Potential for Using Constructed Wetlands to Treat Landfill Leachate: Literature Review and Pilot Study Design

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ABSTRACT

All municipal landfills are susceptible to infiltration by precipitation and runoff. As this water infiltrates the landfill, many substances leach from the fill including oxygen demanding organic compounds, suspended solids, nitrogen compounds, phosphorus, metals, and toxic organics. The composition of the leachate varies tremendously among landfills, but concentrations of several of these components may be quite high. Recent legislation requires all new municipal landfills to be lined and to include a leachate collection system. Collected leachate must either be treated on-site to meet discharge permit limitations or transported to a treatment facility. Many municipal wastewater treatment plants may require pretreatment of leachates before they will accept them. Hauling and pretreating is usually an expensive option for leachate management and may create potential hazards during transportation. Construction and operation of conventional wastewater treatment facilities on-site are also expensive.

Constructed wetlands are engineered treatment systems that make use of the same contaminant removal mechanisms that function in natural wetlands. Constructed wetlands have low capital costs, low maintenance requirements, and provide green space or wildlife habitat in addition to their treatment function. These types of systems have been used for treatment of municipal wastewater, acid mine drainage, urban stormwater runoff, agricultural runoff, animal wastes, and industrial wastewater. A more recently explored application is the treatment of landfill leachate. Several studies have been conducted by various researchers in the eastern United States to assess wetland treatment systems.

The objective of this project is to explore the feasibility of utilizing constructed wetlands as an alternative treatment technology for municipal landfill leachate. Because treatment feasibility depends on the type and concentration of contaminants as well as the flow characteristics of the leachate, characteristics of landfill leachate are discussed. Descriptions of the various types of constructed wetland treatment systems are also presented. Research on the use of these systems to treat leachate are discussed. Relatively few studies have been conducted on landfill leachate, so additional discussion is presented on the likely benefits and problems of this technology based on research using other types of wastewaters. Finally, a description of a research pilot project being conducted at the landfill in New Hanover County, North Carolina, is presented.

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INTRODUCTION

All municipal landfills are susceptible to infiltration by precipitation and runoff. As this water infiltrates the landfill, many substances leach from the waste including oxygen demanding organic compounds, suspended solids, nitrogen compounds, phosphorus, metals, and toxic organics. The composition of the leachate varies tremendously among landfills, but concentrations of several of these components may be quite high. Recent legislation requires all new municipal solid waste landfills to be designed with a single composite liner. This liner system is composed of a leachate collection system, a synthetic membrane liner, and a compacted clay layer (NCSWMR 1994). Collected leachate must either be treated on-site to meet discharge permit limitations or transported to a treatment facility off-site. The high variability of leachate characteristics makes design and operation of treatment facilities difficult. Treatment of the leachate frequently requires a combination of physical, chemical, and biological treatment processes. Pretreatment may be required prior to biological treatment to remove toxic metals (Bagchi 1990; Keenan et al. 1984, 1993). Many municipal wastewater treatment plants will require pretreatment of leachates to reduce high contaminant concentrations or remove toxic substances before they will accept them. Hauling and pretreating is usually an expensive option for leachate management and may create potential hazards during transportation. Construction and operation of conventional wastewater treatment facilities on-site are also expensive. One reason for the expense is due to the need for continuous management by trained personnel during the many years of post-closure leachate collection and treatment.

A number of possible alternatives have been suggested for treating leachate. On-site lagoons have been found to successfully treat high biochemical oxygen demand (BOD) leachate, but long detention times are required (Lavigne 1979). Recirculation of the leachate through the landfill will also provide treatment, particularly anaerobic biological degradation of organic compounds producing carbon dioxide and methane. Biodegradation is frequently limited by moisture in landfills, therefore recirculation of the leachate may stimulate this process. Addition of nutrients may be required to further enhance biodegradation. This process may actually increase concentrations of organics as it speeds up the waste stabilization process. Leachate recirculation may also provide an opportunity for microbially mediated removal of otherwise refractory toxic contaminants. Organic compounds are degraded, but the resulting leachate will still require treatment for remaining organics which may be largely refractory, total ammonia nitrogen (TAN), and other components (Borden and Yanoschak 1989; Diamadopoulos 1994; Pohland and Harper 1986; Pohland et al. 1990; Robinson and Maris 1985; Senior 1990). This process may therefore be used for pretreatment and could be integrated with other on-site technologies. Land application of leachate is another management strategy. However, this disposal option requires treatment to reduce organics, metals, and possibly other contaminants prior to application (Pohland and Harper 1986). Constructed wetlands are another type of on-site treatment that makes use of natural contaminant removal processes and, like pond treatment, they may require some pretreatment, depending on the characteristics of the leachate. Because of the potential for constructed wetlands to treat wastewater with highly variable characteristics, they will be the focus of this report.

Constructed wetlands are engineered treatment systems that make use of the same contaminant removal mechanisms that function in natural wetlands. They have been found to provide treatment for a wide range of contaminant compositions and concentrations and for highly variable flow rates. One reason for this is the diverse environments wetlands provide, including both aerobic and anaerobic microsites. Constructed wetlands have low capital costs, low maintenance requirements, and can be integrated into an urban resource management plan (Reed et al. 1988; Reed 1993; Watson 1992). They provide wildlife habitats, green space, and recreational opportunities in addition to their treatment function. These types of systems have been used for treatment of municipal wastewater, acid mine drainage, urban stormwater runoff,

agricultural runoff, animal wastes, and industrial wastewater. A more recently explored application is the treatment of landfill leachate. Several test studies of constructed wetland systems have been conducted by various researchers in the eastern United States, but many questions remain concerning the applicability of this technology to treatment of landfill leachate.

The objective of this project is to explore the feasibility of constructed wetlands as an alternative treatment technology for municipal landfill leachate. Because treatment feasibility depends on the type and concentration of contaminants as well as the flow characteristics of the leachate, characteristics of landfill leachate will be discussed. Descriptions of the various types of constructed wetland treatment systems will also be presented. Research on the use of these systems to treat leachate will be discussed. Relatively few studies have been conducted on landfill leachate, so additional discussion will be presented on the likely benefits and problems of this technology based on research using other types of wastewaters. Finally, a description of a research pilot project at the landfill in New Hanover County, North Carolina, will be presented.

CHARACTERISTICS OF LEACHATE

Leachate from landfills is created by precipitation and runoff water percolating through waste material in the landfill. As the water moves through the waste, various types of contaminants enter the water in relatively high concentrations but relatively low flow rates. These contaminants include products of decomposition of organic material, such as volatile organic compounds and ammonia, dissolved solids, metals, and toxic compounds. Toxic compounds can be leached from common household hazardous waste, such as cleaners, disinfectants, and car, home, and lawn products, discarded in the landfill. The composition and concentration of the contaminants in the leachate depends on many variables and are highly variable with time and from site to site. One important factor is the character of the waste buried in the landfill. Typical municipal solid waste has been changing in character recently as recycling and alternate disposal of yard waste are implemented. The effect on leachate characteristics has not yet been determined. Also, waste-to-energy facilities at some sites result in landfilling a high percentage of ash from the incinerators. This removes much of the biodegradable waste from the landfill and has a significant effect on leachate characteristics. The age of the landfill also affects the leachate because characteristics of the leachate change over time. Younger landfills tend to leach more volatile fatty acids, due to mostly anaerobic biological activity. As the landfill ages, organics in the leachate tend to be more refractory, which may result in high chemical oxygen demand (COD) but relatively low BOD. The organic acids in younger leachate tend to cause low pH, which results in higher metals concentrations. Iron and manganese are generally present in reduced form at high concentrations. Zinc is also commonly found in landfill leachates. Other more toxic metals, such as nickel, chromium, copper, cadmium, mercury, and lead, are present in lower concentrations. Other factors that affect leachate characteristics include moisture, temperature, pH, nutrient availability, and permeability. Not only will leachate composition vary, but the amount of leachate produced by a landfill will also vary. Important factors include precipitation and runoff reaching the landfill, condition of the waste, and type of cover. Landfills will continue to produce leachate after they are closed and capped, although at a lesser rate, so treatment must continue for many years (Borden and Yanoschak 1989; Diamadopoulos 1994; Pohland and Harper 1986; Pohland et al. 1990; Robinson and Maris 1985; Senior 1990).

The extreme variability of leachate composition is evident in the list of components (Table 1) and the list of organic compounds that have been detected in municipal landfill leachate (Table 2). One of the challenges of effective treatment is finding a system that will work for a wide range of leachate compositions and flow rates, because these characteristics vary with time at any landfill site. Some of the variations are short-term, resulting from seasonal or yearly weather patterns. However, more long-term variations occur as the landfill ages, the biodegradable material stabilizes, the composition of the waste placed in the landfill changes, and the landfill is taken out of service.

There are approximately 20 lined landfills currently operating in North Carolina. Each of these facilities has a leachate collection system and a mechanism for treating leachate. In addition, there are about 25 to 30 other landfills in North Carolina under construction or at various stages of the permitting process (Mussler 1995). State regulations require that all landfills as of January 1, 1998, must have a single composite liner and therefore collect leachate. Since the leachate will need to be treated prior to discharge, constructed wetland systems may serve as a viable treatment option for many of these landfills.

Table 1. Observed Leachate Composition (USEPA 1986)

Constituent	Concentration Range (mg/L)
Biochemical Oxygen Demand (BOD)	5 - 75,000
Chemical Oxygen Demand (COD)	50 - 90,000
Total Organic Carbon (TOC)	50 - 45,000
Total Solids (TS)	1 - 75,000
Total Dissolved Solids (TDS)	725 - 55,000
Total Suspended Solids (TSS)	10 - 45,000
Volatile Suspended Solids (VSS)	20 - 750
Total Volatile Solids (TVS)	90 - 50,000
Fixed Solids (FS)	800 - 50,000
Total Coliforms (CFU/100 mL)	0 - 10^5
Fecal Coliforms (CFU/100 mL)	0 - 10^5
Hardness (as $CaCO_3$)	0.1 - 36,000
Alkalinity (as $CaCO_3$)	0.1 - 20,350
Acidity (as $CaCO_3$)	2,700 - 6,000
Total Phosphorus	0.1 - 150
Organic Phosphorus	0.4 - 100
Phosphate	0.4 - 150
Nitrate Nitrogen	0.1 - 45
Ammonia Nitrogen	0.1 - 2,000
Organic Nitrogen	0.1 - 1,000
Total Kjeldahl Nitrogen	7 - 1,970
Total Volatile Acids (TVA)	70 - 27,700
Turbidity (NTU)	30 - 450
Specific Conductance (μmho/cm)	960 - 16,300
Cadmium	0 - 0.375
Chromium	0.2 - 18
Copper	0.1 - 9
Iron	200 - 5,500
Lead	0.001 - 1.44
Manganese	0.6 - 41
Mercury	0 - 0.16
Nickel	0.2 - 79
Selenium	0 - 2.7
Zinc	0.6 - 220
pH	3.5 - 8.5

Table 2. Organic Compounds Detected in Municipal Landfill Leachate (USEPA 1986)

Constituent	Concentration Range (µg/L)
Acetone	140 - 11,000
Benzene	2 - 410
Bromomethane	10 - 170
1-Butanol	50 - 360
Carbon tetrachloride	2 - 398
Chlorobenzene	2 - 237
Chloroethane	5 - 170
bis (2-Chloroethoxy) methane	2 - 14
Chloroform	2 - 1,300
Chloromethane	10 - 170
Delta BHC	0 - 5
Dibromomethane	5 - 25
1,4-Dichlorobenzene	2 - 20
Dichlorodifluoromethane	10 - 369
1,1-Dichloroethane	2 - 6,300
1,2-Dichloroethane	0 - 11,000
cis 1,2-Dichloroethene	4 - 190
trans 1,2-Dichloroethene	4 - 1,300
Dichloromethane	2 - 3,300
1,2-Dichloropropane	2 - 100
Diethyl phthalate	2 - 45
Dimethyl phthalate	4 - 55
Di-n-butyl phthalate	4 - 12
Endrin	0 - 1
Ethyl acetate	5 - 50
Ethyl benzene	5 - 580
bis (2-Ethylhexyl) phthalate	6 - 110
Isophorene	10 - 85
Methyl ethyl ketone	110 - 28,800
Methyl isobutyl ketone	10 - 660
Naphthalene	4 - 19
Nitrobenzene	2 - 40
4-Nitrophenol	17 - 40
Pentachlorophenol	3 - 25
Phenol	10 - 28,800
2-Propanol	94 - 10,000
1,1,2,2-Tetrachloroethane	7 - 210
Tetrachloroethene	2 - 100
Tetrahydrofuran	5 - 260
Toluene	2 - 1,600
1,1,1-Trichloroethane	0 - 2,400
1,1,2-Trichloroethane	2 - 500
Trichloroethene	1 - 43
Trichlorofluoromethane	4 - 100
Vinyl chloride	0 - 100
m-Xylene	21 - 79
p-Xylene + o-Xylene	12 - 50

WETLAND TREATMENT TECHNOLOGIES

Characteristics of Natural Wetlands

It is difficult to define exactly what a wetland is because the term refers to a rather wide variety of environments. However, wetlands do have many distinguishing characteristics, especially associated with hydrology, soil, and vegetation. One such characteristic is the presence of water, either at the surface or within the root zone. Another distinguishing characteristic is a unique soil condition, due to accumulation of organic material and a saturated subsurface. A third distinguishing characteristic is the type of vegetation found in wetlands. Plants found in wetlands are adapted to wet conditions and are frequently referred to as hydrophytes (Mitsch and Gosselink 1993).

Natural wetlands also perform a number of characteristic functions. Hydrological functions that may be performed by wetlands include flood mitigation and aquifer recharge. Wetlands are important for water quality improvement, because they remove contaminants from water by a number of different chemical, physical, and biological mechanisms. Wetlands also are important for fish and shellfish production, for providing habitat for a wide variety of wildlife and endangered species, and for creating recreational opportunities and aesthetic settings.

Subsurface Flow and Free Water Surface Constructed Wetlands

Constructed wetlands are built for a treatment purpose and designed to maximize the contaminant removal functions of wetlands. Thus, they are regulated as treatment processes rather than as jurisdictional wetlands. Two common types of constructed wetlands are used for treatment of wastewaters are free water surface (also referred to as artificial marsh) and subsurface flow (or root zone method or rock-reed filters). These systems consist of lined basins containing a solid substrate, such as soil or stone, and some type of emergent aquatic vegetation, frequently *Typha* (cattail), *Phragmites* (common reed), or *Scirpus* (bulrush). Wastewater that is applied to the system is treated by natural physical, chemical, and biological processes. The key removal mechanism for BOD and nitrogen in these systems appears to be microbiological (Reed et al. 1988; Watson et al. 1989). Nitrogen is mostly removed through the processes of nitrification (microbial conversion of ammonia to nitrate) and denitrification (microbial conversion of nitrate to N_2), although sedimentation/filtration, ammonia volatilization, and plant uptake may also contribute to removal (Rogers et al. 1991; Schierup et al. 1990). The role of emergent vegetation in contaminant removal is to provide attachment sites for microbial growth, transport oxygen to the subsurface, provide some amount of direct uptake, and perhaps improve permeability of subsurface flow (SSF) systems.

Considerable disagreement exists at this time about whether the free water surface (FWS) system or the SSF is more effective. The FWS consists of a shallow basin containing sand or soil as the solid substrate (Figure 1). The surface is flooded with water to a depth of 6 to 24 inches. Oxygen is provided by diffusion through the water surface. Algae generally do not provide significant oxygen to the water because they are effectively shaded by the emergent vegetation. The SSF system uses a more hydraulically conductive substrate, such as coarse sand or stone (Figure 2). Water applied to the system flows below the surface of the substrate, allowing filtration, sorption, precipitation, and microbial processes to remove contaminants. Microorganisms attach to the substrate surfaces and to plant roots. Oxygen is provided primarily through transport by the plants from the air to the root zones (Armstrong et al. 1990; Gersberg et al. 1989; Reddy et al. 1989). Such systems using reeds are already fairly common in Europe (Cooper 1993).

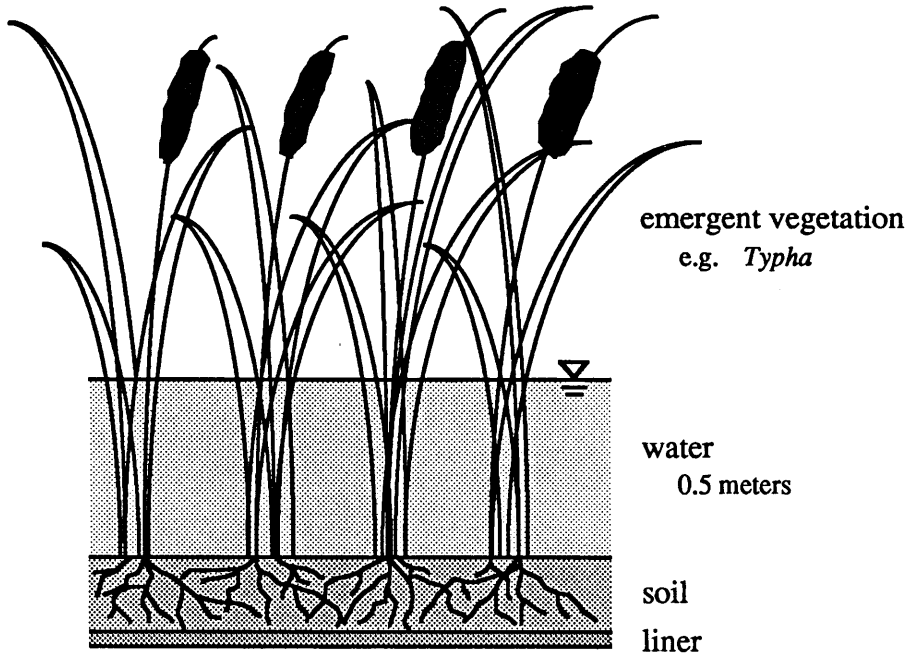

Figure 1. Schematic diagram of a free water surface constructed wetland.

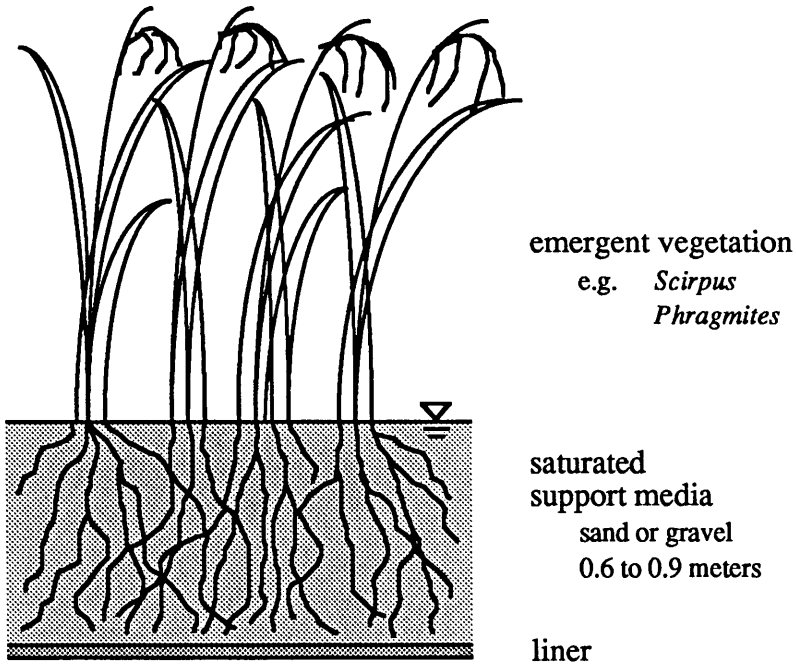

Figure 2. Schematic diagram of a subsurface flow constructed wetland.

Although such systems have been shown to be effective in various applications, design criteria vary widely and are not based on solid theory of removal mechanisms within these systems (Bavor et al. 1989; USEPA 1988; Reed and Brown 1992a; Steiner and Watson 1993; Watson et al. 1989). Most design models are simple first-order relationships. Since many studies of constructed wetlands have not determined removal mechanisms, they do not offer insight into why some systems work well and others do not. They also do not provide much help in determining if results are applicable over a wide range of application conditions. However, guidelines for various aspects of municipal applications for constructed wetland design are available (Cooper 1993; Hammer and Knight 1992; Reed and Brown 1992a; Steiner and Watson 1993; USEPA 1988; WPCF 1990; Watson 1992). The following is a discussion of some of the guidelines given for treatment of municipal wastewater.

Substrate
Differences in soils affect chemical transformations and contaminant removal in wetlands. Contaminants such as nonpolar organic compounds, metals, and phosphorus may adsorb or undergo ion exchange with various components of soils, including organic matter, iron and aluminum oxides and hydroxides, calcite, organometalic complexes, and clay minerals (Faulkner and Richardson 1989). However, adsorption and ion exchange sites can become saturated, and the processes are highly pH and redox dependent. Therefore, these mechanisms are not thought to be reliable for long-term treatment, and constructed wetlands are not currently designed to take advantage of these removal mechanisms. The current recommendation for support media for SSF systems is a one-inch gravel. This size is large enough to minimize problems with clogging and yet small enough to allow plants to survive and to provide surface area for microbial attachment (Cooper et al. 1989; Sanford et al 1990b; Steiner and Freeman 1989; Steiner and Watson 1993). Use of a locally available waste product, such as culled bricks, could be economically favorable and reduce material buried in landfills. However, the effect of these materials on water quality is unknown. Free surface wetlands can usually use on-site fill soil.

Length:Width
Length to width ratios of 4:1 or less are currently recommended for constructed wetlands (Crites 1992; Hammer and Knight 1992). Previous recommendations suggested larger length to width ratios for the purpose of minimizing short circuiting in the beds (Steiner and Freeman 1989). However, this strategy is more costly and can result in overloading the front end of the systems.

Plants
Important characteristics of plants in constructed wetlands are tolerance of fluctuating water levels and high pollutant concentrations, provision of attachment surfaces, and oxygen transport to the subsurface. Macrophytes transport oxygen to their roots and rhizomes in wetlands through their internal lacunal system. During transport, oxygen can leak from the roots and provide a thin aerobic zone in the bulk anaerobic environment of the wetland sediments (Brix 1993b). This creates diverse microbial habitats that stimulate many contaminant removal mechanisms. Specific wetland plant species have other characteristics which make them particularly suitable for constructed wetland systems. *Typha* appears to be well suited to the FWS type of system and provides extensive surface area for attachment of microorganisms (Birkbeck et al. 1990; Reed et al. 1988; USEPA 1988). *Phragmites* and *Scirpus* appear to be most effective for the root zone method of oxygen transfer because of deep penetration of roots and rhizomes into the subsurface (Brix 1993b; Conley et al. 1991; Cooper et al. 1989; Watson et al. 1989; Wood 1990). This makes them suitable for both FWS and SSF systems. Problems with *Phragmites* and *Typha* are that they are considered a nuisance in some places and can crowd out other species that might be desirable for wildlife habitats or community diversity. Other hydrophytes, such as *Juncus* (rush) and *Sparganium* (burreed), have been found to transfer oxygen to the surrounding subsurface (Michaud and Richardson 1989). *Zizaniopsis miliacea* (giant cutgrass), *Panicum hemitomon* (maidencane), *Pontederia cordata* (pickerelweed), and *Sagittaria lancifolia* (arrowhead) have

also been found to thrive in wetland treatment systems (Surrency 1993). Some ornamental plants have also been used, especially for on-site municipal treatment, but data are not yet available regarding their effectiveness.

Water Depth
The depth of the support media in SSF systems is selected based on the depth of root penetration of the plant species, which will be important for effective oxygen transfer by the root zone method. *Scirpus* has the deepest root penetration of the commonly used wetland plants, up to 76 cm (Cooper et al. 1989; Gersberg et al. 1986). In more shallow SSF systems the root penetration is more likely to reach full depth thus preventing flow from bypassing the rhizosphere.

In FWS systems, the soil substrate should be deep enough to allow rooting of the emergent plants. Shallow water depths will provide better oxygen transfer through surface diffusion and therefore are expected to allow better nitrification (Crites 1992). The same conditions will possibly allow greater ammonia volatilization. Since the oxygen transfer per unit area is fixed in a FWS wetland, a reduction in water depth provides proportionately more oxygen per unit volume. Therefore the water depth of FWS wetlands should be maintained between 15 - 60 cm (6 - 24 in).

Detention Time and Loading Rates
A number of different criteria have been suggested for designing constructed wetlands, based primarily on data collected using municipal wastewater or domestic sewage. Detention time (water volume divided by the flow rate) has been used as a general guide for standard types of treatment. Previous studies have shown that between 5 and 14 days detention time is adequate for treatment in most FWS systems, depending in part on the contaminant of most concern. A shorter detention time applies to removal of BOD while a longer time applies to removal of ammonia and total nitrogen (TN). In general, shorter detention times have been recommended for SSF systems (Conley et al. 1991; USEPA 1988; Reed and Brown 1992b; Watson et al. 1989).

Two types of loading rates are important for proper operation of wetland systems. One type is hydraulic loading rate (HLR), which is volume per area per time or flow rate per area. This loading rate is related to detention time in that it is based on flow rate rather than mass of contaminant. The other is mass loading rate (MLR), which is mass of contaminant per area per time, or concentration times flow rate per area. BOD and TAN are the only contaminants for which suggested MLRs have been given.

For SSF systems, the HLR currently recommended guidelines range from 1.5 to 8 cm/day (Hammer and Knight 1992; Platzer and Netter 1992; Watson et al. 1989; Watson 1992). The MLR recommendations are highly variable for both BOD and TAN (Hammer and Knight 1992; Swindell and Jackson 1990; WPCF 1990; Watson et al. 1989). Some design guidelines are given in terms of oxygen demand (for both BOD and TAN), assuming oxygen transfer into the system is the limiting factor for microbial transformation. MLRs of up to 142 kg O_2/ha-day have been recommended, which corresponds to some published values for maximum oxygen transfer by the root zone method (Armstrong et al. 1990; USEPA 1988; Reed et al. 1988; Watson et al. 1989). However, BOD removal can also occur from anaerobic microbial reactions. Therefore, oxygen transfer rate into the system may not be a good determiner of BOD loading rate, especially when denitrification is desired.

For FWS systems, loading rates generally fit within ranges given for SSF systems, with recommended HLRs being at the lower end of the range (i.e., require longer detention times) and MLRs being at the higher end (USEPA 1988; Reed et al. 1988; WPCF 1990). FWS wetlands are not thought to be as limited by oxygen transfer as are SSF wetlands (Hammer and Knight 1992).

An alternative to use of loading criteria for design is the use of empirical design equations or a simple first-order rate equation. These types of equations have been applied to design for both BOD and nitrogen removal (Bavor et al. 1989; Hammer and Knight 1992; Watson et al. 1989; WPCF 1990). However, they should be applied with caution to situations that do not match conditions for which the equations were developed.

Limiting Factors

There are many possible factors that could limit the rate of contaminant removal and thus control the design of constructed wetland systems (Gersberg et al. 1986, 1989; Hsieh and Coultas 1989; Stengel and Schultz-Hock 1989; Watson et al. 1989). The main factor is widely thought to be the rate of oxygen transfer into the system. Oxygen transfer is important for aerobic degradation of organic contaminants and for the nitrification process which converts ammonia to nitrate. Another major limiting factor in nitrogen removal may be organic carbon available for denitrification, which converts nitrate to nitrogen gas (N_2). Supplemental carbon may have to be added if there is insufficient carbon present in the inflow or easily degradable carbon produced by plants in the wetland. Microbiological reactions, especially nitrification, can also be significantly limited by cold temperatures. Removal of contaminants in which the main removal mechanism is adsorption, such as phosphorus, will become limited after operation of the system has continued long enough to saturate the adsorption sites.

A number of other factors could also limit the success of wetland treatment systems (Reed et al. 1988; WPCF 1990). The startup time of two to three years for full efficiency may not be adequate for some applications. Hydrological problems can develop, especially in SSF systems as the pore spaces become clogged with organic debris or inorganic precipitates. Some wastewaters will be especially likely to produce inorganic precipitates, such as waters with high alkalinity, hardness, iron, or manganese concentrations. FWS wetlands may also provide breeding areas for mosquito. This can be controlled by the mosquitofish, *Gambusia*, or may require pesticide application. Odors may also be a problem for some types of wastewater in some areas. The growth of wetland vegetation may be limited by nutrient availability or by toxic substances in the applied wastewater, including ammonia, metals, or trace organics. Toxicity is affected by a number of environmental factors, including pH and the presence of chelating agents (Portier and Palmer 1989). The effects of toxic substances in constructed wetlands are not well understood.

Few constructed wetlands have been studied for a long enough period to determine the long-term problems that will develop as the system ages. Such problems include clogging of SSF wetlands by organic matter or inorganic precipitates which may cause changes in hydraulic conductivity and result in ponding or sedimentation. Another potential long-term problem is the need for plant management by burning or harvesting to maintain desired plant species and control buildup of dead organic matter. Accumulation of metals or other toxic substances after long periods of operation is another potential problem that has not been thoroughly studied.

Each system has advantages and disadvantages. FWS wetlands typically provide better oxygen transfer, do not have clogging problems, and provide for better ammonia volatilization, whereas SSF wetlands typically have better temperature control in the winter and do not require mosquito control (Reed et al. 1988).

Vertical Flow Constructed Wetlands

Vertical flow constructed wetlands are vegetated systems in which the flow of water is vertical rather than horizontal, as in FWS and SSF wetlands (Figure 3). Wastewater is applied at time intervals over the entire surface of the wetland. The water flows through a permeable medium and is collected at the bottom. The intermittent application allows the cell to drain completely

before the next application (Watson 1992). This type of operation allows for much more oxygen transfer than typical SSF systems and thus may be a good option for treatment of wastewaters that have relatively high oxygen demand. This type of system has been recommended for removal of high levels of ammonia through nitrification (Watson and Danzig 1993). High BOD levels may cause clogging, however, due to microorganism buildup. This type of system has the advantage of greatly increasing the oxygen available for microbial reactions, but also increases the mechanical and operational complexity of the system over the more traditional types of wetland treatment processes. Limited research has been conducted on vertical flow wetlands, so design characteristics are not discussed here.

Artificial Wetlands

Artificial wetlands refer to man-made wetlands that are intended to simulate, at least to some extent, natural wetlands. They are typically constructed for mitigation (to replace impacted natural wetlands) or aesthetic purposes. However, they can sometimes be used for treatment as constructed wetlands are. To better mimic natural wetlands, artificial wetlands may include several types of pond and wetland environments (Figure 4). Different environments encourage plant community diversity which increases the wildlife habitat and aesthetic value of the wetland. The diversity of environments can also have a beneficial effect on the treatment function of the wetland (Conway and Murtha 1989; Gearheart et al. 1989; Gearheart 1992; Steiner and Freeman 1989). Different types of environments are best for removal of different pollutants. Open water components, or ponds, may provide greater aeration potential and are usually preferred for aerobic processes, such as nitrification. Dense emergent vegetation may promote anaerobic conditions and thus favor denitrification. Both processes are required for complete nitrogen removal, so the ideal artificial wetland system may include both types of environments. Other types of pollutants, such as some organic compounds and metals, are also best removed by alternate exposure to aerobic and anaerobic environments. The variety of environments in one treatment system also increases the likelihood of successfully removing all contaminants from the wastewater since all types of contaminants may not be removed in any one type of environment.

Systems with ponds as a component may make use of floating or submersed aquatic vegetation. Floating plants, such as water hyacinth, pennywort, and lemna, or plants with floating leaves, such as some *Potamogeton* species, block light and help to prevent excessive algal growth. However, some species, like the water hyacinth, are considered nuisance plants because they reproduce quickly and obstruct waterways. Thus they are not desirable in most systems. Submersed plants, such as *Elodea* and *Ceratophyllum*, or submersed parts of floating plants provide attachment sites for microorganisms. These pond systems can be effective at maintaining aerobic microbial processes, can increase the ambient pH causing precipitation of metals and phosphates, and can become established quicker than wetland emergent vegetation (Brix 1993a,b; DeBusk et al. 1989). They do not seem to be as stable, however, and are best used in combination with wetland environments.

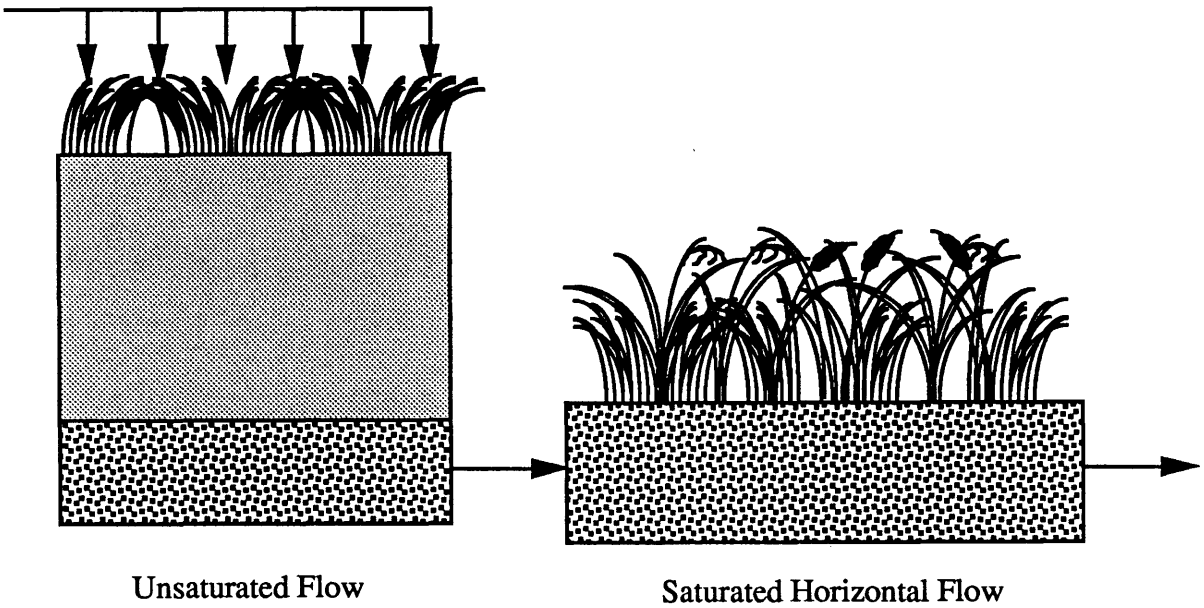

Figure 3. Schematic diagram of a vertical flow constructed wetland.

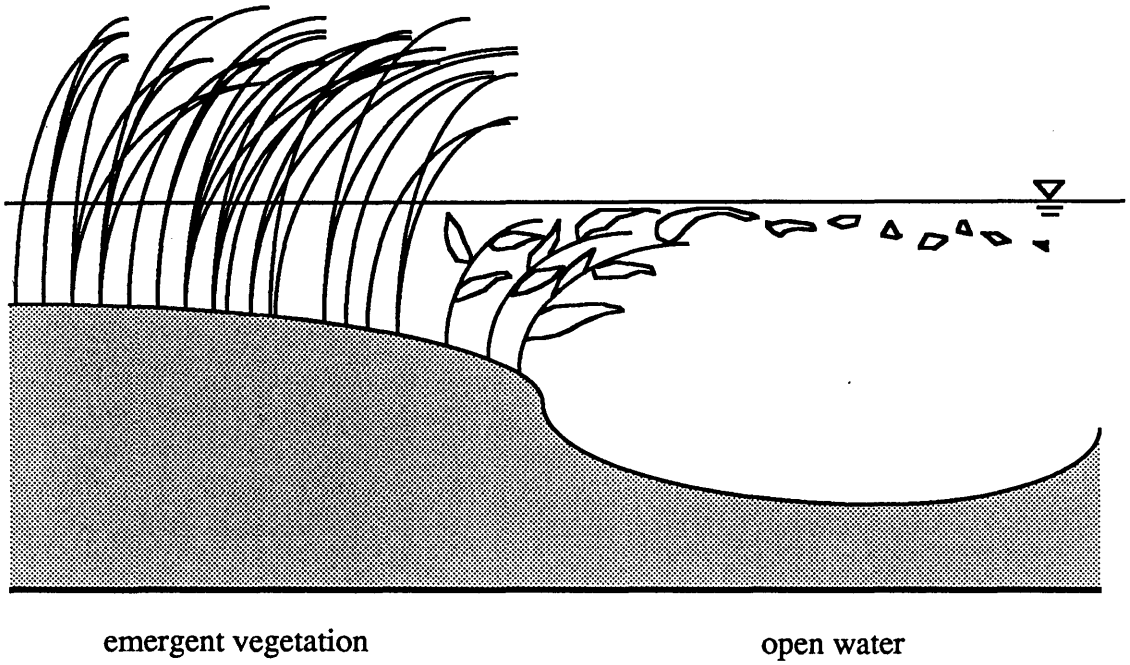

Figure 4. Schematic diagram of an artificial constructed wetland.

TREATMENT OF LANDFILL LEACHATE BY CONSTRUCTED WETLANDS

Evidence of inadvertent treatment

Evidence for removal of contaminants in landfill leachate by wetlands was obtained from a field study of surface water upstream and downstream of the Raleigh landfill in eastern Wake County, North Carolina (Douglass and Borden 1992). The stream receives drainage from the unlined landfill, then flows through a natural wetland area. Baseflow and storm events were sampled upstream and downstream of the wetland. Substantial reductions of conductivity, TAN, COD, total organic carbon (TOC), total suspended solids (TSS), iron, manganese, and zinc were observed in the downstream samples. This study provides evidence that contaminant removal occurred in the wetland. Although the results are not conclusive, this study gives some insight into the possibility of using wetland systems to treat landfill leachate in this region.

Research on treatment of landfill leachate

Studies done under more controlled conditions are needed to get a better indication of the true effectiveness of constructed wetlands for leachate treatment. A number of pilot and full scale systems have been studied in different geographical locations.

Several studies of constructed wetland treatment of landfill leachate have been conducted in the state of New York. A series of studies were conducted in Tompkins County, New York. The first part of the study was conducted in a greenhouse using 62-liter minibeds filled with sandy loam or 1 cm gravel. Half of the beds were planted with *Phragmites* and half were left unplanted. Removal of BOD, COD, nutrients, and metals was examined for a 15-day residence time. Approximate leachate concentrations were 5-day biochemical oxygen demand (BOD_5) and COD of 480 mg/L, TAN of 60 to 136 mg/L, and Fe of 48 to 57 mg/L. BOD and nutrient removals were greater in planted than in unplanted beds; removals of BOD and metals were greater in gravel than in sandy loam beds. In addition, controlled greenhouse studies were conducted in beds with a sand/gravel substrate mixture. Results from these studies led to a recommendation of using coarse substrates to reduce the chance of clogging which causes hydraulic problems (i.e., overland flow) in the system. The second part of this study used four wetland plots constructed at the Tompkins County landfill. Three of the plots were planted with *Phragmites* and contained substrates of coarse 3 cm gravel, pea gravel and a sand/gravel mixture. The fourth plot contained the sand/gravel substrate and remained unplanted. BOD and TAN removals were greatest in the warm months. TAN removal dropped dramatically during the winter, with ammonia being released during the coldest temperature periods. Fe removal was not seasonally affected. Leachate application did not appear to adversely affect plant growth, but there was not strong evidence that planted beds maintained higher removal rates than unplanted beds. Beds containing a mixture of sand and gravel provided the best treatment, but also lost the most hydraulic conductivity. Hydraulic conductivity decreased significantly during the two year study in all of the substrates except coarse gravel. Presence of plants did not prevent this decrease. Plants apparently decreased leachate volume through evapotranspiration (Sanford et al. 1990a,b; Staubitz et al. 1989; Surface et al. 1993).

Another New York study was conducted in Fenton, Broome County, New York (Reis et al. 1994; Trautmann et al. 1989). Two FWS cells were constructed and planted with reed canary grass (*Phalaris arundinacea*). In addition, two SSF cells were constructed and planted with cattail (*Typha glauca*) in a substrate of coarse gravel (1.3 cm maximum size). The FWS cells used overland flow for pretreatment to remove high levels of dissolved iron and manganese, organic matter, ammonia nitrogen, and benzene. Oxidation, precipitation and volatilization of these pollutants reduced clogging in the subsequent SSF cells. The SSF cells used the root zone method for additional removal by decomposition, filtration, and oxidation. The major

contaminants in the leachate were TAN (169 mg/L, mean), iron (31 mg/L, mean), and manganese (1.9 mg/L, mean). BOD_5 levels were relatively low (40 mg/L). Inorganic nitrogen concentrations were reduced by 60 to 100% as a result of dilution by precipitation and by loss of inorganic N, primarily within the FWS cells. Nitrogen loss by nitrification and subsequent denitrification was not analyzed specifically. It was presumed that most of the nitrogen loss was due to volatilization of NH_3, as pH was generally high (7.0 and 8.1). In addition, laboratory studies using the Fenton leachate indicated that the rate of nitrification was slow. However, the data did not rule out the possibility that nitrification was important. Plants in the FWS beds inhibited NH_3 volatilization, so a recommendation was made to eliminate plants from the overland flow beds. As for metals removal, iron was effectively removed by the treatment system, but manganese was not. Unlike the Tompkins City, New York study, these SSF beds did not experience loss of hydraulic conductivity over the one year study period.

Studies were also conducted at landfill sites in the state of Florida. Constructed wetland cells are a component in treatment of leachate collected from the Perdido solid waste facility in Escambia County, Florida (Martin and Moshiri 1994; Martin et al. 1993). The treatment system consists of a series of lagoons, sand filters, and wetland cells. The leachate filters through a sand and gravel collection system into an aerated lagoon. Septage is added at this point in the system. Water from the lagoon is then sprayed onto a compost field. Leachate from the compost is collected in lined holding ponds containing floating aquatic plants, water hyacinth (*Eichhornia crassipes*) and duckweed (*Lemna* sp.). From these ponds, water is discharged into a series of ten FWS constructed wetland cells planted with a variety of wetland plant species, *Sagittaria*, *Pontederia*, *Scirpus*, *Phragmites*, *Typha*, *Juncus*, and *Zizaniopsis*. The effluent then passes through a sand filter. The general characteristics of the untreated leachate from the lagoon, prior to septage addition, were BOD_5 of 240 mg/L, TAN of 482 mg/L, total Kjeldahl nitrogen (TKN) of 526 mg/L, total phosphorus (TP) of 7.7 mg/L, TOC of 395 mg/L, and pH of 8.1. BOD, TOC, and TAN were successfully removed ($\geq$ 95%) by the treatment system some of the time, although the data presented did not indicate which components of the system were most effective at contaminant removal.

A study was also conducted on a project in Orange County, Florida, using natural wetlands to treat diluted leachate (Keely et al. 1992). Leachate is collected from the unlined landfill by perimeter drainage and collection canals, combined with groundwater and stormwater runoff from the site in a holding pond, and applied to a portion of the Wide Cypress Swamp. This swamp is an adjacent wetland that had previously been drained. Results of the study indicate that organic pollutants in the leachate are metabolized or degraded in the wetland. Influent levels of nitrogen and phosphorus were found to be low and were not considered important contaminants in this system. Concentrations of heavy metals were low in the wetland soils and did not appear to be accumulating to a serious extent. This was not a tightly controlled study, however, and it is difficult to draw many conclusions from the information presented. Application of the dilute leachate did not appear to have a detrimental impact on the wetland plant or animal populations and may have been beneficial in restoring the natural wetland.

In Massachusetts, a greenhouse study of SSF systems for treating a high BOD leachate was conducted (Lavigne 1989). The systems consisted of peat moss planted with reed canary grass (*Phalaris arundinacea*) and were operated as both batch and continuous flow reactors. Leachate BOD_5 was approximately 21,000 mg/L and was applied at 50% strength. In the batch systems, the bottom was saturated and moisture was drawn up to a height of about 24 inches, creating an unsaturated environment. Greater than 99% removal of TOC was achieved in these systems. Continuous flow systems were operated with detention times of one to twelve days. The influent was initially toxic to the plants, but they grew back and there were no problems after that. Reed canary grass was found to be essential for maintaining hydraulic conductivity and treatment efficiency. Accumulation of manganese, potassium, iron, zinc, calcium, and magnesium in the

plants was observed over the three year period. Lead and copper, however, did not appear to accumulate. Lagoon systems were also tested in the field for treating this leachate (Lavigne 1979). The lagoons were operated as batch systems with a detention time of 90 days. At the beginning of the study the leachate was toxic to plants, but by the midpoint of the study algae and duckweed were thriving. BOD_5 was reduced from 21,000 to 10 mg/L, TAN was reduced from 437 to 3.5 mg/L, Fe was reduced from 1400 to 1.0 mg/L and pH increased from 5.2 to 7.3.

Natural marsh wetlands and constructed wetlands were tested as part of a system to treat leachate from a closed landfill in British Columbia, Canada (Birkbeck et al. 1990; Hunter et al. 1993). Leachate was first treated by six aerated lagoons. Six individual wetland units were constructed adjacent to these lagoons and filled with support media of sieved sand, 1.6 cm gravel, 2 cm stone, or dredged sand. The units were planted with either soft rush (*Juncus effusus*) or cattail (*Typha latifolia*). Vegetation in the natural marsh area included soft rush, cattail, grasses, birch trees, and several species of shrubs. Full strength leachate was added to the test wetland cells and was primarily analyzed for BOD_5 and TAN. BOD_5 of the influent to the test units was 49 mg/L, which was reduced to between 11 and 23 mg/L with detention times of 1.5 to 2.6 days. The TAN of the influent to the test units was 24 mg/L and was reduced by 8% to 35% by the test cells. TN was not reported, but nitrate levels did increase in the wetland units. Leachate entering the full scale natural marsh was fairly dilute. Average BOD_5 was reduced from 16 to 9 mg/L and ammonia-N was reduced by about 2 mg/L from influent concentrations of 6.4 to 8.0 mg/L. Heavy metals were not a significant problem and phenol concentrations were reduced by 50%. The leachate in this study is more dilute than typical leachate and the results may not be applicable to many situations.

In northern Slovenia, a pilot-scale reedbed treatment system was constructed at a municipal waste dump site (Urbanc-Bercic 1994). A gravel, SSF bed was planted with *Phragmites australis*. Average leachate characteristics included BOD_5 of 98 mg/L, COD of 849 mg/L, TSS of 173 mg/L, nitrate nitrogen of 89.4 mg/L, TKN of 35.5 mg/L, TN of 245 mg/L, traces of heavy metals, and specific organic toxic compounds. A settlement tank with oil trap was used for pretreatment of the leachate before application to the reedbed. Removal of TSS was 73%, COD was 36%, and BOD_5 was 32% in the first 16 months of operation. Vegetation had not become well established during this period. The low BOD efficiency may indicate that the leachate had an inhibitory effect on microorganisms. Toxicity tests conducted on *Pseudomonas putida* were negative.

An integrated approach to landfill leachate treatment that included the use of an anaerobic pond, an aerobic lagoon, two parallel SSF wetlands, and a FWS wetland was researched in Norway (Maehlum 1994). The SSF cells were planted with *Phragmites australis* and *Typha*. One cell had a gravel substrate and the other had light expanded clay aggregates (LECA) as the substrate. The FWS cells were planted with *Scirpus* and *Typha*. The untreated leachate was characterized as low strength with peak values in the winter of COD of 2000 mg/L, 7-day BOD less than 300 mg/L, TAN of nearly 230 mg/L, TP from 0.4 to 1.1 mg/L, and Fe less than 50 mg/L. The reported removal efficiencies of BOD, COD, TOC, N, P, and Fe from the untreated leachate were quite high for the entire treatment system (70 - 95%). Most contaminant removal occurred in the aerated lagoon, where there was a 30 day mean hydraulic retention time. However, effectiveness of the constructed wetlands could not be determined from the data given. In addition, the reported removal efficiencies are not conclusive since dilution effects by surface and groundwater were not considered.

Some other studies have been reported of systems that use constructed wetlands as components of treatment systems receiving landfill leachate, but little data have been reported. In Brookings, South Dakota, trenches and wetland ponds were used to intercept and treat groundwater that was contaminated with leachate (Dornbush 1989). Iron and manganese, along with other trace metals

and toxic organics, were effectively oxidized and precipitated. In Australia, landfill leachate was treated in a series of two holding ponds, followed by a wetland - pond - wetland treatment system (Davies et al. 1993). The wetland cells consisted of *Typha* planted on 10 cm of sandy soil over 25-30 cm of rock. The pond between the two wetland cells provided aeration. With a total detention time of three to four days, TAN concentrations of up to 50 mg/L were reduced to approximately 2 mg/L.

Wetland treatment of wastewaters and implications for leachate treatment

Wetland systems have been demonstrated to be effective for removing BOD, suspended solids, nitrogen, and sometimes metals and toxic organics from a number of different types of wastewater. Since these contaminants are also important constituents of landfill leachate, studies done with other wastewaters may be useful in analyzing the potential effectiveness for treatment of leachate.

BOD

Most BOD removal studies have been done using municipal wastewater, for either secondary or advanced treatment. BOD_5 influents have not been very high (less than 100 mg/L). Removal has been found to follow first-order plug flow kinetics (Watson 1992). The response of systems to higher BOD loadings is unknown. Leachate can have a wide range of BOD_5 concentration, from less than 10 mg/L to tens of thousands mg/L. Studies of constructed wetlands treating the higher levels of BOD have not been widely reported in the literature. Constructed wetlands were included in a natural system for treatment of septage, with COD concentrations of 25,000 mg/L and TKN of 670 mg/L (Ogden 1993). Other components of the system included an equalization tank, an aerated lagoon, and irrigated hedgerows. COD removal of 97% and TKN removal of 92% were observed with an overall detention time of 24 days. Studies of treatment of livestock wastewaters have been reported, but influent BOD levels were not high due to dilution with stormwater (Hammer et al. 1993). Current studies being conducted by the Biological and Agricultural Engineering Department at NCSU on hog wastewater will provide more information on use of constructed wetlands on wastes with relatively high BOD, COD, and TKN.

Nitrogen

The effectiveness of constructed wetlands for removal of nitrogen is particularly controversial. Nitrogen in leachate is frequently in the form of ammonia. Removal of ammonia is important for many systems because of its toxicity in the NH_3 form, which is present at high pH. Removal of nitrogen from the leachate requires either ammonia volatilization, plant uptake, or complete microbial conversion to N_2 by the processes of nitrification and denitrification. The first two mechanisms are thought to be relatively minor in wetland systems. The microbial processes of nitrification and denitrification are significant removal mechanisms in wetland systems and are enhanced by the aerobic and anaerobic environments provided by the interactions between the plants, water column, and substrate. Nitrification is apparently the limiting step in nitrogen removal in many systems, with temperature or the supply of oxygen usually the limiting factor (Choate et al. 1990; Hsieh and Coultas 1989; Watson et al. 1989). Nitrification requires temperatures greater than 6°C to be effective (Shammas 1986; Hammer and Knight 1992). Wetland systems may be able to provide warmer temperatures during cold periods by providing insulation with organic debris. This prevents rapid loss of the heat stored in the ground and that generated by microbial activity. This may be enough to enhance nitrification in the winter in relatively mild climates. Some researchers question whether enough oxygen will be transported to the sediments to achieve nitrification in the root zone method (Brix and Schierup 1990; Schierup et al. 1990). Others believe that the root zone method is effective in providing oxygen to the sediments (Armstrong et al. 1990; Conley et al. 1991; Gersberg et al. 1989). FWS systems were effective at removing TAN and TKN from livestock wastewater with average influent concentrations of 55 mg/L and 70 mg/L, respectively (Hammer et al. 1993). Nitrification has

been enhanced in several studies by providing aeration (Choate et al. 1991; Davies and Hart 1990; Willadsen et al. 1990). Others have found that the limiting factor in nitrogen removal is the organic carbon supply required for denitrification (Gersberg et al. 1983; 1984). Because of these uncertainties, it is unknown over what geographical range nitrogen removal will be effective during the winter, if nitrification will be limited in various types of constructed wetland systems due to oxygen transport, and if the plant material produced will provide a sufficient carbon source for denitrification.

Phosphorus

The removal of phosphorus from wastewaters in constructed wetland systems can be highly variable and dependent upon the substrate and type of wetland used. Mechanisms by which phosphates are removed in a constructed wetland are adsorption, plant uptake, complexation, microbial metabolism, and precipitation. Some removal is temporary since much of the phosphorus accumulated in the algae and macrophytes is released during die-back periods (Watson et al. 1989; Martin and Moshiri 1994). Since the dominant mechanisms are governed by interactions with the substrate, wetland systems that maximize these processes have higher removal rates. Mineral soils with high concentrations of Al, Fe, and Ca have a higher phosphorus retention capacity and thus increase removal (Brix 1993a; Reed 1993; Martin and Moshiri 1994; Kadlec 1987). Phosphorus concentrations are usually low in wastewaters treated by constructed wetland systems and have a range of removal efficiencies from 0 to 90% (Watson et al. 1989). Tanner et al. (1995) investigated the effect of loading rates and planting on the removal of phosphorus and nitrogen from dairy farm wastewaters. Four SSF wetlands were planted with *Schoenoplectus validus* (softstem bulrush) and four other SSF cells were left unplanted. Concentrations of phosphorus were highly variable throughout the 20 month study period, but TP values ranged from 8 to 18 mg/L. As the loading rate was decreased and retention time was increased, TP removal increased from 12 to 36% in the unplanted wetlands and from 37 to 74% in the planted wetlands. This study indicates the importance of macrophytes in phosphorus treatment of wastewaters and their contribution of organic matter and detritus that can adsorb or complex phosphorus compounds.

Metals

Metals removal was observed in some of the leachate studies described above, but was not studied on a long term basis. Constructed wetlands have been used for treating acid mine drainage, which typically contains various metal contaminants (Brodie 1993; Brodie et al. 1989, 1993; Davison 1993; Eger et al. 1993; Hedin and Nairn 1993; Henrot et al. 1989; Witthar 1993). Most of the studies reported observed treatment of water with high iron and manganese and low pH. Some also observed removal of trace metals. Acidity was removed by alkalinity addition from carbonate dissolution and by bacterial sulfate reduction. Iron removal occurred in some cases from bacterial oxidation and precipitation at higher pH. Other mechanisms observed were adsorption/cation exchange and complexation with soil organic matter. In many cases, little metal removal appeared to occur due to plant uptake or precipitation of metal sulfides. Greenhouse and field studies of metals removal from urban runoff have also been reported (Shutes et al. 1993). Copper, zinc, lead, and cadmium were measured in roots, rhizomes, and leaves of *Typha latifolia*. Most metals accumulation was observed in the roots and rhizomes rather than in the leaves. Trace metal removal mechanisms are not well understood. In general, removal mechanisms have not been well quantified and the lifetime of experimental systems is not known because of a lack of long-term data.

Toxics

Many substances in landfill leachate are potentially toxic. Toxics are not only a problem in meeting discharge requirements but potentially could adversely affect the wetland treatment process. Therefore, to be effective, wetland must be able to remove toxic substances and must also be able to thrive in the presence of these substances. Toxic substances that can be found in

leachates include ammonia, metals, petroleum hydrocarbons, household chemicals, and pesticides (Portier and Palmer 1989). High concentrations of ammonia may cause toxicity problems to a wetland and may also cause failure due to excessive loading. Metals can be toxic and could possibly cause toxicity problems for wetland processes as well. Even though no data presented gave any indication of detrimental effect on wetland plants from these substances, metals may inhibit the nitrification process. This may be a direct inhibition or may be caused by the resulting precipitation of phosphorus, leading to phosphorus limitation (Manoharan et al. 1992). Other toxic organic and inorganic substances may present more of a problem in some types of leachate. Treatment of petroleum refinery wastewater with constructed wetlands has been reported (Litchfield 1993; Litchfield and Schatz 1989) and may indicate removal of some toxic substances, including hexavalent chromium and phenols. Wetlands may actually provide a favorable environment for removal of many organic contaminants that biodegrade slowly. Many will adsorb onto organic matter, which will hold them in the system until they are biodegraded. This mechanism was observed in treatment of a textile processing and dyeing wastewater in an experimental gravel reedbed in Australia (Davies and Cottingham 1992). Some organic contaminants biodegrade best in aerobic environments, some in anaerobic environments. A wetland process provides both types of environments and therefore may remove a wide variety of organic contaminants. Some contaminants, such as many chlorinated organics, biodegrade best if they are exposed to anaerobic and aerobic environments sequentially. Wetlands can also meet this requirement, even though evidence of this process was not found in the literature.

PILOT STUDY DESIGN: NEW HANOVER COUNTY, NC

Pilot research using constructed wetlands to treat a leachate in New Hanover County, North Carolina, is currently being conducted. Plots were built with a grant from the Coastal Area Management Association to the New Hanover County Department of Environmental Management. The County is continuing to support this project.

Description of site

New Hanover County lies in the southern region of the North Carolina coastal plain at the mouth of the Cape Fear River. The population is approximately 120,000 with about 60,000 located in the county's largest municipality of Wilmington. Average annual rainfall for the county is 136 cm (53.5 in); the average daily high temperature for the summer months is about 31°C (87°F); the average daily low temperature for the winter months is 4°C (39°F).

New Hanover County began operation of the first double-lined landfill with leachate collection in North Carolina in 1981. The county also operates a 450 ton per day (TPD) waste-to-energy incineration facility adjacent to the landfill. Ash from the incinerator is disposed of in the landfill, along with municipal solid waste, sludges, and mixed construction waste. The leachate collected from the landfill is pumped to a 2.5 acre, 4 million gallon capacity lagoon for pretreatment. From the lagoon the leachate is fed to a 50,000 gallon per day (gpd) extended aeration plant for further treatment. Filter cloth is used to remove solids from the aeration plant effluent, which is then discharged to the Cape Fear River, in accordance with the requirements of the National Pollutant Discharge Elimination System (NPDES) as mandated by the State of North Carolina, Department of Environment, Health and Natural Resources (DEHNR), Division of Environmental Management (DEM).

In order to meet NPDES requirements, the effluent discharged to the Cape Fear River must have 30 mg/L or less BOD_5, 30 mg/L or less of TSS, a pH between 6-9, must pass toxicity test as specified by DEM, and must not cause the turbidity of the Cape Fear River to exceed 50 Nephelometric Turbidity Units (NTU). The facility is also requested to monitor concentrations of arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), iron (Fe), nickel (Ni), zinc (Zn), aluminum (Al), mercury (Hg), manganese (Mn), TN, and TP. Twenty-four-hour acute toxicity testing using fathead minnows is also required.

Leachate influent to the extended aeration plant from the lagoon is characterized by fairly low BOD, TSS and metals and fairly high TAN and TKN. BOD_5 concentrations are generally less than 50 mg/L but on occasion have gone as high as 260 mg/L. Most of the influent nitrogen is in the form of ammonia-N. Concentrations of ammonia-N up to 100 mg/L are common in the summer. Winter concentrations are somewhat higher, occasionally up to 200 mg/L (Figure 5). Effluent concentrations of BOD_5 are generally less than 20 mg/L, with occasional higher levels. In the summer, most of the nitrogen in the effluent is in the form of nitrate, with ammonia-N concentrations generally in the range of 1 to 5 mg/L. During periods of cold weather (November or December to March), nitrifying bacteria slow down in the extended aeration plant, resulting in low to no removal of ammonia in some years. However, the plant has consistently performed well during periods of warm weather. Testing has shown no evidence of toxicity from organics or metals in the effluent. Iron concentrations are frequently less than 0.5 mg/L, with occasionally higher levels, up to 6.0 mg/L. Other metals have consistently been present in very low concentrations. A complete characterization of the leachate influent to the extended aeration treatment plant is presented in Table 3.

There are many potential benefits of using a wetland treatment system at the New Hanover County Landfill. The immediate concern is with meeting their toxicity standard by effectively

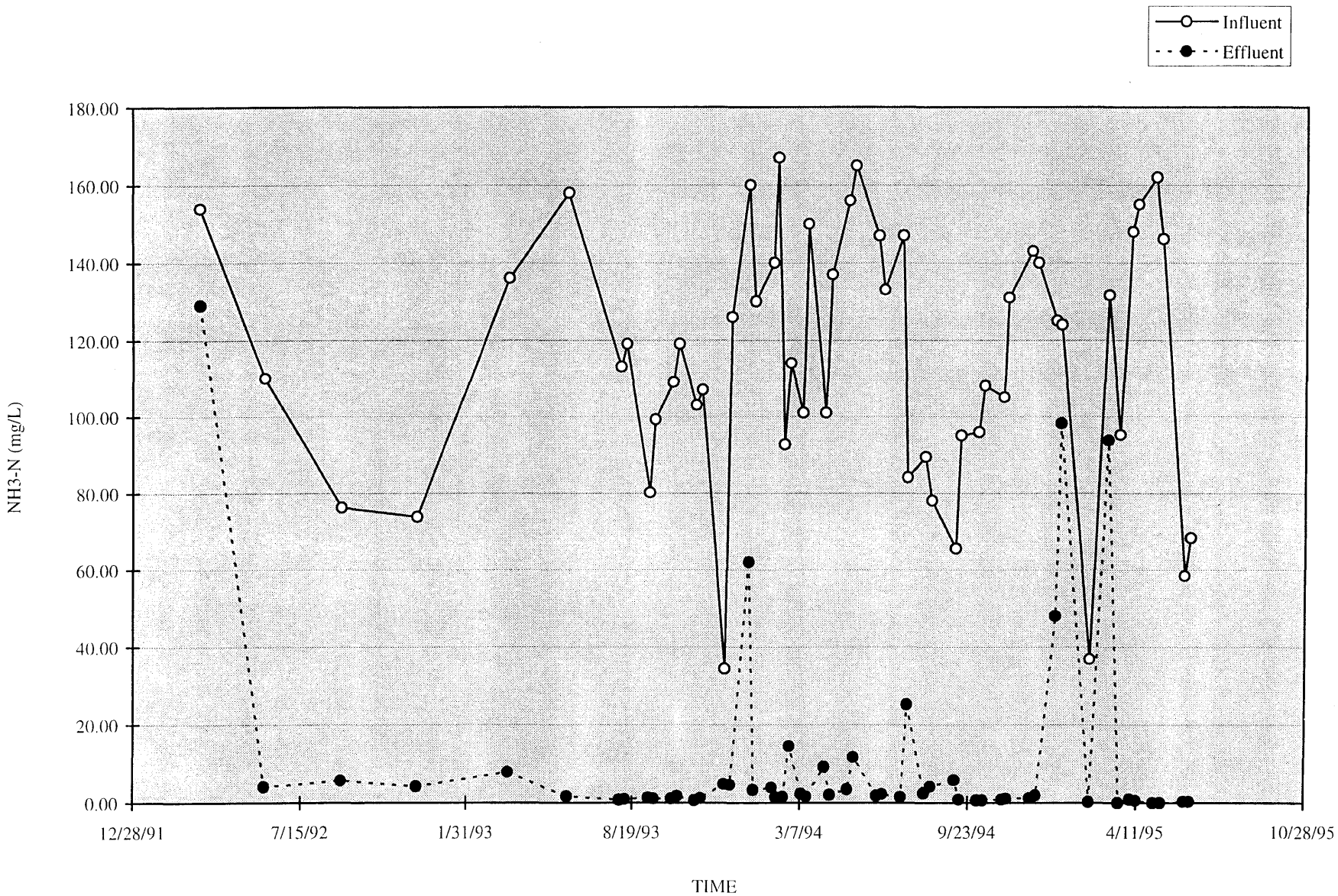

Figure 5. Concentrations of ammonia nitrogen in New Hanover County, NC, landfill leachate at the influent and effluent of on-site treatment plant.

Table 3. Leachate Composition for the New Hanover County, NC, Landfill from March 17, 1992 to August 29, 1995

Constituent	Average Concentration (mg/L)
Biochemical Oxygen Demand (BOD)	32.6
Chemical Oxygen Demand (COD)	484
Total Dissolved Solids (TDS)	3827
Total Suspended Solids (TSS)	27.2
Total Phosphorus (TP)	0.11
Phosphate	0.12
Nitrate/Nitrite Nitrogen	30.2
Ammonia Nitrogen	113
Organic Nitrogen	16.4
Total Kjeldahl Nitrogen	109
Specific Conductance (umho/cm)	6922
Cadmium*	0.002
Chromium*	0.006
Copper*	0.013
Iron*	3.30
Manganese*	0.350
Mercury*	0.000
Nickel*	0.015
Zinc*	0.030
pH	8.01

* Denotes only one data point available for constituent.

removing ammonia during cold weather. The County also wants to replace their extended aeration plant with a technology that requires less maintenance, such as a wetland system. This option is especially attractive for post-closure treatment of leachate, and would fit in well with current plans to develop a "wilderness park" at the site after closure of the landfill. A wetland system may better handle shock loadings of pollutants and may be effective at total nitrogen removal. Standards for total nitrogen are anticipated to be included in future discharge permits. The County is very responsive to public opinion and is interested in developing a wetland treatment technology that would gain public acceptance. They are also interested in the potential for using such a system as an educational tool that will help train environmental professionals and inform the general public about the potential for wetland treatment systems for applications in water quality improvement in coastal North Carolina.

Objectives

The purpose of this project is to evaluate the ability of constructed wetlands to treat a high nitrogen wastewater in a southeastern North Carolina coastal environment by treating leachate from the New Hanover County Landfill. Treatment ability will be determined in pilot scale wetland plots to meet the following objectives:

- Compare effectiveness of SSF wetlands and FWS wetlands for nitrogen removal.
- Evaluate effectiveness of nitrogen removal during winter.
- Determine effect of loading rate and detention time on treatment ability of constructed wetlands.
- Determine the ability of constructed wetlands to treat both landfill leachate in which nitrogen is in the form of ammonia and aerobically pretreated leachate in which nitrogen is predominantly in the form of nitrate.
- Quantify nitrogen removal mechanisms using mass balances to develop rational design strategies.
- Provide a demonstration project of alternative treatment technologies for educational purposes.

Wetland Cell Design

This project will use five pilot scale constructed wetland cells to test the effectiveness for treatment of this high ammonia landfill leachate. Each cell is 46 to 52 feet long and 13 to 16 feet wide. The focus will be on ammonia and total nitrogen removal. Other water quality parameters that will be studied include suspended solids, BOD, COD, organic carbon, phosphorus, metals, alkalinity, and pH. These parameters will not be considered as extensively as nitrogen.

Configuration
Three of the cells will receive untreated leachate from the holding lagoon. One of these cells will be a FWS wetland and two will be SSF, using a standard gravel (river rock) support media (Figure 6). One of the SSF wetlands will remain unplanted as a control cell to determine the importance of vegetation to the treatment function. These cells will receive high ammonia influent all year and will test the ability and limitations of such systems to remove ammonia.

The other two cells will receive treated effluent from the extended aeration plant. This effluent contains high levels of nitrate most of the year when effective nitrification takes place in the treatment plant. One of these cells will be a FWS wetland and the other will be a SSF wetland using gravel support media. This arrangement will allow analysis of the effectiveness of these two types of wetlands for denitrification during most of the year and will demonstrate effectiveness of ammonia removal during the cold season when nitrification is not effective in the treatment plant.

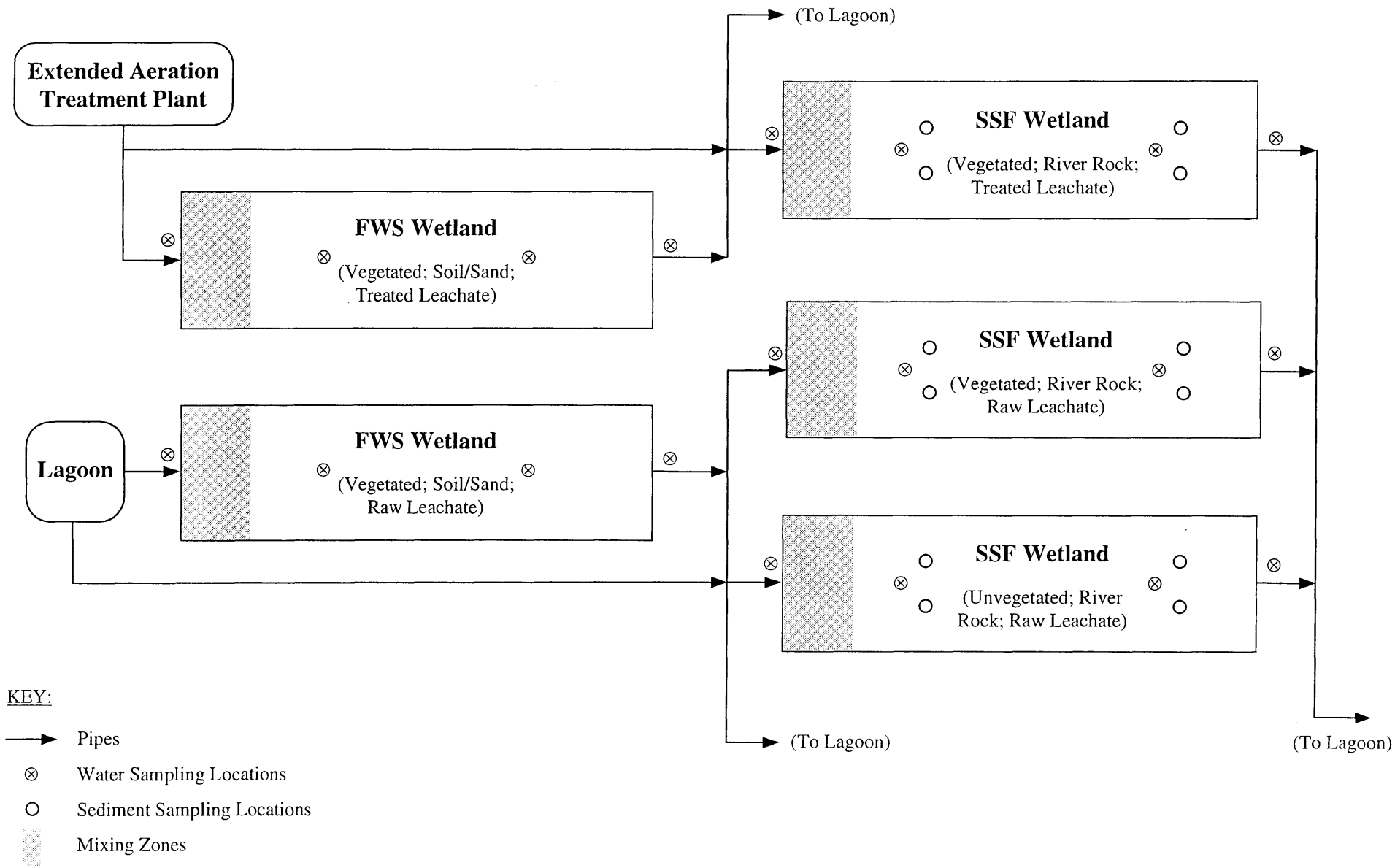

Figure 6. Schematic diagram of pilot wetland plots at the New Hanover County, NC, landfill.

Substrate
One-inch gravel (river rock) was used to minimize problems with clogging in the three SSF cells. The support media for the two FWS wetland cells is a mixture of sand and on-site topsoil.

Plants
A mixture of the following seven wetland plant species will be used in the SSF and FWS systems: *Scirpus validus* (softstem bulrush), *Juncus effusus* (soft rush), *Pontederia cordata* (pickerel weed), *Acorus calamus* (sweetflag), *Peltandra virginica* (arrow arum), *Sarurus cernuus* (lizard's tail), *Sparganium androcladum* (burreed). *Scirpus* has been shown to be effective in the root zone method of treatment in SSF wetland systems and is a locally abundant plant. Therefore, this plant will be used over approximately 30% of the surface area in each of the four planted wetland cells. *Juncus* is also common locally and better at surviving over the winter than many other wetland plants. The other plants will add diversity to the system and provide a source of readily degradable organic carbon for denitrification. As previously stated, one SSF cell will remain unplanted as a control.

Water Depth
In the SSF systems, the support media will be 60 cm deep and the water level will be maintained just below the surface of the support media. This depth was selected based on the depth of root penetration of the *Scirpus validus*. In the FWS systems, the soil substrate will be 30 cm deep, and the water depth will be maintained between 20 - 60 cm, averaging 30 cm, above the soil surface.

Detention Time and Loading Rates
Flow rates through the systems will be determined based on detention time. Detention time for each system will be 14 days. Removal efficiencies at different detention times will be analyzed by sampling at intermediate locations along the test plots. Tracer tests using bromide dye will be performed after the plants become established to determine the actual flow rates required to achieve the desired detention times.

The SSF systems have total volumes of 22 m^3. Assuming a porosity of 0.35 for the gravel, the total volume available for flow is about 8 m^3. To achieve an overall detention time of 14 days, a flow rate of about 0.57 m^3/day is required. This flow rate should result in a HLR of 1.5 cm/day and a MLR of 31 kg N/ha-day (based on a maximum influent concentration of 200 mg N/L). The HLR is within currently recommended guidelines (Hammer and Knight 1992; Platzer and Netter 1992; Watson et al. 1989); the MLR is higher than some recommendations (Hammer and Knight 1992; Swindell and Jackson 1990; WPCF 1990) and lower than others (Watson et al. 1989). Some design guidelines are given in terms of oxygen demand, assuming oxygen transfer into the system will be the limiting factor. The MLR of N corresponds to an oxygen demand loading of 142 kg O_2/ha-day. This is within some published values for maximum oxygen transfer by the root zone method (Armstrong et al, 1990; USEPA 1988; Reed et al. 1988).

The FWS systems with 30 cm water depth contain a total volume of 11.3 m^3. Assuming a porosity of 0.7 with fully established plants, the total volume available for flow is about 8 m^3, which is the same as the SSF systems. Therefore, estimates of required flow rate, HLR, and MLR are the same for both types of systems. Again, the actual flow rate required will be determined in the system after plants have become established. Loading rates for the full 14 day detention time are again similar to some of the recommended guidelines (EPA 1988; Reed et al. 1988; WPCF 1990). FWS wetlands may not be as limited by oxygen transfer as are SSF wetlands (Hammer and Knight 1992).

Operation Alternatives
The five constructed wetland plots will be operated in parallel for the initial phase of the study. This strategy will allow comparison of FWS and SSF systems for both ammonia and total nitrogen removal. The layout of the plots also allows operation of the FWS and SSF cells in series. This strategy may be advantageous for nitrogen removal because FWS wetlands are thought to be more effective at nitrification, while SSF wetlands are thought to be more effective at denitrification. After several years of operation when the plots have become well established, if nitrogen removal is not satisfactory, operation will switch to run FWS/SSF plots in series. The system design also allows for periodic draw-down. This strategy may be tried if it is obvious that the system is severely oxygen limited. However, periodic draw-down adds considerable operational complexity to the system, so it is not a preferred option.

A model of nitrogen and BOD removal in wetlands based on established mathematical descriptions of fundamental removal mechanisms has been developed (Gidley 1995). This model will be upgraded to include results of current studies of oxygen exchange rates into the sediments by wetland plants and the effect of high nutrient loads on wetland plant development and the subsequent impact on treatment function. Utilization of supplemental carbon sources for denitrification to achieve complete nitrogen removal using waste streams from other sources will also be explored.

The data collected will allow determination of effectiveness of nitrogen removal in cold weather, comparison of FWS and SSF wetlands, comparison of ammonia vs. nitrate removal, and development of preliminary design criteria. Development of design criteria will continue as data are collected over a longer period of time. Another important result of this project will be the development of the field site for public education and training of environmental professionals. The County plays an important role in the education of school children, as well as adults, and plans to use this research to educate the public in potential waste treatment alternatives that preserve the quality of the environment.

Project Status
Construction of the 5 pilot scale wetland plots at The New Hanover County Landfill near Wilmington, North Carolina was completed in the spring of 1995. Wetland vegetation was planted on May 9, 1995. Batch additions of raw and treated landfill leachate were introduced to the respective plots beginning in the fall, but full scale operation did not commence until January 1996. A modified monthly sampling schedule was followed during the first 8 months after planting. A complete sampling regime was initiated when full scale operation began in early 1996. Operation and sampling of this constructed wetland system will continue until mid 1997.

CONCLUSIONS

Constructed wetlands are receiving much attention as an alternative treatment method for many types of wastewaters. Constructed wetlands appear to have potential for removal of a number of pollutants, including BOD, TSS, N, P, metals, and toxic organics. Advantages of using wetlands treatment systems include low cost and maintenance requirements, provision of green space and wildlife habitat, and provision of opportunities for public education and recreation.

Many states have not yet developed well-defined policies for the use of constructed wetlands because necessary information is lacking. There is still considerable disagreement about the effectiveness of these systems. Design criteria are generally based on case-by-case input-output observations. Many studies of constructed wetlands have not determined removal mechanisms and thereby do not offer insight into why some systems work well and others do not.

The applicability of these systems for treatment of landfill leachate is difficult to generalize because of the extreme variability of leachate characteristics from site to site. For this reason, it will be necessary to evaluate application on a case-by-case basis. However, some general comments and concerns can be stated.

Landfill leachate has a wide variety of contaminants, many of which vary widely in concentration over time. Therefore, it is important to maintain as much flexibility and diversity in a treatment system as possible. Making use of various types of systems, such as ponds, FWS wetlands, SSF wetlands and vertical flow wetlands, provides a greater diversity of environments and therefore provides greater opportunities for multiple contaminant removal mechanisms to operate. For example, as landfills age, the leachate is likely to contain less BOD, more refractory organics, and more ammonia nitrogen. As this change occurs, physical and chemical processes become more important for organics removal, perhaps associated with biological removal that requires longer reaction time because of the refractory nature of the compounds. Nitrification and denitrification may also become more important.

Some type of pretreatment is likely to be necessary before introducing leachate to a constructed wetland treatment system. Viable possibilities for pretreatment include ponds, lagoons, and leachate recirculation. Requirements will vary with the characteristics of the particular leachate. High BOD concentrations may have to be reduced or diluted. High iron, manganese, and calcium concentrations may require pretreatment to prevent excessive precipitation in the wetland system that would cause clogging.

Design loading rates for constructed wetland systems are still not well established. More research is needed on rate limiting factors in these systems. The rate of oxygen transfer may be the limiting step in many biological removal processes and may determine the appropriate loading rates for BOD and ammonia, although anaerobic BOD removal may be an important process for some systems. Other factors could limit contaminant removal by biological processes. Organic carbon availability could limit denitrification and therefore limit total nitrogen removal by the system. Nutrient availability could also limit biological processes. In particular, phosphorus concentrations may be low in many leachates, partly as a result of precipitation with metals. This could potentially limit treatment of leachates with high nitrogen concentrations. Temperature is also a potentially limiting factor that is not well understood. The wetland environment and warm nature of leachate may actually provide some protection against cold temperatures, particularly in milder climates.

Toxicity is another factor that is not well understood. Some components of landfill leachate may have a toxic effect on wetland systems that could inhibit their treatment function. High ammonia concentrations, although detrimental to most aquatic systems, have not been shown to have a toxic effect on wetland vegetation. Toxic organics and metals, however, may inhibit biological

processes in these systems. More long-term data are necessary to determine the potential for toxic substances to accumulate in the wetland system and present future hazards to humans or wildlife. The data available from relatively short-term studies indicate that such a problem is not likely. Also, wetland systems have good potential for removal of these types of contaminants because of the diversity of environments and removal mechanisms available. Therefore, presence of some toxic contaminants in the leachate should not, in itself, discourage continued study of these systems.

Long-term problems, in general, are also not well understood. Potential metals accumulation in wetland vegetation is only one example of a long-term problem that deserves attention. Hydrological problems, such as clogging of SSF systems and filling of FWS systems, have not been adequately studied on a long-term basis. Current studies should be able to provide data in the future. Management of wetland plant communities may also be necessary to maintain these systems on a long-term basis.

The economics of constructed wetland systems have not been thoroughly investigated. Although there are many benefits in construction, operation, and maintenance costs, there are also expenses. Systems that treat landfill leachate will likely require liners. Land requirements will also be a major expense. Siting of these systems should be planned along with the initial design of the landfill site.

Constructed wetlands have the potential to be part of an effective on-site treatment strategy for landfill leachate. The additional benefits for recreation, aesthetics, and wildlife make these systems worthy of consideration.

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