REPORT NO. 277

# TREATMENT OF DOMESTIC WASTEWATER BY A CONSTRUCTED UPLAND-WETLAND WASTEWATER TREATMENT SYSTEM

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September 1993

Water Resources Research Institute
of The University of North Carolina

UNC-WRRI-93-277

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The research on which this report is based was financed in part by the U.S. Department of the Interior, Geological Survey, through the North Carolina Water Resources Research Institute.



Agreement No. 14-08-0001-G 1580; 14-08-0001-G 2037

WRRI Project No. 70101

USGS Project No. 09 (FY90); 02(FY91)

ACKNOWLEDGMENTS

We would like to acknowledge the assistance of Mr. Jim Baluss and Ms. Vicki Deal of the Pamlico County Health Department; Mr. Fred May, Pamlico County Extension Director, and Mr. Philip Prescott for cooperation in site selection and technical assistance. Mr. Steve Steinbeck, Mr. Steven Berkowitz and Dr. Robert Uebler of the North Carolina Department of Environmental Health and Dr. A. R. Rubin of North Carolina State University, Biological and Agricultural Engineering were very helpful in design advice and technical assistance. Dr. Ernest Seneca, Botany Department Head, and Dr. Larry D. King of the Soils Science Department, North Carolina State University gave expert advice for project evaluation and presentation of the results. We would like to give special thanks to Mr. Carlton L. Campbell of the Soil Science Department and Mr. Larry L. Hobbs of the Botany Department, both of North Carolina State University, for painstaking construction, maintenance and monitoring of the research system. Members of the Water Lab within the Soil Science Department; Ms. Angela Barnes, Ms. Beverly M. Taylor, and Dr. W. P. Robarge were outstanding in response to our research needs and prompt with analysis of research samples.

ABSTRACT

An Upland-Wetland wastewater treatment system was installed in Pamlico County, North Carolina, during September 1989 to test its effectiveness in treatment of wastewater from a single family home. Treatment effectiveness is indicated by representative exit drain samples taken from March 1990 to September 1991 showing the following average concentrations (mg/L): total nitrogen (TN), 13.2; ammonium nitrogen ($NH_4$-N), <0.1; nitrate nitrogen ($NO_3$-N), 11.7; phosphate phosphorus ($PO_4$-P), 0.5; total phosphorus (TP), 0.5; chemical oxygen demand (COD), 34.0 and suspended solids (SS), 8.6. Pathogenic bacteria and viruses as indicated by fecal coliform colonies/100 ml of sample were taken from March 1990 to December 1991. Fecal colonies were lowered from an average of $10^5$-$10^6$ in the influent to 387 from the cell planted with cattail (Typha angustifolia); 1267 from the unplanted cell; and 71 from the cell planted with common reed (Phragmites australis). The Upland-Wetland Wastewater Treatment System has provided low cost, low maintenance and effective wastewater treatment.

**Key Words** (wastewater treatment, constructed wetlands, mound systems, multi-staged wastewater treatment, nitrogen, phosphorus, fecal coliform, chemical oxygen demand, suspended solids, Phragmites australis, Typha angustifolia)

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## SUMMARY AND CONCLUSIONS

An Upland-Wetland wastewater treatment system was installed in Pamlico County, North Carolina, during September 1989 to test its effectiveness in improving quality of wastewater from a septic tank serving a single family home. The upland is a 9 m x 3 m x .75 m (30 ft x 10 ft x 2.5 ft) sand mound capped with 20 cm (8 in) of silt loam topsoil and planted with fescue (Festuca arundinacea). The entire system is underlain by an impervious 10-mil PVC liner that slopes from the mound towards three 3 m x 3 m x .6 m (10 ft x 10 ft x 2 ft) gravel filled wetland cells containing either Typha angustifolia, Phragmites australis, or no vegetation. A wetland edge planted with ink berry (Ilex glabra) and wax-myrtle (Myrica cerifera) provides a transition between the mound and the wetland at its base. Septic tank effluent is dosed into the mound through a pressure distribution network at 40 L/day/$m^2$ (1 gal/day/$ft^2$) loading rate. The efficacy of each system component was evaluated from water samples collected once every two weeks between March 1990 and March 1991 and monthly until September 1991 from the wetland edge and after passing through the wetland cells.

The effects of the system components on water quality are summarized in the table below, which presents mean concentrations during 18 months from March 1990 to September 1991 for all measures except fecal coliform. They include means from March 1990 to December 1991.

Table 1. Summary of Wastewater Treatment by the Upland-Wetland System.

| Analysis | Influent | Wetland Edge | *Typha* | Unplanted | *Phragmites* |
|---|---|---|---|---|---|
| Fecal Coliform colonies/ 100 ml | $3.2 \times 10^5$ | 235 | 387 | 1267 | 71 |
| Total Kjeldahl Nitrogen mg/L | 44.0 | 1.1 | 1.1 | 1.1 | 1.1 |
| Total Nitrogen mg/L | 44.4 | 16.0 | 14.0 | 13.4 | 11.1 |
| Ammonium Nitrogen mg/L | 35.4 | <1 | <1 | <1 | <1 |
| Nitrate Nitrogen mg/L | 0.4 | 14.9 | 12.9 | 12.3 | 10.0 |
| Organic Nitrogen mg/L | 8.6 | 1.0 | 1.0 | 1.0 | 1.0 |
| Total Phosphorus mg/L | 4.4 | .6 | .5 | .6 | .3 |
| Phosphate Phosphorus mg/L | 3.0 | .6 | .5 | .6 | .1 |
| Chemical Oxygen Demand mg/L | 338.0 | 43.9 | 32.2 | 32.1 | 37.7 |
| Suspended Solids mg/L | 95.1 | 11.2 | 9.5 | 9.4 | 6.8 |

The mound provided an aerobic environment that resulted in complete nitrification, reduction of phosphorus and effective filtration to remove organic material. Concentration of TN was reduced 64% by the mound. Nitrogen in wastewater dosed into the mound was in the $NH_4$-N and organic forms, while essentially all the nitrogen present in water that had passed through the mound was in the nitrate form. The mound reduced TP 86%, $PO_4$-P 80%, COD 87% and SS 88%. Fecal coliform colonies were reduced from $10^4$-$10^6$ to $10^1$-$10^3$ colonies per 100 ml.

Conversion of nitrogen to the nitrate form is an advantage of the mound component compared to wetland-only systems in which nitrification is limited by anaerobic

conditions. Another advantage of the mound is reduction of organic load entering the wetland cells. This enhances treatment in the wetland cells by reducing clogging in the substrate and reducing coating of wetland plant roots. Coating of wetland plant roots with organic materials may reduce oxygen transfer from the plant to the substrate. The transfer is desirable for treatment of nitrogen, phosphorus and organic compounds. The root coatings may also reduce the contact between microbial pathogens and root exudates which have been reported to enhance pathogen inactivation.

Treatment by the mound was enhanced by pressure dosing at a rate of 1 gal/$ft^2$/day and using 10 minute cycles at 8 hour intervals to allow soil drying. This resulted in an aerobic environment for nitrification, phosphorus sorption, bacteria attenuation and removal of organic material. A coarse sand material that contained carbonates also improved treatment of all wastewater constituents.

The wetland cells provided additional hydraulic capacity to increase retention time and contributed to reduction in concentrations of nitrate, phosphorus and fecal coliform. The wetland cell planted with Phragmites australis was more effective than both the unplanted cell and the cell planted with Typha angustifolia. Phragmites australis established easily and grew more rapidly. It was more tolerant of variations in water level than Typha angustifolia. Concentrations of $NO_3$-N nitrogen were reduced by 31% of that entering the cell; total phosphorus was reduced 50% and fecal coliform by 1-2 orders of magnitude.

During a 43 day period in which mass balance was measured, the wetland cell planted with Phragmites australis removed substantial quantities of nitrogen as did the mound component. Effluent from the Phragmites australis cell had nitrate nitrogen concentrations below 10 mg/L during 14 of 29 sampling events. Denitrification and plant uptake were probably the mechanisms of nitrogen removal. Carbon may have been a limiting factor in denitrification, and denitrification may increase as below ground biomass increases to provide organic matter as an energy source for denitrifying organisms.

This research has demonstrated the advantages of combining the aerobic environment of a mound with an anaerobic wetland system to improve wastewater treatment. Monitoring of the system should continue to determine if denitrification will improve, if the sand fill of the mound will become saturated with phosphorus and if the wetland plants contribute significantly to the organic load as they mature.

## RECOMMENDATIONS

1. Research is needed to develop methods for enhancing denitrification. Adding carbon sources, particularly before the plants develop, would provide energy for denitrifying bacteria. Potential sources are on-site wastewater, organic waste from agriculture and industry and growing wetland plant species that decompose rapidly.

2. A finer substrate material than gravel should be used in the wetland cells. Gravel was used in this design to prevent physical and biological clogging and therefore a decrease in treatment potential. Since the mound which precedes the wetland cells filters the wastewater and decreases the organic load, clogging is less likely to be a problem in the wetland substrate. Finer material in the wetland cells will also provide a better environment for plant growth.

3. A number of wetland plant species should be investigated to determine which provides the best treatment and to develop propagation and management techniques.

4. Retention time that maximizes wastewater treatment in the wetland cells, the size and depth of cell to best accomplish this and the wetland plants best suited to the resulting configuration should be determined.

5. A more detailed hydraulic monitoring of the system is needed (amount of water that flows into the system, the amount that flows out and the effects of rainfall) to determine mass balance of nutrients.

The Upland-Wetland design approach is effective at reducing all wastewater constituents measured. Wetland only systems are useful for removal of organic constituents such as COD and SS, and for denitrification when preceded by an aerobic component such as a mound, sand filter, mechanical aerobic treatment system or package treatment plant.

The most effective use of the Upland-Wetland design approach to meet regulatory and economic requirements will require either the use of inexpensive subsurface disposal techniques such as trickle irrigation to dispose of the treated water from the Upland-Wetland or increasing the system size and using it in a cluster system for several homes. The cluster approach would be compatible with disposal of the treated water by spray irrigation or discharge into surface waters.

# INTRODUCTION

As development continues in the coastal zone of North Carolina, the need for effective treatment of wastewater becomes more critical. Because of the high cost of municipal wastewater treatment plants and the rural nature of most of the area, on-site treatment is usually the only alternative. This poses a dilemma since much of the land within the coastal zone is unsuitable for effective wastewater treatment using conventional septic tanks and associated absorption fields. Soil limitations are perched or seasonal high water tables, slowly permeable clays or coarse textured sands which offer very little treatment potential (Carlile et al. 1981). Incomplete treatment of wastewater leads to pollution of groundwater, streams and estuaries. Mound systems and constructed wetland systems are technologies that may be of use in these limiting conditions.

A mound system is an above-ground soil filter for treating domestic wastewater. Effluent is distributed within the mound by a pressure dosed pipe system (Cogger et al. 1982). This type of system has been used at sites where soil conditions, such as a high water table and slowly permeable clays, hinder the performance of soil absorption systems. A properly constructed mound provides treatment of effluent by filtration and aeration. The soil or sand also removes phosphorus from the wastewater. A disadvantage is that if the system is overloaded, effluent may be discharged around the edge of the mound on the original soil surface.

Constructed wetlands have been used to treat industrial, municipal, agricultural and on-site wastewater since the mid 1970's. A typical constructed wetland designed to treat wastewater consists of a rectangular cell filled with suitable soil or gravel. Wetland plants are grown within the substrate to enhance the waste treatment processes. The ability of wetland plants to transport air into the primarily anaerobic environment of the wetland cells is thought to be critical to treatment processes. Plant uptake of nutrients, microbial influences and physical and chemical processes are important in the treatment process.

Evaluation of treatment performance of current on-site waste treatment technologies, including constructed wetlands, indicates inadequate treatment of wastewater is likely if used with the soil and water table limitations which frequently occur in the Coastal Plain of North Carolina. However, the integration of a mound system and a constructed wetland system may provide an effective low cost wastewater treatment design.

Objectives

The overall objective was to determine the feasibility and effectiveness of a combination mound and a horizontal flow subsurface constructed wetland system to improve the quality of wastewater flowing from a conventional septic tank. Specific objectives were:

(1). To construct a combination mound and wetland domestic wastewater treatment system in Pamlico County, North Carolina, on a site that is unsuitable for conventional septic tank systems;

(2). To measure the treatment effect of each component of the experimental system on water quality; and

(3). To compare the treatment effects of Phragmites australis, Typha angustifolia and no wetland vegetation.

## LITERATURE REVIEW

### On-Site Wastewater Treatment Systems

Human waste disposal systems have been in existence for several thousand years. Schwiesow (1974) summarized the development of home sewage disposal systems. Toilets were built into castle walls in Ireland in 800 with slots that directed sewage into the streets, the moat or the surrounding tree tops. There were no major changes in waste treatment and disposal technology from the 9th through the 13th centuries.

In 1594 Sir John Harington invented a water closet which was installed in his home. Queen Elizabeth I had a water closet installed in Richmond Palace, and during the early 1700's Queen Anne had one installed in Windsor Castle. William of Orange and Mary II had the first reported modern bathroom with tub, water closet and hot and cold water installed in Chatsworth House during the same period.

"Houses of easement" or privies were a later innovation. Early cities and rural homes in what is now the United States had these facilities, and many are still in use today.

The first modern water supply system in the New World was developed during the 16th century in Antiqua, Guatemala. Stone conduits supplied water under pressure to every dwelling, and the wastewater carried off garbage and sewage. Pressure water supply systems made possible the added convenience of up-to-date waste disposal systems, but wastes from both city and rural sites were discharged into rivers, gullies or ditches. Health and aesthetic concerns prompted the use of treatment systems such as cesspools, septic tanks and Imhoff tanks and later the use of seepage beds, absorption trenches and sand filter trenches. These designs, though better than those in the early castles, needed improvement. Failures due to the variation of soils and hydrology caused contamination of groundwater.

Dosing siphons and tipping bucket aerators were used during the early 20th century to give better distribution of wastewater within these trenches. These were an improvement but as suitable land became less available a more sophisticated waste disposal systems had to be designed.

It has been estimated that only 32% of the total land area in the United States has soil suitable for on-site systems which utilize the soil for final treatment and disposal. In 1980 there were 18 million housing units, or 25% of all housing units, using on-site systems. The number was increasing by 0.5 million each year at the time of data collection (U. S. Environmental Protection Agency 1980).

The potential health hazards brought about by inadequately treated sewage have continually prompted public health officials to seek methods to improve design of treatment systems and purification of water. Initially, public acceptance of on-site systems was poor because they were considered second rate, failure prone or temporary until city sewer could be extended to the area (U. S. Environmental Protection Agency 1980).

Currently several types of systems are used for on-site treatment and disposal: sand filters, aerobic treatment units, fixed film systems and subsurface soil systems. Constructed wetland systems are in the research and development phase (U. S. Environmental Protection Agency 1980).

Sand filters are usually designed to treat wastewater from installations larger than a single family dwelling. The general design includes a septic tank for primary treatment, a filter bed and a disinfecting unit. Wastewater flows from the house into the septic tank from which it flows by gravity or is pumped to a sand filter. The wastewater may pass through the filter once, be recirculated, flow into the filter by gravity or be intermittently dosed by a effluent pump. It is then disinfected, and discharged into a water body or disposed of by a subsurface absorption field. Sand filters are installed either under the ground or on the surface.

Aerobic treatment units are mechanical systems that use a pump to aerate wastewater within a chamber creating an environment to enhance microbial growth. The increased microbiological populations attached to the solids are recycled through the system to enhance the treatment of fresh incoming wastewater.

Fixed film systems use inert media to which microorganisms may become attached to treat wastewater. The water may move through a packed tower of the media by gravity, be pumped up through the media, or flow across a rotating disk or biological contactor.

Subsurface soil absorption systems are trenches, beds or seepage pits. Trenches are shallow, level excavations, usually .3-1.5 m (l-5 ft) deep and .3-.9 m (l-3 ft) wide. A single line of perforated pipe is laid within a gravel envelope within the trench. Beds are very similar to trenches except they are wider than .9 m (3 ft) and may contain more than one line of distribution pipe. Seepage pits are deep excavations used for the subsurface disposal of pretreated wastewater. A covered porous-walled chamber is placed in the excavation and surrounded by gravel or crushed rock.

Mound Systems

A mound system is an aboveground soil filter for treating domestic wastewater. Effluent is distributed within the mound by a pressure dosed pipe system. This type of system has been used at sites where soil conditions, such as a high water table and slowly permeable clays, hinder the performance of soil absorption systems. A properly constructed mound provides treatment of effluent by filtration and aeration. The soil or sand also removes phosphorus from the wastewater. A disadvantage is that if the system is overloaded, effluent may be discharged around the edge of the mound on the original soil surface (Cogger et al. 1982).

The first research toward the development of the mound system began in North Dakota in 1947. The NODAK waste disposal system was developed to treat and dispose of wastewater on sites with heavy clay subsoils. It was thought that by putting the disposal area on top of the ground, percolation rates of sewage would improve and water could be used by grass, shrubs and trees and removed by evaporation. Dimensions of the system typically were 6.1-9.1 m (20-30 ft) wide by 9.1-12.2 m (30-40 ft) long and approximately 0.3 m (1 ft) deep. The filter material consisted of a 0.5 m (1.5 ft) layer of pea gravel placed on and within a 0.3 m (1 ft) layer of coarse sand. A submersible pump was used to supply wastewater to a 0.2 m (0.5 ft) perforated plastic pipe which ran through the center of the filter bed. The system was an improvement over conventional gravity systems but had maintenance problems associated with pumps, electrical controls and plumbing (Witz 1975).

The NODAK design was modified and adapted for use on other soils and under other climatic conditions. Design modifications included improved effluent pumps and electrical controls, a two septic tank system consisting of a pumping tank used in series after a standard septic tank, a wastewater distribution network that spreads the wastewater out over a larger area; intermittent dosing of wastewater into the mound, and installation of the soil mound on the scarified natural soil surface rather than 0.6-0.9 m (2-3 ft) below (Converse et al. 1977).

Specific design criteria have been developed for systems in permeable soils with high water tables, slowly permeable soils with and without seasonally high water tables, and systems in permeable soils with shallow creviced bedrock (Converse et al. 1975 a,b,c). The absorption area configuration, the fill depth, and the slope of the side of the mound are varied for the three different designs (Converse 1978).

Mound systems have been used for cluster type developments (groupings of homes sharing a waste treatment system) as well as single family dwellings (Hantzsche and Fishman 1982). This allows the most acceptable land within a subdivision to be used for wastewater treatment and the remaining land to be used for houses, roads and other construction that is not as limited by soil or water table conditions.

Wastewater characteristics, loading rate, type and interval of wastewater distribution, texture and chemical composition of the soil material, and the microbial composition and their influence on soil crusting, clogging and nitrogen transformation are all important factors in wastewater treatment by mound systems.

The Wisconsin At-Grade absorption system is a recent modification of mound system design. It is designed to be used where soil and site conditions prohibit the use of any subsurface installation but are not as limiting as those that require a sand mound system. A pressure distribution network is installed within a mound of gravel rather than a layer of gravel placed over a layer of sand. The natural soil surface is tilled or plowed before the gravel is placed over it. The gravel containing the pressure distribution network is covered with a synthetic fabric which is in turn covered with a soil cap (Converse et al. 1987).

Studies utilizing columns containing different soil materials have been used to evaluate the potential for the treatment of wastewater by properly designed and constructed mound systems.

A column study conducted by Magdoff et al. (1974 a,b) evaluated simulated environments of a cross section through a mound and a natural soil (silt loam). Two fill materials (sandy loam and sand) were tested. The sand was dosed every 6 hours and remained uncrusted. The sandy loam was dosed once every 24 hours and became crusted. The crusted soil exhibited reduced treatment of nitrogen, phosphorus and COD. Willman (1979) investigated the influence of chemistry and texture of soil material. In his column study, varying amounts of clay (0, 3, 6, and 12%) were added to three types of sand-sized material (limestone, sandstone and shale), and the mixtures were compared to two natural sands (sandstone and glacial outwash). He concluded that sand type is more important than percentage of clay. Limestone sand was found to be the most desirable. Sims (1981) examined the influence of loading rate (0.75 and 1.5 gal/ft$^2$/day) and soil type on the treatment of wastewater. The soil materials used were: builder's sand, Creedmoor loamy sand, Louisburg loamy sand, Suncook sand, or Wagram fine loamy sand. The higher loading rate was found to create a greater potential for system failure due to excessive clogging. Wastewater persisted on the soil surface of the Suncook, Creedmoor and Wagram soils most frequently.

In a field study conducted in Wisconsin, four mound systems were evaluated to determine the influences of design and construction techniques on their treatment performance. It was concluded that mound systems can perform satisfactorily in soils that would be considered unsuitable for the conventional subsurface disposal of septic tank effluent. Special attention should be given to the following construction techniques: the natural soil surface that will form the base of the mound should be plowed to destroy the interface between the mound fill and the soil, construction should be conducted when the weather is dry to prevent compacting and puddling, and the plowed surface should not be driven on by heavy equipment (Bouma et al. 1975).

Thirty-three field systems which had been designed and installed according to official state guidelines were monitored over a two-year period in another study conducted in Wisconsin to determine if mound systems were properly designed and installed and were performing adequately. Dosing volumes to many of the mounds were found to be much higher than recommended though other design and installation criteria were generally met. It was concluded that properly designed and installed mounds treat wastewater better than conventional septic systems in slowly permeable soils, shallow soils over permeable bedrock, and soils with high water tables (Harkin et al. 1979).

Two mounds in the North Carolina Coastal Plain were monitored from the summer of 1979 until the summer of 1980 (Carlile et al. 1981). One of the systems was designed to treat wastewater from a night club with an estimated water use of 1500 L/day (400 gal/day). Extreme water conservation techniques were used (microphore 1.9 L (2 qt) toilets). The mound was built 0.6 m (2 ft) above the natural soil surface and contained 610 m (2000 ft) of distribution line. Wastewater distribution problems developed due to insufficient pumping pressure head.

Another mound system used to treat wastewater from a marina was monitored within the same study. The system was designed to treat >1500 L/day (> 400 gal/day) from toilet and hand washing facilities. The property owner added a camping area and a restaurant, which contributed to the hydraulic overload that occurred when surface water entered the improperly sealed pumping chamber. The distribution lines became plugged with grease which created wastewater distribution problems. The information from these two systems documented the influence of improper operation and maintenance on the performance of mound systems.

Wastewater Treatment by Natural Wetlands

Much of the initial work concerning assimilation of wastewater by wetlands in the United States was conducted in natural systems. Natural wetlands are effective in removing biochemical oxygen demand (BOD) from wastewater; the plants have adsorption and filtration potential, and sediments provide ion exchange and adsorption capacity (U. S. Environmental Protection Agency 1988). However, there are serious constraints to using natural wetlands as components of wastewater treatment systems since wetlands are valuable natural ecosystems. Almost all are "waters of the United States" that require a permit from federal and state agencies to receive any discharge.

Over 250 communities in the North Carolina coastal plain discharge treated wastewater into swamps. However, studies of the effects of sewage wastes on water quality and vegetation have indicated that discharge of wastewater to swamps should be a last resort, not a first choice, and that advantages and disadvantages should be weighed carefully (Kuenzler 1987).

Constructed wetlands provide the positive effects and benefits of natural wetlands (physical entrapment of pollutants, transformation of chemical substances by microorganisms, and low energy requirements) without the environmental concerns and user conflicts associated with discharging effluent into natural ecosystems (U. S. Environmental Protection Agency 1988).

Constructed Wetland Wastewater Treatment Systems

Some of the first work utilizing artificial wetlands for the treatment of wastewater began at Max Planck Institute during 1953. Researchers there tried to alleviate problems of over fertilization, pollution and siltation using wetland vegetation. Their research began with a survey of plants to determine characteristics, and which species possess those characteristics, that are desirable for wastewater treatment. It was found that plant species most desirable for wastewater treatment should have a large rooting zone, grow rapidly, transpire large volumes of water, and have adventitious roots. Phragmites australis was effective for this purpose and was particularly effective in the treatment of sludge (Seidel 1976).

Phragmites australis was used to treat brewery effluents and nuclear laboratory and municipal sludges. Dramatic improvement in effluent quality resulted in all cases and the sludge was reduced to nearly one thousandth of its initial volume. The levels of Salmonella spp. and Escherichia coli were also lowered during the process. As a result of the research at the Max Planck Institute, a wastewater treatment system utilizing aquatic plants bedded in gravel was developed. It is frequently referred to as the Max Planck Institute Process (MPIP). This technology has been used throughout Europe (Seidel 1976).

The MPIP System has been used in the Netherlands for the treatment of wastewater at campgrounds and for new housing developments (deJong 1976; Kok 1974). More recently it has been used to provide tertiary treatment for a municipality with 70,000 people (Greiner and deJong 1982). In Germany this type of system is used to pretreat municipal drinking water (Czerwenda and Seidel 1976).

Dr. Reinhold Kickuth at the University of Hessen in Germany has also developed a wetland wastewater treatment system that utilizes Phragmites australis. This system termed Root Zone Method (RZM), does not rely on the ability of marsh vegetation to assimilate nutrients. Instead, the soil system with its inherent treatment potential, is supplemented by the ability of marsh plants to transport oxygen through the stems and roots to the soil, creating oxidized zones within the low oxygen environment thus providing a suitable environment for nitrification and denitrification. Plant growth also produces carbon, an energy source for the bacteria that are responsible for the nitrogen transformations. This system has been used to treat screened and degritted sewage in Germany since 1974 (Kickuth 1984).

In the United Kingdom, the Water Research Centre conducted a large scale evaluation, research and development project to determine the feasibility of use of the MPIP and RZM systems. There are currently over 500 of these systems in operation in Western Europe. They were referred to by the Water Research Centre as Reed Bed Treatment Systems (RBTS) (Cooper 1990). Countries currently conducting research include Australia, Austria, Belgium, Canada, China, Denmark, France, Germany, Holland, Italy, Luxembourg, New Zealand, South Africa, Sweden, United Kingdom, and the United States (Cooper and Boon 1987). This list is dynamic and not complete due to the wide spread interest in evaluation of the technology and many new programs.

In the United States constructed wetland systems have been used by a number of municipalities, and their value for wastewater treatment has been documented (U. S. Environmental Protection Agency 1988). An example of a successful treatment system for a small town is the marsh/pond/meadow system (MPM). The MPM design evolved from the initial research conducted at Brookhaven National Laboratory (Woodwell 1977; Small 1977). Comparisons were made between different types of aquatic and terrestrial systems' ability to renovate sewage. The wetland configurations tested included a marsh-pond and a meadow-marsh-pond. Typha spp. were used in the marsh components and reed canary grass (Phalaris arundinacea) in the meadow components. Duckweed (Lemna spp.) was found to be a natural companion to the Typha spp. Scirpus lacustris was thought to be a better choice than Typha spp. because of information gained from German and Dutch reports. It was concluded that marshes and ponds in conjunction with terrestrial systems that can be spray irrigated can provide cost effective wastewater treatment (Woodwell 1977). Refinements of the design were used at a small retirement center in Pennsylvania and later in Iselin, Pennsylvania, for the treatment of 12,000 gpd from 62 homes (SMC-Martin, Environmental Consultants 1980; Conway and Murtha 1989).

Treatment at the Iselin facility begins with the removal of large solids from the wastewater with a bar screen. The remaining solids are cut up into a more uniform size by a comminutor. The primarily treated water then flows into a aeration cell with a retention time of 2.9 days. The first wetland component, a lateral flow marsh consisting of a sand substrate planted with Typha spp., then receives the wastewater. A facultative pond is planted with Lemna spp., arrow head (Sagittaria spp.), spatter-dock (Nuphar spp.) and elodea (Anacharis spp.) to remove nutrients from the water. It is also stocked with fish which include carp (Cyprinus carpio), catfish (Ictalurus nebulosis), and sunfish (Lepomis spp.) to feed on the plants. The partially treated water then flows across a meadow planted with Phalaris arundinacea. The chlorinated effluent is discharged into a nearby creek (Conway and Murtha 1989). Sampling over a two year period indicated the system reduced BOD in wastewater by 97%; SS, 89%; fecal coliform, 99.9%; $NH_4$-N, 77%; and TP, 82% (Watson et al. 1987). In 1989 seven similar or modified systems were in the planning, design, and/or construction stages in Pennsylvania (Conway and Murtha 1989).

Research conducted with a similar design at Pembroke, Kentucky, by the Tennessee Valley Authority (TVA) indicates consistent compliance with National Pollution Discharge Elimination System (NPDES) permits for typical secondary and advanced level permit limits for BOD, SS and pH. Consistent compliance for $NH_4$-N, fecal coliform, dissolved oxygen (DO) and phosphorus may require supplemental treatment methods (Choate et al. 1990).

The Pembroke system is designed to treat .09 mgd for a population of 1100. The existing treatment system had a contact stabilization package plant. Parallel cells were designed the same except the substrates and plants were varied. Gravel limestone substrate was used to encourage subsurface flow and natural clay was used to test the effectiveness of surface flow in both the marshes and meadows. The plants used were Typha spp., Scirpus spp. and Phragmites australis in the marsh cells. Lemna spp. were used in the ponds. There was an attempt to establish Phalaris arundinacea in the meadow, but natural vegetation including fall panicum (Panicum dichotomiflorum) and smartweed (Polygonum spp.) became dominant (Choate et al. 1990).

A constructed wetlands demonstration project was installed at Phillips High School, Bear Creek, Alabama, by the TVA. The existing extended aeration package treatment plant had experienced frequent overloads and therefore exceeded the NPDES permit frequently. The constructed wetlands were designed as subsurface washed river gravel cells planted with Typha spp. and Scirpus spp.

In addition to the research by the TVA, many other research and private concerns in North America are currently evaluating constructed wetlands. In Arcata, California, a previously constructed wildlife sanctuary was used to polish the wastewater, and a part of the existing aeration ponds were converted into an intermediate marsh system (Gearheart et al. 1989).

Artificial Typha spp. marshes were constructed in Listowel, Canada, during July of 1980 by the Ontario Ministry of the Environment. The marshes consist of five systems: two channelized serpentine systems, a shallow marsh, and a combination of a shallow marsh, deep pond and serpentine marsh. The marsh cells were lined with clay and filled with a mixture of peat and topsoil to a depth of 15 cm (6 in). The cattails consisted of mainly Typha latifolia with some Typha angustifolia.

**Constructed Wetlands for On-Site Wastewater Treatment:** Only limited research has been conducted with constructed wetlands for on-site wastewater treatment. National Aeronautics and Space Administration (NASA) developed a system at the National Space Technology Laboratory in Mississippi. This research was conducted as a part of the development of Closed Environmental Life Support Systems (CELSS) for space travel.

The system designed for use on the Earth consists of a conventional septic tank that discharges into a 1.2 m wide x 30.5 m long x 0.6 m deep (4 ft x 100 ft x 2 ft) trench lined with plastic and filled with gravel to support growth of plants. Effluent that passes through the trench goes into an absorption field. The scientific basis for the system is the combined influence of the plants and associated microorganisms. Degradation of organic materials is attributed to microorganisms associated with the plant root system. Marsh plants have the ability to translocate oxygen to the roots and create an aerobic zone around the roots allowing aerobic decomposition and nitrification to occur. Plants used in the system in Mississippi were southern bulrush (Scirpus californicus), canna lily (Canna flaccida) and calla lily (Zantedeschia pseudacoras). The system, referred to as the Septic Tank-Rock-Plant Marsh Treatment System, was able to provide treatment of effluent equivalent to advanced secondary levels (Wolverton 1988).

The International Conference on Constructed Wetlands for Wastewater Treatment, sponsored by the TVA was held in Chattanooga, Tennessee, during June 1988. Research and experience reported at the conference demonstrated the practical application of constructed wetland systems to the problem of wastewater treatment. Some of the issues that were thought to need further study are the mechanisms of treatment, the role of various wetland components, refinement of designs, loading rates, retention time and the most effective plant species (Hammer 1989). Researchers with the TVA refer to all the previous designs (RBTS, MPIP, and RZM) as constructed wetlands treatment systems (CWTS). They are currently demonstrating variations of these designs, which are designated either surface flow or subsurface flow CWTS (Steiner and Freeman 1989).

The International Conference on the Use of Constructed Wetlands in Water Pollution Control was held in Cambridge, United Kingdom, during September 1990. Fifty papers and more than 20 poster papers were presented (Cooper and Findlater 1990). The role of the aquatic plant as described in the RZM as proposed by Dr. Kickuth was an issue of primary debate. The lack of sufficient nitrification and removal of phosphorus by current wetland designs were identified as areas for improved design and further research. The RBTS in Europe are designated as either horizontal or vertical flow systems. The vertical flow systems are now receiving renewed attention. The vertical flow systems are variations of the MPIP system developed by K. Seidel and have not received much acclaim in recent years. During this time, variations of Dr. Kickuth's RZM, which are horizontal flow systems, received the most research attention (Cooper 1990).

Vertical flow systems operate as follows: wastewater is applied over the surface of the wetland cells, it flows vertically through the medium, is collected on the bottom in a drain, and discharged to another treatment cell, a subsurface absorption field, or a body of water. Horizontal flow systems are designed so wastewater is distributed into one end and flows horizontally through the medium to the exit drain on the opposite end (Cooper 1990).

Other recent conferences include international conferences held in Pensacola, Florida, during October, 1991, and in Sydney, Australia, during November-December 1992. Both conferences included evaluation of many previous topics but were unique in increased discussion of constructed wetlands in multistage or integrated treatment systems.

**Wastewater Treatment by Aquatic Plant Systems:** Treatment of wastewater with aquatic plants is typically practiced in areas that do not have severe winters. Most of the floating and submerged aquatic species are sensitive to low temperatures. Species used most frequently include water hyacinth (Eichhornia crassipes) and duckweed (Lemna spp., Spirodela spp., and Wolffia spp.) (Reed et al. 1987). Pennywort (Hydrocotyle umbellata), which is not technically a floating aquatic plant, has potential for use in systems in cooler climates (U.S. Environmental Protection Agency 1988). Uptake rates of nitrogen and phosphorus by Hydrocotyle umbellata remain constant regardless of season, and in winter months its uptake is actually greater than that of water hyacinth (DeBusk and Reddy 1987). The treatment of wastewater by floating aquatic species is dependent on the environment which occurs between the plant roots and the water or the root-water interface.

The root-water interface processes are not well understood, but diffusion of oxygen and organic substances from the root into the surrounding water is likely and would be conducive to the nitrification/denitrification of wastewater (Good and Patrick 1987). There is evidence that an anaerobic environment may form around the roots of some floating aquatics, which would encourage denitrification. This has been observed in Eichhornia crassipes (Reddy 1981). A treatment system using Eichhornia crassipes has been reported to remove 40-92% of the influent nitrogen apparently through denitrification (DeBusk et al. 1983). Nitrogen removal by floating aquatics was greater than removal by emergent and submerged aquatic macrophytes (Reddy 1983). Lemna spp. and Hydrocotyle umbellata have potential as plants which may be important in waste management in North Carolina.

### Treatment of Microbial Pathogens

The four main groups of disease causing organisms found in domestic wastewater are enteric bacteria, viruses, protozoans and helminths (Sobsey 1986). These groups include several hundred different pathogens potentially present in human sewage (Feachem et al. 1983).

The importance of ground water contaminated with human waste in the transmission of viral and bacterial diseases by drinking water is well documented. Contamination due to overflow or seepage of sewage, primarily from septic tanks and cesspools, was responsible for 43% of the outbreaks and 63% of the illness caused by the use of contaminated groundwater in the United States between 1971-1979 (Craun et al. 1984). The Housing Assistance Council conducted a survey in 1982 which represented 22 million rural households, that indicated two thirds of all rural households had domestic water judged unacceptable for at least one pollutant. Microbial contamination as indicated by fecal coliform bacteria was the most commonly observed. Failing on-site wastewater disposal systems were thought to be the most likely source (Housing Assistance Council 1984). In 1984, outbreaks of disease caused by contaminated drinking water were at the highest level reported in the United States since 1941 (Craun et al. 1984).

Only limited studies of pathogen reduction by on-site wastewater treatment systems, especially innovative and alternative systems have been conducted (Hagedorn 1984), and little is known about the ability of wetlands to remove microbial pathogens from wastewater (Gersberg et al. 1987). Furthermore, most studies have been conducted using indicator fecal coliform bacteria data that do not accurately assess other bacterial and viral activity (Sobsey 1986).

The ability of mound systems to remove fecal coliform bacteria from wastewater was evaluated in a study in Wisconsin. Eight systems were monitored with variation in sample periods from 2 to 8 times during 2 years. Fecal coliform counts in samples taken from soil at the toe of each mound were typically $<10$ colonies/100 ml of sample. However one system had counts as high as $4.7 \times 10^4$ colonies/100 ml at different times of the day and different locations along the toe of the mound (Converse and Tyler 1985).

Much of the information concerning the ability of constructed wetlands to remove pathogenic microorganisms has been collected during studies of municipal systems or systems larger than those designed to treat wastewater from single family dwellings. Constructed wetlands were more effective at reducing pathogens than conventional treatment processes, which typically exhibit erratic reductions of 1 log (Berg 1983).

The retention time of the wastewater within the wetland and the distance the water travels within the wetland are critical to the removal of pathogens. The results of a study conducted in a natural cypress wetland suggest that initial removal is rapid but is followed by a slower decline. Some of the microbial population is able to undergo long-term survival and may accumulate within the sediment (Scheuerman et al. 1989).

Excretions from the roots of aquatic plants may destroy pathogenic bacteria. During a laboratory study, water containing either Escherichia coli, Enterococci spp., or Salmonella spp. was placed in separate containers with 1 of 7 aquatic plant species. Mortality of the microbial organisms varied within the containers to suggest that some factor associated with the plant roots adversely affected pathogenic microbial survival. The author suggested that root excretions of Mentha aquatica, sweetflag (Acorus calamas), rush (Juncus effusus), Phragmites australis and the root bulbs of Alnus glutinosa could either partially or completely kill disease bacteria in contaminated water (Seidel 1976).

Non-pathogenic microorganisms which occur in constructed wetlands, either mobile or attached to substrate or plant roots, may have an antagonistic affect on pathogenic microorganisms. Actinomycetes in soil were found to be capable of suppressing the growth of Salmonella spp. and dysentery bacilli (Bryanskaya 1966).

## Treatment of Nitrogen

Nitrogen occurs in different forms according to its physical, chemical and biological environments. The major forms of nitrogen of concern for on-site wastewater treatment are ammonium ($NH_4$-N); nitrite, ($NO_2$-N); nitrate ($NO_3$-N); nitrous oxide, ($N_2O$) and dinitrogen gas, ($N_2$). Transformations between these forms are primarily biochemical but they may occur chemically. The transformations within the nitrogen cycle include; mineralization, immobilization, nitrification, denitrification, volatilization and fixation.

**Treatment by On-Site Septic Systems:** The removal of nitrogen from wastewater requires that environments conducive to nitrification and denitrification must be induced and positioned to provide the proper relationships for treatment. The environments may be controlled either temporally or spatially. This requires either design criteria which provide for a periodic change between aerobic and hypoxic conditions within the same space, design criteria which provide an aerobic environment with a hypoxic environment placed immediately after it, or a mosaic of aerobic and hypoxic conditions in close proximity to each other. Adequate quantity and quality of carbon and alkalinity must also be provided for the bacteria which mediate the nitrogen transformations.

Most of the nitrogen in septic tank effluent is in the form of ammonium with lesser amounts of organic nitrogen. Most soil based absorption fields are designed to provide an aerobic environment to convert the ammonium to nitrate and mineralize some of the organic carbon (Lance 1972). Nitrate is water soluble, unaffected by soil cation exchange capacity (CEC) and therefore may leach into nearby ground and surface waters (Preul and Schroepfer 1968; Walker et al. 1973). A waste treatment design goal by many researchers and/or engineers with respect to nitrogen, is to remove nitrate by creating environments suitable for denitrification. The "RUCK" system and the Recirculating Sand Filter Rock Tank system are two approaches currently used to attempt to solve this treatment problem (Laak et al. 1981; Sandy et al. 1987; Gold et al. 1989). Constructed Wetland Treatment Systems when combined with other designs which nitrify well, such as mound systems, have potential to provide for near complete removal of nitrogen.

**Treatment by Mound Systems:** The treatment of nitrogen by mound systems depends on the maintenance of mostly aerobic conditions within the soil material of the mound to enhance nitrification. If the soil underneath and at the base of the mound contains a suitable carbon source and hypoxic conditions occur, then there may be denitrification. Extractable glucose carbon is a useful index of the quantity of carbon sources associated with the loss of nitrate during anaerobic incubation (Stanford et al. 1975). The relationship of carbon to denitrification is 1 ppm of available carbon requirement for the production of 1.17 ppm of nitrogen as $N_2O$ or .99 ppm as $N_2$ (Burford and Bremmer 1975).

Nitrification of wastewater ammonium by properly designed, installed and maintained mound systems is well documented with both field and laboratory column studies. Lack of nitrification is usually associated with hydraulic overloads of part or the entire system.

In a field study conducted in Wisconsin, four mound systems received wastewater containing 33-42 mg/L $NH_4$-N and .5-5 mg/L $NO_3$-N. Complete nitrification occurred in one system with 0 mg/L $NH_4$-N and 17 mg/L $NO_3$-N concentrations in water samples from the toe of the mound. Nitrogen concentration in water taken at similar locations in the other mounds ranged from .4-2.7 mg/L $NH_4$-N and 1.5-2.3 mg/L $NO_3$-N (Bouma et al. 1975).

Thirty-three field systems which had been designed and installed according to official state guidelines were monitored over a 2-year period in another study conducted in Wisconsin. Average influent levels of nitrogen were found to be 82.4 mg/L, of which most was $NH_4$-N. Most of the systems nitrified the wastewater nearly completely. Only one of the systems was found to exhibit anaerobic conditions long enough to deter nitrification. The concentration of $NO_3$-N was found to increase with depth within the aerobic fill of the mound. Nitrate however decreased to a depth of 5 cm; 2 in into the natural soil surface beneath the mound. This was probably due to denitrification. Concentrations of $NO_3$-N in water at the fill-natural soil interface beneath the mounds averaged 50-60 mg/L. Denitrification continued measurably to a depth of 35 cm (14 in) but probably continued to occur to 55 cm (22 in). Groundwater measurements near the mounds did not indicate significant $NO_3$-N enrichment however. Denitrification was responsible for the removal of 44% of the wastewater nitrogen. Mounds with a low dosing rate and a low fill uniformity coefficient had a 86% removal rate (Harkin et al. 1979).

Two mounds were monitored in the Coastal Plain of North Carolina for one year from the summer of 1979 until the summer of 1980. The sequence of events leading to the failure of both systems and the corresponding change in nitrogen concentrations and transformations were documented. One system nitrified the wastewater well until the system owner increased wastewater production by adding a restaurant and a camping area to the existing marina. Ammonium and $NO_3$-N concentrations in water samples taken from wells in the mound were reversed and the system failed. The other mound was improperly dosed due to insufficient pump capacity. This resulted in wastewater distribution problems within the mound. The resulting over loaded areas contained water with high ammonium concentrations and the areas receiving much less wastewater contained nitrate (Carlile et al. 1981).

**Treatment by Constructed Wetland Systems:** Treatment of nitrogen by CWTS is probably most effective if the influent nitrogen is $NO_3$-N rather than $NH_4$-N.

Nitrification-denitrification within the root zone, and sediment water interface in natural systems is well documented but the influence on waste treatment processes within constructed systems is not well understood.

The treatment of wastewater nitrogen by constructed wetlands is influenced by the chemistry of the plant root-water-sediment environment, plant uptake, available carbon, and the type of substrate. The amount of treatment of nitrogen by constructed wetlands is also dependent on the amount of time the wastewater remains within the system. A detention time of 5 to 7 days is usually sufficient to produce a discharge containing <10 mg/L of TKN (Bavor et al. 1987).

The development of oxygenated zones around aquatic plant roots within anoxic substrates and the development of a thin oxygenated layer in anoxic substrates next to oxygenated water induce transformations of nitrogen in hydric environments. The interrelationship between plant and substrate is the basis for the RZM. The development of oxygenated zones between aquatic roots and the water have been studied less than root-sediment and water-sediment interfaces but have important implications in the treatment of wastewater (Good and Patrick 1987).

The aquatic plant root-sediment interface is characterized by an aerobic zone around the plant root where nitrification occurs and adjacent reduced sediments where denitrification occurs. The aerobic zone is maintained by the movement of atmospheric gases through aerenchyma, specialized tissues of the plants which have a low resistance to gas flow, through the plant root walls and into the sediment. The oxidation of the rhizosphere prevents the toxic effects of reduced Fe, Mn and $H_2S$ (Armstrong 1972; Gambrell and Patrick 1986; Ponnamperuma 1965). The mosaic of aerobic and hypoxic conditions created by root-sediment interfaces may be an important factor in nitrification/denitrification in wastewater.

The performance of RBTS in Europe is based on information compiled by the European Community/European Water Pollution Control Association. Total-N removal is typically 20-30%. Research results indicate that nitrification does not generally occur in secondary treatment systems, but if $NO_3$-N is in the influent it can be removed. Nitrification does not generally occur in RBTS probably because of oxygen limitations (Cooper 1990).

The TVA is evaluating constructed wetlands for municipal wastewater treatment with three systems in western Kentucky (Benton, Hardin and Pembroke). Pembroke is a MPM system (Choate et al 1990).

Evaluation of the performance of the Benton system indicates that nitrification was limited by available DO, and there was a reduction in organic nitrogen probably due to ammonification. Results from the system at Hardin indicate little decrease in $NH_4$-N, probably due to the lack of nitrification. The DO levels were typically less than 1 mg/L. Removal of organic nitrogen was relatively high.

The influent of the MPM system had an average TN concentration of 31 mg/L and 6.3 mg/L in the effluent. Most of the removal occurred in the marshes with little difference due to plant species treatments. The pond added TN, and the meadows further removed small quantities. A similar trend was observed for organic nitrogen. Ammonium concentrations in water leaving the system varied seasonally with concentrations during June, September and October exceeding NPDES limits of 4 mg/L. There was a corresponding low DO concentration in the effluent which may indicate a limit in nitrification due to low oxygen.

Constructed wetlands were installed and planted with Typha spp. in Listowel, Canada, during July of 1980 by the Ontario Ministry of the Environment. Treatment efficiencies were lowest in the winter, with maximum average $NH_4$-N concentrations of 13 mg/L and 16 mg/L in effluent from channelized marshes with a serpentine configuration. The $NH_4$-N effluent in one of these often exceeded influent levels. Low water temperatures were thought to contribute to decrease in treatment during the winter. Decrease in treatment during the summer was thought to be associated with increased temperatures and their influence in decreasing dissolved oxygen and therefore nitrification (Herskowitz et al. 1987).

Nitrogen removals of 57-94% have been reported from another municipal system at Santee, California. Denitrification was enhanced by the use of biomass produced within the wetland (Gersberg et al. 1983; 1986).

**Treatment by On-Site Constructed Wetland Systems:** There is little data available evaluating the performance of small on-site constructed wetland systems. There are currently numerous active research projects however. Data from one of the earliest designs in the United States, a Rock-Plant Filter, evaluates treatment of wastewater from a single family home. Influent $NH_4$-N concentration of 28 mg/L decreased to 24 mg/L about half the length of a 21.3 m (70 ft) trench. The $NH_4$-N concentration was 7 mg/L at the discharge end (Wolverton et al. 1984).

## Treatment of Phosphorus

Phosphorus levels are especially important in freshwater systems where phosphorus may be the limiting nutrient for algae and aquatic plant growth and contributes to eutrophication (Dillon and Rigler 1974; Schindler 1974; Wetzel 1975; Medine and Anderson 1983). A critical value of only 0.01 mg/L of inorganic phosphorus contributes to algae blooms during spring overturn (Sawyer 1947).

The phosphorus cycle does not contain an atmospheric outlet as occurs in the nitrogen cycle. On-site wastewater treatment of phosphorus is the result of primarily soil or substrate processes. Natural soil systems have the potential to remove phosphorus from wastewater if they contain adequate cations such as iron, aluminum, calcium and magnesium to fix or precipitate phosphorus. Cations, such as iron, bind phosphorus within an aerobic environment, but when the soil environment becomes reduced due to sustained flooding, the phosphorus becomes mobile. Cations of aluminum are not influenced by reducing conditions and therefore may be important in phosphorus fixation during saturated conditions. Calcium and magnesium also form precipitates with phosphorus but the influence of flooding on calcium and magnesium phosphates is not well understood.

Some soils, especially those of the lower Coastal Plain that are sandy with low organic matter may be low in cations and frequently flooded; therefore, phosphorus removal is limited to sorption reactions and plant uptake. Suitable precipitates and or fixing materials can be mixed into soil materials used for the construction of sand filters, mounds and wetland systems to provide for suitable phosphorus adsorption. In situ soil materials should be used when possible however. Phosphorus is not as mobile as

$NO_3$-N but may become mobile within environments saturated with water, therefore an unsaturated soil environment should be maintained.

**Treatment of Wastewater Phosphorus by On-Site Septic Systems:** The removal of phosphorus by on site septic systems is dependent largely on soil reactions and to a lesser extent plant uptake. Phosphates may be converted into organic phosphorus as plant and microbial tissues which may provide for slower release of $PO_4$-P into the environment.

Septic tank effluent typically contains 18-29 mg/L of TP and 6-24 mg/L of $PO_4$-P. The major sources of phosphorus in domestic wastewater are laundry, dish washing, and toilet wastewaters. The contributions from detergents can be reduced significantly by the use of low $PO_4$-P or $PO_4$-P free detergents. This approach is mandated by law in North Carolina and some other areas (U. S. Environmental Protection Agency 1980).

In situ soil may retain significant quantities of phosphorus. The depth of penetration of phosphorus in sandy soils has been estimated to be 50 cm (20 in)/yr and 10 cm (4 in)/yr in finer textured soils (Sikora and Corey 1976). Sandy loam soils have been estimated to retain 100 to 300 mg of phosphorus per gram of soil (Walker et al. 1973). The potential contamination of water is much higher in sands than soils that contain some percentage of clay.

**Treatment of Wastewater Phosphorus by Mound Systems:** The removal of phosphorus from wastewater by mound systems is dependent on the nature of the fill material from which the mound is constructed, the maintenance of an aerobic soil environment, plant uptake, and the electro-mechanical aspects of design as they influence dosing regime.

Monitoring results of 33 field mound systems in Wisconsin indicated that significant concentrations of phosphorus were not removed by the fill material. The lack of removal of phosphorus was attributed to high dosing rates which caused the wastewater to move too quickly through the sand fill for the reactions of $PO_4$-P within the sand to occur. In addition, biological slimes may have occluded the surfaces of the sands and prevented sorption reactions (Sikora and Corey 1976).

**Treatment of Wastewater Phosphorus by Natural Wetland Systems:**
Removal of phosphorus from a wetland system is accomplished by harvesting plants which contain phosphorus or by the natural flushing action of the water (Prentki et al. 1978). Some forms of phosphorus may become insoluble and be stored within the wetland sediments creating a potential source of phosphorus.

Phosphorus reactions within the sediments and their interactions with the overlying waters are controlled by pH, redox potential and available cations. Wetland plants are but temporary storage areas for phosphorus which results in seasonal exports following plant death (Richardson et al. 1978). Very little uptake of phosphorus from the water column by emergent species is likely because the soil is the primary source of nutrients (Sculthorpe 1967).

**Treatment of Wastewater Phosphorus by Constructed Wetland Systems:**
The chemical, physical and biological processes within constructed wetlands should be similar to that of natural systems. However the reactions of phosphorus with iron, aluminum and calcium; the formation of organic phosphates; and the uptake by

aquatic plants and microorganisms can be manipulated through design variables of a constructed system. Though phosphorus adsorption can be maximized for a particular constructed wetland, aerobic terrestrial environments should be used if feasible. To maximize wetland removal of phosphorus, more actively growing woody species and persistent emergent species may be beneficial. The nitrogen to phosphorus ratio of the wastewater should be manipulated to encourage more plant growth and therefore more phosphorus uptake (Ulrick and Burton 1985).

Reed Bed Treatment Systems are used in Western Europe for the treatment of domestic wastewater. Typical RBTS do not remove significant phosphorus and are therefore not recommended unless the plant substrate has a high aluminum or preferably iron content. The aerobic root zone found around the roots of the wetland plants may provide the necessary environment for the formation of the ferric form of iron which is most effective for phosphorus fixation (Cooper 1990).

Research in the United States has been conducted primarily with large municipal systems. Phosphorus removal in municipal wetland systems is not typically very effective. It is thought that subsurface bed design which contains soils with significant clay content and contain iron and aluminum will enhance phosphorus removal (Reed et al. 1987).

Research by the TVA with municipal systems at Benton and Hardin, Kentucky indicates that phosphorus effluent concentration change seasonally; improved treatment occurs with subsurface flow versus surface flow, and that physio-chemical processes within the gravel substrate may also be important (Choate et al. 1990).

Treatment of Chemical Oxygen Demand and Suspended Solids

Chemical oxygen demand is a measure of the organic matter in a water sample that can be oxidized by a strong chemical oxidant. This is a useful indicator of the effect wastewater may have on levels of DO present in natural systems that receive the organic load. Decomposition processes of organic materials utilize oxygen, decreasing the amount available to aquatic flora and fauna. Transformations of nutrients by microorganisms are also influenced by the amount and type of organic matter present (American Public Health Association 1989).

Solids are material residue which is both suspended and dissolved in water. Suspended solids are a measure of that part that can be filtered out. The increase in solids influences water quality by the reduction in palatability, reduction in aesthetic qualities for such activities as bathing, reduction in usage for industrial applications and increased potential for gastrointestinal irritations in the consumer. Suspended solids may also be an indication of the organic matter present, but COD is a better indicator (American Public Health Association 1989).

Residential wastewater averages 360 mg/L of BOD and 396 mg/L SS. Biochemical oxygen demand measures the biochemical degradation of organic material, the nitrogenous oxygen demand and inorganic oxygen demand as created by sulfides and ferrous iron. Biochemical oxygen demand may be correlated with COD. Chemical oxygen demand measures tend to be higher because chemical oxidation influences more compounds than biological oxidation. To standardize the relationship between COD and BOD, samples must be analyzed for both measures from the same source and compared. Most of the contribution of organic materials come from the garbage disposal and toilet. Laundry and bathing waste contribute the rest. Most of this is soluble however (U. S. Environmental Protection Agency 1980; Bennett and Linstedt 1975).

**Treatment of COD and SS by Mound Systems:** Removal of COD and SS by mound systems is dependent primarily on physical processes of filtration and sedimentation. Adsorption and biological oxidation are also important. These processes are controlled by type and size of filter material and the loading rates of the wastewater. The soil material must remain well drained to provide adequate oxygen supply for biological processes, but must have a fine enough texture to insure contact with particulate materials. Studies evaluating the removal of organic materials from wastewater by mound systems are lacking in the literature. It does appear that the even distribution of wastewater within the mound and the associated unsaturated flow are desirable for treatment of COD.

A field study conducted in Wisconsin monitored four mound systems which were gravity fed by seepage trenches over 60 cm (24 in) of sand fill. Septic tank effluent into the systems ranged from 217-400 mg/L COD. Chemical oxygen demand in liquid sampled at the toe of the mound ranged from 57-166 mg/L. The mound that lowered COD the most also exhibited near complete nitrification of influent $NH_4$-N (Bouma et al. 1975).

**Treatment of COD and SS by Wetland Systems:** Constructed wetlands do not provide as favorable an environment for nitrification as upland environments. They do provide oxidized zones around the plant roots which should enhance organic degradation by oxidation and microbial degradation. Wetlands, both natural and constructed, have the potential to add to the organic load of water as measured by SS and COD. Some relatively non-impacted natural wetland systems contain organic soils or lush herbaceous vegetation and therefore may contribute concentrations of organic materials in excess of regulatory discharge criteria to surface waters. Regulatory requirements will need to be adjusted to quantify the type of organic materials, whether natural or contributed by waste disposal systems. Chemical oxygen demand, BOD and SS are currently measured to monitor the treatment efficiency of constructed wetlands. Most of the constructed systems are relatively new when compared to natural systems and do not typically contribute organic materials beyond discharge criteria. As the systems mature, it is likely that the organic matter accretion within the systems will begin to contribute significantly to the discharge of organic materials. Indications of this can be seen in the older RBTS in Europe.

The RBTS in Europe use BOD loading as a major design parameter. The horizontal flow systems are designed for 5 $m^2$/population equivalent to provide treatment of settled domestic sewage to an average of 30 mg/L BOD (Cooper 1990). Average yearly percentage reductions of BOD for 43 RBTS in the United Kingdom were 76% for systems containing soil substrate and 71% for systems containing coarse substrate. Suspended solids were reduced 59% and 77% by the 2 types of systems. A 20:30 standard for BOD:SS was the research target. Few systems achieved the BOD standard and less than one half achieved the SS standards. Variability in performance was high (Findlater et al. 1990).

Evaluation of the performance of municipal systems demonstrated by the TVA at Benton and Hardin, Kentucky, indicates effective removal of BOD and SS. The BOD of the influent averaged 26 mg/L with a maximum value of 45 mg/L. Effluent concentrations ranged from 1-25 mg/L with an average of 10 mg/L. Most of the BOD was removed within the first quarter of each cell. Sedimentation and filtration were thought to be the primary removal mechanisms. Influent concentration of SS varied from 9-120 mg/L with a mean of 57 mg/L. Mean effluent concentration was10 mg/L. Most of the SS were removed within the first quarter to half of each cell (Choate et al. 1990).

The system at Hardin, Kentucky, also effectively removes BOD and SS. Influent levels of BOD ranged from 11-180 mg/L with a mean of 54 mg/L. System effluent ranged from 1-38 mg/L. Removal efficiencies seem to be dependent on hydraulic loading rate. The lowest removals occurred at the highest loading rates of 8-4 acres/mgd. The highest removals occurred with loading rates of 33-10 acres/mgd. The system also effectively removed SS. Influent concentration ranged from 30-470 mg/L with a mean of 125 mg/L. System effluent concentrations ranged from 1-95 mg/L with a mean of 15 mg/L. Removal of SS did not appear to be related to hydraulic loading rate (Choate et al. 1990).

A MPM system demonstrated by TVA at Pembroke, Kentucky, has also been effective at treating BOD and SS. Performance data since October, 1987 indicates that the average influent concentration of 67 mg/L (22-130 mg/L) was reduced to 9 mg/L (2-24 mg/L). The marshes removed most of the BOD, while the pond and meadow had very little influence. Average influent concentration of SS was 92 mg/L (42-170 mg/L). This was reduced to a mean of 8 mg/L with a high value of 17 mg/L. Most SS were removed by the marshes, with most of that removed within the first half of the marsh cell. The pond contributed SS probably due to Lemna spp. and algae production (Choate et al. 1990).

**Treatment of COD and SS by On-Site Constructed Wetland Systems:**
Wolverton (1989) reported on a Rock-Plant Filter system designed to treat wastewater from a single family home that received a BOD load of 100 mg/L. Influent BOD concentration decreased to 32 mg/L about half the length of a 21.3 m (70 ft) trench.

# MATERIALS AND METHODS

The Upland-Wetland installation studied in this project is located in Pamlico County, N. C., on a study site obtained by Mr. James Baluss, Health Director of Pamlico County and Ms. Vicki Deal, County Sanitarian. The research system is connected to an existing septic tank that serves a family of four. The soil at this site (Leaf silt loam) is not suitable to obtain a permit for septic tank installation under present regulations. The research system was installed during September 1989 to test the upland-wetland design concept (House and Broome 1990) (Fig. 1-2). The upland is a 9 m x 3 m x 0.75 m (30ft x 10 ft x 2.5 ft) sand mound capped with 20 cm (8 in) of silt loam topsoil and planted with Festuca arundinacea. The sand textured soil material of the upland contains 91.3% sand, 6.2% silt and 2.5% clay. Chemical analysis of the upland sand by the Agronomic Division, N. C. Department of Agriculture indicates that it has a cation exchange capacity of 29.5 (meq) milliequivalents/100 $cm^3$. This is a measure of the soil's ability to hold cations such as hydrogen, aluminum, calcium, magnesium and potassium. Magnesium satisfies 4.4 milliequivalents (meq)/100 $cm^3$ of the charge and calcium, 25.0 meq/100$cm^3$. These base cations are probably contributed by marl and shell fragments. The high base status probably contributes to the slightly basic pH of 7.5. The entire system is underlaid by an impervious 20-mil PVC liner that slopes from the mound towards three 3 m x 3 m x 0.6 m (10 ft x 10 ft x 2 ft) gravel filled wetland cells containing either Typha angustifolia, Phragmites australis, or no vegetation. A wetland edge planted with Ilex glabra, and Myrica cerifera provides a transition between the mound and the wetland at its base. Septic tank effluent is dosed into the mound through a pressure distribution network at a 40 Lpd/$m^2$ (1 gpd/$ft^2$) loading rate (Table 2). The efficacy of each system component was evaluated from wastewater samples collected every two weeks between March 1990 and March 1991 and monthly to September 1991 from within the pumping chamber, wetland edge and wetland. Biweekly samples taken at locations (points) indicated by Fig. 3 were averaged.

For most constituents wastewater treatment was evaluated from analysis of 24 influent samples (point 1, Fig. 3), 72 wetland edge samples (avg. of points 2-4) and 24 samples from the distal end of each wetland cell (Points 5-7). Analysis of 16 influent samples (point 1), 48 wetland edge samples (points 2-4 avg.) and 48 effluent samples (points 5-7 avg.) was used to evaluate COD treatment (Fig. 3). Total Kjeldahl nitrogen (TKN), $NH_4$-N, $NO_3$-N, TP, $PO_4$-P, COD, SS and fecal coliform bacteria were determined using standard laboratory methods (American Public Health Association 1989). Total nitrogen was determined by adding TKN and $NO_3$-N values. Organic nitrogen was determined by subtracting $NH_4$-N from TKN values.

An electrical impulse counter and a elapsed time indicator were wired into the control circuit for the effluent pump which doses the mound component. The volume of wastewater pumped during a 10 minute cycle was measured during the start up of the system. The wastewater load into the system was determined by multiplying the number of minutes of pump operation times the volume pumped per minute. Pump cycle time was checked periodically. The total pump operation time was divided by the number of pump cycles to determine if the pump cycle remained constant. This information was also used to estimate the volume of wastewater pumped into the system daily and the intervals between cycles.

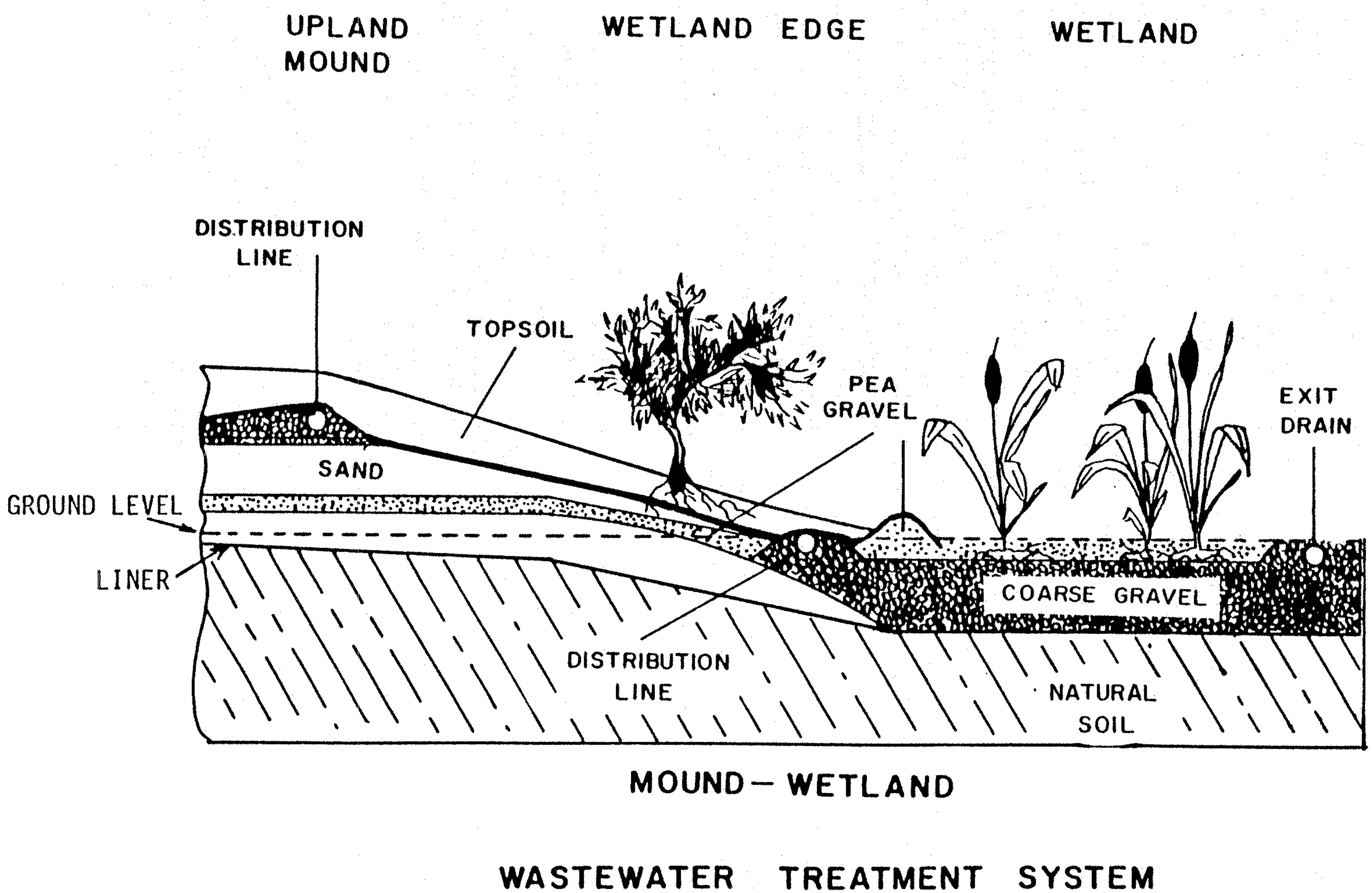


Fig. 1. Cross Section View of Constructed Upland-Wetland System

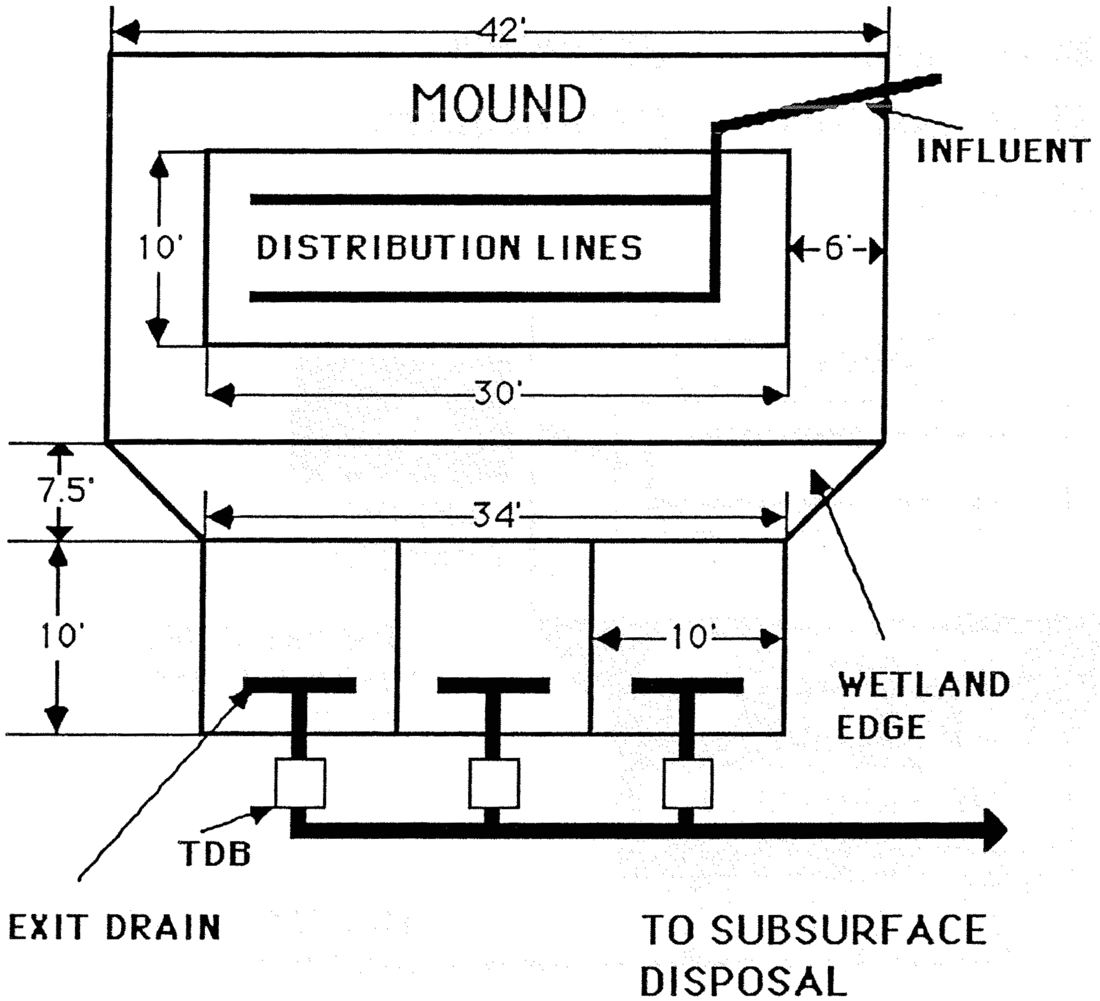


Fig. 2. Surface View of Constructed Upland-Wetland System. Tipping Distribution Boxes (TDB) or "Dippers" are Manufactured by High Point Concrete of North Carolina and United Concrete of Connecticut.

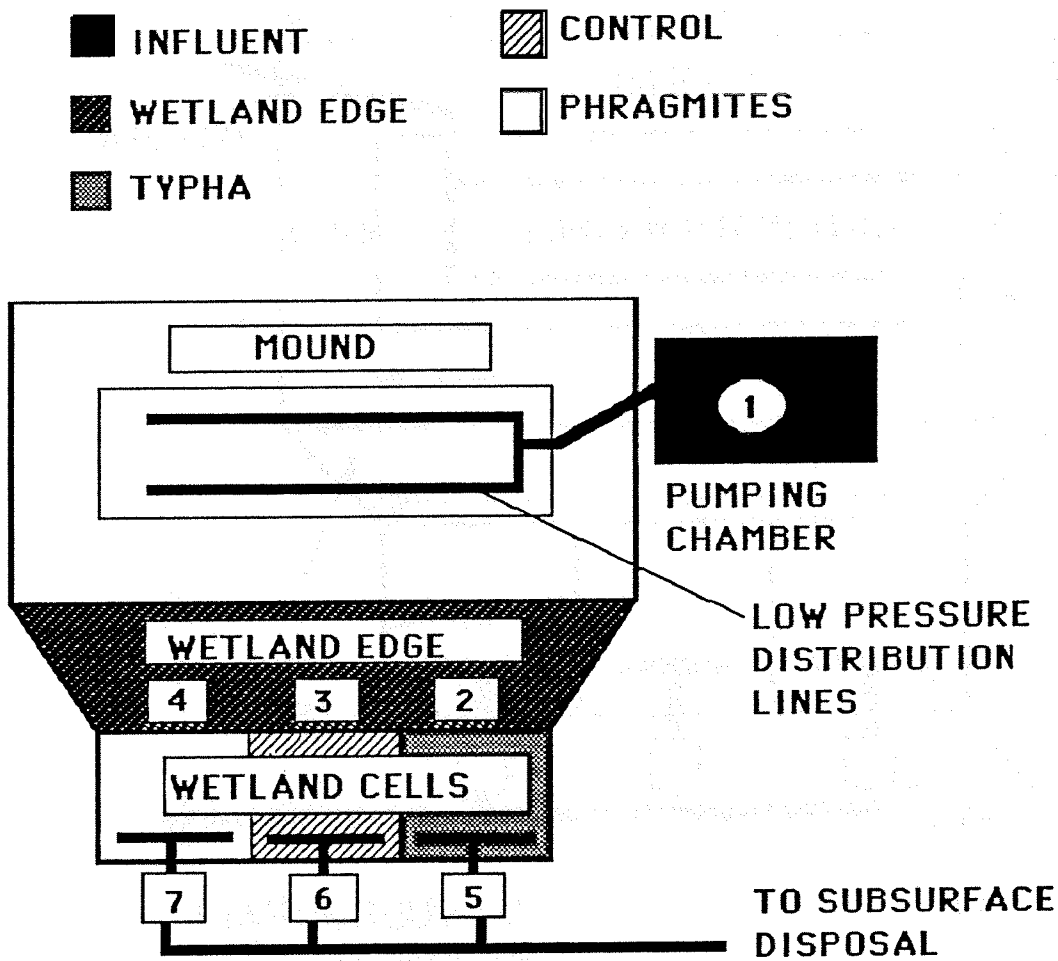


Fig. 3. Sample Locations for Water Samples Used to Evaluate Constructed Upland-Wetland Efficacy.

"Tipping buckets" equipped with mercury switch sensors were used to estimate flow from each wetland cell. The mercury switch was wired into a circuit containing impulse counters. The "tipping buckets" activate the mercury switch when they are filled to a critical volume of 4,000 ml. The switch closes and therefore activates the impulse counter. This technique provides an estimate of the number of "tips" and therefore the volume of water which leaves each cell.

An in-line water meter was installed within the supply manifold from the pumping chamber which collects the water from the 3 cells and distributes it to the final absorption field. The volume indicated by the in-line meter was compared to the total volume from the 3 wetland cells.

Mass balance calculations were made for the 43-day period from July 29, 1991, to September 10, 1991. Volume flow as measured by electro-mechanically monitoring the 3 "tipping buckets" corresponded within 8% of the volume flow of water pumped from the pumping chamber receiving water from the 3 cells and measured with the in-line meter. The following assumptions were made for these calculations: the agreement in volume flow between the two monitoring methods indicates that they both were functioning properly, the volume into a wetland cell is equal to the volume out, and the difference between TN removed by the system and the sum of the wetland cells equals that removed by the mound component.

Chloride concentration was measured to assess dilution within the system. Chloride is relatively unaffected by the biological and chemical processes within the system and therefore provides a conservative value with which to compare other nutrient concentrations. Also, chloride concentration of the influent when compared with chloride concentrations at other points within the system is a good indicator of dilution. This approach has been used to assess the hydrology of wetland systems when adequate direct hydrologic information is not available (Rigler 1979; Peterjohn and Correll 1986).

The cost of the construction and maintenance of the research system is competitive with other on-site systems used in Pamlico County for soils which are unsuitable for conventional septic tanks and associated absorption fields (Table 3). Installation cost for mound systems in the area are $10,000-$15,000.

Table 2. Mound Wastewater Dosing Network Design.

| | |
|---|---|
| Number of Lines | 2 |
| Length of Line | 30 ft/line |
| Hole Spacing | 5 ft centers-2.5 ft from ends |
| Number of Holes/Line | 6 |
| Hole Size | 7/32 inch |
| Pressure Head | 5 ft |
| Flow Rate/Hole | 1.25 gal/min |
| Total Number of Holes | 12 |
| Total Flow Rate | 15.12 gal/min |
| Dosing Length and Interval | 10 min every 8-12 hours |
| Dosing Volume | 150 gal/dose=450-300 gal/day |
| Pump and Control | 0.4 HP Effluent Pump<br>Timer Float Control Combination |

(Cogger et al. 1982; House and Cogger 1985).

Table 3. Cost of Installation of Constructed Upland-Wetland System During 1989.

| | |
|---|---|
| Asphalt Sand @4.50/ton | 30.65 tons @ 137.93 |
| Concrete Sand @ 4.50/ton | 61.15 tons @ 275.18 |
| # 67 Granite Stone | 40.59 tons @ 340.96 |
| #78 M Granite Stone | 24.29 tons @ 204.04 |
| 8 Loads Hauling Cost | 1070.00 |
| Pump Out Existing Septic Tank | 100.00 |
| 1 Pumping Tank Installed | 675.00 |
| 1 Effluent Pump | 400.00 |
| 1 Float Switch | 50.00 |
| 3 Distribution Boxes @ $50.00 | 150.00 |
| PVC pipe, valves, fittings | 400.00 |
| Electrical Hardware | 400.00 |
| Salaries 3 people/40 hrs/$30/hr | 3600.00 |
| 9 shrubs @ $5.00/40 plants @$4.00 | 205.00 |
| Back Hoe Rental 1 week | 400.00 |
| TOTAL COST | $8,408.00 |

RESULTS

Fecal Coliform Bacteria

The commonly accepted standard for water quality for point source discharges into surface waters is 2.3 $\log_{10}$ colonies/100ml (200 colonies/100ml). The mean number of colonies in samples from the influent applied to the Upland-Wetland during 1.75 years of operation ranged from 4.3-6.3 $\log_{10}$ (19,953-1,995,262) Millipore Filter (MF) colonies/100ml (Fig. 4). The upland component (as measured at the wetland edge) reduced this measure to 1.0-3.2 $\log_{10}$ (10-1,585) MF colonies/100ml. This represents an average decrease of 3.8 $\log_{10}$ (6310) by the mound (Fig. 5). This value is higher than the 2.3 $\log_{10}$ (200 colonies)/100 ml required for NPDES.

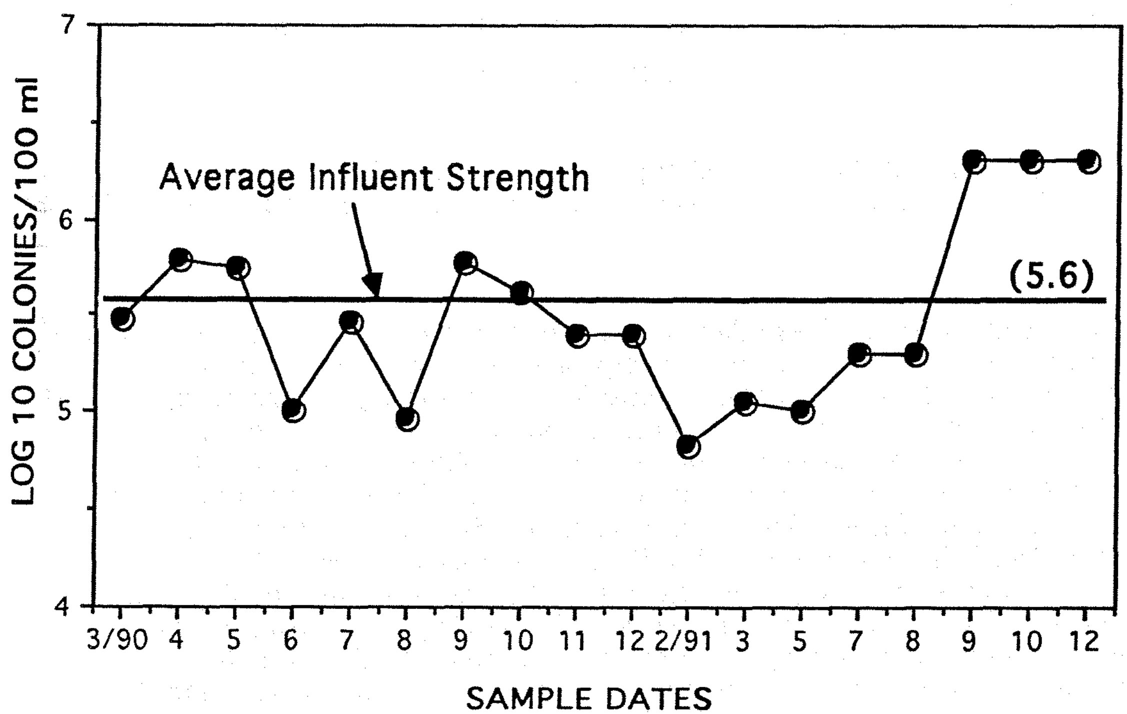


Fig. 4. Influent Fecal Coliform Colonies/100 ml from 3/90 to 12/91. Samples Were Taken from Point 1 in Fig. 3.

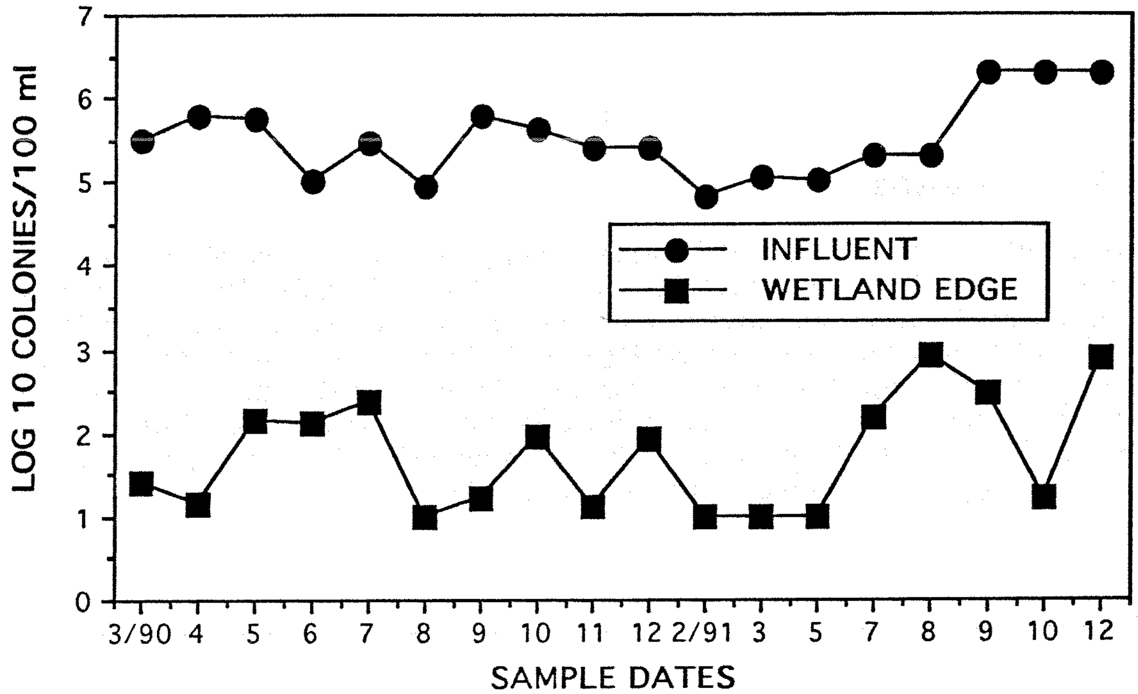


Fig. 5. Influence of the Mound Component on Fecal Coliform from 3/90 to 12/91. Samples Were Taken from [Points (1) ( 2, 3, and 4 avg.) in Fig. 3].

Coliform counts in water samples taken at the base of the mound provide evidence of even distribution of influent from the PVC pipe distribution system and in the mound. The wells at the wetland edge (points 2, 3, and 4 in Fig. 3) are evenly spaced along the long axis of the mound. Fecal coliform values from the three wells generally increased or decreased in concert (Fig. 6). Coliform colonies within the wells peaked 5 times during the study: June, October and December of 1990; and August and December of 1991. The peaks occurred only in the center well in December 1990 and the center and end wells in December 1991 (Fig. 6). The peak in colony numbers in water coming from the base of the mound is also evident within the wetland cells during August and December of 1991 (Fig. 7). The amount of water leaving the system as estimated by an in line meter, indicated that during the August 1991 peak, the flow from the system was more than 100 gal/day greater than the average. Meter measurements did not indicate unusual flow through the system to correspond with peak coliform values during December 1991. Hydraulic outflow measurements from the system were not made during 1990.

Geometric mean of fecal coliform colonies in liquid from the cell planted with Typha angustifolia ranged from 1.0-3.7 $\log_{10}$; in the unplanted cell, 1.0-4.3 $\log_{10}$ ; and in the cell planted with Phragmites australis, 1.0-3.0 $\log_{10}$. This represents very little influence by the wetland cells (Fig. 7).

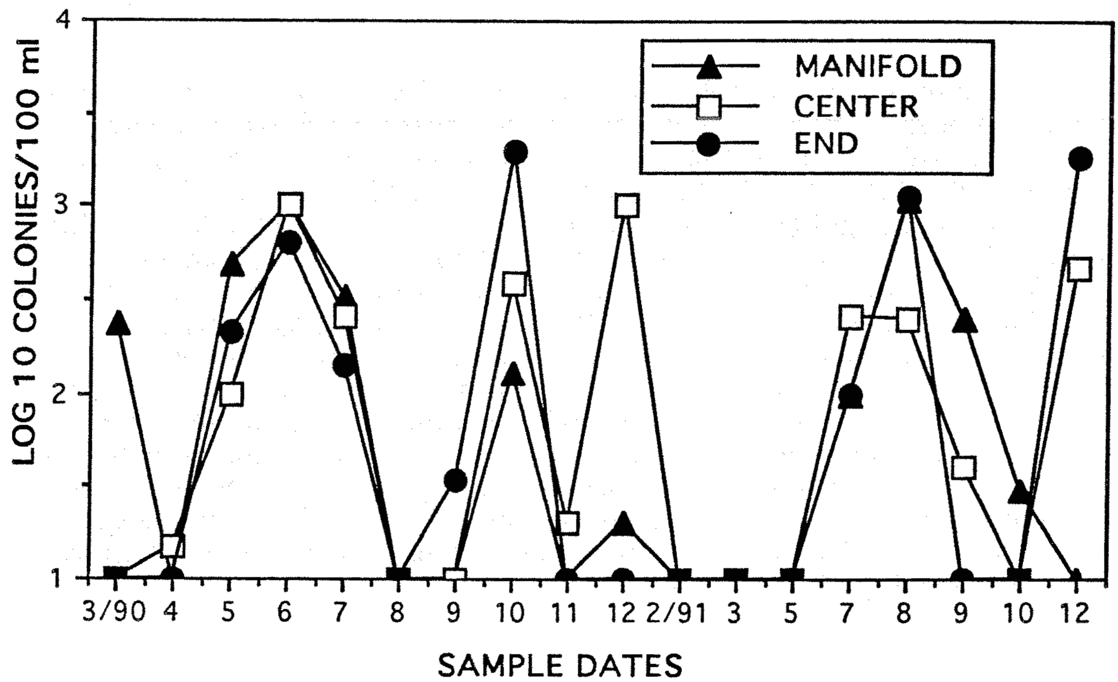


Fig. 6. Distribution of Fecal Coliform Within the Mound from 3/90 to 12/91. Samples Were Taken from the Wetland Edge (Points 2, 3, and 4 in Fig. 3).

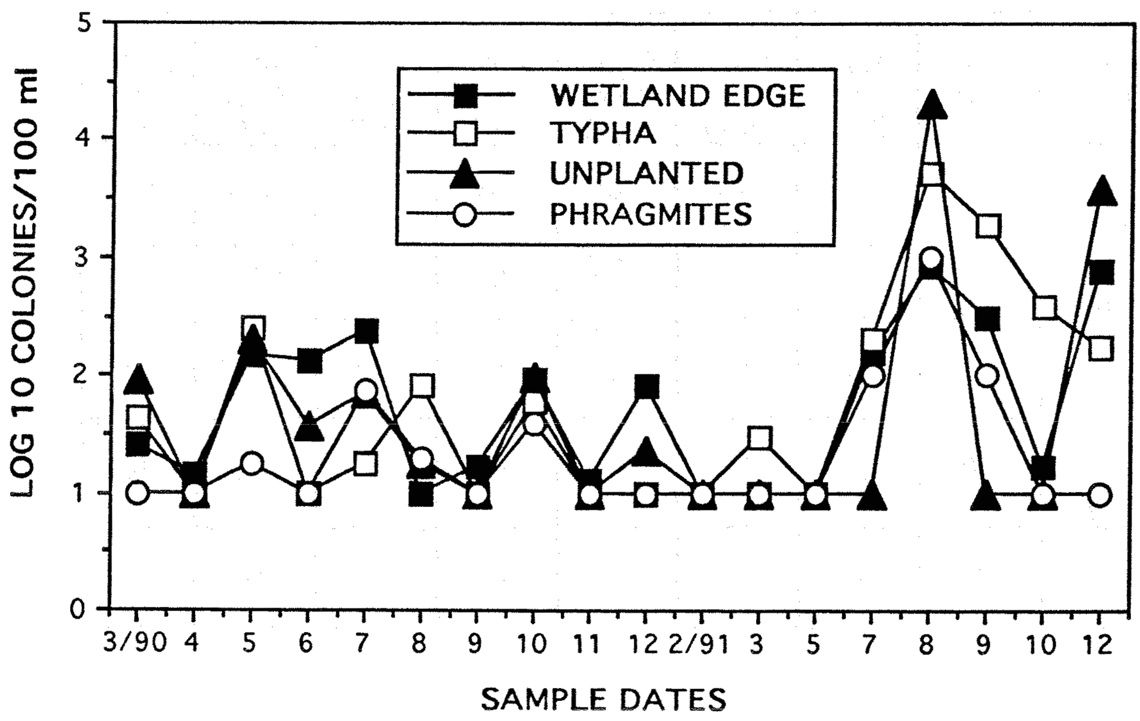


Fig. 7. Influence of the Wetland Cells on Fecal Coliform. Samples Were Taken from the Wetland Edge and the Exit Drains of the Wetland Cells [Points (2, 3 and 4 avg.)(Points 5, 6 and 7) in Fig. 3].

Comparisons of fecal coliform colony counts in samples averaged quarterly for 1990 and 1991 do not show seasonal trends but suggest that the cell planted with Phragmites australis was more effective than the other cells in decreasing fecal coliform bacteria (Table 4). The highest average value (90 colonies/100 ml) for the cell planted with Phragmites australis during 1990 is associated with heavy rainfall on July 17 when there was an increase in fecal coliform colonies (from a previous high of 30 colonies to 540 colonies/100 ml) in water from the exit drain. The high value during June of 1991 from the cell planted with Phragmites australis (373 colonies/100 ml) was associated with mechanical failure. This value was substantially lower than fecal coliform in samples from the other wetland cells during this same period. The mechanical failure of the effluent pump caused wastewater to flood the septic tank that supplies the system. An increase in fecal coliform throughout the system was observed in samples taken after the system became fully operational. This was probably due to the overflow and subsequent mixing of raw unsettled sewage from the septic tank into the pumping chamber that supplies the Upland-Wetland.

Table 4. Influence of the Wetland Cells on Fecal Coliform. Samples Were Taken from the Wetland Edge and the Exit Drains of the Wetland Cells [Points (2, 3 and 4 avg.)(5, 6 and 7) in Fig. 3]. (The NPDES Standard is 200 colonies/100 ml).

| Sample Points | Jan-Mar 1990 | Apr-June 1990 | July-Aug. 1990 | Sept.-Dec. 1990 | Jan-Mar 1991 | Apr-June 1991 | July-Aug. 1991 | Sept.-Dec. 1991 |
|---|---|---|---|---|---|---|---|---|
| Wetland Edge | 80 | 390 | 80 | 396 | 10 | 10 | 364 | 553 |
| Typha | 60 | 127 | 76 | 100 | 15 | 10 | 2433 | 275 |
| Un-planted | 880 | 365 | 84 | 250 | 20 | 10 | 6673 | 1855 |
| Phrag. australis | 10 | 13 | 90 | 48 | 10 | 10 | 373 | 10 |
| Average of 3 Cells | 317 | 168 | 83 | 133 | 15 | 10 | 3160 | 713 |

Nitrogen

The concentration of influent nitrogen was within the 30-80 mg/L TN (primarily ammonium) range typically found in domestic wastewater (U. S. Environmental Protection Agency 1980). All of the ammonium-N in the influent was converted to nitrate-N and the concentration of nitrogen was lowered substantially as the wastewater passed through the mound (Fig. 8). The low point on the graph that occurred during November was due to system down time for pump replacement. Concentrations of influent ammonium have been generally higher since pump replacement, but complete nitrification by the mound has continued.

The wetland cell planted with Phragmites australis lowered the concentration of nitrogen in the wastewater more than did the unplanted cell or the cell planted with Typha angustifolia. Nitrate concentrations in water leaving the cell planted with Phragmites australis were below the Environmental Protection Agency drinking water standard of 10 mg/L 14 of 29 sampling events (Fig. 9). The nitrate concentrations leaving the cell planted with Typha angustifolia were below 10 mg/L 9 times. Nitrate concentrations in effluent from the unplanted cell were below 10 mg/L 12 times.

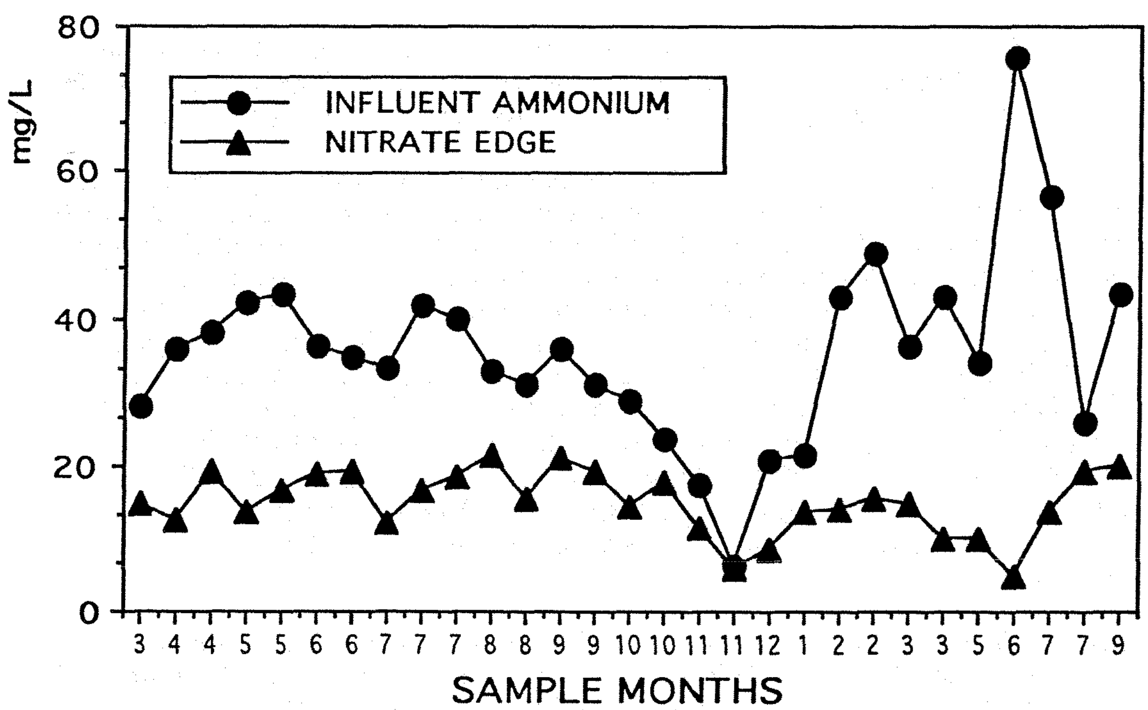


Fig. 8. Nitrification by the Mound Component from 3/90 to 9/91. Influent Values Compared to the Wetland Edge [Points (1)(2, 3 and 4 avg.) Fig. 3].

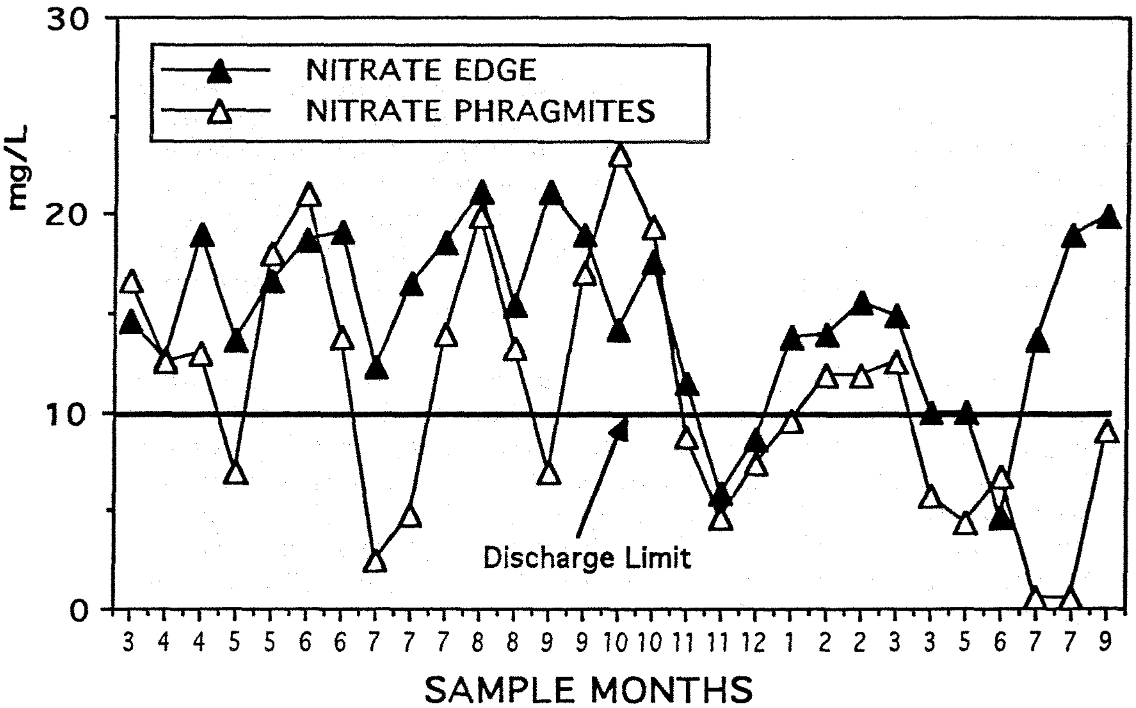


Fig. 9. Treatment of Nitrate Nitrogen by the Phragmites australis cell from 3/90 to 9/91. Samples Taken from the Wetland Edge and the Exit Drain of the Phragmites australis cell [Points (2, 3 and 4 avg.) (7) in Fig. 3].

Average concentration of TN in the influent was 44.4 mg/L, primarily ammonium-N (35.4 mg/L). Other forms of nitrogen were organic nitrogen (8.6 mg/L) and nitrate-N (0.4 mg/L). Analysis of water samples taken from wells at the base of the mound averaged 16.0 mg/L of TN primarily in the form of nitrate-N (14.9 mg/L) and <0.1 mg/L of ammonium-N. This is a 64% reduction in total nitrogen concentration caused by the mound component probably by nitrification-denitrification, plant uptake, volatilization and dilution. Average concentration of TN entering the wetland cells was 16.0 mg/L and water leaving the cells had an average concentration of 13.2 mg/L which was primarily nitrate-N. (Fig. 10). Average nitrate-N concentrations (mg/L) in water leaving the three cells was: Typha angustifolia, 12.9; unplanted cell, 12.3; and Phragmites australis 10.0. Organic nitrogen concentration in the influent averaged 8.6 mg/L and was approximately 1 mg/L in samples at all other sampling points.

The ratio of chloride in samples from the wetland edge and the exit drains of each wetland cell, compared to chloride in the influent indicates that the cell planted with Typha spp. and the unplanted cell were diluted about the same relative to the average of the three sampling points at the wetland edge. Less dilution occurred within the cell planted with Phragmites australis. The chloride/chloride values for the wetland edge compared with that from the exit drain of each of the three cells by regression analysis suggest this. (Typha angustifolia, $R^2$ =0.788; unplanted cell, $R^2$ =0.768; and Phragmites australis, $R^2$ =0.073). The drop in the graph on June 1991 suggests that there was an increased hydraulic load on all of the system except the cell planted with Phragmites australis (Fig. 11).

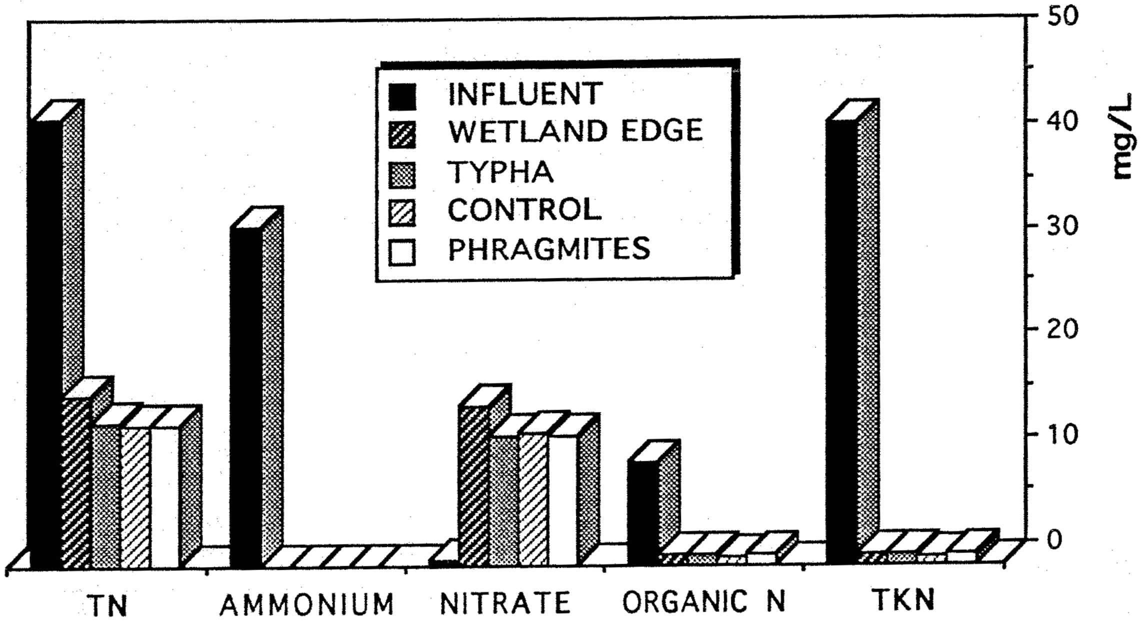


Fig. 10. Average Concentration of Nitrogen Within the Upland-Wetland from 3/90 to 9/91. Samples Were Taken from the Influent, Wetland Edge and the Exit Drains [Points (1) (2, 3 and 4 avg.) (5, 6 and 7) in Fig. 3].

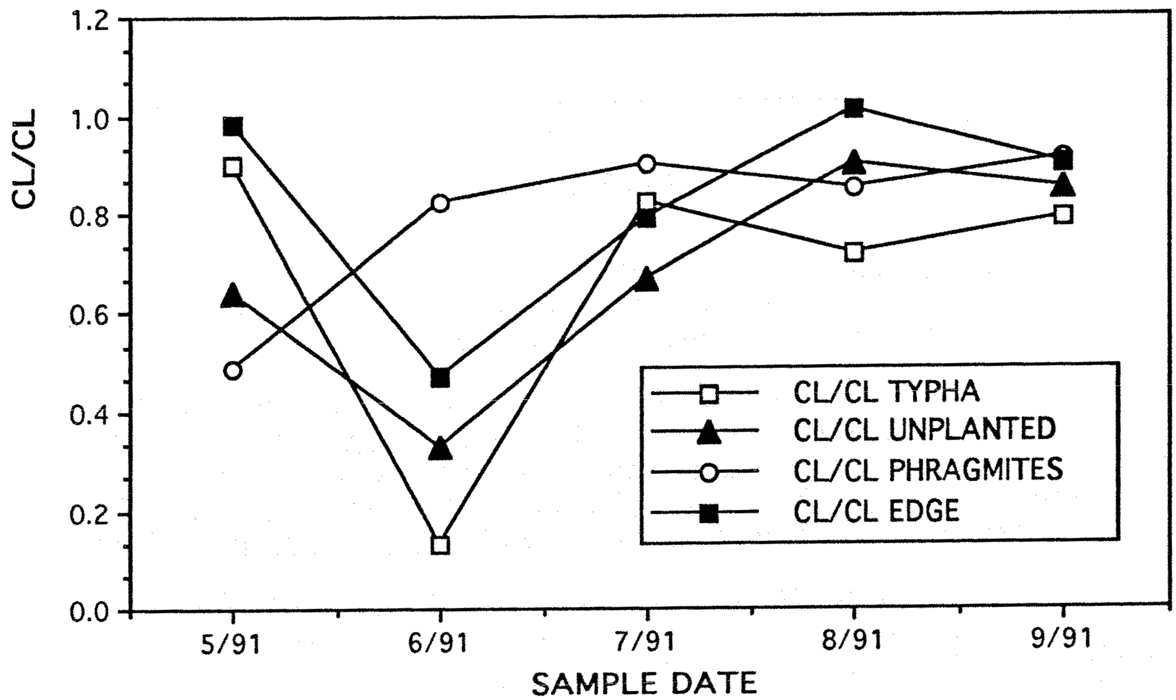


Fig. 11. Comparison of Chloride/Chloride Ratios from 5/91 to 9/91. Samples Were Taken from the Influent, Wetland Edge and the Exit Drains of the Wetland Cells [Points (1) (2, 3 and 4 avg.) (5, 6, and 7) in Fig. 3].

Influent ammonium-N and chloride are strongly related as indicated by regression analysis ($R^2$=0.967). The line which represents the relationship between nitrate and chloride in the cell planted with Phragmites australis ($R^2$=0.058) suggests the change in chloride and nitrate-N are not related and therefore any change in nitrogen concentration which occurs is not related to dilution. Nitrate-N to chloride comparisons at other points within the system suggest that change in nitrogen and chloride are related and therefore nitrogen concentrations are probably impacted to a greater extent by dilution. Comparisons of nitrogen/chloride by regression analysis gave the following: wetland edge, $R^2$=0.758; Typha angustifolia cell $R^2$=0.867; and the unplanted cell, $R^2$=0.670 (Fig. 12).

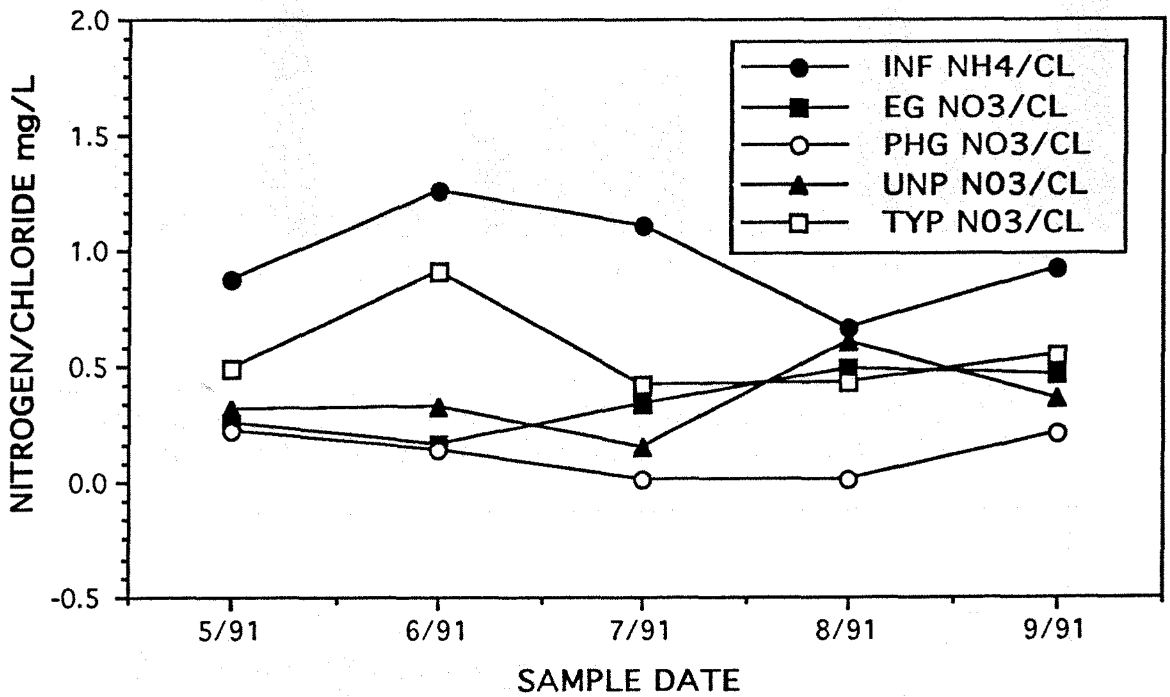


Fig. 12. Nitrogen/Chloride Ratios Within the Upland-Wetland from 5/91 to 9/91. Samples Were Taken from the Influent, Wetland Edge and the Exit Drains [Points (1)(2, 3 and 4 avg.)(5, 6, and 7) in Fig.3]

Mass balance calculations from July 29 to September 10, 1991, indicate that the mound component and the cell planted with Phragmites australis were the most effective, removing 12.6 kg and 13.2 kg respectively. Note however that the mound received more total nitrogen than the cell planted with Phragmites australis. The unplanted cell removed more than the cell planted with Typha angustifolia (Table 5).

Table 5. Total Nitrogen Mass Balance from July 29 to September 10, 1991.

| Sample Component | Influent kg | Effluent kg | kg Change |
|---|---|---|---|
| Mound | 73.4 | 60.8 | 12.6 |
| Typha | 14.0 | 11.9 | 2.1 |
| Unplanted | 22.0 | 14.6 | 7.4 |
| Phragmites | 21.0 | 7.8 | 13.2 |

Phosphorus

The average concentration of TP entering the system was 4.4 mg/L, of which 3.0 mg/L was phosphate (Fig. 13). Most of the decrease in phosphorus concentration within the system occurred in the mound. The mound component reduced the concentration from 4.4 mg/L to an average of 0.6 mg/L. The wetlands decreased the concentration to an average of 0.5 mg/L, all of which was phosphate (avg. 5, 6, and 7 in Fig. 3). The cell planted with Phragmites australis decreased the concentration more than the other two cells (Fig. 13).

Chloride concentration was measured to differentiate lowering concentration due to dilution versus that of other processes within the system. Influent TP and chloride are strongly related as indicated by regression analysis ($R^2$=0.981). There is no relationship between TP and chloride at any other location within the system. The TP/chloride ratio of the cell planted with Phragmites australis is the most different from the influent ratio (Fig. 14).

Mass balance calculations from July 29 to September 10, 1991, indicate that the mound component was the most effective and removed 4.4 kg of the 4.88 kg TP removed. The wetland cell planted with Phragmites australis was the most effective of the wetland components and removed .63 kg TP. The unplanted wetland cell contributed .42 kg TP to the discharge (Table 6).

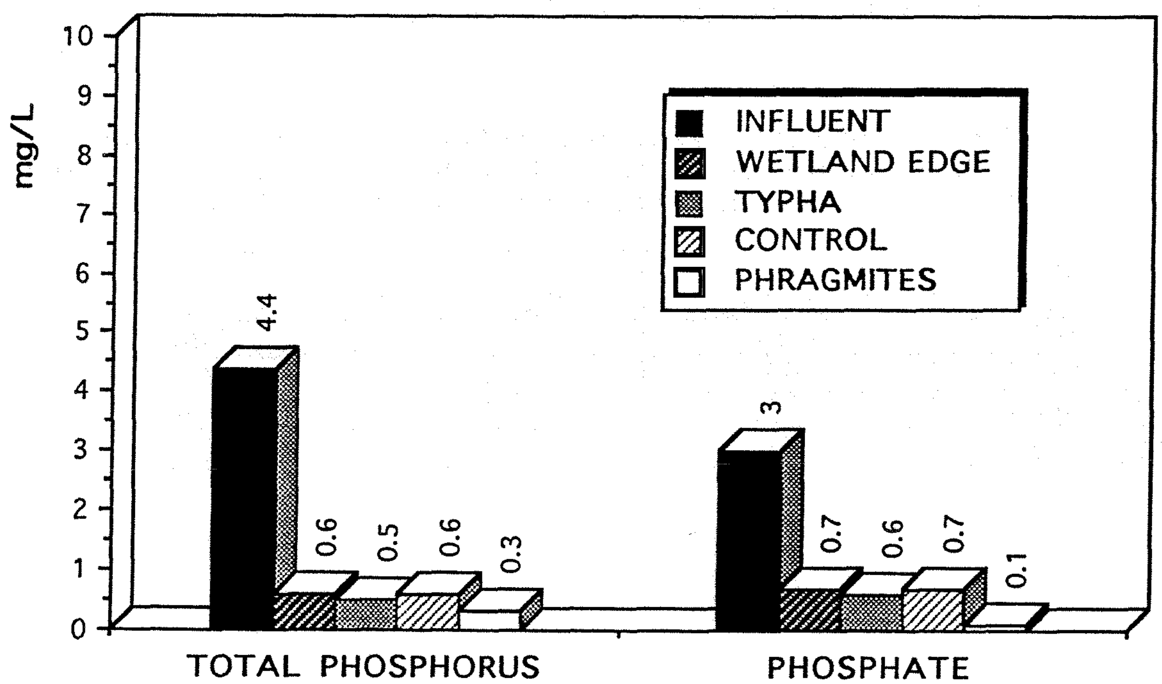


Fig. 13. Average Concentrations of Total and Phosphate Phosphorus Within the Upland-Wetland. Samples Were Taken from the Influent, Wetland Edge and the Exit Drains [Points (1)(2, 3 and 4 avg.)(5, 6, and 7) in Fig. 3].

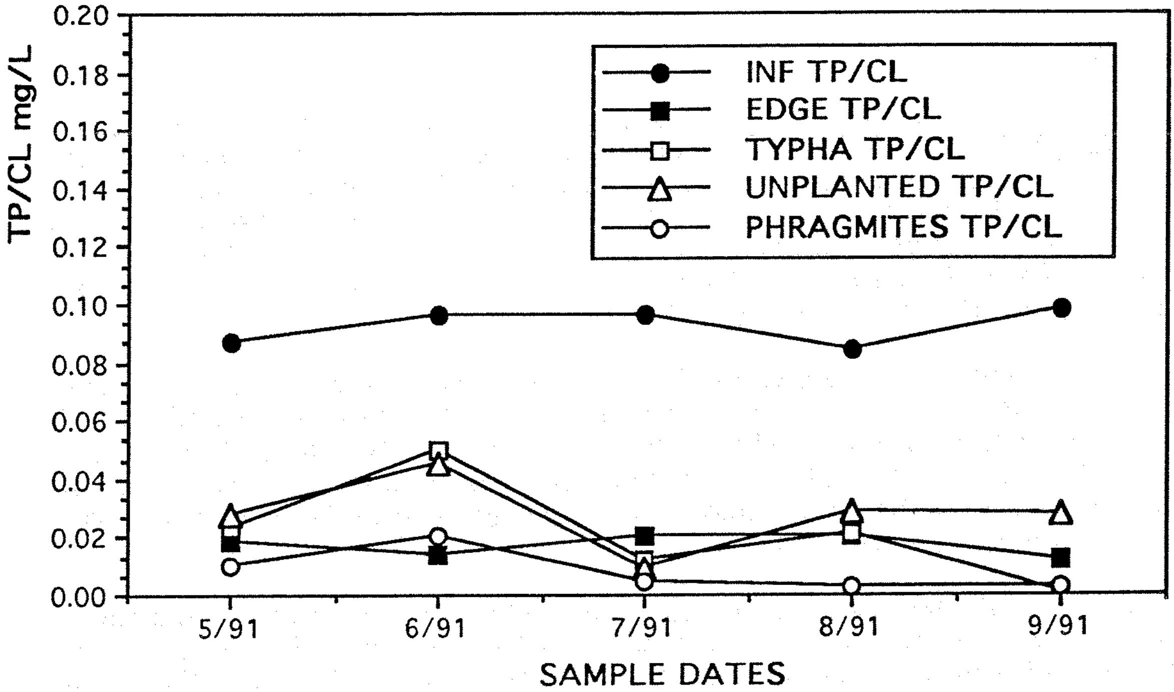


Fig. 14. Total Phosphorus/Chloride Ratios Within the Upland-Wetland. Samples Were Taken from the Influent, Wetland Edge and the Exit Drains [Point (1) (2, 3, and 4 avg.) (5, 6, and 7) in Fig. 3].

Table 6. Total Phosphorus Mass Balance from July 29 to September 10, 1991. Volume Flow Measurements were from the Influent, Wetland Edge and the Exit Drains [Points (1) (2, 3 and 4 avg.) (5, 6 and 7) in Fig. 3].

| Sample Component | Influent kg | Effluent kg | kg Change |
|---|---|---|---|
| Mound | 6.14 | 1.74 | 4.40 |
| *Typha* | .49 | .22 | .27 |
| Unplanted | .38 | .80 | (+) .42 |
| *Phragmites* | .73 | .10 | .63 |

Chemical Oxygen Demand and Suspended Solids

Chemical oxygen demand influent concentration ranged from 75-492 mg/L and average exit drain sample concentration ranged from 10-76 mg/L. Most of the decrease in COD concentration by the system occurred in the mound (from 337.8 mg/L in the influent to 43.9 mg/L in the wetland edge). The wetlands decreased the concentration to an average concentration of 34 mg/L (Fig. 15). There was very little difference among the effects of the 3 wetland cells.

Suspended solids influent concentration ranged from 16-122 mg/L and average exit drain sample concentration ranged from 2-17 mg/L. The decrease of SS concentration by the Upland-Wetland system occurred mostly within the mound (from 95.1 mg/L in the influent to 11.2 mg/L in the wetland edge). The average decrease by the 3 wetland cells was from 11.2 mg/L to 8.6 mg/L (Fig. 16).

Mass balance calculations from July 29 to September 10, 1991, indicate that the mound component removed 137.7 kg of SS. The unplanted wetland cell and the cell planted with Phragmites australis contributed to SS in the discharge (Table 7).

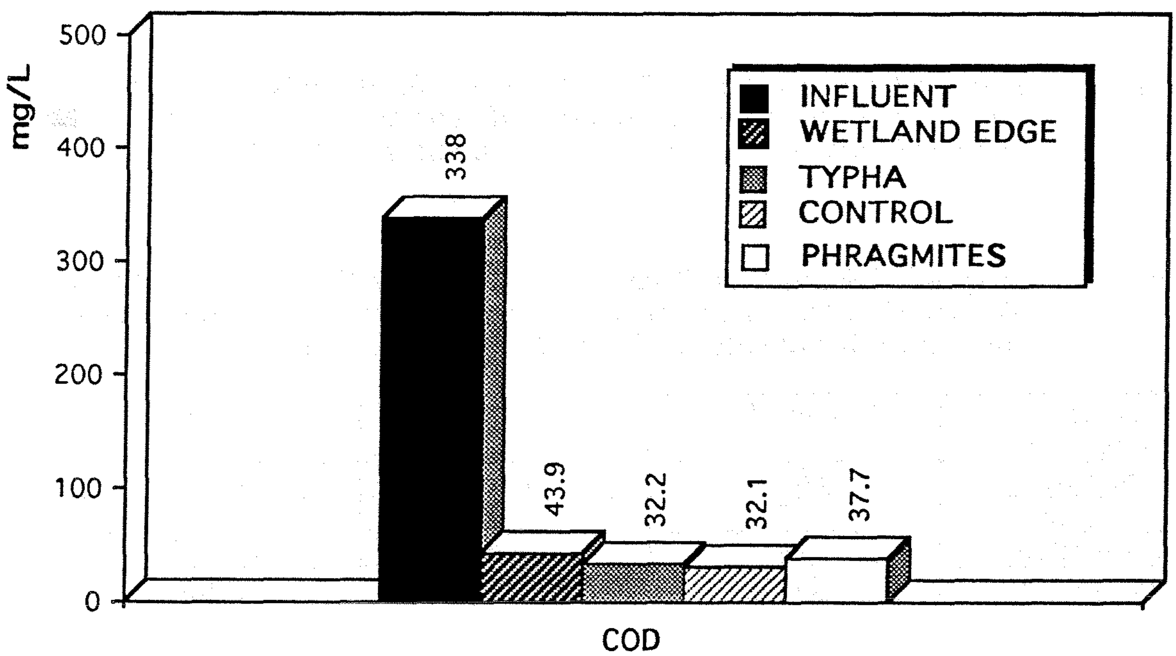


Fig. 15. Average Concentration of COD Within the Upland-Wetland. Samples Were Taken from 3/90 to 11/90 from the Influent, Wetland Edge and the Exit Drains [Points (1)(2, 3 and 4 avg.)(5, 6 and 7) in Fig. 3].

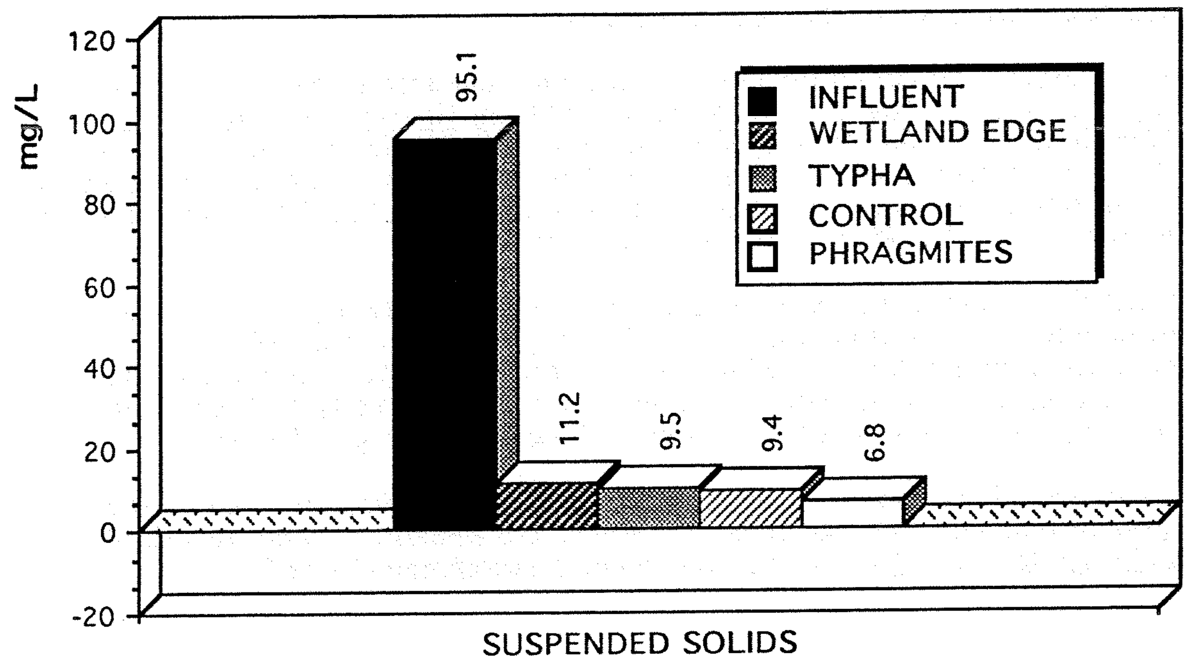


Fig. 16. Average Concentration of SS Within the Upland-Wetland. Samples Were Taken from 3/90 to 9/91 from the Influent, Wetland Edge and the Exit Drains [Points (1) (2, 3 and 4 avg.)(5, 6 and 7) in Fig. 3].

Table 7. Suspended Solids Mass Balance from July 29 to September 10, 1991. Volume Flow Measurements were from the Influent, Wetland Edge and the Exit Drains [Points (1)(2, 3 and 4 avg.)(5, 6 and 7) in Fig. 3].

| Sample Component | Influent kg | Effluent kg | kg Change |
|---|---|---|---|
| Mound | 166.2 | 28.5 | 137.7 |
| Typha | 9.7 | 7.7 | 2.0 |
| Unplanted | 5.7 | 9.5 | (+) 3.8 |
| Phragmites | 12.4 | 12.9 | (+).5 |

## DISCUSSION

### Fecal Coliform Bacteria

The results indicate that the mound was the most effective system component in removing pathogenic bacteria; that there is little difference between the influence of the cell planted with Typha angustifolia and the unplanted cell; and that the cell planted with Phragmites australis was more effective than the others in decreasing coliform. Decrease of coliform bacteria by the system was the result of the combination of influences by the septic tank, the pumping chamber, the wastewater distribution system, the soil within the mound and the hydraulics of the system.

A 3,785 liter (1000 gal) septic tank collects wastewater from the house. The wastewater then flows by gravity into a second tank of the same size, the pumping chamber. This 7570 liter (2000 gal) capacity provides 6 to 7 days of retention for sedimentation to reduce microbial pathogen populations. A typical residential septic tank contains a fecal coliform population of $10^8$-$10^{10}$ colonies/100 ml (U.S. Environmental Protection Agency 1980). Analysis of samples taken from within the pumping chamber of the research system contain $10^5$ -$10^6$ colonies/100 ml of fecal coliform. This is an estimated 3 to 4 magnitude decrease in indicator bacteria from the septic tank to the pumping chamber.

The mound component provides an unsaturated soil environment to help retain and inactivate microbial pathogens. The 2 feet of sandy loam textured materials used to construct the mound reduced the numbers of indicator bacteria by a magnitude of almost 4. The system is designed to pressure dose 378.5 liters (100 gal) of wastewater 3 to 4 times/day into the mound. This regime allows for 6 to 8 hours of soil drainage, which improves pathogen attenuation and destruction. The pressure dosed design gives better distribution of the wastewater than a gravity fed system and therefore helps to prevent saturated flow and resultant microbial movement (Converse et al. 1974; Green and Cliver 1974; Bouma et al. 1972; Bouma et al. 1973).

The measure of fecal coliform colonies in wells at the base of the mound indicates a fairly even distribution of wastewater. The hydraulic capacity of the mound was apparently overloaded periodically however, as indicated by peaks in coliform in all wells.

Soil factors which may influence pathogen removal include pH, moisture, temperature, organic matter content of the substrate, type and amount of clay, and the depth of soil material through which the wastewater passes (Sobsey 1986).

During this study the pH of the influent wastewater ranged from 7.7-8.3; water taken from the wetland edge, 5.7-7.5; from the cell planted with Typha angustifolia, 6.4-7.9; from the unplanted cell, 5.4-8.1; and from the cell planted with Phragmites australis, 5.4-7.9. Though the soil water was occasionally acid within the wetland edge and the wetland cells, it was generally basic which is favorable to pathogen survival (Sobsey 1986; Cuthbert et al. 1950; Beard 1940; Kligler 1921).

The soil and water environments within the system are probably not subject to temperature extremes that would increase pathogen destruction (Bitton and Gerba 1984). During winter cold, the warmth of the wastewater, the depth of the soil of the mound, the insulating effect of the vegetation, the southern aspect of the mound and the dark colored gravel within the wetland cells probably maintain a fairly uniform temperature. During summer the vegetation provides insulation and transpires water which helps to cool the system.

The soil materials used to build the system were very low in organic matter. As the system becomes older, the organic matter from the wastewater and that from plant and microbial decay will probably have more of a influence on the pathogenic microbes. It is likely that the increase in organic matter will enhance survival of microbial pathogens (Tate 1978; Sobsey 1986).

The wetland provided additional hydraulic capacity to minimize flushing of pathogens from the system. To help reduce flushing of pathogens that sediment within the wetland cells, discharge pipes from the wetland cells were placed 15 cm (6 in) beneath the surface rather than on the bottom of the cells. The 2 feet of 0.5-0.75 inch gravel used in the wetland cells probably does not contribute to the removal of bacteria and viruses as much as finer textured materials would have. The size of the gravel within the wetland cells was chosen to prevent clogging by biological and physical processes. Since the mound reduces BOD and SS, a finer material probably could be used as substrate in the wetland, which may improve treatment of microbial pathogens.

The flushing of pathogens, as indicated by fecal coliform, was more dramatic from the unplanted wetland cell and corresponds to flushing events from the mound. The cells planted with Typha angustifolia and Phragmites australis reduced the impacts of hydraulic overload to the system. The cell planted with Phragmites australis was most effective. It is probable that the root mass of Typha angustifolia and Phragmites australis increased the removal of coliform bacteria by filtration. The growth of Phragmites australis was more extensive than Typha angustifolia which probably contributed to its greater effectiveness.

Periodic pulses of microbial pathogens are a concern, though generally the system lowered the bacteria to values acceptable for discharge under NPDES requirements. Phragmites australis performed better than Typha angustifolia or the unplanted cell and therefore should be considered for use in future system designs. The hydraulic capacity of the wetlands should be increased by making the cells deeper to reduce the potential for flushing microbial pathogens from the system. Plant species that can tolerate a wide moisture range should be used with this design. Possibly coarse sand should be used in the wetland cells rather than gravel to increase the filtration of microbial pathogens.

## Nitrogen

The aerobic environment of the mound was effective at both nitrifying the $NH_4$-N and lowering nitrogen concentrations. The decrease in nitrogen concentration by the mound was probably the result of volatilization, plant uptake, denitrification and dilution.

The increase in $NH_4$-N in the influent after November 1990 was probably due to the flooding of the septic tank that supplies the system prior to effluent pump replacement. This caused materials that would normally be contained in the septic tank to flow into the pumping chamber.

In general, the reduction of nitrogen concentration within the wetland cells was probably due to plant uptake, denitrification and dilution. Removal of nitrogen by the wetland cells may increase as the root system of the plants develop to provide organic matter as an energy source for denitrifying microorganisms. The low concentrations of organic nitrogen found within the system may be an indication of the low available carbon for denitrification.

The wetland cell planted with Phragmites australis appears to have lowered nitrogen concentration by processes other than dilution (i.e., plant and microbial uptake or nitrification-denitrification) to a greater extent than other locations within the system. It also appears that the cell planted with Phragmites australis was less diluted by precipitation than were other areas and therefore may have received less hydraulic load or evapo-transpired more water.

The removal of more TN by the unplanted wetland cell than the cell planted with Typha angustifolia as indicated by mass balance, may have been the result of an increase in temperature in the unplanted cell due to lack of vegetative cover. Dissolved oxygen within the cell would be expected to decrease with increasing temperature and therefore enhance denitrification. Carbon necessary for microbial denitrification may have been supplied by the wastewater.

The current data suggest that: (1) a properly designed and maintained mound system is effective at nitrification of wastewater ammonium; (2) Phragmites australis does influence the treatment of wastewater nitrogen either by plant uptake or by enhancing nitrification-denitrification; (3) the wetland cells are probably carbon limited as indicated by low organic nitrogen concentrations; and (4) $NO_3$-N concentrations fluctuate widely.

A more detailed hydrologic monitoring of the system is desirable. Continued comparisons between mass balance and indicator ratios of chloride and nitrogen would better substantiate these results.

## Phosphorus

Sand in the mound and wetland edge contains marl that helps remove phosphorus by precipitation as calcium and magnesium phosphates. The wetland cells, other than the one planted with Phragmites australis had little influence on the treatment of phosphorus. The cell planted with Phragmites australis produced extensive above-ground biomass as compared to the other cells and therefore probably contributed to accretion of phosphorus as organic matter.

There is no apparent correlation between TP and chloride in the wetland edge or any of the wetland cells. Therefore, any change in phosphorus concentration is probably due to precipitation, adsorption and plant and microbiological uptake rather than dilution.

The contribution of TP to the system discharge by the unplanted cell during July to September 1991 is probably an indication that the granite gravel substrate in the cell has reached its limits for phosphorus adsorption and the microbial population in the cell has reached a near maximum population.

Domestic wastewater phosphorus contains mainly phosphate. The mound component is most effective at the removal of phosphorus and the wetlands contribute very little to phosphorus removal. The wetland cell planted with Phragmites australis was most effective at TP removal. This is consistent with results from other studies with domestic wastewater and constructed wetland systems. The length of time that the mound will continue to effectively decrease phosphorus concentrations in wastewater should be determined through long term monitoring.

Chemical Oxygen Demand and Suspended Solids

The system effectively lowers COD and SS. The removal of COD and SS by the system was primarily a function of the mound component. The processes involved are probably filtration, sedimentation, oxidation, adsorption and biological degradation. The mound component has consistently lowered both of these parameters throughout system monitoring. Mass balance calculations suggest that the wetland component may become a source of organic material in the discharge. The organic levels within the Upland-Wetland should continue to be monitored to determine if equilibrium will be reached at a level high enough to be considered as water quality degradation. Laboratory analyses which evaluate wastewater treatment effectiveness should be refined to better reflect the nature of organic constituents and their impacts on natural aquatic systems.

Finer substrate materials should be used in the wetland component in future designs to enhance filtration and plant growth. Physical and biological clogging of the substrate was known to be a problem with other constructed wetland systems when the Upland-Wetland system was designed; therefore, coarse gravel was used in the wetland cells to prevent this problem. The removal of organic load, which contributes to clogging, by the mound prior to the wastewater's entering the wetland cells and the fairly even distribution of water from the mound to the wetlands by base flow should permit the usage of finer materials in the wetland cells in future designs.

Decreasing the organic load into the wetland cells probably has the added benefit of preventing the coating of wetland plant roots with particulates which could reduce their function in oxygen transfer and pathogen deactivation. This decreased function has been observed in other constructed wetlands receiving wastewater without prior organic removal.

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