight (1996).

The success of nitrogen removal in CW is more variable. Nitrogen may be stored by adsorption and sedimentation processes or plant uptake, and may be removed from the system by denitrification. Since influent to many wetlands contains primarily NH3-N, a nitrification step is usually necessary to precede denitrification. The nitrification step often appears to be limiting, as nitrifying organisms do not compete well with heterotrophs for limited oxygen supplies (Blicker, 1997). Nitrogen removal by nitrification-denitrification has been observed in various cases, including the work of Wolverton et al. (1983) and Burgoon et al. (1995).

Modeling OM Degradation

As a primary design target for reduction, organic matter has received the most attention with regard to degradation modeling. Most such efforts have involved adaptation of equations governing the performance of conventional treatment systems. For degradation of organic matter, often quantified using a bulk parameter such as BOD, COD, or TOC, the simple first-order model has been the most commonly accepted (WCPF, 1990; USEPA, 1988):

$$\frac{C_e}{C_o} = e^{-k_T t_H} \qquad \textbf{(2.1)}$$

where C_e is the effluent concentration, C_o is the influent concentration, k_T is a temperature-dependent rate parameter and t_H is the hydraulic residence time in the wetland cell. Hydraulic residence time for a subsurface-flow wetland can be determined from:

$$t_H = \frac{Vn}{q} \tag{2.2}$$

where V is the gross volume of the wetland, n is the porosity of the rock media and q is the design flow rate. The temperature-dependent rate parameter k_T can be described by the empirical modified Arrhenius relation (WPCF, 1990):

$$k_T = k_{20}(1.06)^{(T-20)} \tag{2.3}$$

where k_{20} is the degradation rate parameter at 20°C and T is temperature in °C.

Many authors have documented some success in fitting the first-order model to BOD data. For example, Tanner et al. (1995a) found that removal of total BOD in a subsurface wetland system treating dairy wastewater could be described by the above equations. Crites et al. (1994) supported use of the first-order model but suggested that k_{20} was between 0.8 d^{-1} and 1.1 d^{-1}. Burgoon et al. (1991) fit the first-order model to planted batch microcosms and noted that fitted values of k_T decreased as the wastewater residence time in the microcosm was increased. Further work with a similar system led to the conclusion that the first-order model remained valid for only the first 18 to 24 hours of the batch, after which time the degradation slowed markedly (Burgoon et al., 1995). Similar results were obtained by Allen (1999) in the present research.

Initial investigation of COD data from batch microcosms in the present study indicated that COD degradation curves appeared to approach a nonzero asymptote, which was termed Cr. Kadlec and Knight (1996) have proposed the existence of such a non-zero asymptote for BOD degradation in constructed wetlands. They attribute this residual to a background inherent in the wetland system. Sources might include carbon fixation, degradation of plant materials or other storage compartments, such as litter. The first-order model was modified as follows to include the existence of a nonzero asymptotic value of COD:

$$\frac{C - Cr}{Co - Cr} = e^{-kt} \tag{2.4}$$

C = COD concentration, mg/l
Co = initial COD concentration, mg/l
Cr = residual COD concentration, mg/l
k = first-order decay parameter, d^{-1}
t = time, days

A model of the same form was proposed by Kadlec and Knight (1996) for BOD degradation. Treatment of the residual Cr may be difficult. Though some residual values have been reported (Kadlec and Knight 1996), these are sparsely supported and thought to vary with many factors, including season, wetland age and ecology, and nutrient loading. Through a regression of data, Kadlec and Knight (1996) found only weak dependence of Cr upon influent load. However, there is evidence that Cr may be related to input OM and not just to wetland background levels. Orhon et al. (1993) conducted experiments on the biological treatability of COD from both dairy effluent and sucrose in model, complete-mix, activated sludge reactors. Although neither influent source contained an appreciable inert fraction, the effluent produced contained inert microbial products amounting to 6 to 7% of the influent COD.

Factors Affecting Wetland Performance

As pseudo-natural systems, constructed wetlands are subject to a greater variety of controls and influences than conventional engineered treatment technologies. Some of these factors, such as hydrology, can be understood in terms of traditional engineering assessments, whereas others require new research and perhaps contributions from other disciplines. The regulating factors most important to this study are discussed below.

Hydraulic Format and Media

There are two common classifications of constructed wetlands: free water surface operation (shallow pond) and subsurface operation (water surface is maintained below that of the soil or other rooting medium). Free water wetlands are usually operated in a continuous flow mode, with one or more influent and effluent points. Subsurface wetlands may be operated in a variety of fashions, including batch, continuous horizontal flow (saturated) or continuous vertical flow (unsaturated). A discussion of some of these may be found in Kadlec and Knight (1996), Breen and Chick (1989) and Rogers et al. (1991). Subsurface-format CW (both continuous-flow and batch) were deemed more appropriate for operation in cold climates, so these were chosen for the present research.

A variety of porous media are used in constructed wetlands. After early failures with fine soils in the "Root Zone Method," focus has shifted to the use of sands and gravels. These media are sufficient for most rooting wetland plants, and have lesser tendency to plug. One drawback is that coarse media have much lower surface-to-volume ratio, and therefore become saturated with sorbents much more quickly. It is recognized that sorption may aid removal of various wastewater constituents, and that it is the only net sink for phosphorus (WPCF, 1990).

Climate

The effects of cold-temperature operation on subsurface wetland performance are debated. Operational issues, such as freezing are of concern. However, several demonstration projects have functioned successfully without major operational problems (see review by Jenssen et al., 1993). It is well known that biological processes, such as those that accomplish breakdown of organic matter, may slow down or stop as temperatures decrease. However, examples of effective biological treatment at cold temperatures are not uncommon, (i.e. Margesin and Schinner, 1998). Excellent tertiary treatment of wastewater in CW was reported by Gubricht (1992) for a project in southern Sweden, and generally good results have been reported by Doku and Heinke (1995) for CW operating in the Northwest Territories, Canada. Little is known about the effects of temperature on breakdown kinetics of OM and nitrogen in constructed wetlands. Current information suggests that nitrogen kinetics may be strongly affected, but that OM kinetics are relatively insensitive to temperature (Kadlec and Knight, 1996).

Plant Selection

Emergent or floating macrophytes (plants) are usually considered an important component of CW systems. Their role may include transport of oxygen into the wetland, filtration of solids, nutrient uptake, insulation against freezing in winter, and the provision of suitable habitat for microorganisms and larger species (Reddy et al., 1989a). In much of the earlier CW research, plants were given the majority of the credit for treatment (Wolverton, 1976; Bastian, 1979). In the 1980s, however, the major role of microorganisms was recognized (Wolverton, 1983; Gersberg et al., 1984), and some of the presumed benefits of plants were even called into question (Davis, 1984; Brix, 1987).

The enhancement of dissolved oxygen in the root zone is one of the primary reasons for using plants in CW systems (Armstrong et al., 1992; Reddy et al., 1989a). The ability to transport oxygen to roots from aerial parts has been observed in various plants whose rooting systems are submerged for part or all of the year. It is believed that this adaptation allows plants to supply respirational oxygen to roots and to combat the deleterious effects of an otherwise reduced environment (Armstrong, 1964, Sand-Jensen and Prahl, 1982; Moorehead and Reddy, 1988). Justin and Armstrong (1987) describe the physiological traits that represent this adaptation. If the oxygen transport to the root zone exceeds that required by the plant to satisfy its respiratory needs, then the excess may be available for release or "leakage" into the surrounding solution. It is thought that this oxygen may then become available for use by heterotrophs or by autotrophic nitrifiers. Burgoon et al., (1995) documented an apparent oxygen transport rate of 28.6 g/m^2 of wetland area per day by Scirpus *pungens* in a subsurface-flow wetland for BOD removal. Several authors have reported evidence of increased nitrification activity in the root zone of wetland plants (Bodelier, 1996; Engelaar, 1995). If the bulk solution remains largely anaerobic while the root zone is oxygenated, the situation might prove ideal for the sequential nitrification and denitrification necessary for permanent net nitrogen removal. Reddy et al. (1989b) introduced ^{15}N-labeled ammonia into wetland microcosms and subsequently detected ^{15}N in the air above planted columns, but not in the air above unplanted columns. This was taken as direct evidence of nitrification-denitrification in the root zone. Rogers et al. (1991) found that although dissolved oxygen was supplied from the roots of several wetland plants, nitrification-denitrification was not enhanced, and plant uptake accounted for the majority of nitrogen removal. Likewise, Farahbakhshazad et. al. (1997) monitored profiles of ammonia, nitrate, and dissolved oxygen in the root zone of Phragmites *australis* (common reed) and concluded that the plant influence on nitrification-denitrification had been overemphasized.

The beneficial effect of oxygen transport by plants in CW was generally accepted for several decades after the oxygen-transport work of Armstrong (1964). However, the magnitude of the transport has been questioned and debated hotly in the last decade (Armstrong 1990, 1992; Bedford et al. 1991, 1994; Sorrel and Armstrong 1994). Some of the debate has focused on whether it is appropriate to measure dissolved oxygen or redox potential, and on the distance from the roots to take such measurements. Flessa and Fisher (1992) clearly demonstrated increased redox potential in the root zones of submerged rice plants. However, Howes and Teal (1994) showed that a net loss of oxygen from roots was absent in the salt marsh plant Spartina *alterniflora* even when respiration was inhibited, and that a net uptake of oxygen was in some cases noted in the uninhibited plants. The work of Gries and Kappen (1990) added evidence that temperature and season influenced such behavior. Though much of this work shed light on the fundamental nature of wetland plants, little of it could be directly applied to the constructed wetland environment.

The laboratory experiments of Reddy et al. (1989a) were accepted as evidence of oxygen transport by plants in a wastewater treatment setting. The authors sealed the roots of several species of wetland plants into flasks of primary sewage effluent, leaving the aerial parts exposed, and calculated the transport of oxygen as follows:

$$O_2 \text{ Transport} = BOD_5 \text{ (Initial} - \text{Final)} + DO \text{ (Final} - \text{Initial)} \qquad \textbf{(2.5)}$$

BOD5 = 5-day biochemical oxygen demand (mg/l)
DO = dissolved oxygen concentration (mg/l)

Use of this equation, however assumes that any decrease in BOD_5 may be attributed to oxygen-consuming aerobic respiration. That assumption may be far from reasonable in a wetland environment. Burgoon et al. (1995) contended that methanogenesis was the major removal pathway for carbonaceous BOD in model constructed wetlands. If degradation of organic substances is accomplished through anaerobic or a mixture of aerobic and anaerobic pathways, then accurate kinetic modeling may be more complex than the current efforts. Also, the particular transport characteristics and suitability of a particular species may vary greatly from one setting to another. Augmentation of the oxidation status by plants should, in any case, be expected to affect treatment, and the redox depth profile data of Allen (1999) may shed additional light on this subject.

Uptake of nutrients from wastewater is thought to be another advantage to the use of plants in constructed wetlands. All plant species require both nitrogen and phosphorus for tissue development, and compounds containing these elements are often of concern as limiting nutrients downstream from waste treatment processes. Typical tissue concentrations of above-ground biomass in Typha *sp.* and similar plants are 0.4 to 4.0% dry weight nitrogen and 0.1 to 1.0% dry weight phosphorus. Documented annual rates of uptake for these same plants range from about 30 to 60 g/m^2 for nitrogen and from 8 to 11 g/m^2 for phosphorus (Cary and Weerts, 1984; Davis, 1984; Gopal and Sharma, 1988). These amounts are sufficient to impact annual effluent loads in lightly loaded wetlands (Brix, 1997). Unfortunately, plant uptake does not represent a steady, long-term sink for these nutrients. Upon the senescence and death of plant tissues, most nutrients are returned to solution (WPCF, 1990). The only net removal is the nutrient content of undegraded refractory organic residue (Brix, 1997). Removal of these nutrients by plant harvesting has been deemed impractical for several reasons. First, it is difficult to determine when the most efficient time for harvesting occurs. Plants actively redistribute nutrients among their above- and below-ground parts throughout the year. Graneli et al., (1992) found that Phragmites *australis* began to move nutrients and carbohydrates from leaves into rhizomes as early as June, in the midst of the growing season. Davis (1984), however, reported higher leaf tissue nutrient concentrations in fall than in spring for Typha *domingensis*. A study of Typha *elephantina* in warm conditions found 35% of N and 50% of P was translocated to rhizomes for winter storage and then reused in the spring (Gopal and Sharma, 1988). Harvesting may therefore be relatively ineffective during the senescent season, though harvesting during the growth season might interrupt other beneficial plant influences. Plant uptake is therefore considered an active storage compartment rather than a true removal mechanism (Brix, 1987).

Use of Batch Reactors to Mimic Plug-Flow CW

In ideal terms, a finite element of fluid in a plug-flow reactor may be considered a batch reactor that moves through space. For this reason, batch reactors have often been used to pilot the behavior of plug-flow reactors. This is true in treatment wetlands research, as many researchers have used batch-mode "microcosms" or "columns" to predict or understand continuous-flow CW or "flow cells" (Burgoon et al., 1991, 1995; Breen and Chick, 1989).

One aim of this project was to evaluate the comparability of flow cells and columns operated under similar conditions with the same wastewater. Several factors related to hydraulics should be considered when making this comparison. First, flow cells do not always operate in a true plug-flow fashion. Although water does enter at one end and leave at the other, mixing, dispersion, detention and short-circuiting may all cause deviation from ideal behavior (Kadlec and Knight, 1996). Second, the two reactors are quite different from a microbial ecology standpoint. The flow cell, whether truly plug-flow or not, can be expected to contain gradients in microbial ecology in response to the steady-state gradients in chemical concentrations. That is, each region of the wetland is likely specialized in response to a certain suite of chemical concentrations generally found there. The microbes in batch-mode columns, on the other hand, see cycles of changing conditions over time. This scenario is likely to produce a mixed consortia of organisms, some of which are likely to be facultative, in order to respond to the rapidly changing chemical concentrations and redox status. There may be some spatial gradients in the column, but the data of Allen (1999) indicate that such columns are fairly well mixed. So, an element of wastewater in a flow cell moves through various semi-permanent microbial niches, while one in a column may drift slowly through a fairly homogeneous, but temporally variable environment.

The plants introduce another major difference between the flow cells and columns. If the column is well mixed with regard to the batch duration, then most of the wastewater can be expected to pass through the immediate root zone. In flow cells, the possible reduction in porosity caused by the plant roots may cause preferential flow beneath the densest rooting zone, limiting root zone contact (Breen and Chick, 1989).

METHODS AND MATERIALS

Experimental Overview

Two experimental systems were designed. The first experiment consisted of eight bench-scale constructed wetlands with continuous horizontal subsurface flow, termed "flow cells". This system was established in December 1995. Two planted treatments, termed "typha" and "scirpus" were compared to an unplanted "control" treatment as the system was subjected to seasonal changes in photoperiod and ambient temperature. Influent and effluent samples were collected approximately once per hydraulic residence time (five days) and analyzed for various wastewater parameters. The second experiment involved the use of smaller, subsurface wetland microcosm columns operated in batch mode. This column experiment was located in the same greenhouse as the flow cells and experienced the same environmental conditions, but its operation started 16 months later (April 1997). The experiment involved unplanted controls and the same plant treatments as the flow cells, with the addition of treatment planted to sedge, termed "carex". Each treatment had four replicates, and the treatments were randomly placed in a rectangular array. A series of 20-day batches were conducted using the same wastewater as was applied to the flow cells. Time series data were collected from the columns for modeling of waste-degradation kinetics.

Both experimental systems were observed at different temperatures and stages of plant growth to assess seasonal variability and plant influence. The final 14 months of the study, termed the kinetic study period, were used to assess the effects of temperature modulation on each of the systems. Ambient temperatures were 4 °C in January and February 1998, increasing in 4 °C increments to a high of 24 °C in August, and decreasing again in 4 °C increments to reach 4 °C in January 1999. Relative humidity ranged from 30 to 70%, with no seasonal pattern. Supplemental lighting was not used; cumulative daily net solar radiation ranged from 1 to 8 $MJ/m^2/d$ and, due to greenhouse shading, was about 25% of locally recorded net solar radiation (Towler, 1999). The environmental conditions appeared to produce normal seasonal cycles of plant dormancy and growth.

Bench-Scale Flow Cells Constructed Wetlands

Flow Cells Design and Construction

During late 1995, eight flow cells were constructed and placed in an environmentally controlled greenhouse at Montana State University's Plant Growth Center. Each CW cell was 152 cm long, 76 cm wide, and 53 cm deep and filled with gravel (13-19 mm) to a depth of 47 cm (Figure 1). Cells were constructed of 16-mm polypropylene and fitted with 0.75" (19 mm) PVC distribution and collection manifolds. The manifolds were constructed with 3-mm slots on the crown and were placed horizontally across the flow cell to ensure an even flow distribution. The distribution manifold was placed 41 cm, and the collection manifold 5 cm, above the bottom of the cell. The collection manifold led to a separate water surface-regulating tank to allow calibration of the wetland phreatic line. This was set at 46 cm for the duration of the study. All cells were set on customized pallets so that they could be transported if necessary. Initial porosity was 0.40.

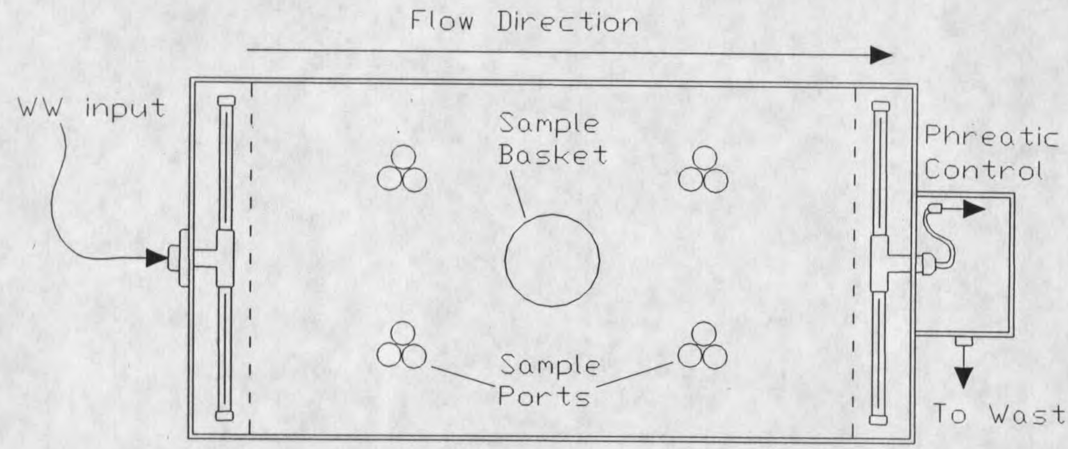

Figure 1. Plan view schematic of a constructed wetland flow cell

Sampling port clusters were located on 25-cm width by 51-cm length centers within each cell . Each cluster was comprised of three individual ports to allow for monitoring of water quality at depths of 15 cm, 30 cm, and 41 cm measured downward from the gravel surface. Each port consisted of half-inch (13-mm) PVC pipe drilled at the appropriate depth and a length of 3-mm flexible tubing secured to the pipe. The former allowed insertion of probes while the latter allowed withdrawal of water samples using a standard Leur-slip syringe. One sampling port of each cell was fitted with a differential thermocouple at a depth of 36 cm from the upper gravel surface. Other than for