To the Graduate Council:

I am submitting herewith a thesis written by Matthew W. Bowling entitled "Monitoring and Evaluation of Constructed Wetlands for Wastewater Treatment". I have examined the final copy of this thesis for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Master of Science, with a major in Agricultural Engineering Technology.

Dr. D. Raj Raman, Major Professor

We have read this thesis and recommend its acceptance

Dr. C. Roland Mote

Dr. Neal S. Eash

Accepted for the Council:

Associate Vice Chancellor and Dean of The Graduate School

Monitoring and Evaluation of Constructed Wetlands for Wastewater Treatment

A Thesis

Presented for the

Master of Science

Degree

The University of Tennessee, Knoxville

Matthew Warwick Bowling

August, 1996

To:

My parents

David and Judy Bowling

for their persistence and support throughout my education.

And my wife Becky

for her love and remarkable patience.

Acknowledgments

The author would like to express his sincere gratitude to his major professor Dr. D. Raj Raman for his support and patience throughout this project; Dr. Neal S. Eash and Dr. C. Roland Mote for their comments and assistance; Dr. Ron Yoder and Dr. Dan Yoder for their for their guidance and technical assistance.

In addition the following individuals are recognized for their contributions: Mr. Johnny Fain provided invaluable support to the research both in the form of site information and in technical assistance at the site, he was also the primary liaison between the UT research team and M&M Mars. Mr. Richard Roy was responsible for overseeing the lab analyses that were critical to the successful completion of the project. Thanks is also extended to Deron Jaggers, Alex Lowery, Scott Snyder, Craig Easterly, Andy Arthur and other fellow graduate students who spent long hours assisting in the project work.

The author would like to recognize the financial support of M&M Mars Corporation, and of the Tennessee Agricultural Experiment Station at the University of Tennessee, Knoxville, which made this work possible.

ABSTRACT

To better understand how constructed wetland treatment systems work, the treatment performance of four constructed wetlands (two overland flow and two subsurface flow) was evaluated from December, 1994 through December, 1995. The wetlands provide tertiary treatment of a Southeast Tennessee food processing plant's wastewater. All four wetlands were established with a variety of wetland plants. Wetland influent and effluent were sampled weekly. Samples were tested for nitrate, ammonia, TKN, BOD_5 and other pollutants. Multiple samples were collected to maintain quality control.

Bench-scale test cells were established on-site. Each (1.5 m x 0.5 m) (5 ft by 1.7 ft) bench-scale cell corresponded to a full-scale wetland, and was set up with similar plant populations, plant types and horizontal velocities. The bench-scale cells were sampled in a similar manner to the full-scale wetlands.

The results show that a high degree of treatment is being realized for the measured pollutants. The system appeared to be lightly loaded for the pollutants measured. During the course of the study the average loading for nitrate, ammonia, TKN and BOD_5 was 25, 17, 17, and 34% (respectively) of the maximum observed loading. Even during the periods of maximum loading, the system achieved high removals for the monitored compounds.

Air, substrate and water temperatures, photosynthetically active radiation (PAR), and precipitation were monitored in order to correlate treatment performance with environmental conditions. However, hydraulic loading to the wetlands varied considerably due to disturbances in the wastewater treatment plant that provides influent to the constructed wetlands. Because of these variations, correlations between treatment performance and environmental conditions could not be made. However, the environmental data was used in calculating evapotranspiration (ET), which was needed in order to compute areal mass removal rates (AMRR).

No significant differences in treatment performance between plant types or wetland types were seen. The bench-scale cell performance appeared to be slightly lower (per unit area) than the full-scale marshes, but generally followed the same removal trends. The removal kinetics of nitrate, TKN, ammonia, and BOD_5 fit a first order model; i.e., the removal rates went up in proportion to inlet concentration.

In order to determine flow characteristics through the two wetland types, a tracer test was performed. Results of the tracer test show that the soil marsh tested exhibited plug-flow. The gravel marsh tested also exhibited plug-flow characteristics, but with some short circuiting.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER I

INTRODUCTION

Agriculture, industry, and municipalities are constantly searching for more effective, cost-efficient methods of dealing with wastewater. One such method that is currently gaining attention is the use of wetlands. Reed and Calkins (1994) define wetlands as land in which the water surface is near the ground surface for a period long enough each year to maintain saturated soil conditions along with the related vegetation. Furthermore, they define a constructed wetland as "a wetland specifically constructed for the purpose of pollution control and waste management, at a location other than existing natural wetland."

As opposed to natural wetlands, constructed wetlands can be considered part of the wastewater treatment system instead of part of the receiving waters and thus not subject to certain laws and regulations (Reed, 1991). Three basic functions of wetlands that make them a potentially good wastewater treatment system (reproduced from Davis et al. 1992) are as follow:

1. physical entrapment of pollutants;
2. utilization and transformation of constituents by microorganisms;
3. low energy and low maintenance requirements to attain consistent treatment levels.

Brix (1993) claims that constructed wetlands may also be a better alternative to conventional secondary and advanced wastewater treatment systems for several reasons, as follow:

1. because they are low technology systems, they can be established and operated by relatively untrained personnel
2. low cost of construction
3. constructed wetland treatment systems are usually more adaptable to variations in loading than are conventional systems

Many constructed wetland studies have been conducted to determine the capabilities of these treatment systems. Constructed wetlands are primarily used as polishing systems (Rogers et al., 1995), therefore, the full potential of these systems has not been determined. Reed and Brown (1992) report that there is no definite consensus on certain wetland parameters such as water and media depth, plant and media type or system configuration. Knight et al. (1993) found that there are certain unanswered questions about constructed wetlands such as longevity of the system and effects of these systems on biota. Reed and Brown (1992) report that tracer studies have revealed that ideal plug-flow conditions do not occur in most constructed wetland systems. This short circuiting can limit the system's ability to trap pollutants and also reduces contact time, thus limiting treatment efficiency (Kadlec, 1995). Environmental conditions can also have adverse effects on wetland treatment efficiency. Kadlec (1995) states that evaporation can concentrate pollutants and cold temperatures can slow biological processes. The study conducted on the Cleveland, TN wetlands was developed to address some of these issues. The specific objectives of this study were as follows:

1. determine the treatment efficiencies of the existing wetlands;
2. compare the treatment efficiencies between the different wetland types;
3. compare the treatment efficiencies between test cells and the existing wetlands;
4. determine the effect of environmental conditions on treatment efficiency;
5. determine the flow patterns in the wetlands.

To address these issues, water quality parameters were recorded for eight months and data was analyzed on a mass removal rate basis. The water quality parameters examined were; nitrite, nitrate, TKN ammonia, TOC, BOD_5, phosphorous, sulfate, chloride and total solids. In addition, light and temperature measurements were recorded at the site.

CHAPTER II

REVIEW OF LITERATURE

A. Wetland Types

There are two main types of constructed wetlands for use in wastewater treatment. The first of these is a wetland in which wastewater flows over land through a constructed marsh (Gearhart et al., 1989). This type of wetland is known as a free water surface or a soil marsh (SM). The second type of wetland is one in which the wastewater flows through some type of media with a high hydraulic conductivity, usually gravel or sand (Brix, 1993). This type of wetland is referred to as a subsurface flow or gravel marsh (GM). Wetland plants are used in both systems to provide a rhizosphere for bacterial attachment and oxygen transfer (Gross et al., 1995).

Five principal components in constructed wetlands (reproduced from Hammer and Bastian, 1991) are, as follow:

1. substrates with various rates of hydraulic conductivity;
2. plants adapted to water-saturated anaerobic substrates;
3. a water column (water flowing in or above the surface of the substrate);
4. invertebrates and vertebrates;
5. an aerobic and anaerobic microbial population.

Reed and Brown (1992) hypothesize that one advantage of SM's over GM's is increased oxygen transfer rates due to the exposed water area. DeShon et al. (1995) found that a disadvantage of the GM wetland is its tendency to develop overland flow, or to mound, at the inlet. Construction cost may also vary with wetland type. Reed and Brown (1992) find an average construction cost for 19 existing SM wetlands of $55,000/ha, compared to an average construction cost for 18 existing GM wetlands of $215,000/ha. However, due to differences in system depth, they find if the cost of wetland construction is calculated on a volumetric basis, for SM and GM wetlands, the costs are $\$206/m^3$ and $\$163/m^3$, respectively.

In contrast, Reed and Calkins (1994) find that there are several advantages to the GM wetland over the SM. The first of these advantages is less risk of odor, mosquitoes and human contact, because the wastewater level is usually maintained below the media surface. The second of these advantages is the greater available surface area for treatment. This causes treatment responses to be faster, thus requiring less land area. The final advantage to GM wetlands is greater thermal protection in cold climates. The position of the water below the surface of the media and the accumulation of plant debris insulates the wastewater from cold weather.

In this work we addressed these issues by examining both soil and gravel marshes for treatment efficiency. No specific correlations were found relating to odor control due the close proximity of the wetland types to each other. At one point during the study organic loading was increased to the point where a slight odor was detected. However, odor differences between wetland types were indistinguishable. At no time during the study were mosquitoes found in excess. It is evident that human contact with the wastewater is more likely in the soil marshes than in the gravel marshes, however, the wetlands described in this study are on isolated, privately owned property, the waste was non-domestic in origin, and human contact was not considered to be a major concern.

B. Organic Loadings/Removal Rates

1. Units

Many different terms were used in the literature to describe wetland performance e.g., pounds removed per acre, percent removed, and concentration reduction. The most universal unit for wetland performance was determined to be on a mass of pollutant removed per unit surface area per unit time, or Areal Mass Removal Rate (AMRR), expressed as kg/ha/yr throughout this work. Whenever possible, values found in the literature were converted into these units. Outlet concentrations were multiplied by outlet flows, then subtracted from inlet concentrations multiplied by inlet flows to determine total mass change. In some cases only inlet flows were given. In these cases AMRR was calculated assuming that there were no net gains or losses of water from the system (inflow = outflow) via evapotranspiration or other mechanisms. Total mass change was divided by the total system surface area to obtain treatment per unit area. In order to determine how the system loading varied in the literature studied, an areal mass loading rate (AMLR) was calculated whenever possible. This was calculated by multiplying the inlet concentration by inlet flow and dividing by the area of the system.

2. BOD_5

Metcalf and Eddy (1991) state that constructed wetland BOD_5 loading must be limited so that oxygen demand does not exceed the oxygen transfer capacity of the system. Plants transport oxygen from the aerial leaves and shoots through the aerenchyma (gas transport passages) to the roots and rhizomes (Conley et al., 1991). Metcalf and Eddy (1991) state that oxygen transfer rates for emergent plants range from 18,000 to 164,000 kg/ha/yr, with an average value of 74,000 kg/ha/yr. Using oxygen criteria based on a 1.5 BOD_u: BOD_5 ratio, they find a maximum BOD_5 loading rate for a gravel marsh to be 49,000 kg/ha/yr. However, loading is concentrated at the inlet of the system, so to avoid anaerobic conditions they recommend limiting gravel marsh BOD_5 loading rates to 24,000 kg/ha/yr. In gravel marshes the plant roots are in contact with the flow of wastewater, thus making the oxygen available to organisms that degrade soluble BOD_5 (Metcalf and Eddy 1991). However, they state that in soil marshes, BOD_5 removal will be limited because the root zone is in the soil profile below the water column.

In a study conducted by Davis et al. (1992) in Mississippi, six cells measuring 4 m x 30 m (15 ft x 98 ft) and six measuring 4 m x 15 m (15 ft x 49 ft) were studied for treatment of two-stage dairy lagoon wastewater. Each long marsh was placed in series with a short marsh and planted with a variety of wetlands macrophytes. One system was left unplanted for use as a control. Using the Metcalf and Eddy (1991) oxygen criteria, they state that the upper limits of BOD_5 loading rates for this system should be between 21,900 and 25,550 kg/ha/yr with a 5.5 day retention time. During the two-year study, they loaded the cells at much lower rates than the calculated maximums: BOD_5 loading ranged from 3,900 kg/ha/yr to 19400 kg/ha/yr. First stage BOD_5 concentration reduction was poorest at the greatest loading (16%), and best (64%), at intermediate loading. This characteristic may be an indicator that for the greatest loading, the system was overloaded and began anaerobic transformations.

Reed and Brown (1994) found that in a California climate, gravel marshes and soil marshes could achieve high levels of treatment even at loading rates as high as 36,500 kg/ha/yr BOD_5. In a Southeastern climate, McCaskey et al. (1994) reported on organic loading rates as high as 43,500 kg/ha/yr BOD_5, still the results show an 87% reduction in BOD_5 concentration. These studies were, however, conducted on swine lagoon wastewater and in different climates than were used in our study.

Knight et al. (1993) compiled loading and removal data from several existing wetlands. The following table (Table 1) lists the BOD_5 results for wetlands less than or equal to 1/2 hectare in size. The AMLRs range from 936 to over 50,000 kg/ha/yr. Even with the maximum of these loadings a 61% treatment efficiency was achieved. However, the wetland type that achieved this removal was a vegetated submersed bed, (VSB) and was not further described. Values in Table 1 were calculated using inlet and outlet concentrations, flow rates and system areas. Calculations made for AMLR should therefore be accurate. However, no adjustments were made for system losses through ET or other mechanism, making AMRR and percent removed rough estimates. A kinetic chart was produced using the Knight et al. (1993) data in order to graphically represent the relationship between AMLR and AMRR. The data exhibits first order kinetics (R^2 =0.91). Three data points, all from the Leaf River Pond wetlands, deviate rather extremely from this relationship. For this system, BOD_5 concentration reduction averaged just over 1 mg/L. This could be due to a short HRT or some other unknown factor about the system. The wastewater type used in the Leaf River pond study was

Table 1: BOD_5 removal data compiled from Knight et al. (1993)

A	B	C	D	E	F	G	H
	TYPE	AREA	FLOW	BOD_5 IN	BOD_5 OUT	AMLR	AMRR
NAME		(ha)	(m3/day)	(mg/l)	(mg/l)	(kg/ha/yr)	(kg/ha/yr)
Leaf River Pond 1	SM	0.13	225	15.8	14	9981	1137
Leaf River Pond 2	SM	0.13	254	15.8	15.7	11268	71
Leaf River Pond 3	SM	0.13	220	15.8	13.9	9760	1174
Cobalt	SM	0.1	49	20.7	4.6	3702	2879
Shelbyville	SM	0.16	250	73.6	31.8	41975	23839
Gustine 1A	SM	0.39	163	130	49.8	19832	12235
Gustine 1B	SM	0.39	82	130	26.8	9977	7920
Gustine 1C	SM	0.39	41	145	24.2	5564	4635
Gustine 1D	SM	0.39	164	141	30.5	21642	16960
Gustine 2A	SM	0.39	174	151	44.8	24590	17294
Modna Basin	SM	0.15	57	52.8	17.7	7323	4868
Listowel #1	SM	0.34	17	19.6	8	358	212
Listowel #2	SM	0.09	17	19.6	11.3	1351	572
Listowel #3	SM	0.13	17	19.6	7.6	936	573
Listowel #4	SM	0.13	17	56.4	9.6	2692	2234
Listowel #5	SM	0.094	17	56.4	14.6	3723	2759
Phillips High School	VSB*1	0.2	58.7	15.3	1	1639	1532
Monterey	VSB	0.023	83	38	15	50053	30295
Ann Arundal Co.	HYB*2	0.385	464	13.3	9	5851	1892

*1 Vegetated Submersed Bed
*2 Free Water System Hybrid

Columns A - F reproduced from Knight et al. (1993)
Columns G - H are calculated values

industrial and only had a BOD_5 concentration of 15.8 mg/L. Flow rates into this system were higher than for any other system in this database (both in liters/day and in liters/ha/day). When these points are removed from the chart, it enhances the linear relationship even further with an R^2 value of 0.98 (Figure 1.1).

Tennessee agriculturists are concerned with BOD_5 output from the constructed wetlands since excessive BOD_5 to receiving waters can harm aquatic populations. BOD_5 is also a good indicator of system performance, therefore, it is widely studied in the literature. Finally, we were concerned with BOD_5 because the client for whom the study was performed, discharges directly into the city sewer and could be fined if organic loadings become too high.

3. Nitrogen

In a study conducted on natural and constructed wetlands in Florida, Gale et al. (1993) found that the major nitrogen removal mechanism appeared to be nitrification/denitrification. The process of nitrification requires oxygen to be present. Reed and Brown (1994) state that the conversion of ammonia to nitrate, by the process of nitrification, may be limited in gravel marshes due to low oxygen availability. They believe this is due to the lack of penetration by the root/rhizome into depths below 0.3 meters. In a Santee, CA pilot study, *Scirpus* roots penetrated to the bottom of a 0.76 m deep bed and ammonia removal of 94% was achieved (Gersberg et al., 1985). Davis et al. (1992) also agreed that oxygen availability should be a deciding factor in ammonia loading. Using the Metcalf and Eddy (1991) oxygen criteria they determined an ammonia loading rate of 6,570 kg/ha/year for 12 soil marshes in Newton, Mississippi. Only once during the study did ammonia loading reach this level, and during this period,

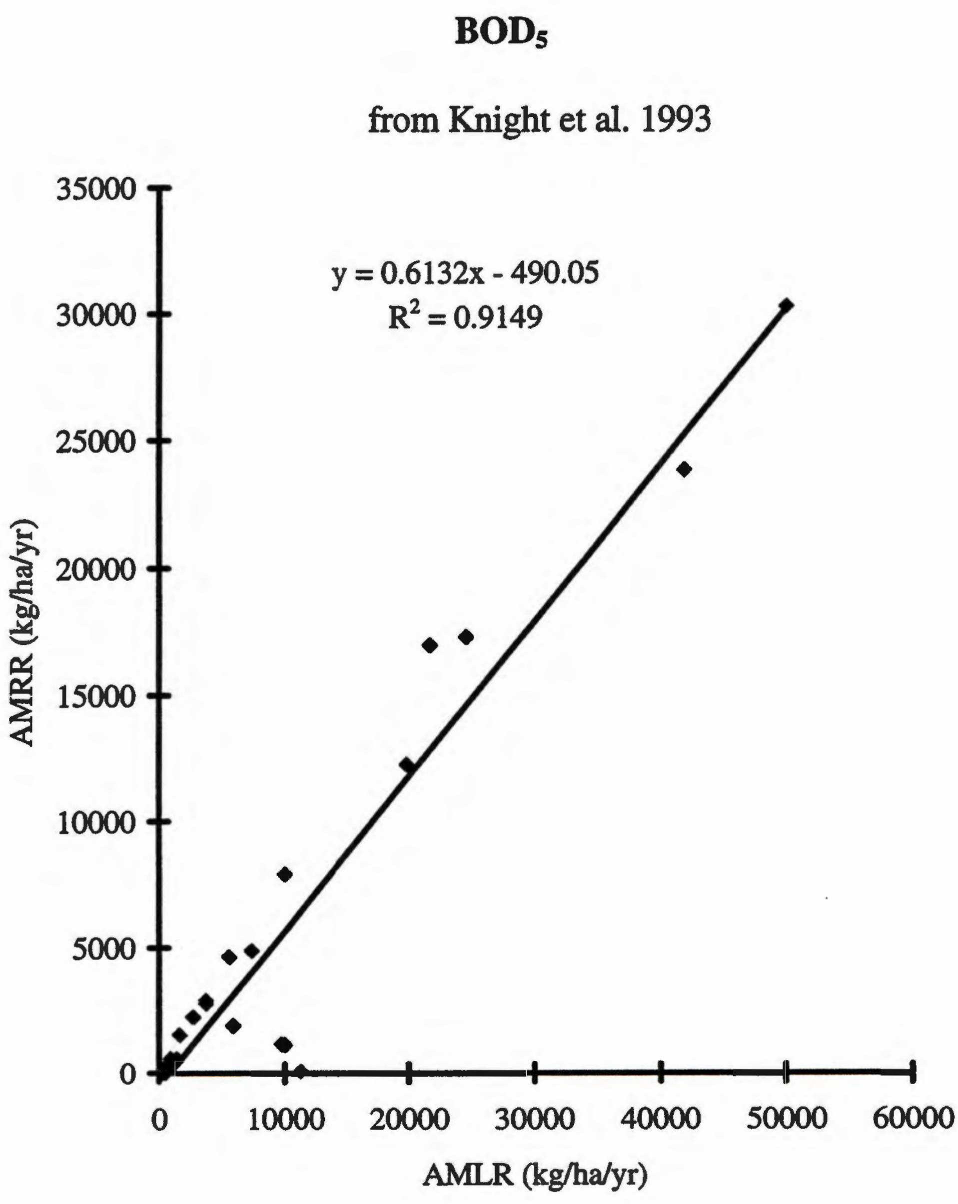

Figure 1.1. BOD_5 removal kinetics calculated using data from Knight et al. (1993)

there was very little ammonia concentration reduction and outlet concentrations were highest.

In loading experiments on a 3.2 m^2, horizontal flow gravel cell, Watson and Danzig (1993) used a wastewater with low ammonia concentrations (7.1 to 14 mg-N/L) to determine nitrification capabilities. The inlet and outlet concentrations, and hydraulic loading data were used to calculate the following values for AMLR (Table 2). The system capacity of 3.2 m^2 was multiplied by the media depth (0.3m) and then multiplied by the estimated porosity (.32) in order to determine system capacity. System capacity was divided by flow in order to estimate a system hydraulic retention time (HRT). No out-flow quantities were given so system losses were assumed to be zero.

Table 2: Ammonia removal data from Watson and Danzig (1993).

Ammonia						
Inlet (mg/L)	Outlet (mg/L)	Flow (liters/day)	Capacity (liters)	HRT (days)	AMLR (kg/ha/yr)	AMRR (kg/ha/yr)
10	8.6	117	300	2.6	1353	74
7.6	3.3	117	300	2.6	1029	227
6.8	0.3	42	300	7.1	334	46
7.8	1.3	322	300	0.9	2900	2593
7.1	0.1	109	300	2.7	897	324
7.4	0.8	234	300	1.3	2003	1389
9.9	0.01	54	300	5.5	618	111
10	1.1	450	300	0.7	5205	6958
10.7	1.6	360	300	0.8	4456	4553
11	10	1673	300	0.2	21266	10785
8.1	7.5	862	300	0.3	8071	1719
11	3.1	347	300	0.9	4417	3675
14	4.5	309	300	1.0	4997	3492
14	8	425	300	0.7	6871	4170
13	6	412	300	0.7	6187	4574

The following kinetic graph (Figure 2.1) was generated in order to better understand the changes that occurred in AMRR with the changes in AMLR. The relationship appears to be linear. One noticeable outlier occurred the week following the maximum organic loading rate (OLR) and hydraulic loading rate (HLR), when HRT had fallen to less than one-half day.

Knight et al. (1993) compiled ammonia loading and removal data for several wetlands. The results of their findings for wetlands less than or equal to 0.5 hectare are listed in Table 3 (0.5 maximum was chosen by us to more closely represent the wetlands in our study) This table was calculated using inlet and outlet concentrations, flow rates and system areas. Calculations made for AMLR should therefore be accurate. However, no adjustments were made for system losses via ET or other mechanism. Therefore AMRR and ultimately percent removed are rough estimates. Ammonia mass removal rates for the soil marsh systems appear to be low compared to the Watson and Danzig (1993) test gravel marsh system. The maximum AMRR for all of these soil marshes is 2,550 kg/ha/day. The Watson and Danzig (1993) study observed one AMRR as high as 10,000 kg/ha/yr and several AMRR in the 4,000 to 6,000 kg/ha/yr range.

Hammer and Knight (1994) summarized results from a Lakeland, Florida gravel marsh. The TKN removal rates and efficiencies are listed in Table 4. The results show very high treatment efficiencies for TKN. However, the loading rates are very low compared to others found in the literature.

Reed and Brown (1992) cited previous studies to determine the role of plants in the removal of nitrogen in constructed wetlands. These studies found that harvesting of plants from wetlands accounted for less than 10% of the nitrogen removed from the system. On the other hand, using vegetated and non-vegetated pilot cells, Rogers et al. (1991) reported average nitrogen removals of 50% for the non-vegetated buckets and

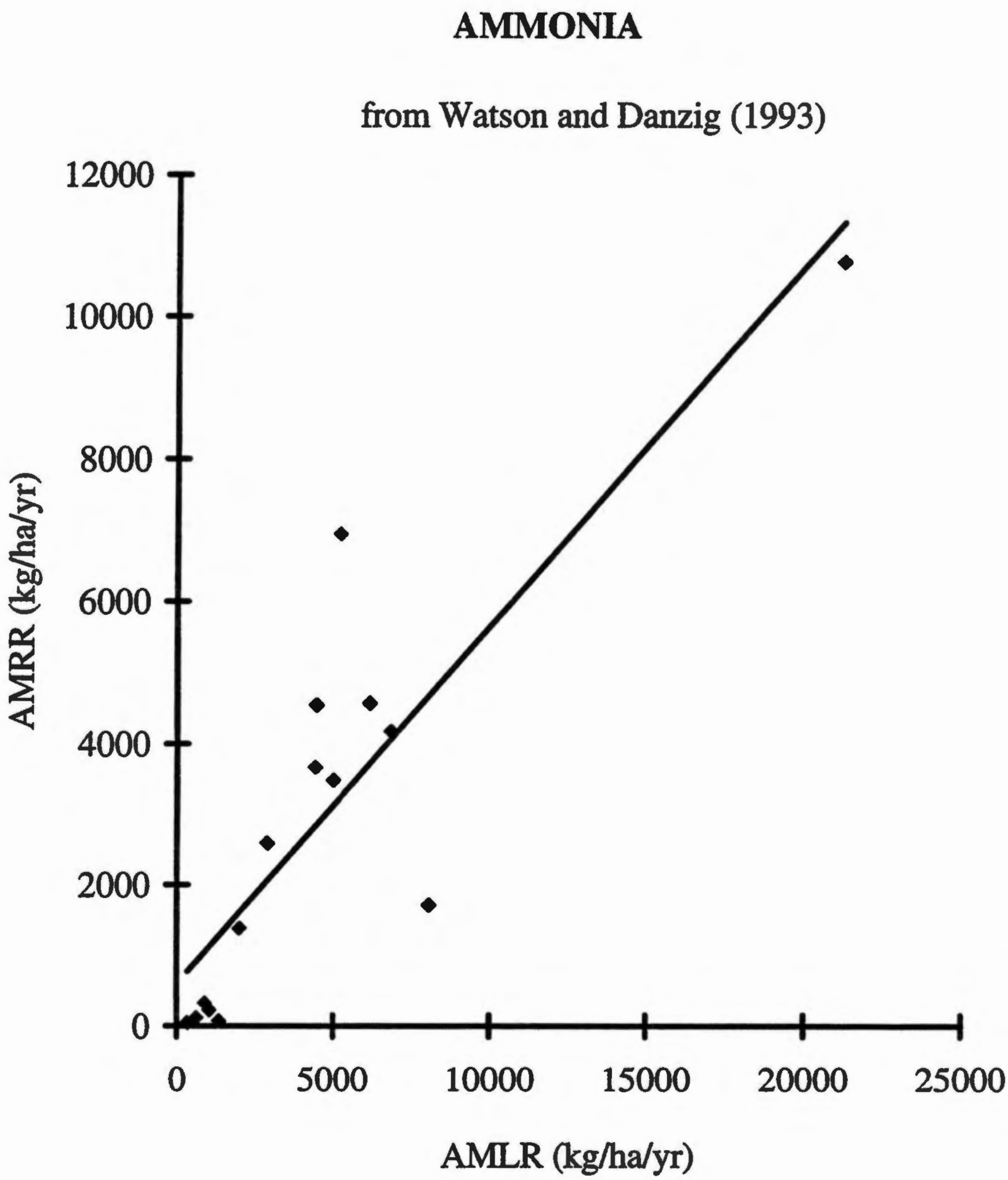

Figure 2.1. Ammonia removal kinetics calculated using data from Watson and Danzig (1993)

Table 3: Ammonia removal data compiled from Knight et al. (1993)

A	B	C	D	E	F	G	H
	TYPE	AREA	FLOW	NH^3 IN	NH^3 OUT	AMLR	AMRR
NAME		(ha)	(m^3/day)	(mg/l)	(mg/l)	(kg/ha/yr)	(kg/ha/yr)
Clermont L	SM	0.2	4.44	1.7	0.2	14	12
Clermont M	SM	0.2	11.05	1.7	0.3	34	29
Clermont H	SM	0.2	27.4	1.7	0.3	84	71
Leaf River Pond 1	SM	0.13	225	9.9	7.2	6260	1706
Leaf River Pond 2	SM	0.13	254	9.9	6.3	7067	2553
Leaf River Pond 3	SM	0.13	220	9.9	6.8	6121	1927
Cobalt	SM	0.1	49	3.0	1.0	528	342
Shelbyville	SM	0.16	250	6.8	2.9	3901	2264
Gustine 1A	SM	0.39	163	17.0	16.1	2599	143
Gustine 1B	SM	0.39	82	16.3	17.9	1252	-122
Gustine 1C	SM	0.39	41	18.5	20.4	708	-75
Gustine 1D	SM	0.39	164	19.7	22.9	3016	-499
Gustine 2A	SM	0.39	174	18.0	23.2	2931	-847
Modna Basin	SM	0.15	57	20.4	11.4	2829	1248
Listowel #1	SM	0.34	17	7.2	4.9	130	42
Listowel #2	SM	0.09	17	7.2	5.1	493	141
Listowel #3	SM	0.13	17	7.2	3.8	341	161
Listowel #4	SM	0.13	17	8.6	6.1	410	117
Listowel #5	SM	0.094	17	8.6	7.9	566	44
Phillips High School	VSB*1	0.2	58.7	11.0	1.7	1178	996
Monterey	VSB	0.023	83	9.3	8.7	12289	869
Ann Arundal Co.	HYB*2	0.385	464	9.7	6.5	4263	1412

*1 Vegetated Submersed Bed
*2 Free Water System Hybrid

Columns A - F reproduced from Knight et al. (1993)
Columns G - H are calculated values

98% for the vegetated ones. However, Crites and Tchobanoglous(1993) note two problems with the results of Rogers et al.'s (1991) study. The first problem is that the plants used were immature and in maximum growth phase. Therefore they were

Table 4: TKN removal data compiled from Hammer and Knight (1994)

Inlet Concentration (mg/L)	Outlet Concentration (mg/L)	AMLR (kg/ha/year)	AMRR (kg/ha/year)	Removal Efficiency (%)
1.91	1.17	43.8	18.3	43
3.14	1.69	58.4	25.6	47
3.23	1.51	62.1	47.5	74
2.97	1.29	69.4	43.8	64
2.47	1.31	58.4	40.2	71

competing for available nitrogen at a rate that would not be maintained in a mature wetland. Secondly, Rogers et al.'s (1991) measurements of plant uptake of nitrogen was made by harvesting the entire plant, which is not a practical method of nitrogen removal. Crites and Tchobanoglous (1993) estimate that only 33% of the total nitrogen is in the harvestable aerial shoots.

For the Cleveland study, mature plants from the existing wetlands were transplanted into the test cells. The root systems expanded rapidly, and shoots filled the entire cell in a matter of weeks. There were no apparent advantages in cell performance during these weeks of rapid growth and later in the study when the plants were more mature.

4. Phosphorus

Davies and Cottingham (1993) reported that phosphorus could be lost into the air in the form of phosphine, but only to a limited extent. Brix (1993) states that phosphorus loss into the air is unlikely due to thermodynamic reasons. Debusk et al. (1995) state that one way to enhance phosphorus removal in a constructed wetland is to amend the soil with a compound such as iron or aluminum which promotes phosphorus adsorption. Higgens et al. (1993) studied constructed wetlands treating cropland run-off. During two consecutive spring study periods, the authors recorded exporting more phosphorus from the system than was imported. They attribute this effect to the decay of plants and biota in the system. For our study, phosphorous results were not reported because the wastewater tested had very low concentrations of phosphorous in all of the samples tested.

C. Hydraulic Loading

The pollutant removal efficiency of a wetland is determined by several factors. One of these factors is hydraulic retention time (HRT). Since the HRT of a cell is determined by the amount of flow into a system, hydraulic loading is an important factor in pollutant removal. Tanner (1994) found that removal of BOD_5, total nitrogen (TN) and total phosphorus (TP) were positively correlated with HRT. Gale et al. (1993) stated that a minimum five day HRT was required for 50% reduction in nitrogen load. However in a study they conducted in Florida, constructed and natural wetlands consistently failed to remove 50% of the nitrogen load in five days.

In order to estimate the hydraulic retention time (days) of a system, the system flow rate (cubic meters per day) was divided by volume (cubic meters). However, this estimate assumes no dead space, and ignores the system losses, such as ET. Dead spaces cause short-circuiting and can greatly reduce effective HRT. In order to determine if the system in our study exhibited plug-flow, or if short circuiting was occurring, a tracer test was performed.

D. Effect of Plant Species on Treatment Efficiency

In a study conducted by Gersberg et al. (1985) four systems with different plant species were compared to determine treatment efficiencies. They found bulrush to consistently outperform reed and cattail for both ammonia and BOD_5 reduction. A graphical illustration of their results is presented in Figure 3.1. In contrast, Davis et al. (1992) found no pronounced BOD_5 reduction difference in six constructed wetlands used in treating dairy wastewater. These cells were planted with a variety of wetland plants including: cattails (*Typha latifolia*), bulrush (*Scirpus validus*), pickeral weed (*Pontoderia cordata*), arrowhead (*Sagittaria latifolia*) and maidencane (*Panicum hemitomen*).

Surrencey (1993) found that bulrushes tolerated high ammonia levels (160 to 170 mg-N/L) whereas cattails showed severe stress. He also states that giant reed (*Phragmites* sp.) was intolerant of the fifteen cm water level which was maintained in the cells.

Some wetland plants may be more efficient at treating one pollutant and less efficient at another. Kuehn and Moore (1993) reported on results from the Pope and

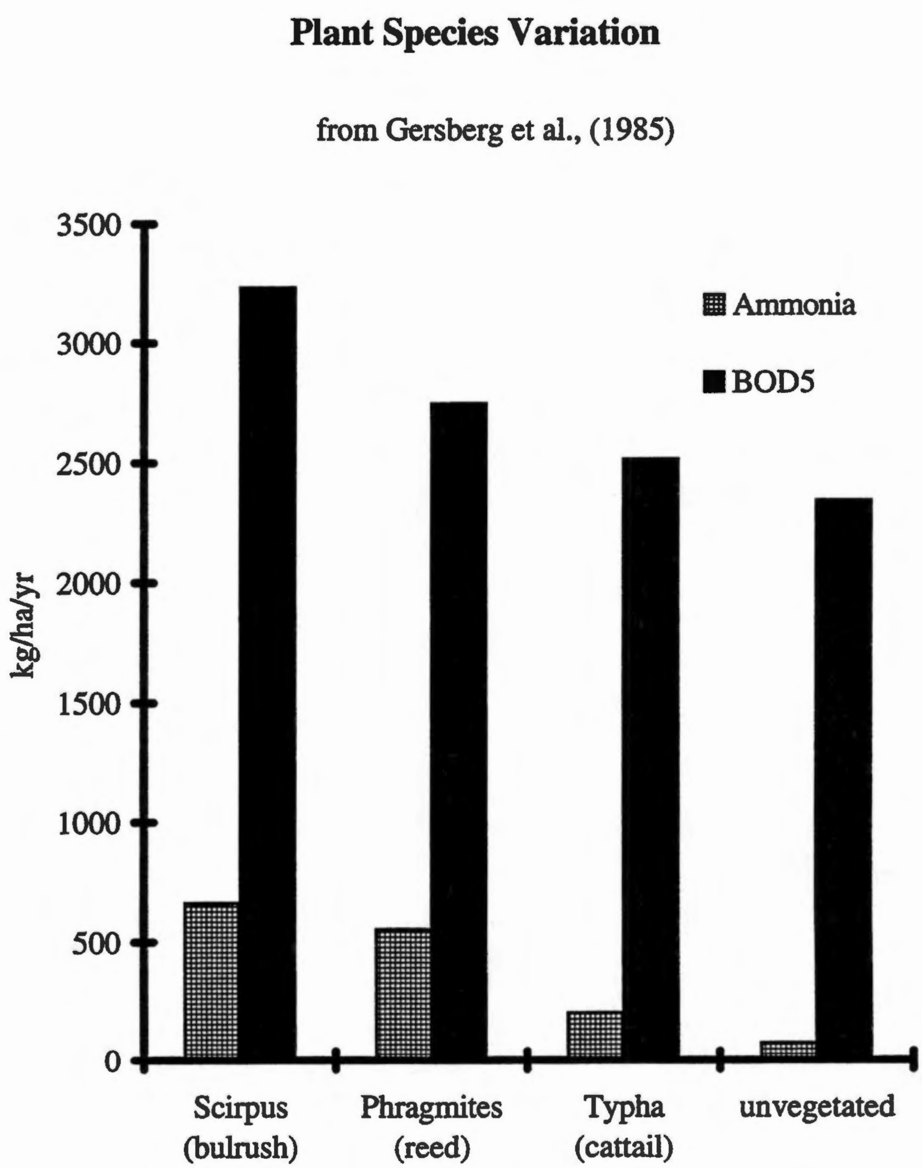

Figure 3.1. Ammonia and BOD_5 removal efficiencies reported by Gersberg et al. (1985)

Talbot wetlands in Hasley, Oregon. In this study, replicate wetlands (treating pulp mill effluent) were designed for comparison purposes. They found that BOD_5 was reduced 27.1% in bulrush ponds, while only 17.8% in cattail ponds. Inversely, TSS removals were 39.0% in the bulrush ponds and 53.7% in cattail ponds. For their study, both BOD_5 and TSS inlet concentrations were below 30 mg/L.

Vegetated wetlands appear to be more efficient at removing waste than unvegetated wetlands, perhaps, because of the lower oxygen transfer in an unvegetated cell (Watson and Danzig, 1993). They found that in three identically constructed and loaded vertical flow cells, the two that were established with reeds provided slightly higher treatment efficiency of ammonia nitrogen than the third, unvegetated cell. This difference could also be attributed to the lack of microbial attachment sites (MAS) in an unvegetated cell. Benham (1995) found that cells with inert MAS (wooden dowels) removed total organic carbon (TOC) at the same rate as unvegetated cells. However, this study was conducted indoors using artificial light. With artificial light, which was much less intense than normal sunlight, plant functions may have been reduced.

For our study, four different wetlands and a water garden were evaluated for treatment efficiency. The primary vegetation type in soil marsh 1 and soil marsh 2 was hard stem bulrush (*Scirpus acutus*) and narrow leaf cattail (*Typha angustifolia*), respectively. Gravel marsh 1 and gravel marsh 2 were planted primarily with broad leaf cattail (*Typha* sp.) and reed (*Phragmites*), respectively.

E. Pilot Studies

In order to test the feasibility of constructed wetlands for use in wastewater treatment it may be necessary to first conduct a pilot study (Metcalf and Eddy, 1991).

By doing this researchers can answer important questions but also save resources that would otherwise be spent by starting off with full-scale operation.

Watson and Danzig (1993) conducted a study to determine design information for a full-scale constructed wetlands treatment system. The study was conducted at the municipal wastewater treatment facility located in Benton, Kentucky. Six septic tanks were modified for use as tertiary "polishing" units to treat effluent from an existing wetland. Five of the cells were set-up for vertical flow while one was horizontal subsurface flow. Their results showed that ammonia removal efficiencies on a mass loading basis were not nearly as high in the horizontal flow cell than in the vertical cells. They attribute this to less oxygen availability due to saturated conditions and less surface area.

Rogers et al. (1991) used twenty, 25 L buckets for up-flow and down-flow wetland test cells. Each cell was filled with 32 kg of 3-7 mm diameter washed gravel. The operating volume of the cells was 22 L with 32% porosity. Twelve of the buckets were planted with club rush (*Scirpus* sp.)and eight of the buckets were left unplanted. The twenty cells were then divided in half and set-up for either up-flow or down-flow (six planted and four unplanted for each). The results showed that unplanted buckets were inefficient at nitrogen removal and became overloaded quickly.

Surface et al. (1993) tested four parallel constructed wetlands installed at the Tompkins Co. municipal solid waste landfill located near Ithaca, New York. Cell dimensions were 3m by 30m and filled to a 0.6 m depth of assorted sand and gravel sizes. The purpose of this study was to determine the efficiency of landfill leachate by constructed wetlands.

One goal of our study was to determine if wastewater treatment efficiency in full-size constructed wetlands could be predicted using test cells. If so, experimentation could be performed on test cells, before making loading changes in the full-size marshes.

F. Harvesting

Reed and Brown (1994) suggest that harvesting is not necessary when dealing with species such as *Scirpus*, *Typha* or *Phragmites*. van Oostrom and Russel (1994) state that unless plants are harvested and biomass removed, nitrogen removal via plant uptake from constructed wetlands will not normally exceed 1,826 kg/ha/yr. Debusk et al. (1995) state that one method of enhancing phosphorus removal in a constructed wetland is by harvesting the plant material. They do, however, acknowledge that this is a more intensive and costly management technique than is normally associated with constructed wetlands. If harvesting is to become part of a management regime for constructed wetlands, Debusk et al. (1995) suggest using floating macrophytes, since they effectively remove pollutants such as phosphorus from wastewater and can be easily harvested. In our study, the systems are young (three years) and no attempts were made at harvesting the vegetation.

G. Evapotranspiration

Surface et al., 1993 calculated ET in gravel marshes planted with *Phragmites austrailus* with the following formula: ET = inflow + precip - outflow. The results are presented in Table 5.

Table 5: Evapotranspiration data compiled from Surface et al. (1993)

	ET mm (in)		
	July 27 - Oct 9 1990	Oct 10 - April 30 1991	May 1 - Aug 1 1991
Bed 1 (coarse gravel, planted)	297 (11.7 in)	208 (8.2 in)	485 (19.1 in)
Bed 2 (sand and gravel, unplanted)	287 (11.3 in)	43 (1.7 in)	292 (11.5 in)
Bed 3 (sand and gravel, planted)	246 (9.7 in)	44 (1.4 in)	221 (8.7 in)
Bed 4 (pea gravel, planted)	112 (4.4 in)	208 (8.2 in)	282 (11.1 in)

Netter (1994) reported ET losses in a primarily reed inhabited gravel marshes from 0 to 88% of the inflow. Using a lysimeter, Allen et al. (1992) recorded maximum and minimum ET rates in cattails of 15.0 and 6.1 mm per day. They reported also maximum and minimum ET rates in bulrush marshes 15.0 and 7.9 mm per day.

Evapotranspiration, which decreases wastewater volume has a concentrating effect on pollutants. Conversely, an increase of water volume, due to precipitation can dilute pollutants. Ignoring either of the above effects yields inaccurate AMRRs. In order to address this issue, hourly weather data were collected to be used in estimating ET.

H. Weather Effects

Rogers et al. (1995) reported that pollutant treatment efficiencies at the Sand Mountian Experiment Station in Alabama seem to remain constant even after the frost killed the wetland plants. He further states that removal efficiencies did not drop until the water temperature fell to the point of inhibiting microbial growth. Holmes et al. (1994) reported on constructed wetlands used for treating milk-house wastewater in a cold climate (south of Green Bay, Wisconsin). Problems with wetland design limited some of the results; however, average concentrations for BOD_5, TSS and TP were reported (Table 6).

Table 6: Concentration discharge from Holmes et al. (1994)

	BOD_5 (mg/L)	TSS (mg/L)	TP (mg/L)
Inlet	168	509	16.9
First Cell Discharge	14	37	8.1
Second Cell Discharge	10	80	5.8
Third Cell Discharge	17	200	2.8

Seasonal variations may be more prominent with specific types of wastewater than with others. For bulrush and cattail wetlands operated at two day HRT, Kuehn and Moore (1993) reported highly significant seasonal BOD_5 variation with poorer performance in summer months. The authors speculate that this could be attributed to high influent temperatures of pulp mill wastewater which increased anaerobic conditions.

The effects of cold weather were of particular interest in this study, in order to determine whether loading rates should vary between summer and winter. The plant currently distributes wastewater to the wetlands year round. Before reaching the wetlands the wastewater is held in outdoor aeration basins for several days, therefore, heat pollution is not a concern.

CHAPTER III

MATERIALS AND METHODS

A. Site Description

In September, 1993 the M&M Mars processing plant located in Cleveland, Tennessee began using four constructed wetlands and a water garden to treat effluent from its wastewater treatment plant. Plant effluent consists primarily of wash-water from cleaning and sanitizing operations in the plant. Under normal operation, the primary wastewater facility produces approximately 190-300 m^3 (50,000 to 80,000 gal) of wastewater per 16 hour day, along with sludge which is composted off-site. During the course of this study, approximately two-thirds of this water was discharged directly into the city sewer, while the other third was diverted into the wetlands for secondary treatment.

The effluent distributed to the wetlands is pumped by a centrifugal pump through 15 cm (6 in) diameter PVC pipe into a splitter basin/flow measurement station. This basin consists of five adjustable weirs, one for each of the four wetlands and one that can send water directly into the water garden. During this study, no water was diverted directly into the water garden. After discharge from the splitter basin, the wastewater travels by gravity flow through 10 cm (4 in) diameter PVC pipe into a distribution manifold in each wetland. The distribution manifold consists of a 10 cm (4 in) diameter aluminum pipe which has discharge ports every 1.2 m (4 ft). Water depth in the cell can be varied by adjustment of the discharge standpipe.

There are two soil marshes (SM1 and SM2) and two gravel marshes (GM1 and GM2) at the site; soil marshes have overland flow, while the gravel marshes are subsurface flow. Each gravel marsh receives one third of the total flow, while the soil marshes split the remaining third of the flow, thus receiving one-sixth of the total flow each. The soil marshes are 61 m long by 12 m wide with a surface area of 0.073 ha. (200 ft long by 40 ft wide, 0.18 acre). The edges of the soil marshes are tapered at a 2:1 slope and 0.46 m (18 in) of topsoil fill the bottom of the marshes to enhance the growth of aquatic vegetation. The average water depth in SM1 is 7.6 to 10.2 cm (3 to 4 in) with exception of a 6.1m long deep zone excavated into the center of the cell. The average water depth of SM2 (excluding the deep zone) is 7.6 to 12.7 cm (3 to 5 in). The deep zone is approximately 1.2 m (4 ft) deep and was added in an attempt to enhance the habitation of aquatic life. Soil marsh 1 currently has an established colony of hard stem bulrush (*Scirpus acutus)* throughout the cell except inside the deep zone. Soil marsh 2 has an established colony of narrow leaf cattail (*Typha angustifolia*) except inside the deep zone. Rip rap gravel has been placed at the beginning and end of both cells to prevent the edges from degrading and also to prevent plugging of the inlets.

Each of the gravel marshes is 30.5 m long 61 m wide with a surface area of 0.19 ha. (100 ft x 200 ft, 0.46 acre). The sides of the gravel marshes are sloped at a 2:1 slope. A 0.46 m (18 in) layer of smooth-river rock with an average diameter of 10 to 15 mm (0.4 - 0.6 in) was added to the gravel cells for planting media and to provide the desired subsurface flow. Gravel marsh 1 has been planted with broad leaf cattail (*Typha* sp.) on the south side and yellow water iris (*Iris pseudocharis*) on the north. Gravel marsh 2 has been planted with reed (*Phragmites* sp.*)* on the south-side and soft stem bulrush (*Scirpus* sp.) on the north. During construction a large boulder was discovered in GM2. Additional clay packing was added to prevent leaching around the boulder.

After discharge from each of the wetlands, the wastewater travels by gravity flow through 10 cm (4 in) diameter PVC pipe into the water garden. The water garden is approximately 0.9 ha. (2.25 acres) in size. The depth of the water garden ranges from 0.9 m (3 ft) on the south side to 1.8 m (6 ft) on the north side. There are several types of aquatic vegetation currently in the water garden, including water lily (*Nymphaea odorata*), zebra rush (*Junicus gymnocarpus*), cattail (*Typha* sp), pickerel weed (*Pontederia cordata*), and fire flags (*Iris fulva*). Discharge from the water garden travels through 25.4 cm (10 in) diameter PVC pipe by gravity flow into the city sewer.

Initial calculations were made to estimate the hydraulic retention times (HRT) for the marshes and for the entire system (marshes + water garden), the results are given in Table 7. The minimum HRTs were calculated based on a maximum flow rate of 208 m^3 per day (55,000 GPD), while the average HRT was based on a flow of 550 m^3 per day (14,500 GPD). For the purpose of this estimation, rainfall and evapotranspiration losses were ignored.

Table 7: HRT calculations at the Cleveland, TN wetlands

	Minimum HRT (55,000 GPD)	**Average HRT (14,500 GPD)**
Soil Marsh	5 Days	19 Days
Gravel Marsh	4.5 Days	16.5 Days
Total System	40 Days	152 Days

B. Test Cell Description

The four 0.79 m^2 (8.5 ft^2) test cells were made of 19 mm (3/4 in) plywood, lined with fiberglass resin and matting (Benham, 1995). They were originally intended for indoor use only. To prepare them for outdoor use an additional layer of fiberglass resin was added to the exterior of each of the cells. In addition, the cells were insulated with 127 mm (5 in) of Styrofoam to minimize the effects of the exposed sides on cell temperature. A 19 mm (3/4 in) PVC pipe with eight drainage ports was attached to the head of each of the cells for use as an influent manifold. A perforated, 19 mm (3/4 in) PVC pipe, attached to an outlet was used to drain the test cells.

On 5/25/95 the cells were transported to the Cleveland, TN wetland site. The cells were placed approximately 0.6 m (2 ft) apart and leveled. Two of the cells, test soil marshes 1 and 2 (TSM1 and TSM2) were filled with 0.38 m (15 in) of soil taken from the south side of the M&M wetland location. The other two cells, test gravel marshes 1 and 2 (TGM1 and TGM2) were filled with 0.46 m (18 in) of polished river stone taken from M&M's gravel marshes. Measured amounts of water were added to the test gravel cell in order to determine the porosity of the media for future calculations (0.32). Plants from each of the full-size wetlands were removed and transplanted into the respective test cell. Within two weeks of establishment, new growth from transplanted vegetation had begun.

A circulating flow of water was maintained in the test cells to simulate the flow of water along the length of the full-scale cells using Teel ™ submersible pumps (model 2p873). In order to do this, water velocities in the full-size marshes were calculated and converted to flow rates to be used in the test cells ($v_x \cong 9.1$ m per day for the soil marshes and 4.6 m per day for the gravel marshes):

$$Qsm = 9.1\frac{m}{day} \times 0.07m^2 \times 1000\frac{L}{m^3} \times \frac{day}{1440\min}$$
$$Qsm = 0.44LPM$$

$$Qgm = 4.6\frac{m}{day} \times 0.08m^2 \times 1000\frac{L}{m^3} \times \frac{day}{1440\min}$$
$$Qsm = 0.26LPM$$

A segment of 15.2 cm (6 in) diameter perforated pipe was attached to the end of each cell to house a Teel™ submersible pump. Four submersible pumps were used to circulate the wastewater in the test cells. Since no low cost pump could be purchased to deliver low enough flow rates, the pump outlets were split, sending only a portion of the flow to the head of the test cell. Furthermore, the discharge end of the distribution hose could also be raised and lowered to fine-tune the circulation rates. Flow rates were measured using a graduated cylinder and stop-watch. After several weeks of operation the clear plastic hoses between the pump and the distribution manifold became clogged with algae; to prevent this from occurring a black plastic conduit was placed around the hoses.

C. Sampling

Water quality samples were collected weekly from the M&M site beginning on 12/8/94. A 1.5 m (5 ft) PVC extension arm was constructed in order to access the outlets. Samples were manually collected from the inlet weir, and from the outlets of each of the wetlands and the water garden. Grab samples were also collected from the deep zones of each of the two soil marshes. Samples were collected in 500 ml Nalgene™ sample bottles.

Test cell "final" samples were collected from the discharge outlet of the submersible pumps. The cells were then drained and refilled with wastewater from the inlet weir using two little giant™ (model 3e-12n 500 GPH) sump pumps. The Teel™ circulating pumps were then restarted and allowed to run for at least 30 minutes. After the minimum 30 minute period the test cell "initial" samples were collected from the discharge end of the Teel pump. This 30 minute sample lag was used to allow mixing of the new wastewater and any residual wastewater which remained after the draining period.

On each sampling day, quadruple samples were pulled from one location to assure quality control. Samples were iced and returned to The University of Tennessee water quality lab for immediate analysis. The final set of water samples was collected on 12/13/95 for a total of 48 sample sets. The first BOD_5 test was performed on 04/13/95. Therefore, the period of this study that will be reported is 04/13/95 through 12/13/95 for a total of 35 sample collections.

D. Weather Data

On 12/20/94 a Campbell Scientific 21X data-logger was placed at the site of the wetlands. Thermocouples were placed to allow the following temperature readings to be recorded:

- air temperature
- substrate and water temperature from the inlet of SM2
- substrate and water temperature from the outlet of SM1
- substrate and water temperature from TSM2

A quantum sensor photometer was used to measure light availability at the site (photosynthetically active radiation or PAR). Weekly rainfall data were recorded using a manual rain-gauge. Rainfall data were also obtained from the weather station at the Cleveland, TN filter plant for comparison purposes and to fill gaps in data. Percent possible sunlight, relative humidity and wind speed could not be obtained for the Cleveland, TN location. These readings were, however, collected from the National Weather Service in Chattanooga, TN for use in evapotranspiration (ET) calculation. ET was calculated using the Blaney-Criddle method (Cuenca, 1989). Wind speed values were collected at ten meters above the ground, so the log-wind law (Cuenca, 1989) was used to approximate wind speed at two meters. The Blaney-Criddle method was developed using grass as the reference crop. Allen et al. (1992) conducted ET studies on bulrush and cattail marshes using lysimeters. Their results showed that these wetland plants had ET rates that were 1.8 times higher than grass values determined using the

Blaney-Criddle method. The Blaney-Criddle results were multiplied by 1.8 to account for the increase in ET from wetland plants.

E. Lab work

Five day biochemical oxygen demand (BOD_5) was detected using a five day test as described in *Standard Methods for the Examination of Water and Wastewater* (Greenberg, 1992). Samples were diluted and decanted into a 300 ml BOD bottle with nitrification inhibitor and seed solution. On each run the BOD dilution water was checked, and a standard glucose-glutamic acid check was performed. Seed controls were conducted using 10, 15, 20 and 25 ml volumes in a 300 ml BOD bottle. Seed control solution was obtained from the addition of one polyseed BOD inoculum capsule in 250 ml of dilution water. Dissolved Oxygen (DO) levels were detected using an Orion™ model 860 O_2 meter. Samples were incubated for five days @ 20°C (+/- 0.5°C) in a Revco™ environmental chamber/incubator. Check samples and seed control were performed in conjunction with actual samples.

The inorganic ions nitrite (NO_2^-), nitrate (NO_3^-), chloride (Cl^-), ortho-phosphate (PO_4^-), and sulfate (SO_4^-) were determined using EPA method 300.0, which is the determination of inorganic ions using ion chromatography. Samples were run on a Dionex #2000I ion chromatograph after injection through a IONPac AS4A-SC analytical column.

Total Kjeldahl Nitrogen (TKN) was determined using Lachat Quik Chem method 10-107-06-2-D. 20 ml of sample was digested with sulfuric acid, potassium chloride and mercuric oxide. Samples were distilled at 160°C for one hour. Samples were then digested at 380°C for 2.5 hours. Samples were injected into the instrument and reacted with reagents to produce a blue color which was then analyzed at 660 nm to determine TKN concentration.

Ammonia (NH_3) concentrations were determined using Lachat Quik Chem method 10-107-06-1-C (USEPA, 1983). Samples were injected into the instrument and reacted with reagents to produce a blue color which was then analyzed at 630 nm to determine ammonia concentration.

F. Tracer Study

M&M's wastewater treatment plant uses aluminum chloride to remove solids and adjust the pH of the wastewater during primary treatment. Initial considerations for a bromide (Br^-) tracer test on the wetlands showed that the high concentration of chloride ions present in the wetlands would disrupt analyses for bromide ions. A bromide test could have been performed, but extremely large quantities of bromide would have been necessary to over-shadow the chloride. Such high quantities of bromide could have adversely affected the wetlands.

However due to an unusual condition at the wastewater treatment plant, the tracer test was conducted using M&M's waste stream. During the week of 8/24/95, the wastewater treatment plant experienced a disruption in its primary treatment system. While the system recovered, pure tap water was diverted to the wetlands for a period long enough to flush the system of wastewater thus decreasing the chloride concentrations by 100x. Monitoring of the wetland systems using Isco™ model 3700 samplers (set to sample every 6 hours) was performed during the no-load period and the restart period. Chloride ions were used as the tracer.

CHAPTER IV

RESULTS AND DISCUSSION

A. Tracer Study

Figure 4.1 illustrates chloride tracer concentrations for the duration of the tracer study. Day 0 is several days after the wastewater treatment plant (WWTP) plant began discharging tap water to the wetlands. Tap water samples were analyzed for chloride ions and all were below five mg/L. The wetland sampled, soil marsh 2 (SM2), appeared to be flushed of residual chloride ions after approximately 30 days from the start of the study. Grab samples pulled from gravel marsh 1 (GM1) showed that it had also been flushed of residual chloride. On day 62 of the tracer study, the WWTP began discharging wastewater to the wetlands. Multiple samples of the wastewater were analyzed for chloride ions, showing an average concentration of 400 mg/L. Samples were collected at the outlet of SM2 and GM1 every six hours. The sudden chloride increase from SM2 after five days indicates plug-flow (Figure 4.2). In contrast, pulses in chloride concentration from GM1 (Figure 4.3) indicates that some short circuiting occurred, though the overall pattern is primarily plug-flow.

B. Organic Loading

Because of the effectiveness of the WWTP, the average influent concentration of wastewater treated by the marshes is much lower than most of the previously cited

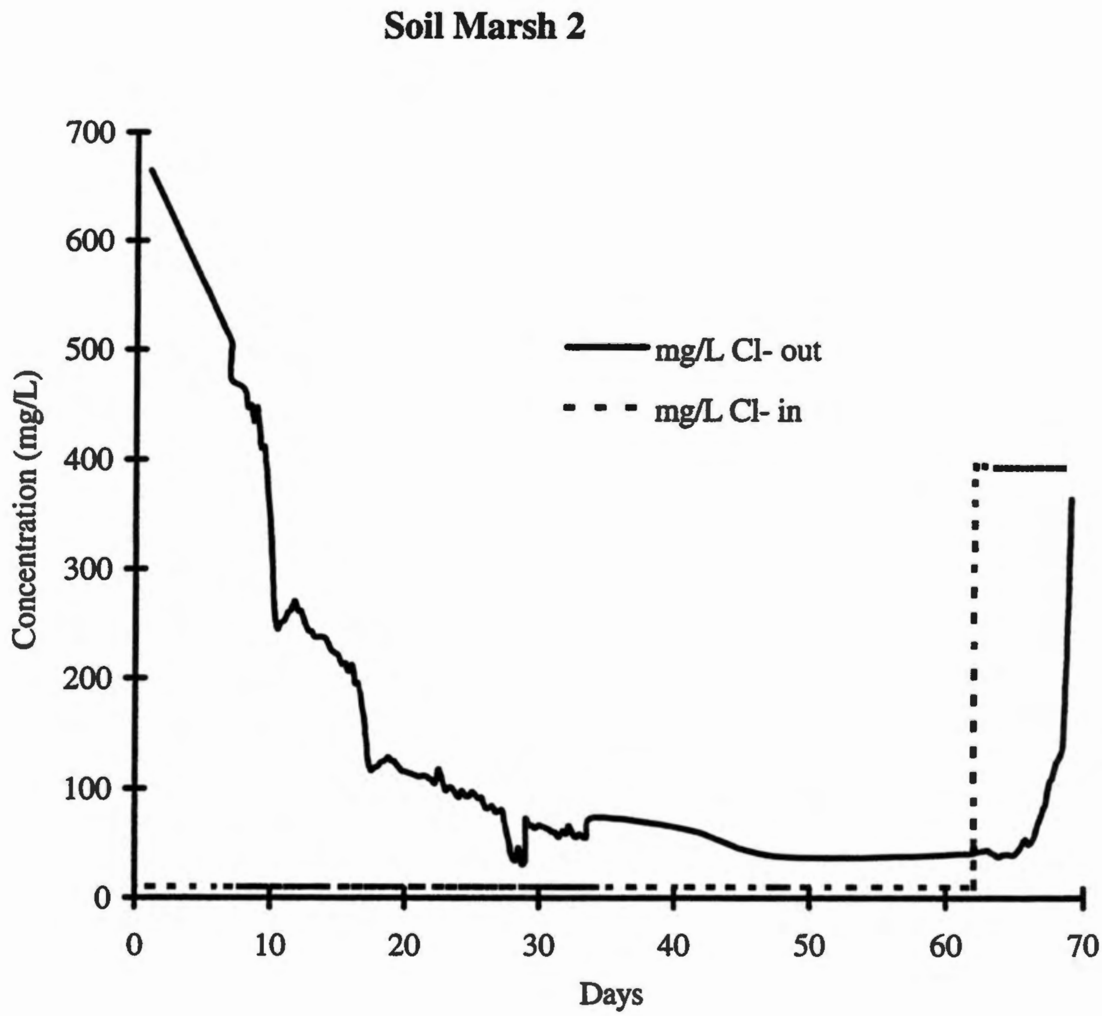

Figure 4.1. Soil marsh 2 inlet and outlet chloride concentrations during the entire tracer study

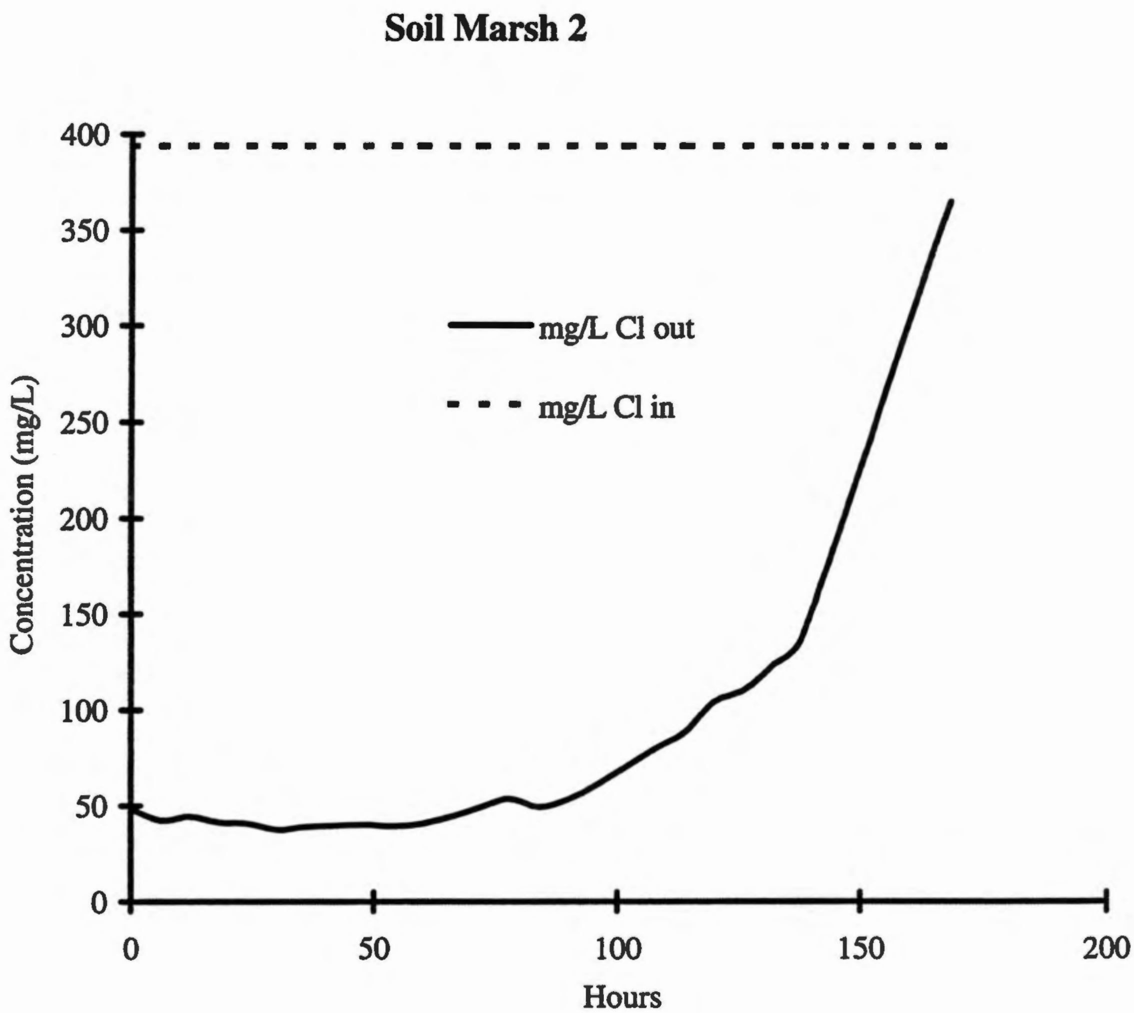

Figure 4.2. Soil marsh 2 inlet and outlet chloride concentrations during the load period of the tracer study

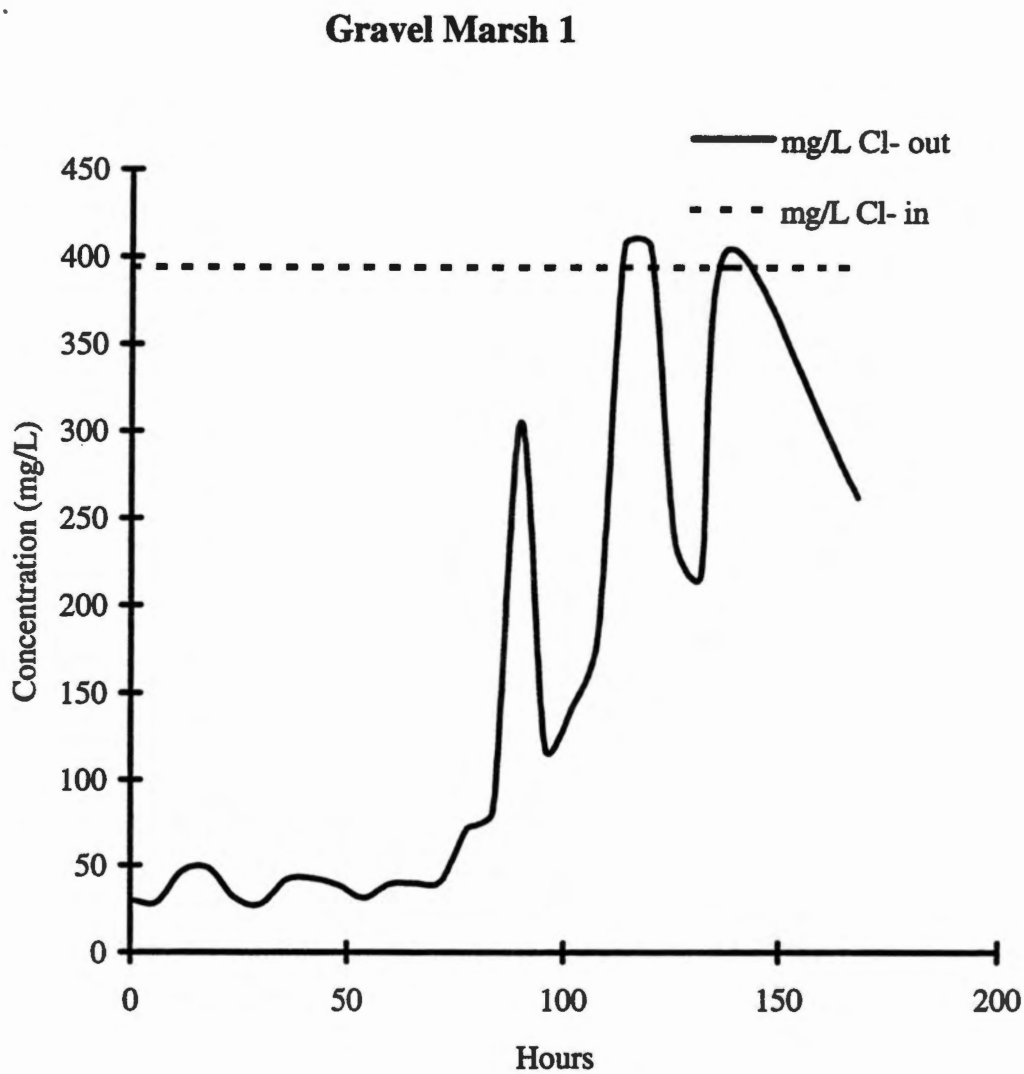

Figure 4.3. Gravel marsh 1 inlet and outlet chloride concentrations during the load period of the tracer study

studies. Table 8 lists the average concentration of measured pollutants from the inlet of the marshes, excluding no-load weeks.

Table 8: Average inlet concentration for the Cleveland, TN wetlands

Pollutant	Average Concentration (mg/L)
Nitrate	18.5
Ammonia	3.6
TKN	6.2
BOD	8.9

Gersberg et al. (1985) reported extremely low average nitrate plus nitrite concentrations of 0.3 mg-N/L in the Santee, CA study. The only values reported in the Cleveland study are for nitrate. However, for comparison purposes, these nitrate concentrations can be considered as nitrate plus nitrite since no detectable nitrite concentrations were found in the duration of the study. Watson and Danzig (1993) used effluent from existing wetlands for nitrification studies. The effluent used was described by the authors as organically "light" in ammonia and BOD_5. Ammonia concentrations for their study ranged from 7 -14 mg-N/L, while BOD_5 concentrations were between two and 40 mg/L.

Figures 5.1, 5.2, 5.3, and 5.4 illustrate average influent concentrations plotted against time. They show that the concentration of pollutants in the wastewater sent to the wetlands varied considerably during the study. Some of this could be attributed to normal fluctuations in WWTP performance. The major cause of this variation, however, can be attributed to the dilute nature of the wastewater. A variation of 10 mg/L BOD_5 in a wastewater which is normally 20 mg/L is an extreme variation. However, a 10 mg/L BOD_5 variation in a wastewater which is normally 200 mg/L is hardly noticeable.

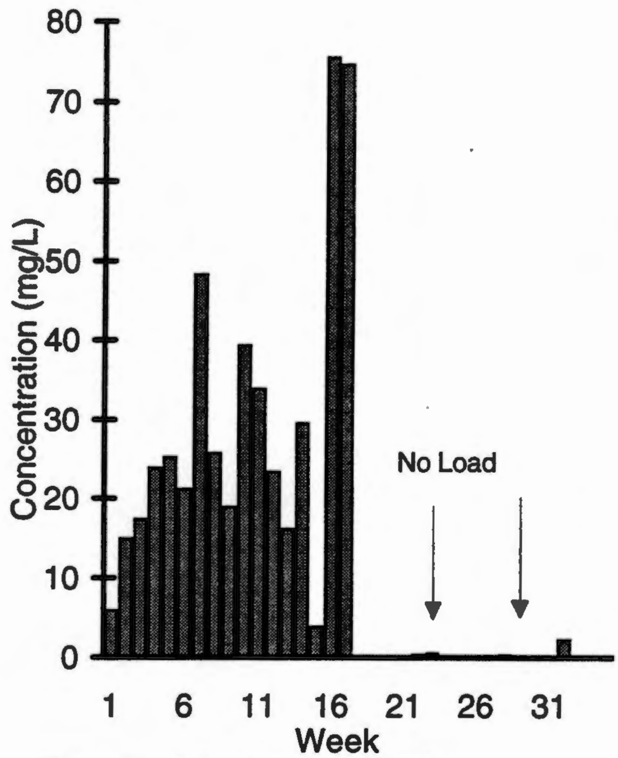

Figure 5.1. Inlet nitrate concentration

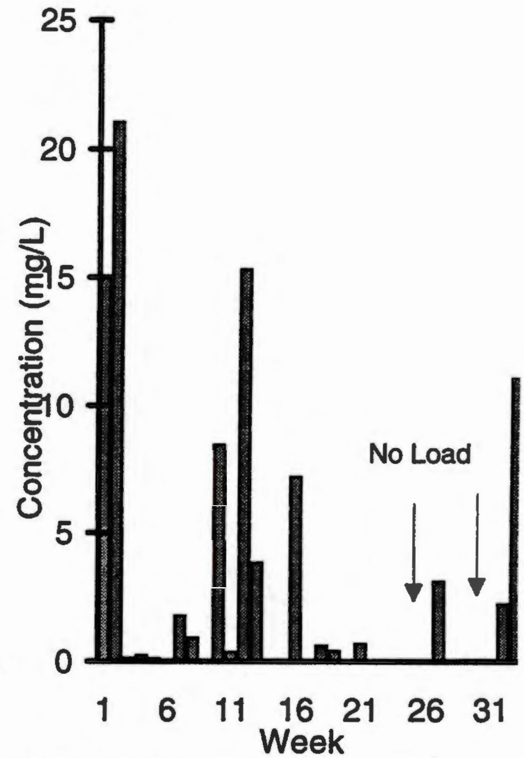

Figure 5.2. Inlet ammonia concentration

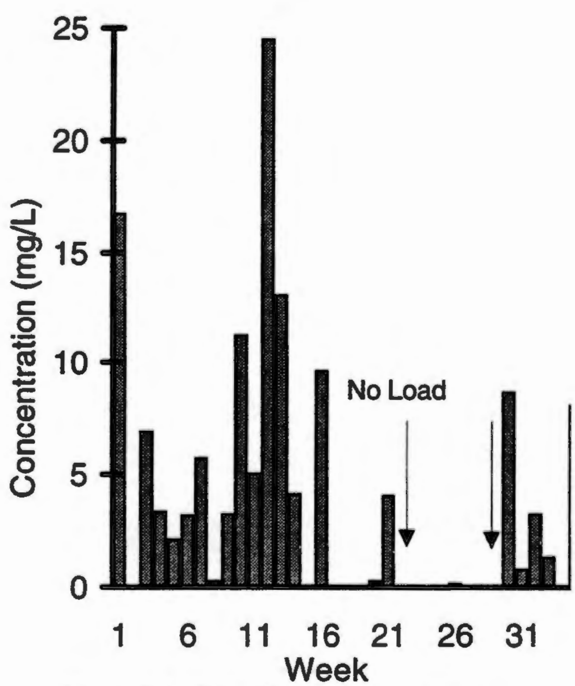

Figure 5.3. Inlet TKN concentration

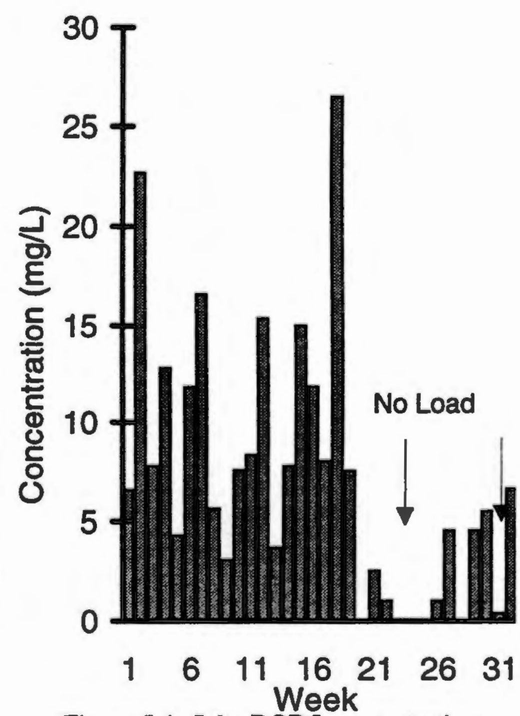

Figure 5.4. Inlet BOD5 concentration

C. Concentration Reduction

1. Full-size wetlands

Nitrate

Full-size wetlands and water garden average initial and average final nitrate concentrations are shown in Figure 6.1. Concentrations were reduced from 14.7 mg-N/L to 3.8 mg-N/L. There are no noticeable differences between soil marsh and gravel marsh reductions. Wastewater leaving the wetlands went directly into the water garden where nitrate concentrations were reduced to below 1 mg-N/L.

Ammonia

Figure 6.2 represents average inlet and outlet ammonia concentrations for the wetlands and the water garden. The wetlands reduced the ammonia concentrations from 2.8 mg-N/L to 0.9 mg-N/L. It appears that the gravel marshes are more efficient at lowering the ammonia concentration.

TKN

TKN average inlet and outlet concentration for the wetlands and the water garden are shown in Figure 6.3. Inlet concentration was reduced from 4.7 mg-N/L to 2.9 mg-N/L. Again, the gravel marshes appear to outperform the soil marshes. One

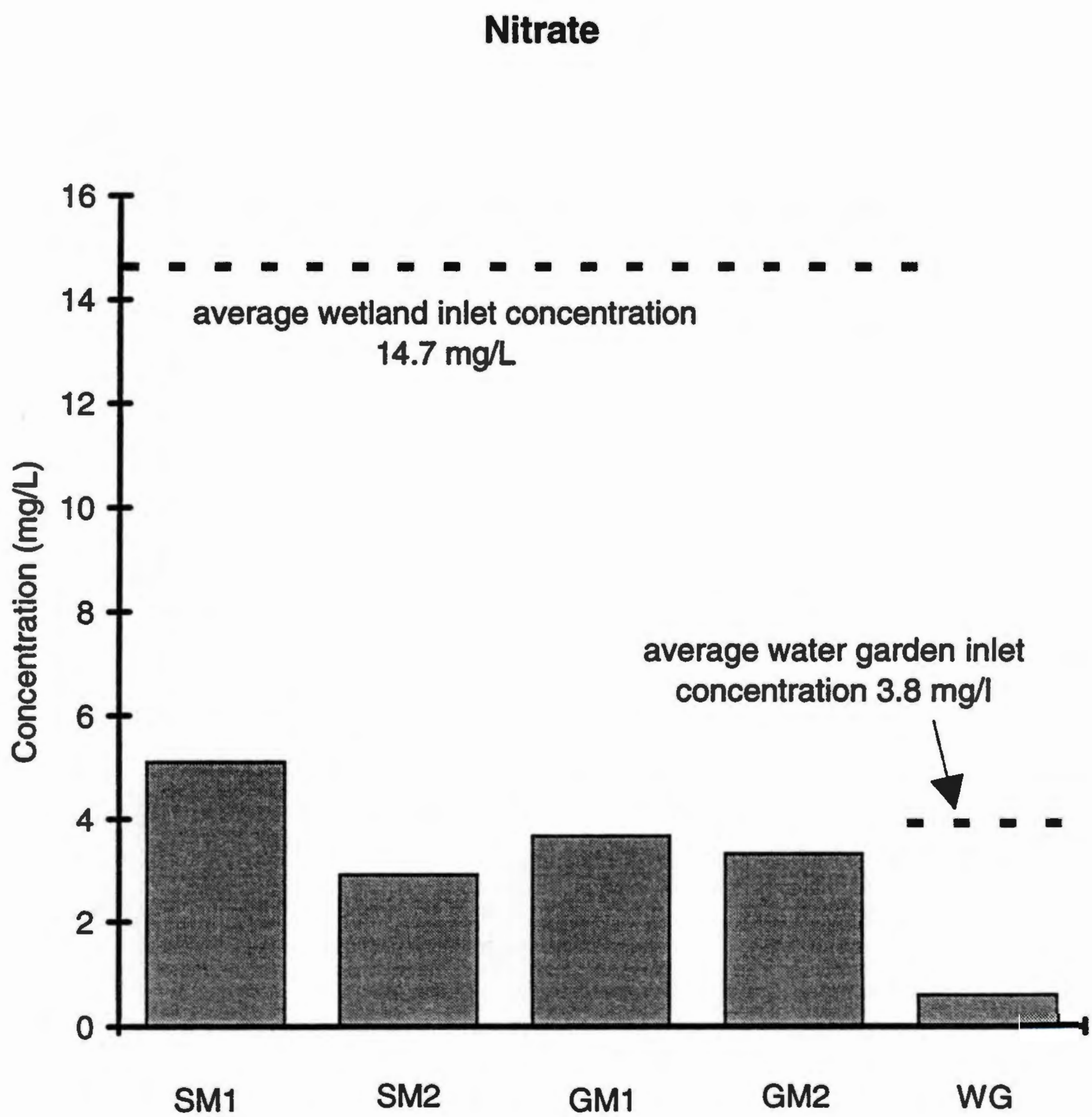

Figure 6.1. Inlet and outlet nitrate concentrations for the full size marshes

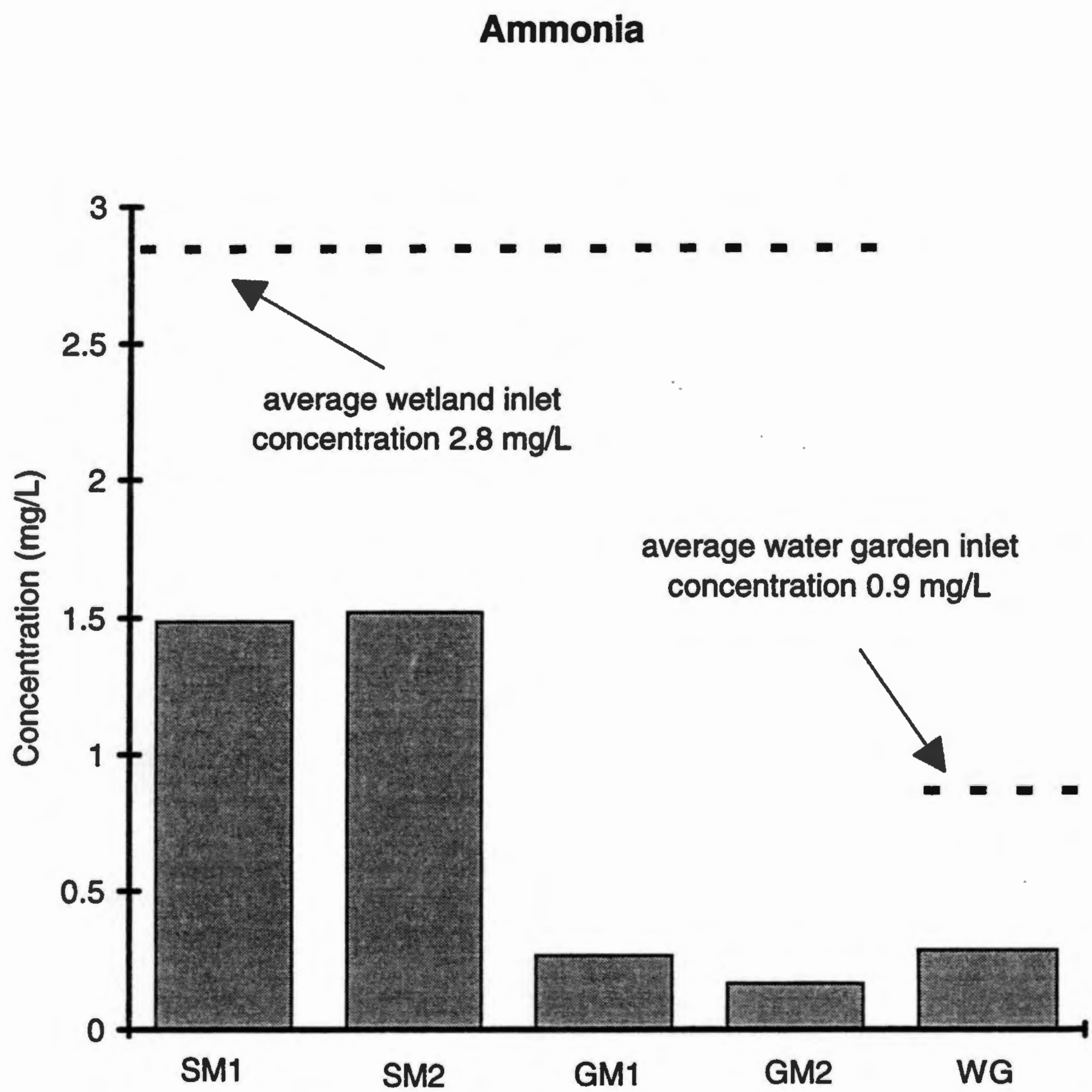

Figure 6.2. Inlet and outlet ammonia concentrations for the full size marshes

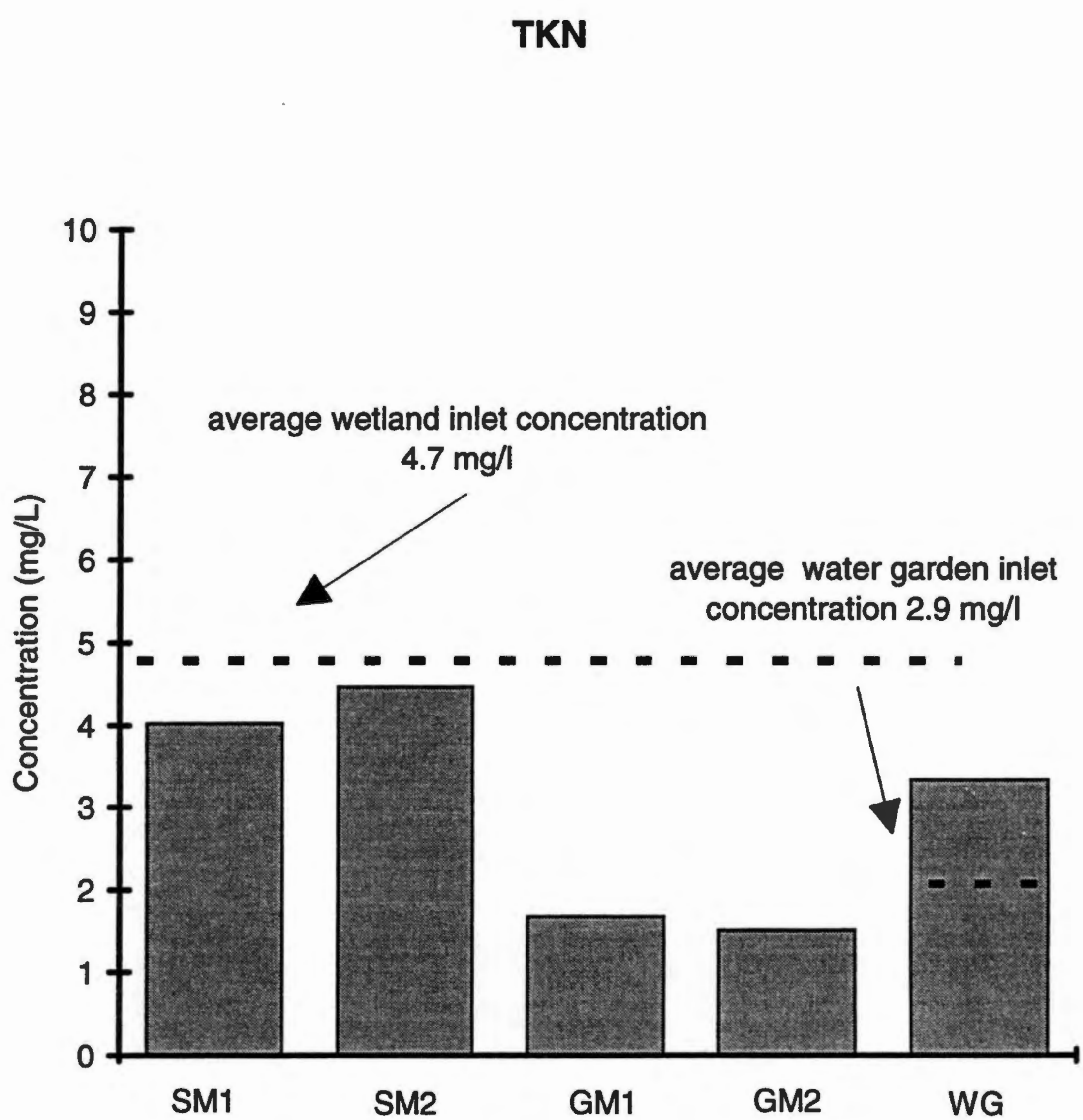

Figure 6.3. Inlet and outlet TKN concentrations for the full size marshes

noticeable attribute of this figure was the increase in TKN concentration in the water garden. One possible explanation of this could be from the addition of algae and leaf litter which was present in the water garden output.

BOD_5

Figure 6.4 represents the average inlet and outlet BOD_5 concentrations for the wetlands and the water garden. BOD_5 inlet concentrations were reduced from 7.3 mg/L to 3.9 mg/L. Outlet BOD_5 concentrations were again lower in the gravel marshes than in the soil marshes. The reduction of BOD_5 concentration in the water garden was low.

2. Test cells

Nitrate

Test cell average initial and average final nitrate concentrations are shown in Figure 7.1. There are no apparent advantages of one cell type over another. Even with the concentrating effects of ET, the concentration was reduced in all cells.

Ammonia

Ammonia initial and final ammonia concentrations are represented in Figure 7.2. Inlet concentrations were relatively stable among cells, with the inlet concentrations averaging around 1.5 mg-N/L. Test soil marsh 1 (TSM1) performed worst of any cell for this pollutant, lowering the concentration to just over one mg-N/L. The test gravel

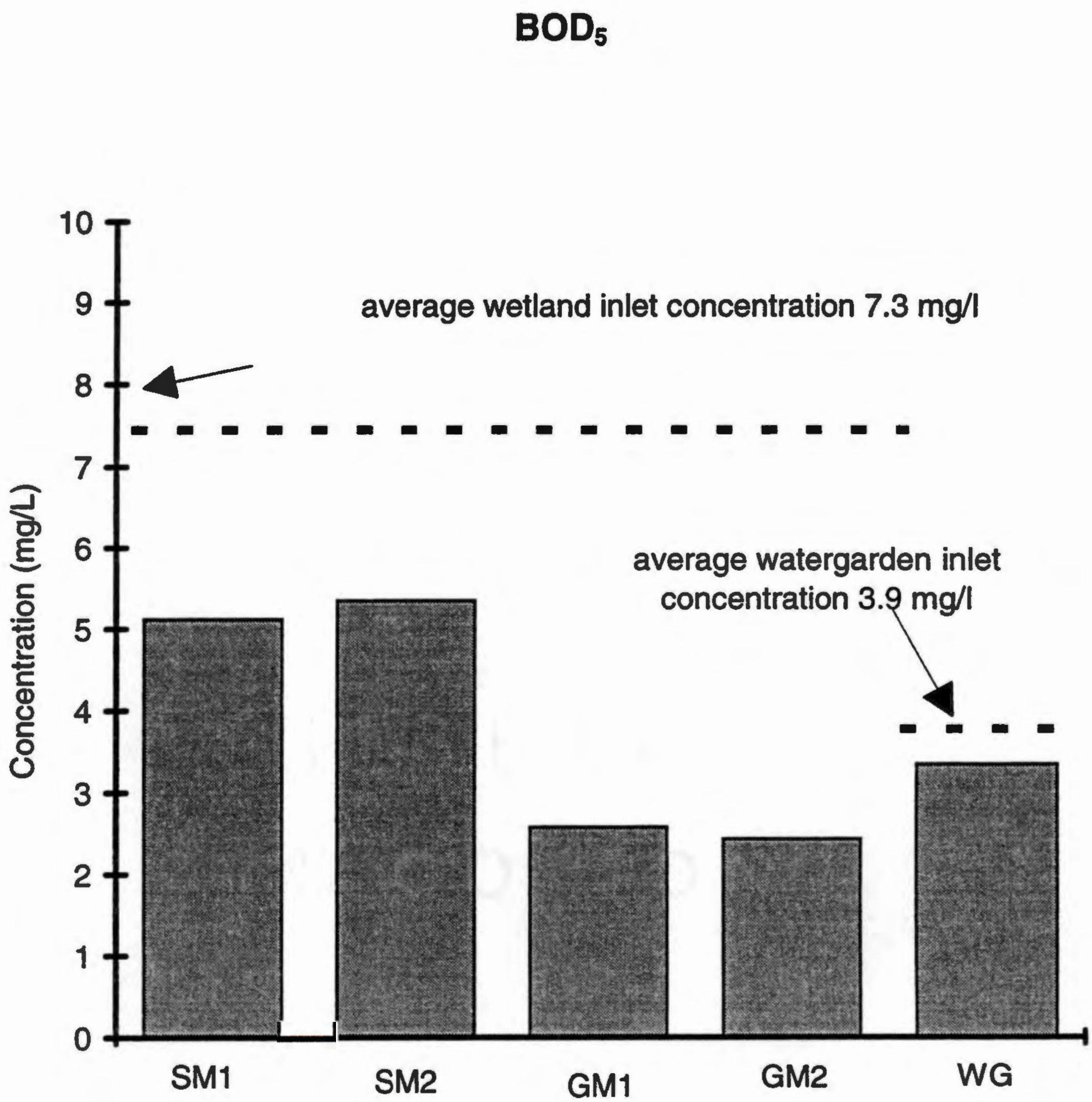

Figure 6.4. Inlet and outlet BOD_5 concentrations for the full size marshes

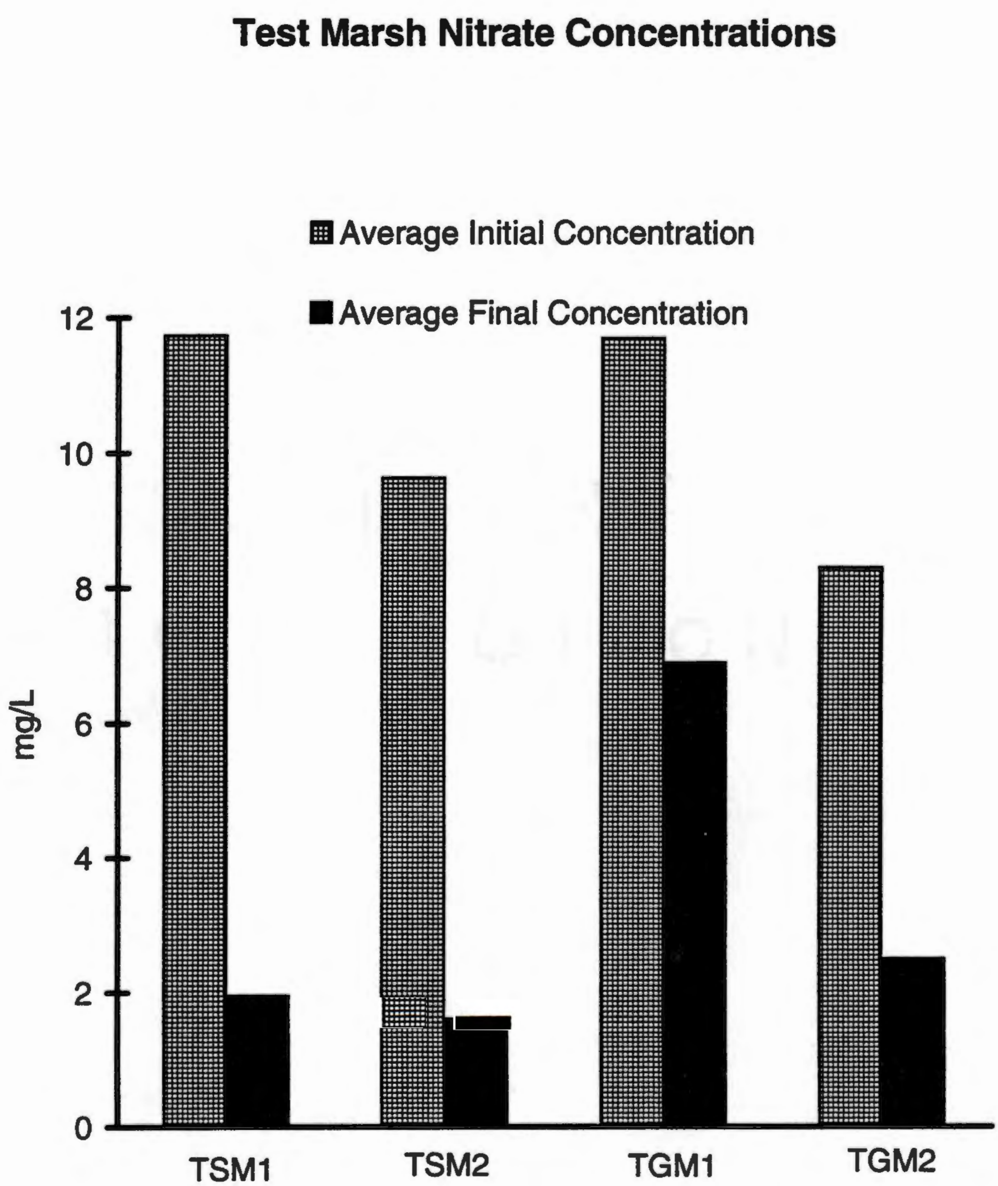

Figure 7.1. Test marsh inlet and outlet nitrate concentrations

Test Marsh Ammonia Concentrations

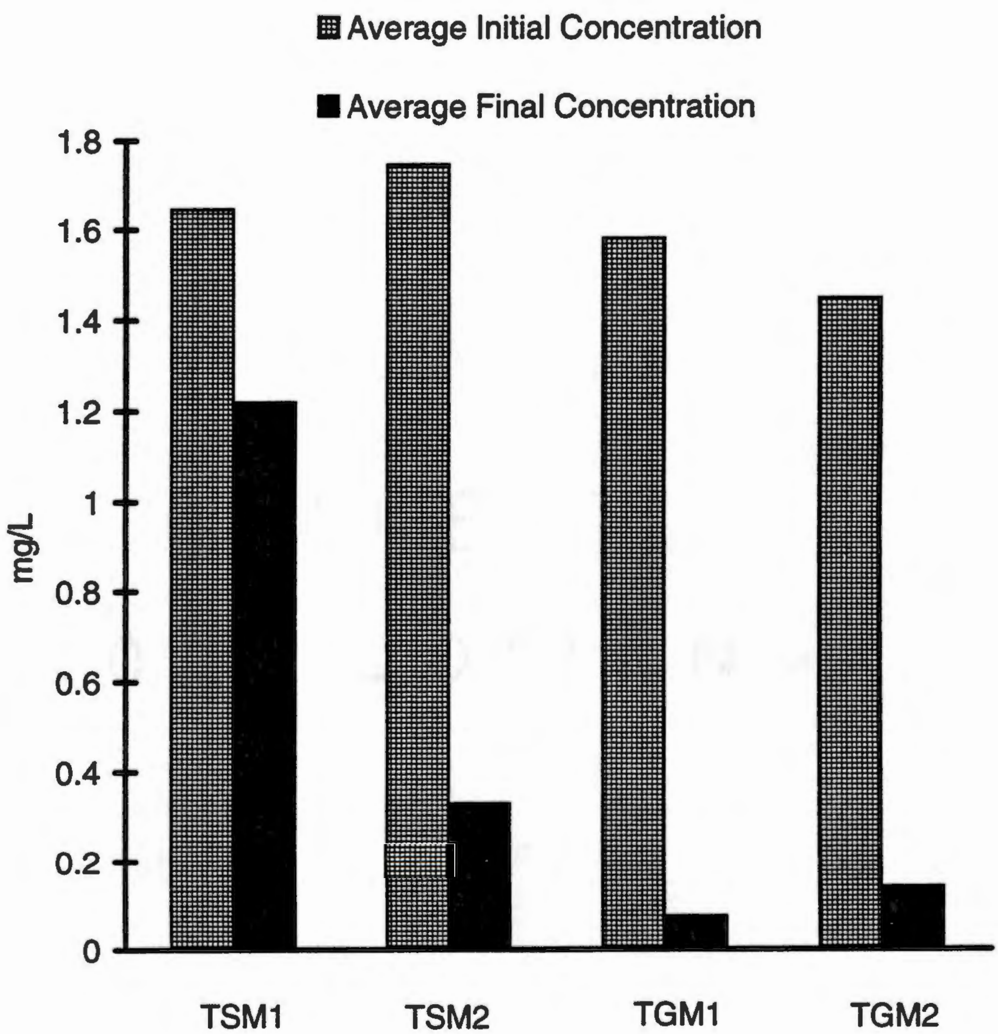

Figure 7.2. Test marsh inlet and outlet ammonia concentrations

marshes appear to have performed best at ammonia concentration reduction. These concentration reductions are very similar to the reductions observed in the full-scale wetlands, with the gravel marshes outperforming the soil marshes.

TKN

Test cell average initial and final TKN concentrations are shown in Figure 7.3. The initial concentrations for all the test cells were approximately the same at about 4 mg-N/L. No cell type was noticeably more efficient at TKN concentration reduction.

BOD_5

Test cell average initial and final BOD_5 concentrations are shown in Figure 7.4. Concentration reductions were approximately equal in all cell types. BOD_5 concentrations were reduced about 2.5 mg/L in all cells.

3. Variations in inlet concentrations

Differences in the inlet concentrations between test and full-size wetlands can be attributed to residual treated wastewater which remained in the test cells before the refilling process. The test cells and the wetlands appear to reduce the concentrations of the tested pollutants with the same efficiency.

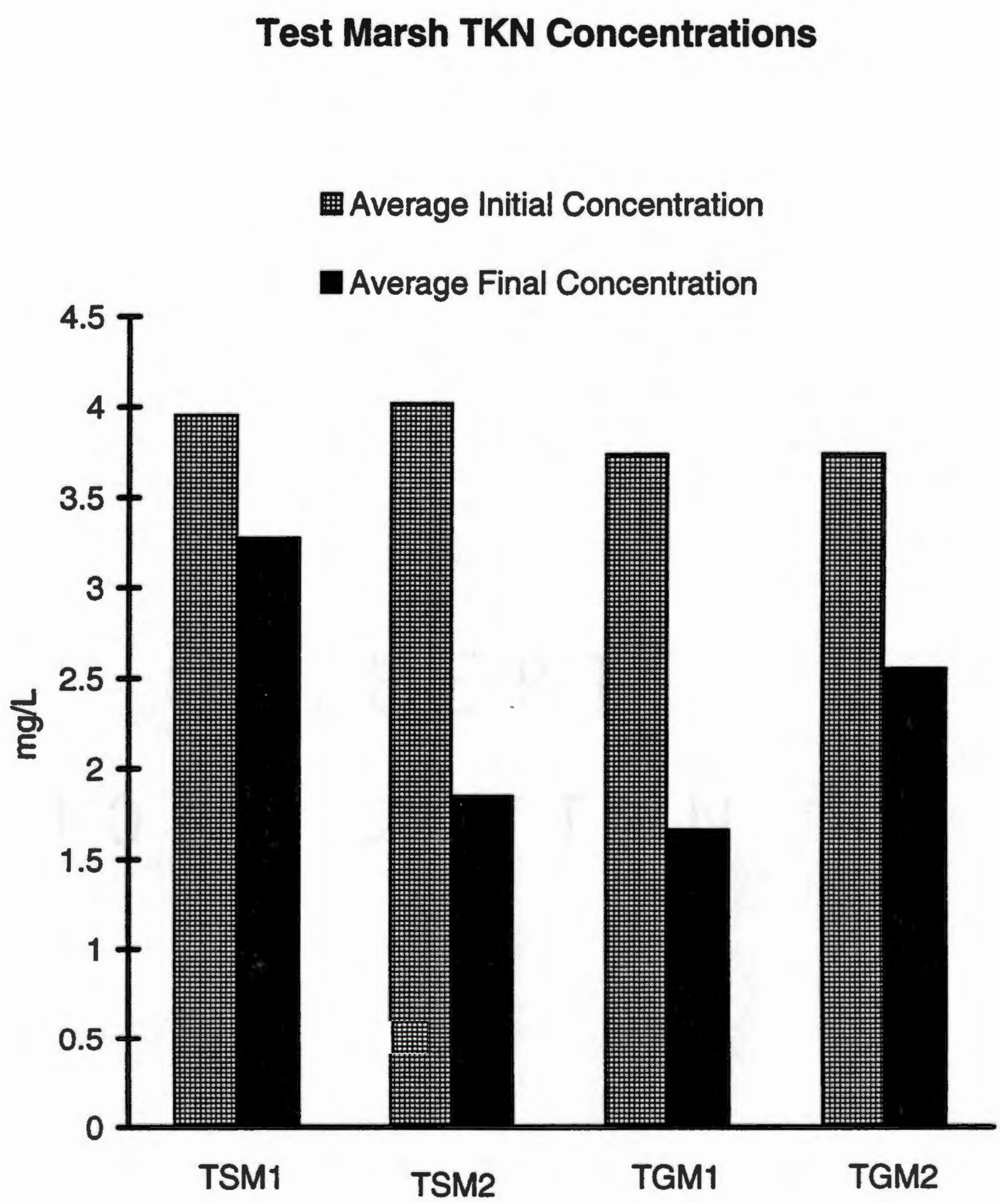

Figure 7.3. Test marsh inlet and outlet TKN concentrations

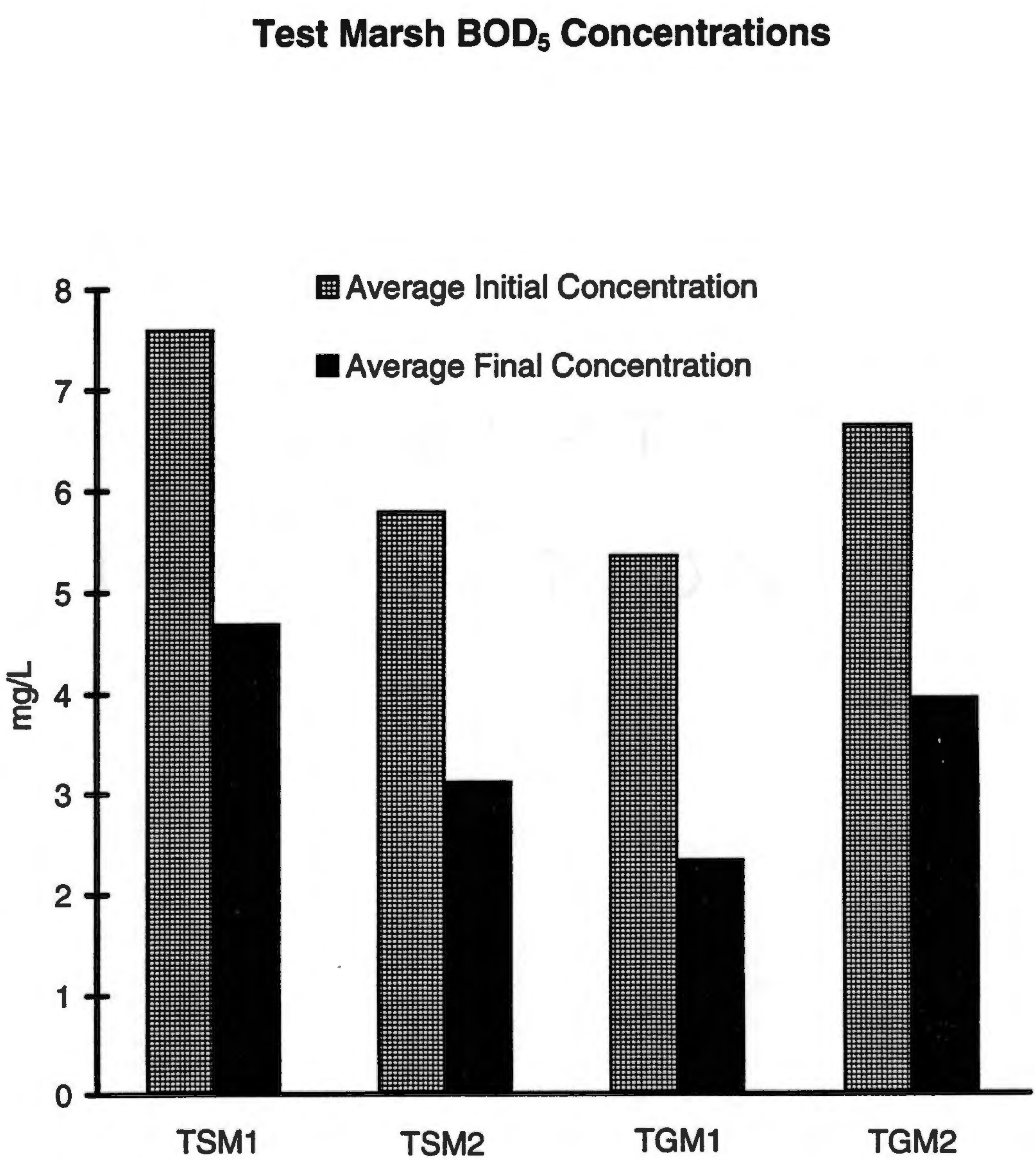

Figure 7.4. Test marsh inlet and outlet BOD_5 concentrations

D. Hydraulic Loading

Hydraulic loading of the M&M wetlands varied throughout the duration of the study (Figure 8.1). This variation is due to the wastewater treatment which occurs prior to application to the wetlands. As mentioned above, only a small portion of wastewater is diverted to the wetlands. Therefore, wastewater wasn't sent to the wetlands unless certain water quality parameters were met at the outlet of wastewater treatment plant.

E. Weather

One goal of this project was to determine the effects of weather on the treatment performance of the M&M marshes as well as the test cells. Weather data collection ran smoothly for the entire duration of the project. However, due to the variations in hydraulic and organic loading, no relationships between removal rates and weather conditions could be recognized.

F. Full-size vs. Test Cells

Nitrate

Average nitrate areal mass removal rates (AMRR) are shown in Figure 9.1. All full-size marshes performed evenly among themselves as did the test marshes. There appears to be an advantage in nitrate treatment efficiency in the full-size marshes.

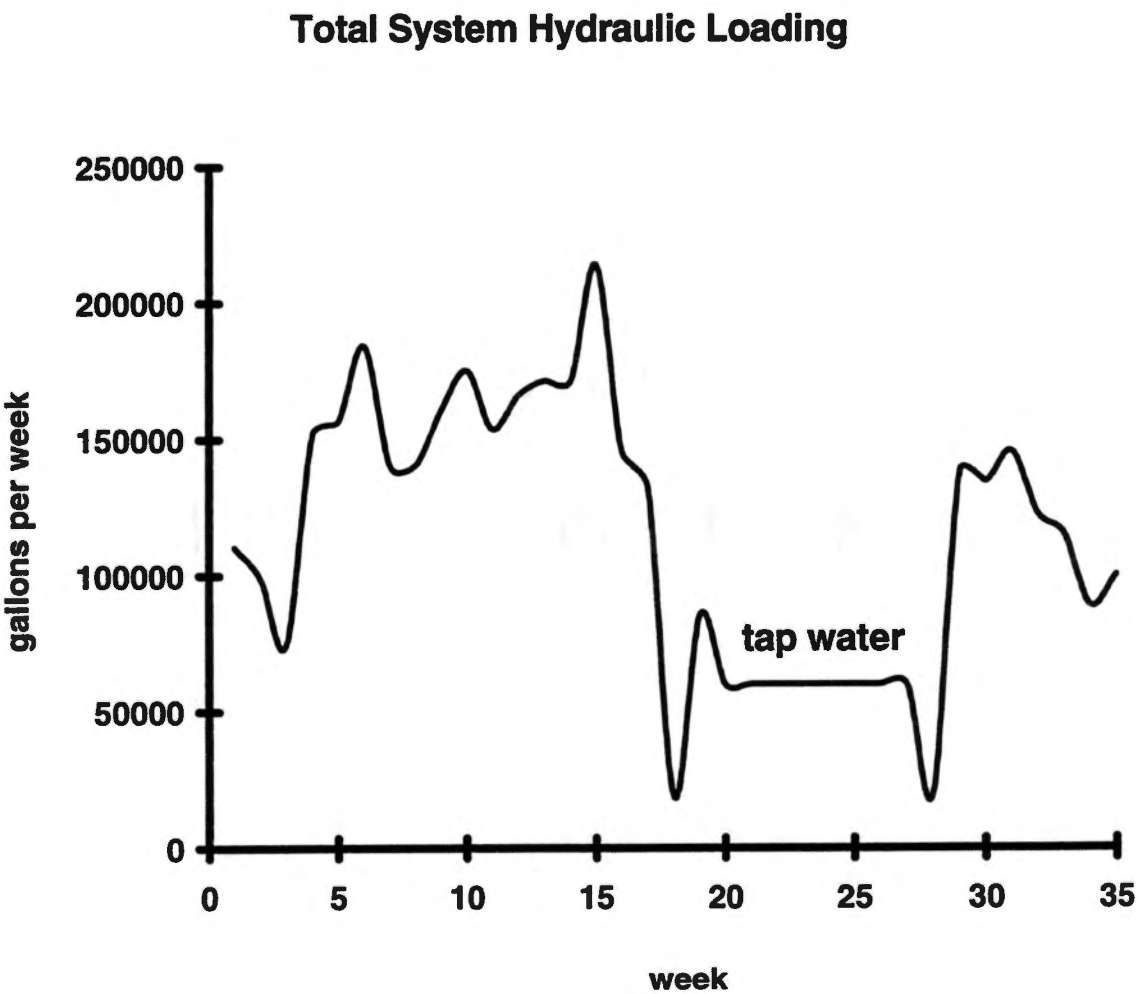

Figure 8.1. System hydraulic loading for the entire study

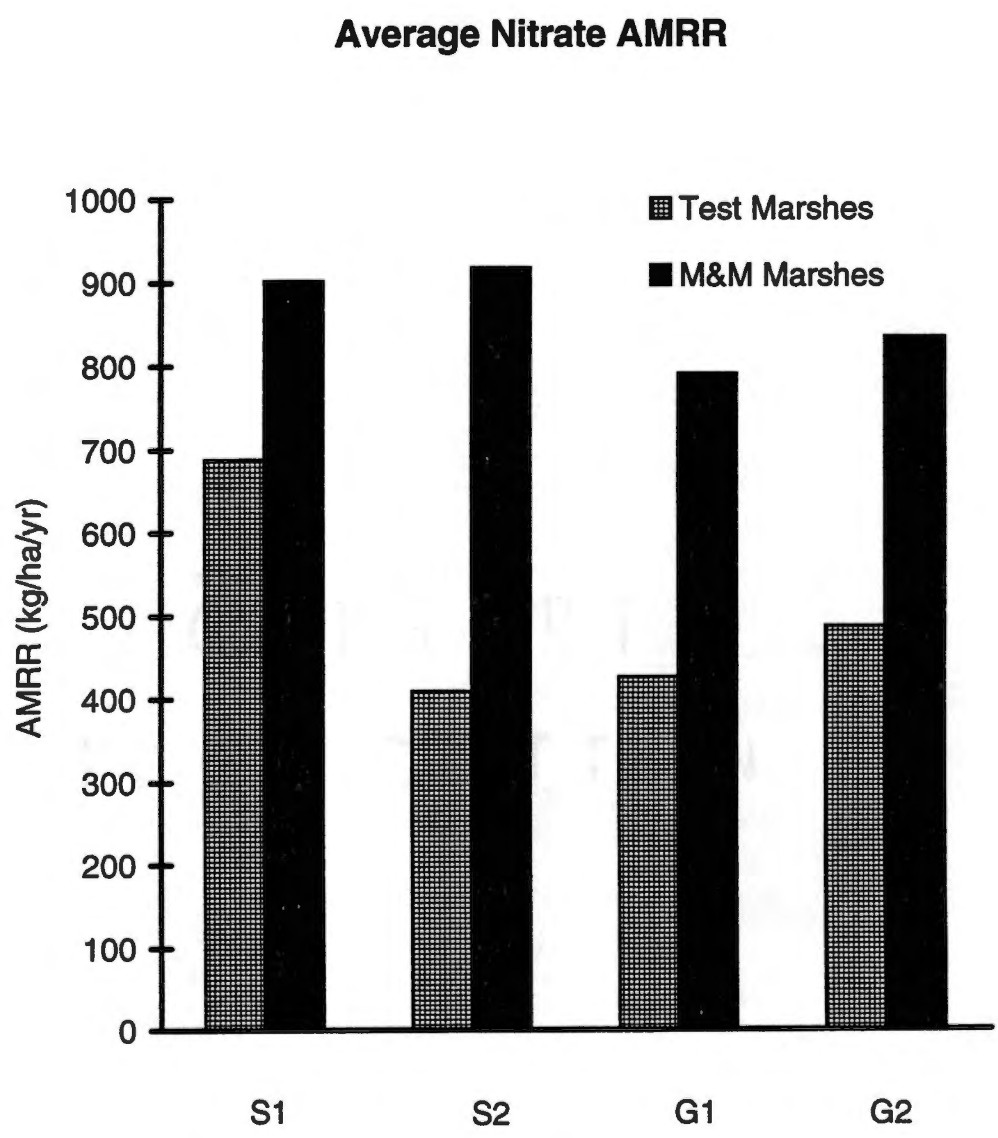

Figure 9.1. Test and full size marsh average nitrate aeral mass removal rate

Ammonia

Average ammonia AMRR are given in Figure 9.2. Again the full-size marshes appear to outperform the test marshes. However, even the maximum AMRR (M&M)soil marsh 1) is only 160 kg/ha/yr. This relatively low AMRR (compared to 7,000 kg/ha/yr reported by Knight et al. (1993), is due to the extremely low loading.

TKN

TKN average AMRR for full-size cells and test cells are given in Figure 9.3. The full-size marshes were again more efficient at removing the tested pollutant.

BOD_5

Average BOD_5 AMRR for the full-size marshes and the test marshes are given in Figure 9.4. There are no apparent differences in performance between full-size and test cells. Test gravel marsh 2 did outperform its full-size counter part slightly.

Full-size wetland AMRR could have been affected by the variability in loading. As noted in earlier discussions, influent concentrations varied greatly from week to week. It is uncertain how much loading variation occurred during the weekly sample lags. This variation would not have affected the test cells since they were batch loaded.

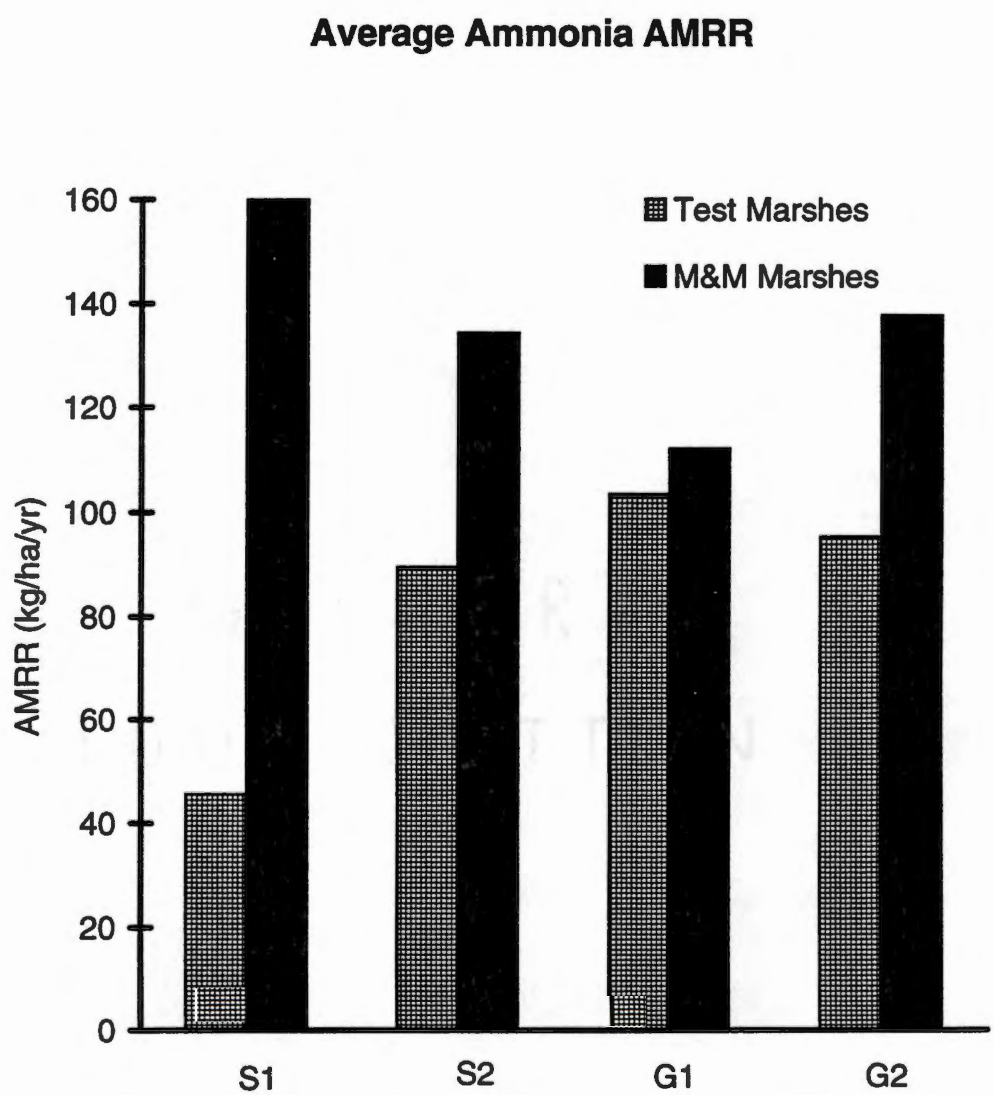

Figure 9.2. Test and full size marsh average ammonia areal mass removal rate

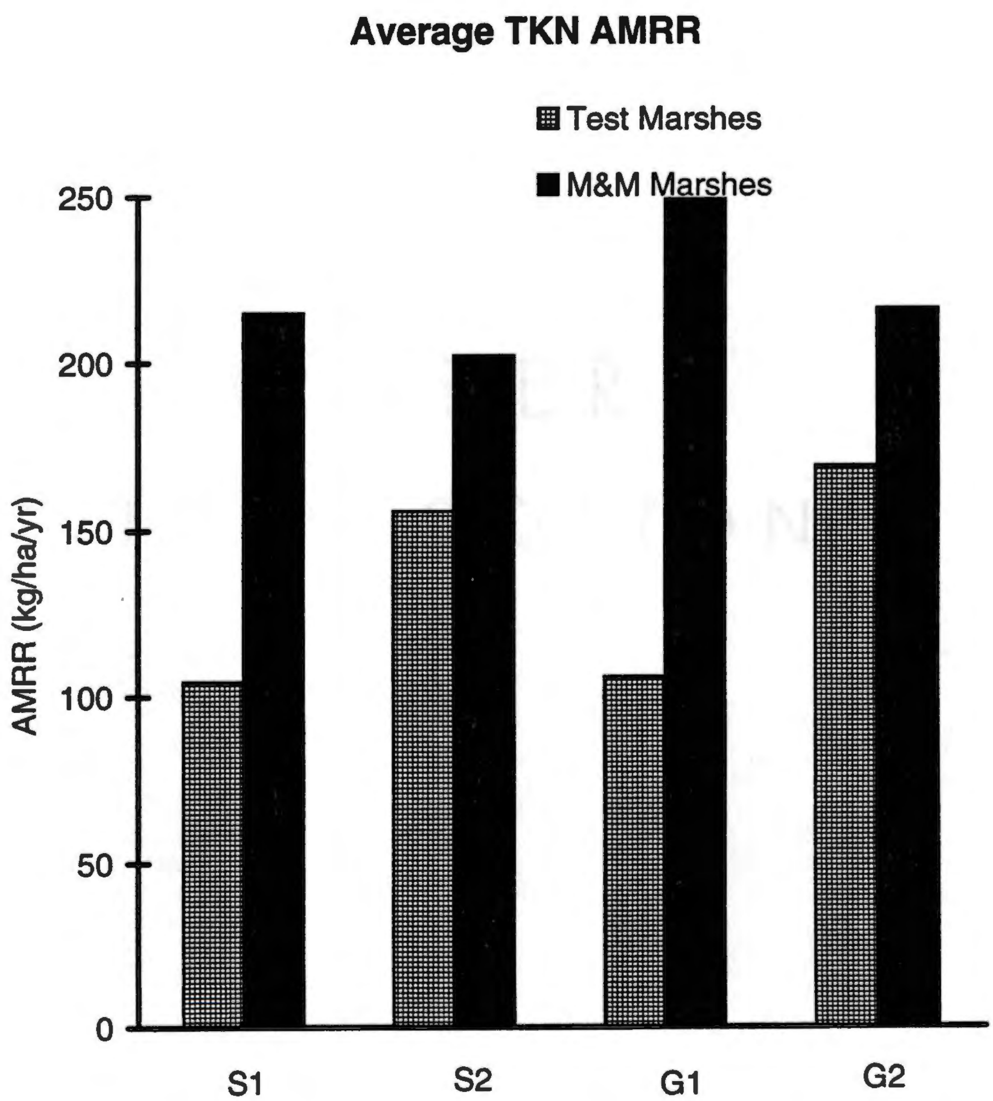

Figure 9.3. Test and full size marsh average TKN areal mass removal rate

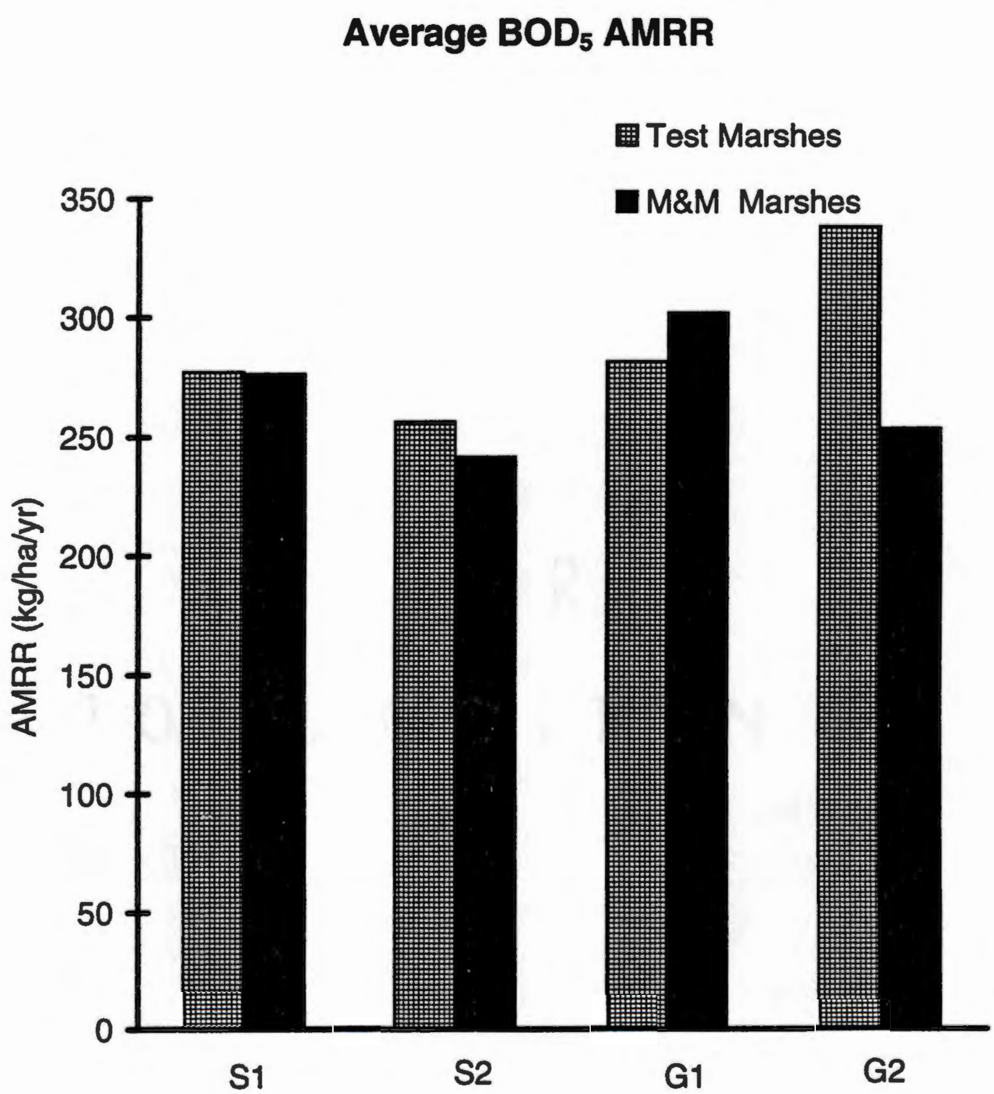

Figure 9.4. Test and full size marsh average BOD_5 areal mass removal rate

G. Test Cells as Predictors of Full-size Performance

Figures 9.1, 9.2, 9.3 and 9.4 indicate that the pollutants monitored at the tested concentrations, the test cells are good predictors of the performance of the full-size wetlands. There are differences in the mass removal rates of the systems which indicate that the full-size wetlands performed better than the test cells during this study. As mentioned earlier, these differences could be attributed to slight under-estimation of evapotranspiration rates in the test cells. Although they were insulated, their elevation and arrangement made them more susceptible to the elements.

Another issue which could cause variation between the test and full-size cells is the overhang of plants within the test cells. After the plants became well established, they began to hang over the cell edges, thus, increasing their area. This could make the test cell AMRR higher than it actually is.

H. Effect of Plant Species on Treatment Efficiency

Based on areal mass removal rates (Figures 9.1, 9.2, 9.3, 9.4) there appears to be no difference in treatment performance between plant species. This is in contrast to Gersberg et al.'s (1985) results. Difference between our results and this study are probably due to differences in loading rates and climates between the two studies.

I. Harvesting

Annual harvesting of the plants in the M&M system is probably not necessary. As reported in the literature harvesting is probably not necessary when dealing with species such as *Scirpus*, *Typha* or *Phragmites* (Reed and Brown 1992). The wetlands may, however, become clogged with plant litter after several years and require some type of harvesting to maintain adequate flow through the bed.

J. Kinetics

Inlet concentrations of measured pollutants were plotted against AMRR in order to determine the kinetics of the system. Gravel marsh 1 and 2 were combined as well as soil marsh 1 and 2 to determine if there were any distinct differences in kinetics between like cells. The SAS (statistical analysis system) procedure GLM (general linear model) was used to determine if the slopes of these data were similar. The results of this procedure show that for all practical purposes, GM1 and GM2 perform identically, as do SM1 and SM2. There are several data points which reflect negative AMRR; i.e. an addition of pollutant to the system. However, this is simply a representation of extremely light loading weeks in which very little or no pollutant was added to the system. With little or no AMLR, even a slight detection of an outlet pollutant concentration results in a negative AMRR.

Nitrate

Nitrate removal kinetics for both marsh types are illustrated in Figure 10.1. A first-order kinetic model fits the data well giving high R^2 values for the soil marshes and gravel marshes (0.95 and 0.98, respectively); i.e. there is a linear relationship between influent concentration and mass removal rate. There is also very little difference between the two marsh types.

Ammonia

Ammonia kinetics for the gravel and soil marshes are given in Figure 10.2. R^2 values for the soil marshes and gravel marshes were 0.88 and 0.94, respectively. Again, the data fits a first order kinetic model, and there is very little difference between cell types.

TKN

Figure 10.3 illustrates TKN removal kinetics for the soil and gravel marshes. R^2 values for the soil and gravel marshes were 0.84 and 0.94, respectively. Removal kinetics are linear and there are no apparent differences between soil marsh and gravel marsh removal kinetics.

Nitrate Removal Kinetics

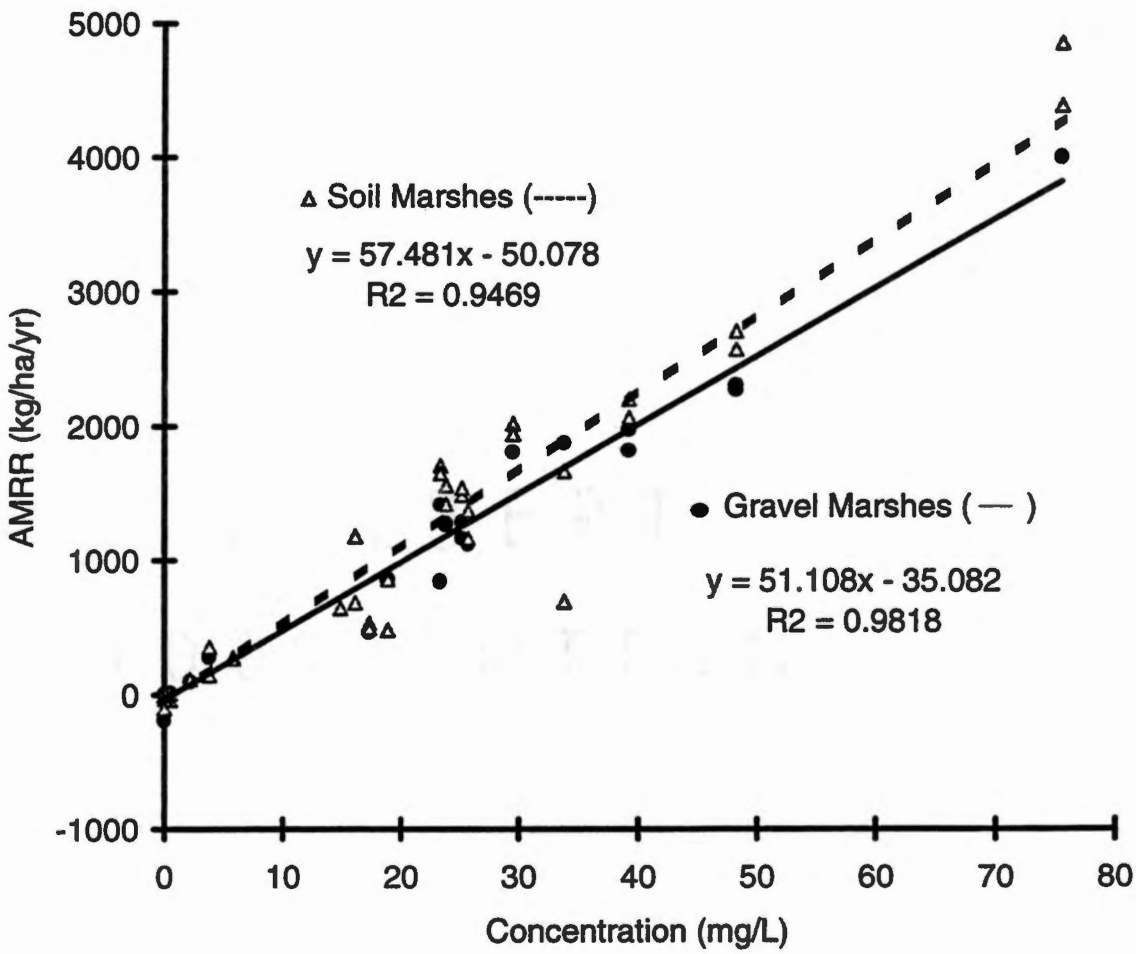

Figure 10.1. Soil and gravel marsh nitrate removal kinetics

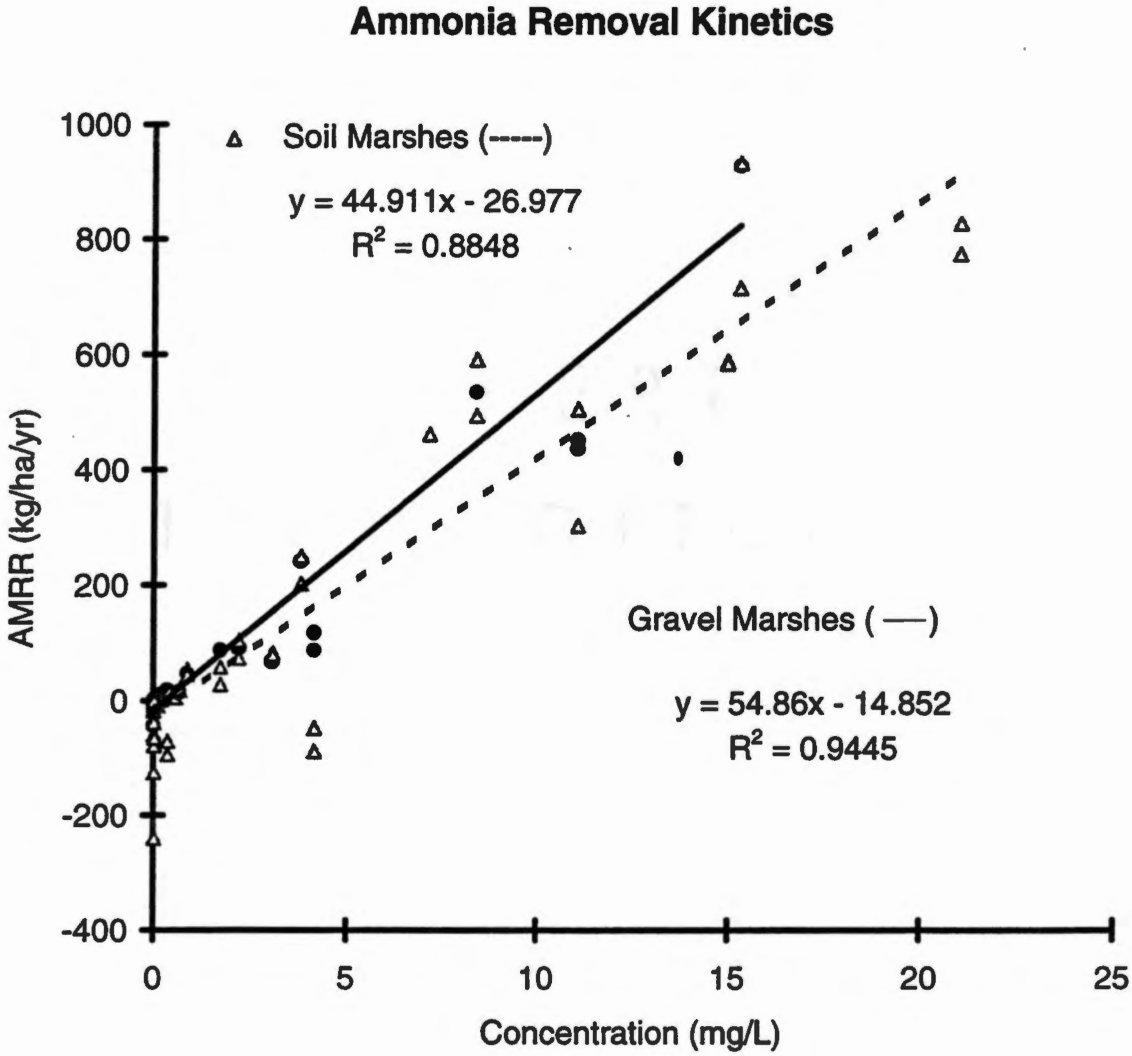

Figure 10.2. Soil and gravel marsh ammonia removal kinetics

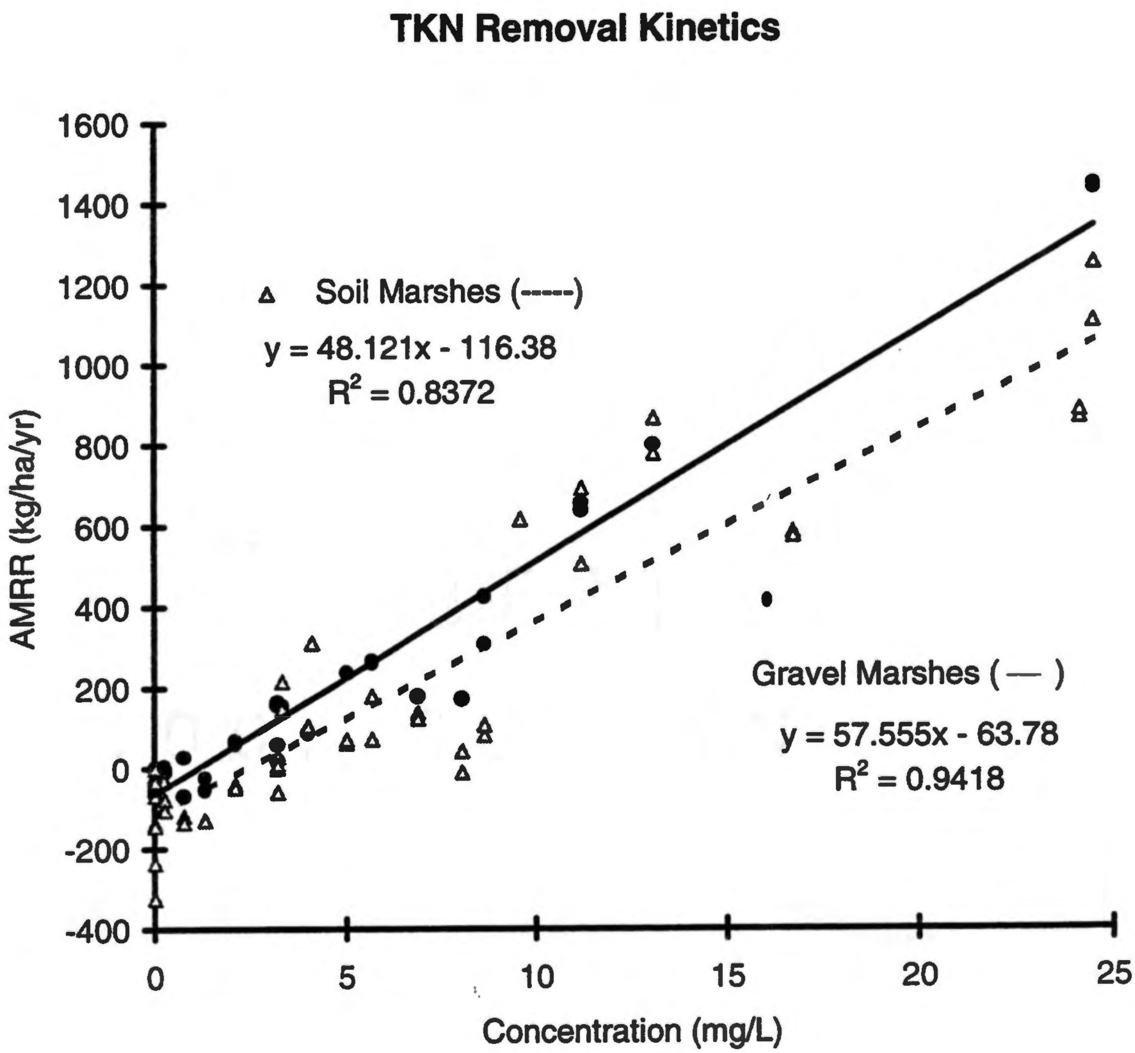

Figure 10.3. Soil and gravel marsh nitrate removal kinetics

BOD_5

BOD_5 removal kinetics for the soil and gravel marshes are given in Figure 10.4. R^2 values for soil marsh and gravel marsh BOD_5 removal kinetics were lower than any other tested pollutant (0.77 and 0.89, respectively). However, the overall relationship is still fairly linear and there is little or no difference between cell type.

K. Predictive Values

Based on observations made during this study, rough estimates can be made as to outlet concentrations for various flow rates. The following equation was based on a mass balance on the wetlands.

$$c_{out} = c_{in} - \frac{(AMRR)(\text{wetland area})}{Q}$$

Maximum inlet nitrate, TKN, and BOD_5 concentrations (35, 25, and 30, respectively) were used. Figure 11 was developed using maximum areal mass removal rates for the pollutants nitrate, TKN and BOD_5 observed in the M&M soil marshes. Outlet concentrations were calculated using these maximum AMRR and hypothetical flow rates. These values are estimated rates and should not be used for loading criteria before tested on pilot scale or under close monitoring.

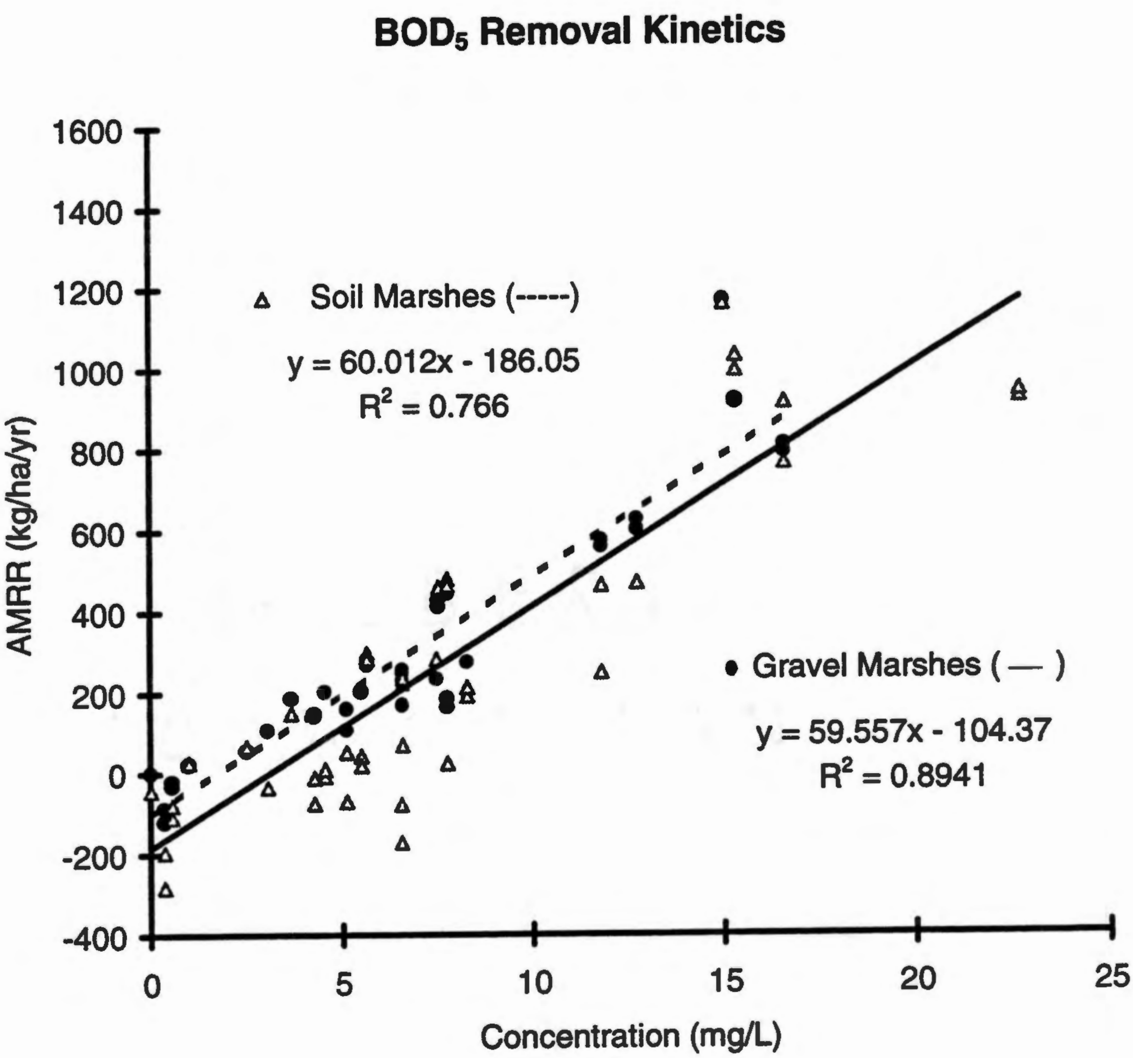

Figure 10.4. Soil and gravel marsh BOD_5 removal kinetics

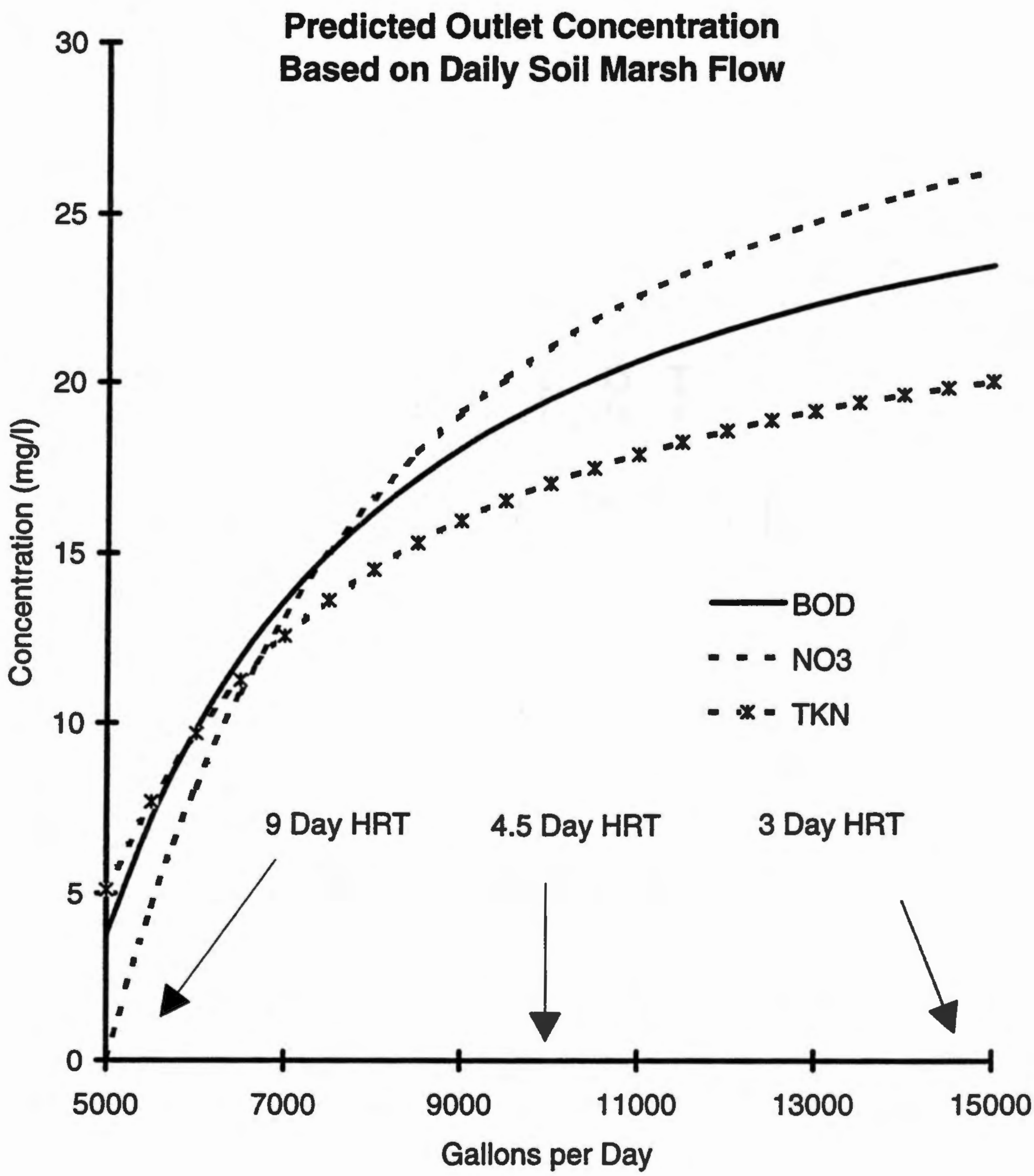

Figure 11. Predicted outlet concentration using a fixed AMRR and hypothetical flow rates

CHAPTER V

CONCLUSIONS AND RECOMMENDATIONS

This research provided necessary information about the constructed wetlands studied. However, certain aspects of the wetlands made it difficult to attain all of the objectives set forth during the initial phases of the project. One obstacle encountered was the variation in quantity and strength in the wastewater distributed to the wetlands. In order to determine if one wetland type or plant species is more effective at wastewater renovation, there must be a constant loading used for a period long enough to establish trends in treatment. Another issue which made it difficult to establish treatment differences was the low concentration of wastewater used in the study. In most cases, wastewater concentrations were reduced to non-detectable levels. This made it impossible to determine the wetland's full capabilities.

Although the test cells performed comparably with the full size cells, further studies should be conducted using wastewater with higher pollutant concentrations. Furthermore, if a waste such as dairy lagoon effluent were used, comparisons could be made to several studies previously conducted.

REFERENCES

REFERENCES

Allen, R. L., J. H. Prueger, and R. W. Hill. 1992. Evapotranspiration from stands of isolated hydrophytes: cattail and bulrush. Transactions of the ASAE 35(4):1191-1198.

Brix, H. 1993. Wastewater Treatment in Constructed Wetlands: system design, removal processes, and treatment performance. In *Constructed Wetlands for Water Quality Improvement*, ed G. A. Moshiri, pp. 9-22. Ann Arbor, MI: Lewis Publishers Inc.

Benham B. L. 1995. Utilization of Laboratory-Scale Constructed Wetlands to Evaluate the Applicability of Constructed Wetland Technology to the Treatment of a Livestock Waste. Doctor of Philosophy Dissertation, The University of Tennessee, Knoxville, TN.

Conley, L. M., R. I. Dick, and L. W. Lion. 1991. An assessment of the root zone method of wastewater treatment. Research Journal WPCF, 63(3):239-247.

Crites, R. W. and G. Tchobanoglous. in K. H.Rogers, P. F. Breen, and A. J. Chick. 1991. Nitrogen removal in experimental wetland treatment systems: evidence for the role of aquatic plants. Research Journal WPCF, 63(7):934-941.

Cuenca, R. H. 1989. Irrigation System Design. Prentice Hall Inc., New Jersey. pp. 125-127.

Davis, S. H., R. Ulmer, L. Strong, T. Cathcart, J. Pote, and W. Brock. 1992. Constructed wetlands for dairy wastewater treatment. ASAE Paper No. 92-4525. ASAE, 2950 Niles Rd., St. Joseph, MI 49085-9659.

Davis, T. H., and P. D. Cottingham. 1993. Phosphorus removal from wastewater in a constructed wetland. In *Constructed Wetlands for Water Quality Improvement*, ed. G. A. Moshiri, pp. 315-320. Ann Arbor, MI: Lewis Publishers Inc.

Debusk, T. A., J. E. Peterson, and K. R. Jensen. 1995. Phosphorus removal from agricultural runoff: an assessment of macrophyte and periphyton-based treatment systems. In *Versatility of Wetlands in the Agricultural Landscape*: an International Conference jointly sponsored by AWRA and ASAE. September 17-20, 1995.

DeShon, G. C., A. L. Thompson, D. M Sievers, and A. T. Hjemfelt. 1995. Hydraulic Properties and Relationships for the Design of Subsurface Flow Wetlands. In *Versatility of Wetlands in the Agricultural Landscape*: an International Conference jointly sponsored by AWRA and ASAE. September 17-20, 1995.

Gale, P. M., K. R. Reddy, and D. A. Graetz. 1993. Nitrogen removal from reclaimed water applied to constructed and natural wetland microcosms. Research Journal WPCF, 65(2):162-168.

Gearhart, R. A., F. Klopp and G. Allen. 1989. Constructed free water surface wetlands to treat and receive wastewater: pilot project to full-scale. In *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural*, ed D. A. Hammer, 121-138. Chelsea, MI: Lewis Publishers Inc.

Gersberg, R. M., B. V. Elkins, S. R. Lyon and C. R. Goldman. 1985. Role of aquatic plants in wastewater treatment by artificial wetlands. Water Resources, 20(3):363-368.

Greenberg, A. E. , L. S. Clesceri, A. D. Eaton. 1992. Standard methods for the examination of water and wastewater, 18 ed. American Public Health Association., Washington, D. C. Section 5-6.

Gross, M. A., R. Burson, and A. Brown. 1994. Sand filtration of constructed wetland effluent. In *Versatility of Wetlands in the Agricultural Landscape*: an International Conference jointly sponsored by AWRA and ASAE. September 17-20, 1995.

Hammer, D. A. and R. L. Knight. 1994. Designing constructed wetlands for nitrogen removal. Wat. Sci. Tech., 29(4):15-27.

Hammer, D. A. and R. K. Bastian. Wetlands ecosystems: natural water purifiers? Constructed wetlands for wastewater treatment. pp. 5-19.

Higgins, M. J., C. A. Rock, R. Bouchard, and B. Wengrezynek. 1993. Controlling agricultural runoff by use of constructed wetlands. In *Constructed Wetlands for Water Quality Improvement*, ed G. A. Moshiri, pp. 359-367. Ann Arbor, MI: Lewis Publishers Inc.

Holmes, B. J., G. D. Bubenzer, L. R. Massie, and G. Hines. 1994. A constructed wetland for treating milkhouse wastewater in a cold climate--status report. Proceedings: Constructed Wetlands for Animal Waste Management, Lafayette, IN pp. 53-63.

Kadlec, R. H. 1995. Performance and Design of Treatment Wetlands. In *Versatility of Wetlands in the Agricultural Landscape*: an International Conference jointly sponsored by AWRA and ASAE. September 17-20, 1995.

Knight, R. L., R. W. Ruble, R. H. Kadlec, and S. Reed. 1993. Wetlands for wastewater treatment: performance database. In *Constructed Wetlands for Water Quality Improvement,* ed G. A Moshiri pp. 35-56. Ann Arbor, MI: Lewis Publishers Inc.

Kuehn. E., and J. A. Moore. 1993. Variability in treatment by constructed wetlands. Presented at the 1993 ASAE International Winter Meeting, Paper No. 932578. ASAE, 2950 Niles Rd., St. Joseph, MI 49085-9659.

Metcalf and Eddy, Inc. 1991. Wastewater Engineering: Treatment, Disposal, and Reuse. 3rd Ed. McGraw-Hill Book Company, New York. pp. 995-1001.

McCaskey, N. E., T. C. Hannah, R. M. Krotz, J. T. Eason, D. A. Hammer and V. W. E. Payne. 1994. Constructed wetlands for the treatment of wastewater: operational and maintenance parameters. Proceedings Eighth Annual Alabama Water Resources Conference and Symposium, Gulf Shores, AL. Aug. 31-Sept. 2, 1994.

Netter, R. 1994. Flow characteristics of planted soil filters. Wat. Sci. Tech., 29(4):37-44.

Reed, S. C. 1991. Constructed wetlands for wastewater treatment. BioCycle, Jan. 1991.

Reed, S. C. and D. J. Calkins. 1994. Constructed wetlands for wastewater treatment. Presented at: Alternative Waste Water Treatment Conference, Burlington, Vermont, Oct. 5, 1994, pp. 29-39.

Reed, S. C. and D. S. Brown. 1992. Constructed wetland design--the first generation. Water Environment Research, 64(6):776-781.

Reed, S. C. and D. S. Brown. 1994. Subsurface flow wetlands--a performance evaluation. Water Environment Research, 67(2):244-248.

Rogers, J. W., D. T. Hill, V. W. E. Payne, and S. R. Kown. 1995. A biological treatment study of constructed wetlands treating poultry waste. In *Versatility of Wetlands in the Agricultural Landscape*: an International Conference jointly sponsored by AWRA and ASAE. September 17-20, 1995.

Rogers, K. H., P. F. Breen, and A. J. Chick. 1991. Nitrogen removal in experimental wetland treatment systems: evidence for the role of aquatic plants. Research Journal WPCF, 63(7):934-941.

Sawyer, C. N. and P. L. McCarty. 1978. Chemistry for Environmental Engineering. McGraw Hill Book Company, New York. pp. 417-439.

Surface, J. M., J. H. Peverly, T. S. Steenhuis, and W. E. Sanford. 1993. Effect of season, substrate composition, and plant growth on landfill leachate treatment in a constructed wetland. In *Constructed Wetlands for Water Quality Improvement*, ed G. A. Moshiri, pp. 461-472. Ann Arbor, MI: Lewis Publishers Inc.

Surrency, D. 1993. Evaluation of aquatic plants for constructed wetlands. *In Constructed Wetlands for Water Quality Improvement*, ed G. A. Moshiri, pp. 349-357. Ann Arbor, MI: Lewis Publishers Inc.

Tanner, C. C. 1994. Treatment of dairy farm wastewater in horizontal and up-flow gravel-bed constructed wetlands. Wat. Sci. Tech., 29(4):85-93.

U.S. Environmental Protection Agency, 1983. Methods for Chemical Analysis for Water and Wastes, EPA-600/4-79-020. method 10-107-06-1-C.

U.S. Environmental Protection Agency, 1983. Methods for Chemical Analysis for Water and Wastes, EPA-600/4-79-020. method 10-107-06-2-D.

van Oostrom, A. J., and J. M. Russell. 1994. Denitrification in constructed wastewater wetlands receiving high concentrations of nitrate. Wat. Sci. Tech., 29(4):7-14.

Watson, J. T., and A. J. Danzig. 1993. Pilot-scale nitrification studies using vertical-flow and shallow horizontal-flow constructed wetland cells. *In Constructed Wetlands for Water Quality Improvement*, ed G. A. Moshiri, pp. 301-313. Ann Arbor, MI: Lewis Publishers Inc.

VITA

Matthew Warwick Bowling was born on April, 8 1970 in Knoxville, TN. After graduating from Anderson County High School in 1988, he entered The University of Tennessee, Knoxville and received a B.S. degree in Wildlife and Fisheries Science, with a minor in Forestry in May, 1992. The following January he entered graduate school at The University of Tennessee, Knoxville. In August, 1996 he received a M.S. degree in Agricultural Engineering Technology. He and his wife, Rebecca Glover Bowling currently reside in Powell, TN.