1984

Transformation of Nitrogen in Highway Runoff Management Systems

Robert D. Tolbert
University of Central Florida

TRANSFORMATION OF NITROGEN IN HIGHWAY RUNOFF MANAGEMENT SYSTEMS

BY

ROBERT D. TOLBERT
B.S., Troy State University, 1981

THESIS

Submitted in partial fulfillment of the requirements
for the degree of Master of Science
in the Graduate Studies Program of the College of Engineering
University of Central Florida
Orlando, Florida

Spring Term
1984

ABSTRACT

The operation and maintenance of highways contributes a variety of pollutants to surface and subsurface waters. Solids, nutrients, heavy metals, oil and grease, pesticides and bacteria can all be associated with highway runoff. Although the full extent of the effect of all of these runoff constituents upon the quality of surrounding waters is not well defined, this study will mainly concentrate on nutrient contaminants (essentially nitrogen).

The last decade has seen increasing efforts in research and development to abate contaminant discharges from highway runoff using a number of treatment facilities such as retention/detention basins, swales and wetland systems. An evaluation of the effectiveness of these systems in removing nitrogen by physical, chemical and biological uptake is the aim of this research endeavor. This information could prove invaluable in an overall assessment of the effectiveness of highway runoff treatment systems.

ACKNOWLEDGEMENTS

The author is grateful to the Faculty and Staff of the Civil Engineering and Environmental Sciences Department of UCF for their help and guidance. Special acknowledgement is given to Dr. Yousef A. Yousef, Dr. Martin P. Wanielista and Mr. Harvey H. Harper for reviewing this thesis and serving on my committee.

Special thanks are accorded to the following:

To Dr. Yousef A. Yousef for his patience, assistance and understanding in the past three years.

To Ms. Sharon Darling whose kindness, patience and expertise in typing and editing this thesis were deeply appreciated.

To my parents for providing love and understanding for 25 years.

And finally to my wonderful wife, Louan, whose devotion, patience and self-sacrifice made this all possible and worthwhile. I leave her with a poem that expresses my love much better than I could ever hope to:

THE SHINING BARRIER

This present glory, love, once-given grace,
The sum of blessing in a sure embrace,
Must not in creeping separateness decline
But be the centre of our whole design.

We know it's love that keeps a love secure,
And only by love of love can love endure,
For self's a killer, reckless of the cost,
And loves of lilactime unloved are lost.

We build our altar, then, to love and keep
The holy flame alight and never sleep:
This darling love shall deepen year by year,
And dearer shall we grow who are so dear.

The magic word is <u>sharing</u>: every stream
Of beauty, every faith and grief and dream;
Go hand in hand in companionship -
In sober death no sunduring of the grip.

And into love all other loveliness
That we can trace from time we shall impress:
Slow downs and lilacs, treasures of the trees,
The spring and poems, stars and ancient seas.

This splendor is upon us, high and pure
As heaven: and we swear it shall endure:
Swear fortitude for pain and faith for tears
To hold our shining barrier down the years.

C.S. Lewis

TABLE OF CONTENTS

TABLE OF CONTENTS (Continued)

LIST OF FIGURES

LIST OF FIGURES (Continued)

LIST OF FIGURES (Continued)

LIST OF TABLES

LIST OF TABLES (Continued)

CHAPTER I

INTRODUCTION

A number of treatment practices, including roadside swales and detention/retention ponds, have been used to abate contaminant discharge from highway runoff. However, the effectiveness of these systems in removing these contaminants, namely nitrogen, phosphorus and heavy metals, is not known. This study will concentrate on the capacity of roadside swales and detention/retention basins for nitrogen removal from highway runoff. Nutrients (nitrogen and phosphorus) contained in highway runoff may cause adverse effects on water quality of receiving waters through: (1) their shock and acute loadings, and (2) their long-term accumulation within the water body as well as associated sediments. Both of these mechanisms may result in water quality deterioration outside the acceptable limits of the general water quality criteria for aquatic life, water use, and recreational uses of receiving water. Receiving water impacts are often site specific and the extent of the problems will depend heavily on rainfall quantities, pollution point sources and their treatment, land use and the sensitivity of the receiving waters.

The use of stormwater detention/retention ponds for storage and attenuation of water peaks is well established. However, the efficiency of these ponds in removing stormwater pollutants has not been fully investigated as a basis for design to improve water quality. The design and operation of these ponds generally depend upon the climatic and geophysical characteristics of the area involved. The actual quantity and quality of the runoff received by the pond is a function of the area's topography, soil characteristics and land use.

Highway pollutants released to adjacent detention/retention ponds may be in particulate and soluble forms. These pollutants may settle to the bottom sediments, adsorb on existing particulate matter, plant and animal species, chemically precipitate out of solution and/or interact with other ions and compounds present. The majority of these pollutants are expected to settle down to the bottom sediments and may recycle or resuspend back to the overlying water when the environmental conditions are favorable.

Thus, the role of the bottom sediments as a nutrient sink and the exchange processes of uptake and release of nutrients in the sediment-water interface will be closely monitored. The importance of bottom sediments and their impact on the overlying surface water has been investigated by many researchers (Lee 1975 and Nichols 1983).

CHAPTER II

LITERATURE REVIEW

Concentrations and Loadings in Highway Runoff

The discharge of nutrient materials which can fertilize or stimulate growth can create substantial problems in receiving waters. Overstimulation of aquatic weeds or algae (eutrophication) can cause dissolved oxygen problems, be aesthetically objectionable, and in severe cases can interfere with recreational activities, causing foul odors and heavy clumps of floating masses at shorelines.

In general, highway runoff characteristics are similar to that of urban runoff and may cause severe shock pollutional loads on receiving waters. The quantity of pollutants on highway surfaces varies widely, however. According to an in-depth study (1974) by URS Research Company on the water pollution aspects of highway surface contaminants where street runoff samples were collected from eight representative cities across the U.S., industrial areas contained substantially higher loadings than commercial areas. Using average time periods of 2 to 10 days between successive street cleanings, either by rain or sweeping, the average nutrient pollution accumulation for the eight cities studied was found to be 0.094 lb NO_3-N/curb mile (0.026 kg NO_3-N/curb km), and 2.2 lb Kjeldahl-N/curb mile (0.61 kg Kjeldahl-N/curb km)(FHWA 1981).

Highway surface contaminants are deposited on roadways via mechanisms which may be related or unrelated to traffic. Loadings of the traffic related depositions will be proportional to total traffic and may arise directly (tire rubber, motor oil) or indirectly (abraded materials from roadway surfaces) from the motor vehicle. The bulk of the traffic-related materials deposited on roadways does not originate directly from the motor vehicle. Products abraded from the highway surfaces are largely inorganic in nature. Most of the nitrogen originates from sources other than the motor vehicle itself and the low levels of traffic-related nitrogen are contributed by soils and plant materials carried onto the roadway by the vehicles.

Mattraw and Miller (1979) studied runoff quality from three land-use areas of Broward County, Florida. The average sampled storm size for the highway site was 0.68 inches and the water quality constituents included total organic nitrogen, total ammonia nitrogen, total nitrite nitrogen, and total nitrate nitrogen, showing average concentrations of 0.53, 0.13, 0.02, and 0.28 mg/l, respectively.

Organic compounds in highway surface contaminants can be classified as: (1) greases and oils from vehicles (including exhaust hydrocarbons), (2) bird and other animal wastes, (3) food litter, and (4) organic nutrients consisting of wood, leaves, grasses and other vegetation wastes. According to a study conducted by Sartor, Boyd and Agardy (1974), algal nutrients were found consisting of phosphates, nitrates and Kjeldahl nitrogen. Kjeldahl nitrogen was found to have the highest loading. This agreed with the findings of Pitt and Amy (1973). Highway runoff contributes the highest loadings of total nitrogen and nitrate nitrogen of the three types of roadways studied.

Precipitation is also a major source of nitrogen in runoff. Kluesener and Lee (1974) found that rainfall contributed anywhere between 20 to 90 percent of the nitrate-N loading in urban stormwater runoff. Thus, most of the ammonia-N and about one-third of the nitrate-N in urban runoff seem to originate from rainfall itself. Precipitation will contribute varying amounts of nutrients, which accumulate through natural atmospheric processes. Qualitative data on nutrients added to precipitation by the leaching of vegetation is lacking, but it is known that some nitrogen and phosphorus are added to the storm runoff through leaching of the ground vegetation (Chancellor 1975).

Fate of Nitrogen in Natural Systems

Surface water bodies with long detention times, such as lakes and estuaries, tend to concentrate nutrients and other pollutants in both the water column and bottom muds. These pollutants can be resuspended and become available to plant growth when anoxic conditions and favorable environments exist. In Lake Eola, in Orlando, Florida, urban runoff was found to be the major source of lake degradation. Concentrations and loading rates discharged to Lake Eola indicate an average total Kjeldahl nitrogen (TKN) loading of 32 kg/ha-yr and an average total phosphorus loading of 4.8 kg/ha-yr (Yousef 1981).

Control of flow of a limiting nutrient is essential if control of the eutrophication process is to be achieved. The limiting nutrient in a water body or a segment of the water body can be determined by measuring the available nitrogen, phosphorus and other elements during the period of maximum phytoplankton biomass. The available nitrogen concentrations are generally the nitrates (NO_3-N) and ammonia (NH_4-N). If the available N concentrations are reduced to about 30 to 50 μg/l or so, N is likely to be the limiting nutrient (Lee et al. 1980).

Nitrogen in the bottom sediments of natural systems is usually unavailable to the overlying water (McPherson et al. 1976). Under anaerobic conditions, when the sediment is agitated, when urea is excreted from benthic organisms, or when organic cells break down, ammonia may be released. The bottom materials act as a nutrient sink by accumulating organics that have settled and by absorbing ammonia onto clays, metal oxides, hydroxides, and organic colloids. The process of denitrification under anaerobic conditions will reduce the accumulation of nitrogen in bottom sediments.

Transformation Mechanisms

The nitrogen cycle is a complex one since it involves more than a cycling from the atmosphere through the producer, consumer and decomposer organisms. Although organisms live in a nitrogen-abundant atmosphere, nitrogen gas cannot be used directly by most forms of life. Thus, the apparent main nitrogen source, the atmosphere, is not the crucial one; the inorganic forms of nitrogen (ammonia, nitrite and nitrate) and the organic forms (urea, protein and nucleic acids) are the essential reservoirs in the nitrogen cycle. The atmospheric nitrogen gas must be fixed into these organic and inorganic forms before it can be used for biological processes (Buthonex 1977).

Although nitrogen fixation (the conversion of N_2 gas into inorganic or organic nitrogen compounds) can occur by both physiochemical and biological reactions, biological fixation is the most significant process. Bacteria, fungi and algae are the major nitrogen fixers.

The nitrates are assimilated and converted into organic forms in nucleic acids, protein, and other complex molecules in the producer, consumer and decomposer cycles. These, in turn, form the wastes of dead organisms on which bacteria and fungi act. The nitrogen is converted to ammonia as it is released from the organic wastes. Other organisms convert the ammonia into nitrites, then into nitrates, and finally back into nitrogen gas to complete the cycle.

Organic wastes containing organic nitrogen or ammonia are assimilated by bacteria that consume oxygen in the streams. When the organic content is too high, the rate of oxygen consumption exceeds the replenishment rate from the atmosphere and an oxygen depletion occurs. Game fish usually require approximately 4 to 5 mg/l of dissolved oxygen to thrive. Other rougher fish species, such as shad and gar, can exist at approximately 1 to 2 mg/l, but such a low level is highly undesirable.

Only a few species of bacteria can consume oxygen through the biological oxidation of nitrogen compounds. Ammonia can be oxidized to nitrite by Nitrosomonas species and nitrite is oxidized by Nitrobacter species.

Under anoxic conditions, many facultative and strictly anaerobic bacteria are able to use nitrate as a terminal electron acceptor reducing it to nitrous oxide (N_2O) and dinitrogen gas (N_2). These products are, generally speaking, not available for biological incorporation and, therefore, denitrification represents a nitrogen sink for most ecosystems. Denitrification generally occurs at or near an interface between anoxic and oxic environments, such as a sediment-water interface. Nitrate produced by nitrification in the oxygenated zone diffuses into the anoxic zone, where denitrification takes place (Snyder 1981).

Fate of Nitrogen in Swales

The State of Florida requires that stormwater originating within a new project or development should be managed and treated within the boundaries of the development. Most of the available management techniques, however, are based on hydraulic considerations of the flow and either ignore or minimize quality parameters. For example, roadside swales are designed to drain, transport and control highway runoff waters from paved surfaces. They are built to reduce the volume of runoff from a drainage basin and attenuate the peak of runoff hydrographs. A survey of various grassy swales in the Central Florida area showed that they are built with longitudinal slopes between 0.06 and 3.14 percent and lateral slopes between 8 and 29 percent (Anderson 1982).

Very little or no qualitative investigations have been undertaken to determine the fate of pollutants in these systems. Therefore, two swale sites representative of this area were selected to study removal efficiencies of highway pollutants through roadside swales.

Fate of Nitrogen in Detention/Retention Basin Systems

The fundamental purpose of a stormwater detention basin is to reduce the rate of runoff from a contributory sub-basin by providing temporary storage of excess runoff. Therefore, the primary design considerations are, of course, the volume of storage required and the maximum permitted release rate (Poutneu 1974). Also considered a major purpose of detention basins is the reduction of pollutional content to improve the water quality of runoff before further discharge to receiving bodies. Among these pollutants are nutrients, particularly nitrogen and phosphorus. Nutrients present in urban stormwater runoff could cause significant water quality deterioration in receiving water bodies. Surface water bodies with long detention times, such as lakes and estuaries, tend to concentrate nutrients and other pollutants in both the water column and bottom sediments. Some of these pollutants can be resuspended and become available to the plant growth when anoxic conditions and favorable environments exist. In Lake Eola, urban highway runoff was found to be a source of lake degradation (Wanielista et al. 1981). Concentrations and loading rates discharged to Lake Eola indicate an average total Kjeldahl nitrogen (TKN) loading of 32 kg/ha-yr and an average total phosphorus loading of 4.8 kg/ha-yr.

Control of flow of a limiting nutrient is essential if control of the eutrophication process is to be achieved. The limiting nutrient in a water body or a segment of the water body can be determined by measuring the available nitrogen, phosphorus and other elements during the period of maximum phytoplankton biomass. The available nitrogen concentrations are generally the nitrates (NO_3-N) plus ammonia nitrogen (NH_4-N). If the available N concentrations are reduced to about 30 to 50 μg/l or so, N is likely to be the limiting nutrient.

Several methods are used to determine the limiting nutrients. The ratio of available N to available P can be used to indicate the potential limiting nutrient. Theoretically, the required weight ratio of these nutrients by algae is 7.5 N to 1 P. If N:P is less than or equal to 5:1, the limiting nutrient is most likely to be nitrogen. If N:P is greater than or equal to 10:1, the limiting nutrient is most likely to be phosphorus. In between these ranges, the limiting nutrient can be either one.

A study was conducted by Yousef et al. (1981) to determine responses of Lake Eola water to pollutional loadings, particularly phosphorus and nitrogen. Available nitrogen was estimated at 70 percent of total nitrogen. The values of available N:P varied between 8.0 and 52.1 and averaged 21.5. These data suggested that Lake Eola was phosphorus limited most of the time.

Sediment retention for nitrates and total Kjeldahl nitrogen on an annual basis were estimated to be 85.8 and 77.6, respectively.

The stormwater quality characteristics in detention basins were investigated in a study by Ferrara and Witowski (1981). The drainage area of 637 acres to the detention basin is characteristic of rural-suburban development. The study presented the results of a stormwater quality sampling program conducted to describe the time variable influent and effluent concentrations of chemical oxygen demand, total phosphorus, total Kjeldahl nitrogen (TKN), and solids during various storm events. Concentrations in three separate particle size ranges for each of the four parameters were determined. This allowed for the development of a relationship between particle size distribution, settling velocity and efficiency removal.

The data demonstrated that pollutant reduction on an average concentration basis is higher for the most intense storms which carry a greater percentage of the larger size fractions. Therefore, percent reduction in total solids is and must be a function of the influent particle size distribution. Percent solids reductions were 36.2, 14.7 and 46.7 for Storms One, Two and Three, respectively.

The TKN data show that only Storm Two provides a reduction of TKN on an average concentration basis. Storms One and Three exhibit an increase in total TKN concentration of approximately twenty percent. A similar conclusion was drawn by Oliver and Grigoropouls (1981) in studying the control of storm generated pollution using a small urban lake. They demonstrated that, on an average basis over their study period, although organic-N decreased by 31%, NH_3-N was found to increase by 13%. The increase in nitrogen concentrations may be a result of water column-sediment interaction within the detention basin. Settleable organic nitrogen may be entering the basin, settling, and during dry weather converting to dissolved NH_3-N. An additional factor may be a separate input of nitrogen to the basin as a result of nitrogen fixation by resident algae.

The conclusions of the study revealed that the basin was generally effective in reducing TP, particularly for the largest storm (over 40% reduction). However, TKN concentrations and loadings were generally increased.

Randall (1982) concluded that quality management could be easily incorporated into flood management strategies such as the utilization of detention ponds, and in a few cases, the protection of water quality may be the most critical need. The Oceoquan Watershed of Northern Virginia is an area where water quality was concluded to be the primary reason for stormwater runoff management.

Enactment of stormwater runoff management ordinances resulted in the installation of a wide variety of stormwater management procedures including the use of dual purpose detention ponds, both wet and dry. Wet ponds are facilities which retain a column of water between storm events. Unfortunately, while it seemed obvious that such facilities would reduce stormwater runoff pollution, the actual effectiveness of the various procedure was not known and, therefore, extensive monitoring was implemented.

Most pollutants of concern have a high affinity for the suspended solids in runoff and for soil particles. Thus, the most logical way to achieve pollutant removal is through sedimentation and infiltration. Consequently, detention pond design for water quality control should maximize settling to the extent possible, and the considerations may alter typical design. In general, quiescent short-circuiting should be minimized.

As part of the Nationwide Urban Runoff Program (NURP), the efficiencies of three in-operation detention ponds located in the greater Washington, D.C. area were determined by inflow and outflow monitoring. Two were wet ponds with full pool volumes of 33,000 m^3 and 36,000 m^3, and mean depths of 2.6 m and 2.3 m, respectively. The third was a dry pond with a storage volume of 3,520 m^3. The wet ponds drained single family residential areas of 11.0 and 19.4 ha with respective dwelling unit densities of 7.4 and 3.0 per ha. The dry pond drained an area of 13.9 ha with a dwelling density of 15.1 per ha. Percent impervious was 24.2 for the low density wet pond area, 33.5 for the high density one, and 30.5 for the dry pond area. A total of 259 storms were monitored at the three sites.

The percent removals of nitrogen and phosphorus forms by the wet ponds were reasonably similar, but the pond with the greatest percent TSS removal produced the best quality effluent for all parameters measured, as shown in Table 1. In contrast to the wet ponds, the dry pond had negative removals of all nitrogen forms and was significantly poorer in removals of phosphorus forms.

It appears that the better nutrient removals in the wet ponds were accomplished by biological activity. Both wet ponds had substantial emergent vegetation in the shoreline areas and there was evidence of biological activity in the water column. The principal conclusion is that wet ponds are considerably better for the removal of nutrients from stormwater runoff, but no better than dry ponds for suspended solids removal.

The trap efficiency of detention basins for sediment has been studied and has been correlated with laboratory determinations of grain size and particle settling velocity in still water (Chen 1974; Bondurant 1975; and Ward, Haan, and Barfield 1977). However, these results cannot be satisfactorily translated into pollutant settleability results because of differing specific gravities of the various particles, and because the association of several pollutants with particulates has not been adequately described. In fact, very little information is available regarding the actual settleability characteristics of urban runoff pollutants. Virtually the only study available in the literature is one conducted by Whipple and Hunter (1981).

The results obtained by Whipple and Hunter indicate a close association between hydrocarbons and suspended solids. The percent removals were the same for a 32-hour settling period, and the range of values was very nearly the same. Lesser removals of other pollutants

TABLE 1

PROJECTED ANNUAL PERFORMANCE BY THREE IN-PLACE PONDS
STUDIED FOR REMOVAL OF NITROGEN AND PHOSPHORUS

Pollutant	Burke Pond - Wet			Westleigh Pond - Wet			Stedwick Pond - Dry		
	Inflow (kg/ha/yr)	Outflow (kg/ha/yr)	Removal (%)	Inflow (kg/ha/yr)	Outflow (kg/ha/yr)	Removal (%)	Inflow (kg/ha/yr)	Outflow (kg/ha/yr)	Removal (%)
Total Suspended Solids	51.3	32.4	36.8	72.0	9.5	86.8	99.0	22.5	77.3
Total Kjeldahl Nitrogen	5.58	3.51	37.1	2.16	1.17	45.8	3.15	4.32	-37.1
Soluble Kjeldahl Nitrogen	3.96	2.38	39.8	1.48	0.76	48.6	2.38	3.15	-32.1
Nitrite and Nitrate	2.38	0.40	83.6	0.81	0.23	71.1	1.17	1.98	-69.2
Total Phosphorus	0.79	0.32	59.2	0.40	0.12	69.8	0.79	0.58	26.2
Total Soluble Phosphorus	0.40	0.17	56.2	0.23	0.12	50.9	0.62	0.45	27.4

SOURCE: William DeGroot, ed. Stormwater Detention Facilities, Proceedings of the Conference on Stormwater Detention Facilities - Planning, Design, Operation and Maintenance, 1982.

were obtained with lead and phosphorus having the next higher removals, respectively. Even though less than 50% of the other pollutants were removed by sedimentation, the removals were significant and would be beneficial in efforts to control the water quality effects of urban runoff. It may be further noted that most of the removal for all pollutants was accomplished during the first 16 hours of settling.

To further investigate urban runoff sedimentation, a laboratory study was designed and implemented by Randall (1982). The purpose of the study was to provide information that could be used to develop specific designs for urban stormwater runoff sedimentation basins. In this study, urban runoff pollutant removal by sedimentation was investigated using seven separate test samples.

Total nitrogen removals during the experiments ranged from 9 to 77%, with an average of 32.5%. Removals were substantially higher for the samples with initial TSS concentrations of 100 mg/l or greater, ranging from 29 to 77% with an average of 47%. The final concentrations achieved did appear to be strongly related to the initial TSS concentration because the percent removed increased as the initial TSS concentration increased. The final nitrogen concentrations were also influenced by the initial total nitrogen concentrations. Surprisingly, the two highest initial total nitrogen concentrations observed were in two samples that had very low TSS concentrations. Removals from these two samples occurred at approximately the same efficiency.

The percent removals of TKN were very similar to those of total nitrogen. Nitrate removals were generally poor as would be expected because they are highly soluble and have very little affinity for sorbing on particulate surfaces. The average reduction of nitrate concentrations in the experiments was less than 10%, although one test achieved a removal of 27%. Ammonia removals were very small in terms of concentration and frequently increased during the settling test. The largest concentration decrease was 0.12 mg/l, whereas the largest increase was 0.19 mg/l.

Characterization of the urban runoff samples demonstrated the highly variable nature of the pollutant concentrations that may be observed. The data also showed that while most of the parameters measured have a general tendency to increase in concentration as the TSS increase, there were some notable exceptions. Thus, generalizations relating pollutant loads to the TSS loads should be done cautiously, if at all. In fact, the ratio of pollutant to TSS tended to decrease as TSS increased for all parameters except COD and BOD_5, and materials subject to rapid change by microbial metabolism such as ammonia and nitrate.

The pollutants in urban stormwater runoff can be substantially reduced by gravity sedimentation but the concentrations remaining in the water column may still be quite high after settling has been accomplished. TSS, lead and BOD, in that order, were removed the most efficient way and substantial reductions were obtained for all detectable pollutants measured except soluble nitrogen forms.

Wetlands Storage/Treatment Systems

Detention-retention marshes are transitional ecosystems located in watersheds between terrestrial and aquatic ecosystems. They are usually wet, dominated by herbs, and have a highly mineralized soil. These wetlands receive a substantial amount of organic and inorganic matter. They act as nutrient sinks, removing nutrients (nitrogen and phosphorus) from the water leaving the wetlands, thereby improving the water quality (Winkellman 1981).

The use of wetlands for nutrient removal has come into focus in the last decade. Wetlands have been utilized for storage and treatment of both stormwater and secondary wastewater effluent. Wastewater effluent has been applied to many types of natural wetlands from Florida to Canada's Northwest Territories. In all of these studies, some nitrogen and phosphorus was removed from the wastewater as it flowed through the wetland. However, a more quantitative assessment is needed of the capacities and limitations of wetlands to remove nutrients.

Nichols (1983) reviewed the mechanisms by which wetlands remove nutrients and, at the same time, attempted to develop a relationship between wastewater N and P application rates and efficiency of N and P removal by wetlands.

The N cycle in wetlands is extremely complex. Nitrogen exists in a multitude of organic forms, as inorganic NH_4^-, NO_2^-, NO_3^-, and as gaseous NH_3, N_2 and N oxides, and is converted from one form to another by a variety of biochemical and chemical processes.

Denitrification is an obvious mechanism for removing N from wastewater in wetlands. When oxygen is lacking, facultative anaerobic bacteria use NO_3^- in place of free O_2 as the terminal exogenous H acceptor in respiration. Organic carbon compounds serve as H donors. In this process, NO_3^- is the first converted to NO_2^-, then to gaseous N_2O and N_2. Except for the terminal enzymes, the electron transport system is the same under anaerobic and aerobic conditions. Many facultative anaerobic bacteria - primarily those in the genera *Pseudomonas*, *Achromobacter*, *Bacillus* and *Micrococcus* - are capable of this reaction.

Denitrification occurs much more slowly under acid conditions than at neutral or alkaline pH. At pH of less than 6, the further reduction of N_2O to N_2 is strongly inhibited. Below pH 5, chemical rather than biochemical reactions can convert N to gaseous forms. At low pH, NO_2^- is unstable and will react with amino acids, ammonia and urea to form N_2 gas. Soil organic matter, or some component of it, seems to increase NO_2^- instability. The disappearance of N from acid organic soils may result as much from the chemical breakdown of NO_2^- as from microbial denitrification. Chen et al. suggested that the loss of NO_3^- added to an acid lake sediment (pH 4.9) may have been caused by microbial reduction of NO_3^- to NO_2^- followed by chemical conversion to gaseous N.

In studies in which NO_3^- was mixed into lake sediments or wetland soils that were then maintained under anaerobic conditions, as much as 90% of the added NO_3^- disappeared within a few days. This was either from denitrification alone or denitrification plus some microbial immobilization. Under more natural conditions, the rate of NO_3^- diffusion to the anaerobic portion of the sediment or soil is often limiting. Nitrification of NH_4 to NO_3^- takes place in the oxygenated surface layer of the soil or in the overlying water. The NO_3^- then diffuses through the aerobic layer to the anaerobic portion of the soil where it is denitrified. The rate of NH_4 diffusion or a lack of oxygen for nitrification can also limit denitrification.

The wetland vegetation serves a number of important functions in purifying wastewater: (1) filtering and settling of inorganic and organic particulate matter and the nutrients associated with it as the wastewater spreads out and passes slowly through the wetland community, and (2) a substrate for the attachment of decomposer microorganisms, behaving somewhat like a trickling filter in breaking down dissolved organic material.

For emergent wetland vegetation, the soil rather than the water is the major source of nutrients. Klopatek (1975, 1978) calculated a nutrient budget for a stand of Scorpus fulviatilis in a Wisconsin marsh and found that about 17.5 g/m^2 of N and 3.8 g/m^2 of P per year were translocated from the wetland soil to the plant shoots. At the end of the growing season, about 12% of this N and P was transferred to the below-ground portions of the plants and stored over winter, 42% of the N and 58% of the P was leached into the water, and the remainder was found in the dead plant material. Prentki et al. (1978) reported a similar budget for P in a Wisconsin cattail marsh.

Under natural unharvested conditions, the growth of wetland vegetation represents only a minimal annual nutrient sink. Nutrient retention by wetlands is generally the greatest during periods of active vegetation growth and is low during the non-growing season. Release from dead vegetation often results in a net export of nutrients from wetlands at certain times of the year. The death of wetland vegetation is typically followed by the rapid release to the water of 35 to 77% of the plant tissue P and somewhat smaller amounts of N. Lee et al. (1975) concluded that much of the P assimilated by two Wisconsin cattail marshes during the growing season is flushed out during the fall and spring. Cattail marshes adjacent to Lake Wingra, Wisconsin retained 83% of the P input from storm sewers during the summer, but only 1% in the fall and 8% in the spring for annual retention of only 10%. In a study of small artificial marshes to which wastewater effluent was applied, almost all of the P that accumulated in the marshes during the growing season was lost during the fall.

Even this temporary storage of wastewater N and P by wetland vegetation may benefit downstream water quality. Because these nutrients are tied up during the growing season and released during the non-growing season, and because part of this N and P is converted from available to non-available forms, the eutrophication potential of wastewater may be lessened or retarded by its interaction with wetland vegetation.

At low loading rates, wetlands have the capacity to remove much of the P applied, and to continue to do so for many years. However, as the loading rate is increased, efficiency of P removal declines rapidly. At a loading rate of 1.5 g P/m^2-yr, about 68% or 1.0 g/m^2-yr would be removed from the wastewater. If the loading rate quadrupled to 6 g P/m^2-yr, P removal would increase only 2.8 times to 2.8 g/m^2-yr, or a 47% removal. A 10-fold increase in P loading to 15 g/m^2-yr increased P removal only 4.5 times to 4.5 g/m^2-yr, or a 30% P removal (Tilton and Kallec 1979 and Burke 1975).

Data from three different wetlands indicated that for the first year or two of wastewater application, a higher P retention efficiency can be expected, with 90 to 95% retention at low loading rates of 2 to 5 g P/m^2-yr and up to 70% retention at loadings as high as

10 to 15 g/m^2-yr. This efficiency will not be maintained, however. The N removal pattern of wetlands is similar to that for P with high removal efficiency, >70% at low loadings, <10 g N/m^2-yr, and rapidly declining efficiency as loading rates increase (Tilton and Kallec 1979 and Burke 1975).

The rapidly declining N removal efficiency with increasing rates may be because of limits on the rate of denitrification, nitrification, oxygen availability, or NH_4^+ or NO_3^- diffusion. The cattail marsh that was studied in Wisconsin achieved only a 31% N reduction averaged over the whole year at a loading rate of 54 g/m^2-yr, while a Massachusetts cattail marsh removed only a small percentage when loaded at the very high rate of 428 g/m^2-yr.

Hydrologic conditions in a wetland can also effect the removal of N and P. Higher N and P loading rates are generally accompanied by higher hydraulic loadings so that retention times in the wetland are reduced and less time is allowed for N and P removal reactions to occur. At very high loading rates, nutrient removal may be limited primarily to the sedimentation of particulate forms. Wetland morphology is also important. As the depth of water in a wetland increases, the chance for reactions between wastewater nutrients and the wetland soil decreases. On the other hand, a deep-water wetland will have a longer retention time than a shallow-water wetland, given the same hydraulic loading.

A study was performed in which large quantities of stormwater were pumped into the Everglades Conservation Areas, Florida, and the nitrogen and phosphorus uptake was determined. In much of the water pumped into the norther Everglades, concentrations of inorganic nitrogen and phosphorus were relatively high. These nutrients were transported to the canals or into the peripheral marshes. Concentrations decreased sharply within 100 meters (330 ft) or less of the canals, whereas specific conductance remained essentially unchanged within this distance. The sharp decrease in inorganic nitrogen and phosphorus near the canal edge indicated net uptake in these shallow waters (McPherson, Waller and Mattraw 1975). Concentrations of total phosphorus and inorganic nitrogen decreased an average of 3% and 4% per kilometer, respectively, downstream in canals in the conservation areas. This decrease was due partly to net uptake in the canals and peripheral marshes.

CHAPTER III

FATE OF NITROGEN IN ROADSIDE SWALES

Roadside swales are drainage areas for highway stormwater runoff. They are generally covered with grass and they may not have standing water. They are designed according to their ability to percolate water and there is scarce or no information on the fate of runoff pollutants discharged to these swales. Qualitative and quantitative analyses of pollutants discharged do not exist. Therefore, controlled experiments were designed to investigate the fate of nutrients and heavy metals discharged to swale areas located along Interstate 4 at Maitland Interchange and at the Epcot Center Interchange.

Runoff water from adjacent retention/detention pond was dosed with nutrients (N and P) and heavy metals (Pb, Ni, Cr, Cd, Cr, Cu and Fe) and pumped up to flow over a selected area of the swale for approximately 200 feet long. The input and output hydrographs were measured and losses due to percolation and evaporation were measured. Concentrations of nutrients and heavy metals at 25', 75', 125' and 175' from discharge point were measured and removal efficiencies were quantified. The experiments were repeated using three different flow rates and water flow cross-sections were measured to examine the effect of changing hydraulic parameters and flow-through time on removal efficiencies of pollutants.

Experimental Sites

Maitland Interchange Site

The site for this investigation is located at the Interstate 4 and Maitland Boulevard interchange north of the city of Orlando, Orange County, Florida. Maitland Boulevard crosses over Interstate 4 by means of a bridge overpass created during the construction of the interchange in 1976. The traffic lanes on the interstate are separated by a 6.0 m grassy median, as they approach the interchange, which widens to 13.5 m through the interchange. The Maitland Boulevard bridge consists of two sections, one carrying two lanes of east bound traffic plus one exit lane with the other section carrying two lanes of west bound traffic plus one exit lane. The section carrying west bound traffic spans 168 m with a 16 m roadway and also a 16 m horizontal clearance. The section carrying east bound traffic spans 163 m and also has a 16 m roadway and a 16 m horizontal clearance. The traffic volume on Maitland Boulevard approximates 15,000 ADT.

Interstate 4 has three lanes of through traffic east and west bound through the Maitland Interchange. The traffic volume on Interstate 4 through the Maitland Interchange is approximately 39,000-48,000 ADT east and west bound.

Three borrow pits were dug to provide fill for the construction of the overpass, as depicted in Figure 1, and remain in existence serving as stormwater detention/retention facilities. The total design drainage areas for those three ponds are shown in Table 2.

TABLE 2

TOTAL DESIGN DRAINAGE AREAS FOR PONDS LOCATED AT MAITLAND INTERCHANGE

Pond	Location of Pond	Total Drainage Area (ha)
A	Southwest (west)	19.8
B	Northeast	48.6
C	Northwest	10.1

Stormwater runoff from the interstate is delivered by overland flow over grassy swales to storm drain inlets or detention ponds A, B and C. Stormwater runoff from the Maitland Boulevard bridge crossing over Interstate 4 is conveyed directly off the roadway surface through stormwater inlets to culverts that discharge ultimately directly into Pond A. The ponds are interconnected so that the water from the two northernmost ponds flow into the southwest pond (referred

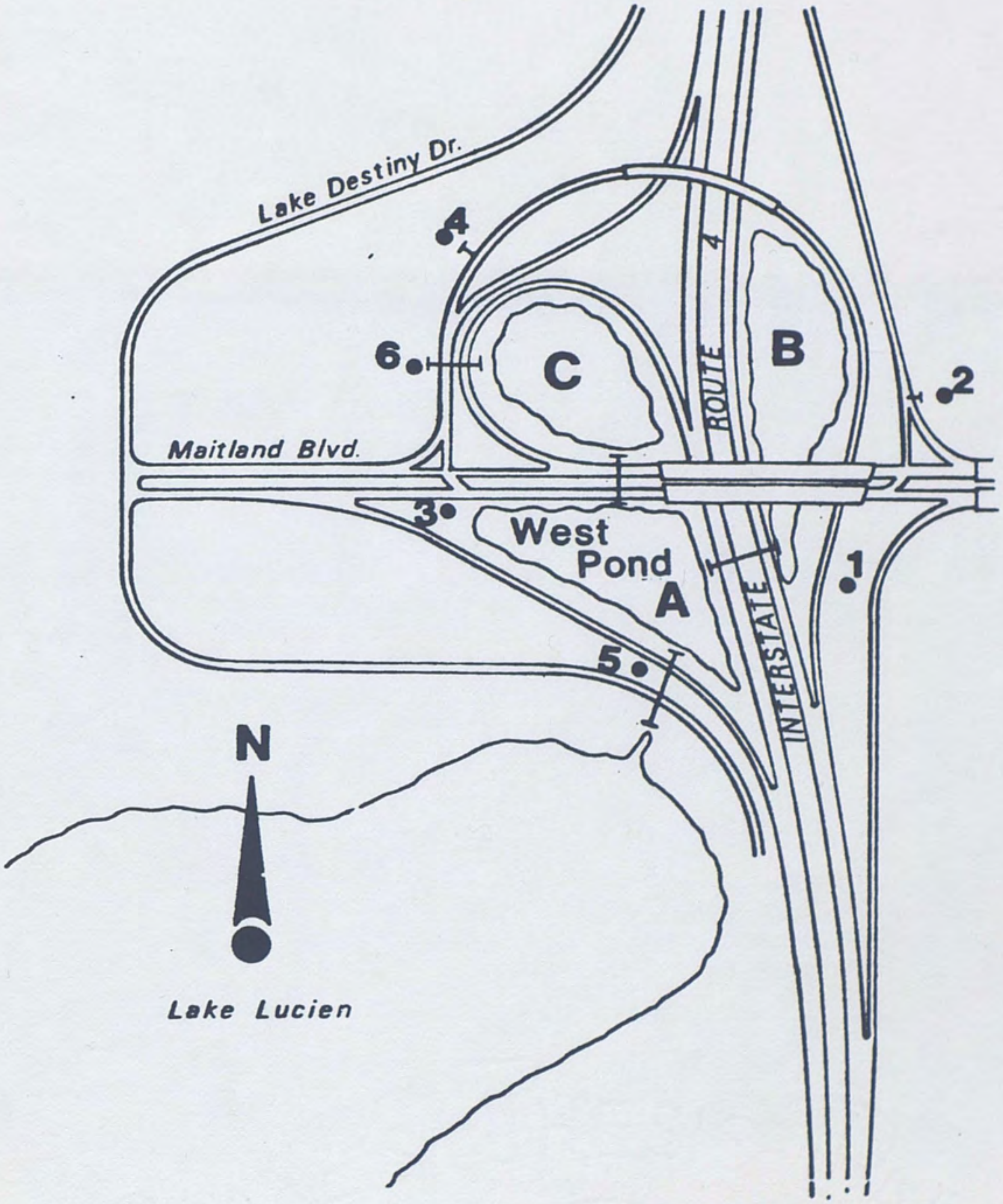

Fig. 1. Sampling site for highway runoff at Maitland Interchange and Interstate 4.

to hereafter as the west pond) when they reach a certain design level. The water from the west pond flows over a wooden weir at its southern end which is connected to Lake Lucien by means of a culvert and a short, densely vegetated ditch.

The west pond is triangular in shape with an approximate surface area of approximately 3 acres or 1.2 ha. The eastern side is parallel to I-4, the northern side is parallel to Maitland Interchange and the third side is parallel to ramp A leading from Maitland Interchange to I-4 west. A grassy swale along the western side of ramp A was selected for this investigation. This swale was ideal because of its accessibility and the availability of a continuous source of runoff water from a drain located at the bottom of the swale and connected with the west pond by a 36 inch diameter RCP, as shown in Figure 2.

During each of the four experiments conducted at this site, water was pumped up through a 2 inch PVC pipe connected to a submersible pump which was placed inside the existing stormwater inlet. A concentrated solution of heavy metals and nutrients was dosed into the water flow at a constant rate to produce a solution with pollutants similar to those normally detected in highway runoff. The spiked water travelled a distance of 175 ft (53 m) through the PVC pipe and then was allowed to flow by gravity over the bottom of the swale and return back to the inlet. The swale area was covered with grasses, predominantly bahia. Cross-sectional areas were taken at

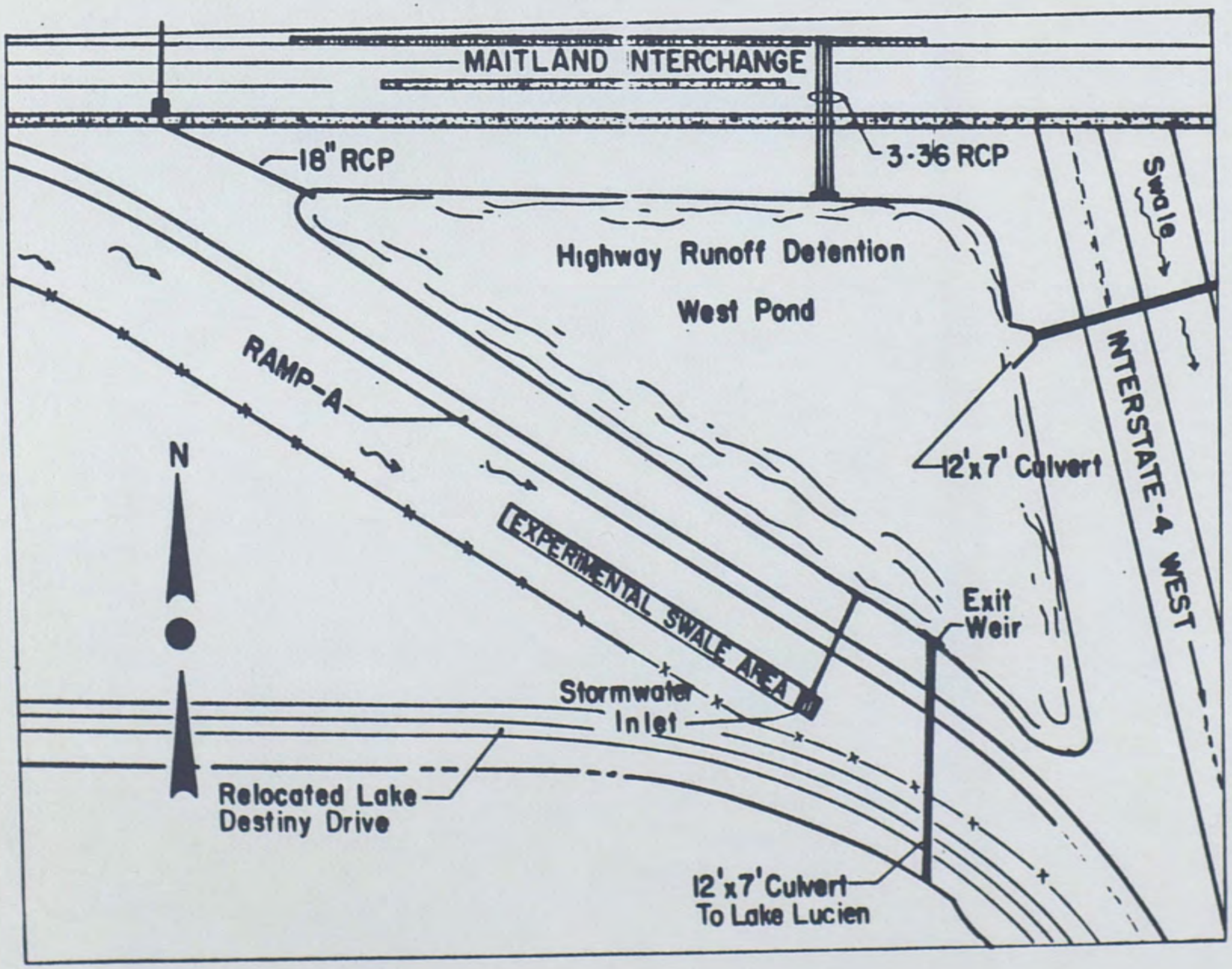

Fig. 2. Location of experimental swale along ramp A at Maitland Interchange and I-4.

25 ft (8 m), 75 ft (23 m), 125 ft (38 m) and 160 ft (49 m) from the water flow outlet at the end of the PVC pipe. The cross-sections were used to determine longitudinal slopes and cross-sectional areas of the water flow, as shown in Figures 3 and 4. A sharp crested 90° V-notch weir was placed before the inlet to measure the flow leaving the swale area at various times during the experiment.

Analysis of Highway Runoff

The Maitland Interchange stormwater sampling program covered an eight month period between August 1982 and March 1983. Six sampling stations were included in the program: Station #1 - a low grassy swale area receiving runoff from I-4; Station #2 - direct highway runoff from Maitland Boulevard; Station #3 - direct bridge and highway runoff from Maitland Boulevard into a detention pond; Station #4 - direct highway runoff from an I-4 exit ramp; Station #5 - a sandy, grassy swale which receives runoff predominantly from an I-4 entrance ramp; and Station #6 - a grassy swale which receives highway runoff from Station #4. A total of 17 storm events were included in the stormwater sampling program, with various numbers of samples taken at each station. The sampling locations are depicted in Figure 1. Collection of these samples should assist in differentiating between runoff quality from highways before and after flowing over adjacent swales.

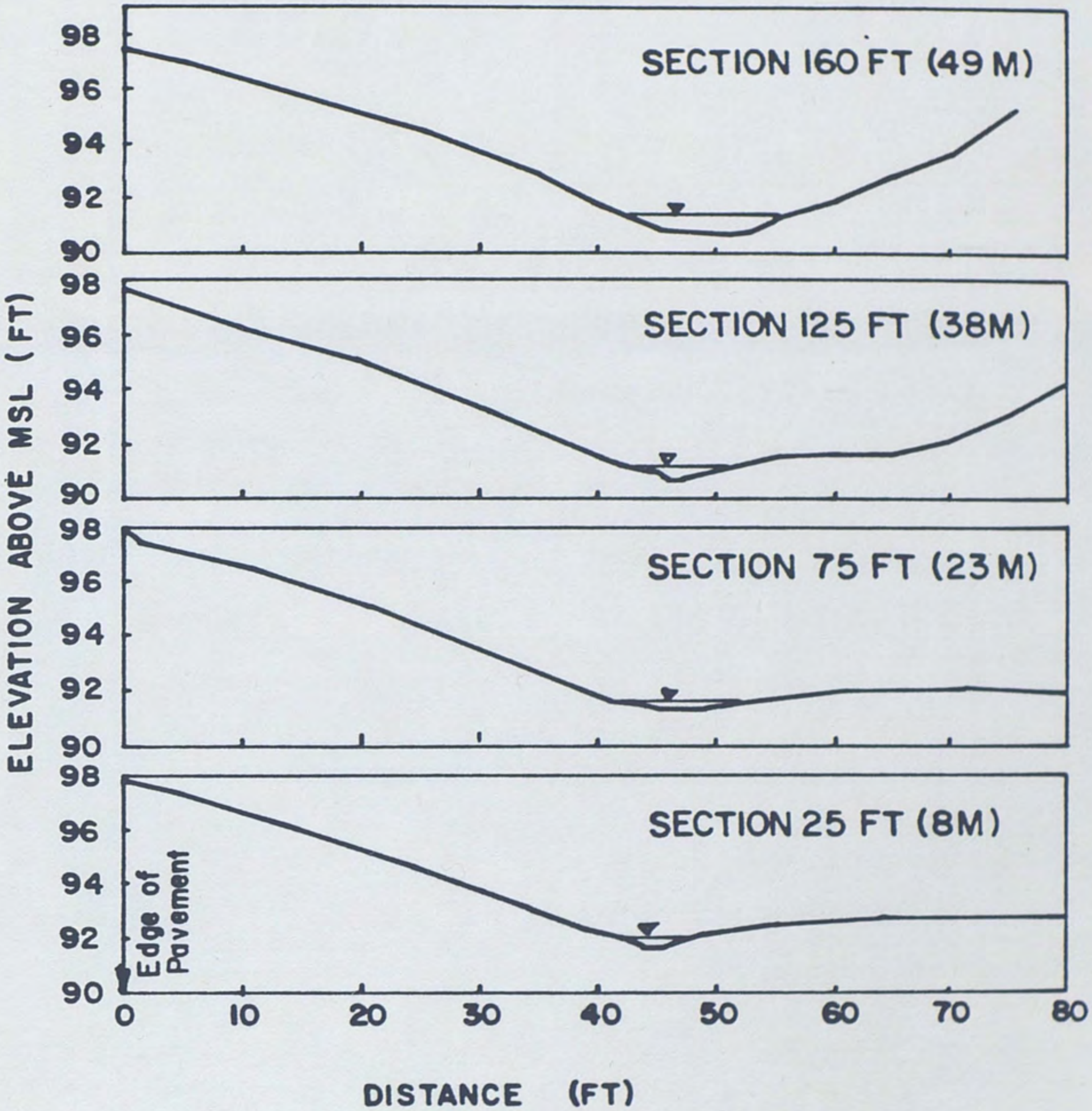

Fig. 3. Cross-sections of grassy swale along ramp at Maitland Interchange and I-4.

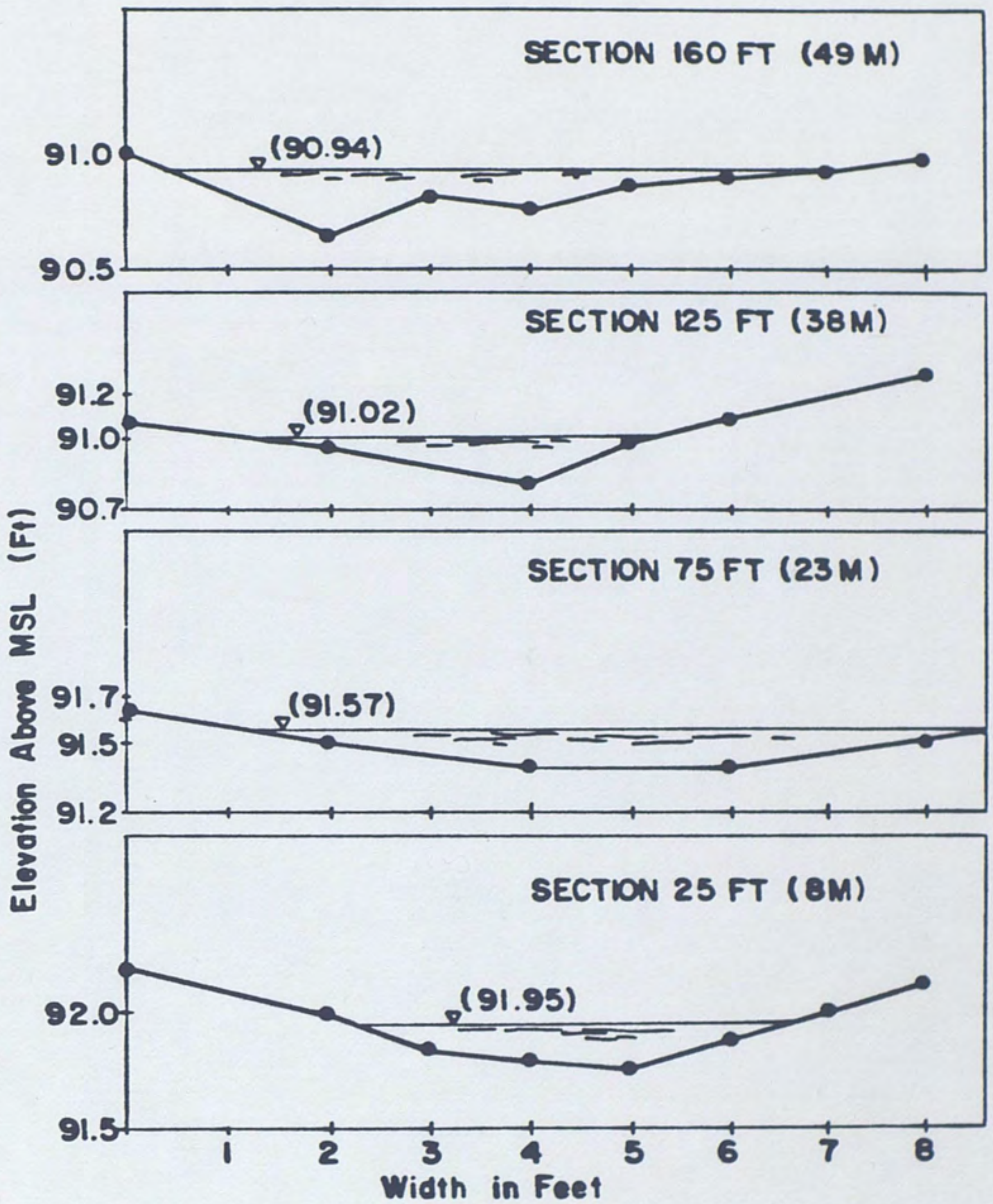

Fig. 4. Cross-sections for water flow through swales adjacent to Maitland Interchange and I-4.

Epcot Interchange Site

Epcot interchange was constructed during 1982-83 as an 0.8 mile multi-lane connector between the Epcot entrance and SR 535, constructed approximately 1.5 miles southwest of the I-4/SR 535 interchange and 2.4 miles northeast of the I-4/US 192 interchange, near Lake Buena Vista in Orange County, Florida, as shown in Figure 5. The swale area selected for the study was a newly constructed swale along ramp A which connected the Epcot Center exit to the west bound lanes of I-4. A detailed map of the swale area is shown in Figure 6. Cross-sectional profiles of the swale area were taken at 10 ft (3 m), 100 ft (30 m), 200 ft (60 m), 315 ft (95 m), 400 ft (120 m) and 535 ft (160 m) to determine longitudinal slopes for the swale and also to calculate cross-sectional areas of the water flow through the swale as shown in Figures 7 and 8. Also, two sharp crested 90° V-notch weirs were installed at 300 ft (90 m) and 550 ft (170 m) from the inlet culvert (S-411) for flow measurements. Two experiments were conducted at this site, one in a predominantly earthen state before the establishment of vegetation in the swale, and the other after vegetation had become established. Grass cover was estimated at approximately 20% in the first experiment and 80% in the second. Similar to the procedure followed at the Maitland site, water was pumped up by a Gould submersible pump from the adjacent detention area through a 2 inch PVC pipe to the 18" RCP connector culvert to the swale area. Hence, it flowed by gravity over the swale area as shown in Figure 6.

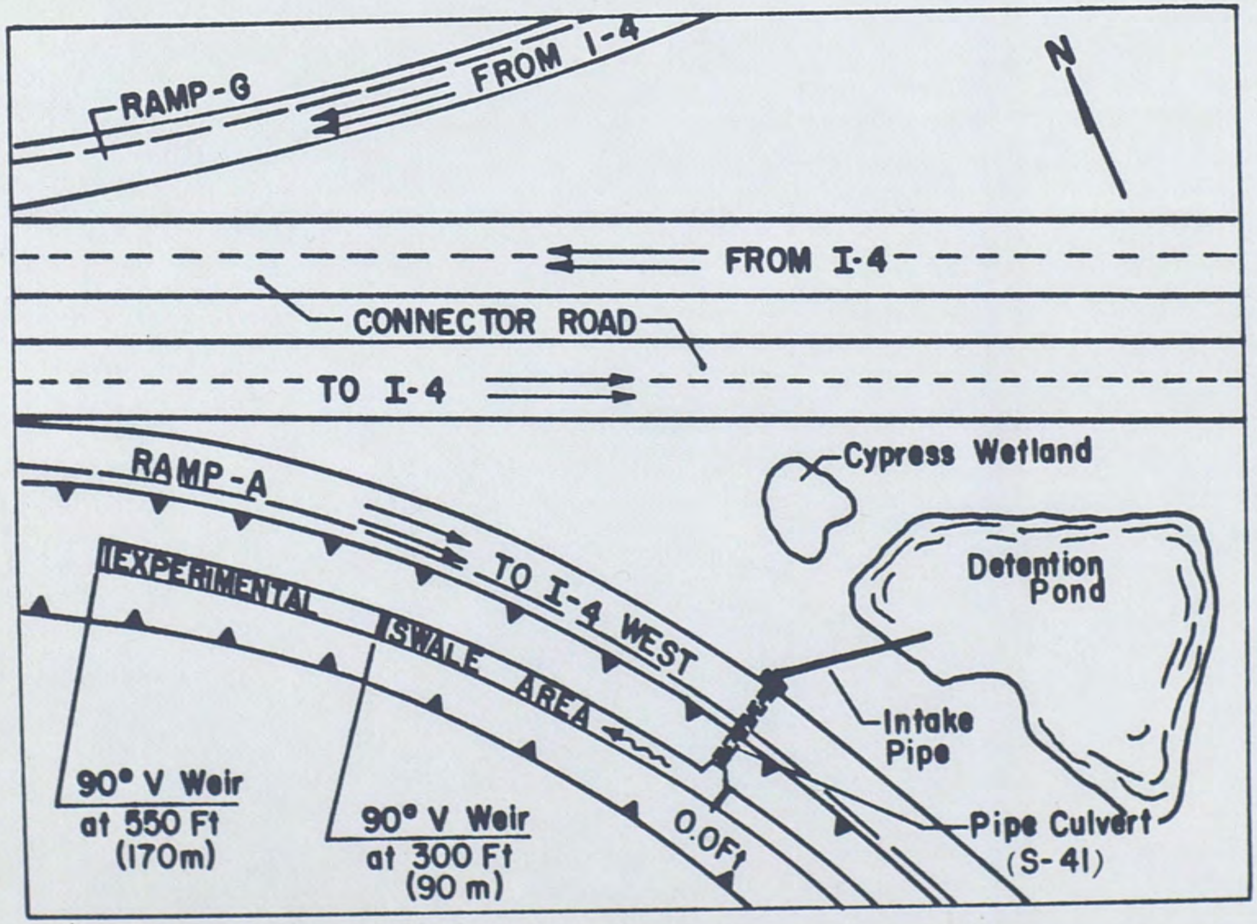

Fig. 5. Highway runoff study area at I-4 and Epcot Interchange.

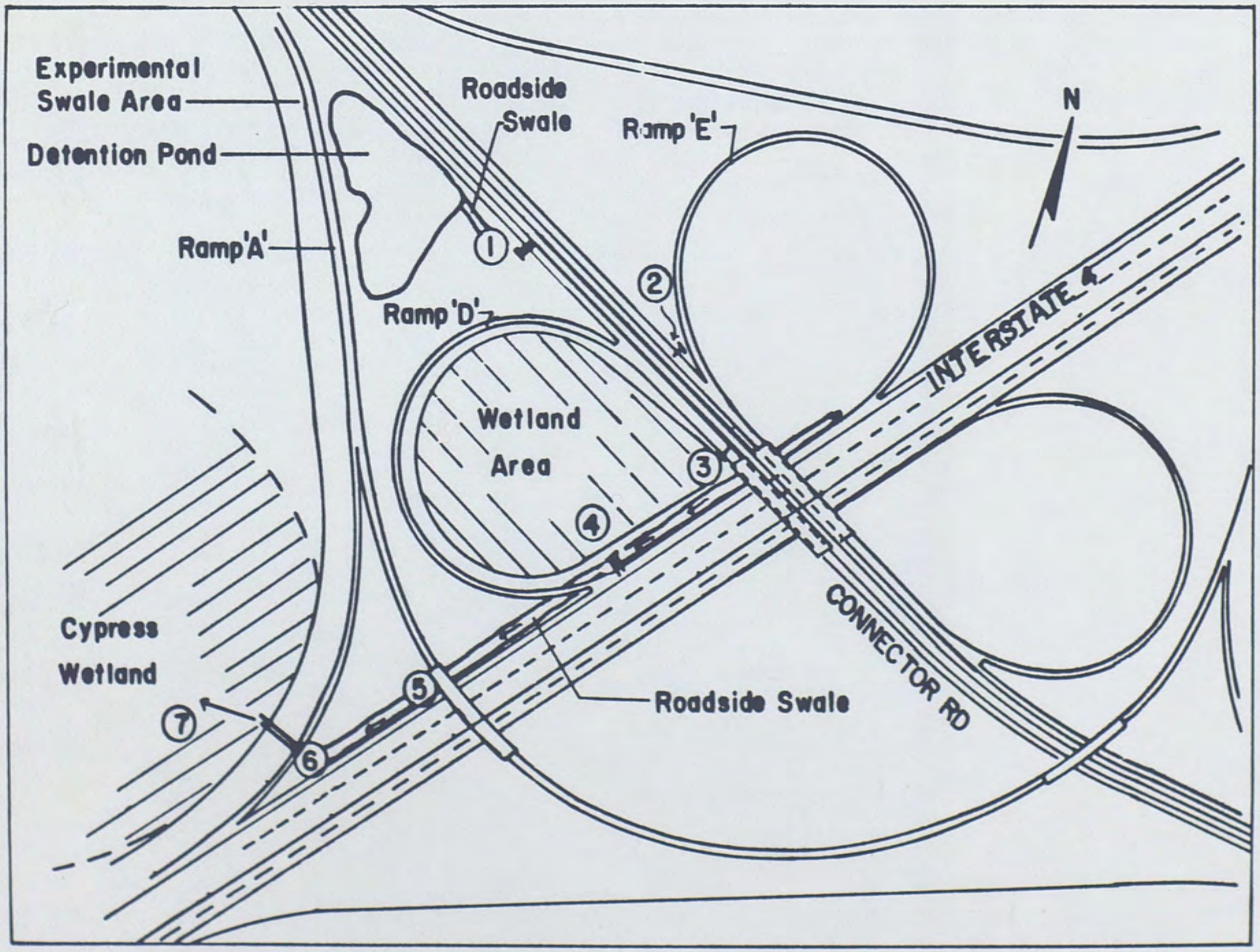

Fig. 6. Location of experimental swale along ramp A at I-4 and Epcot Interchange.

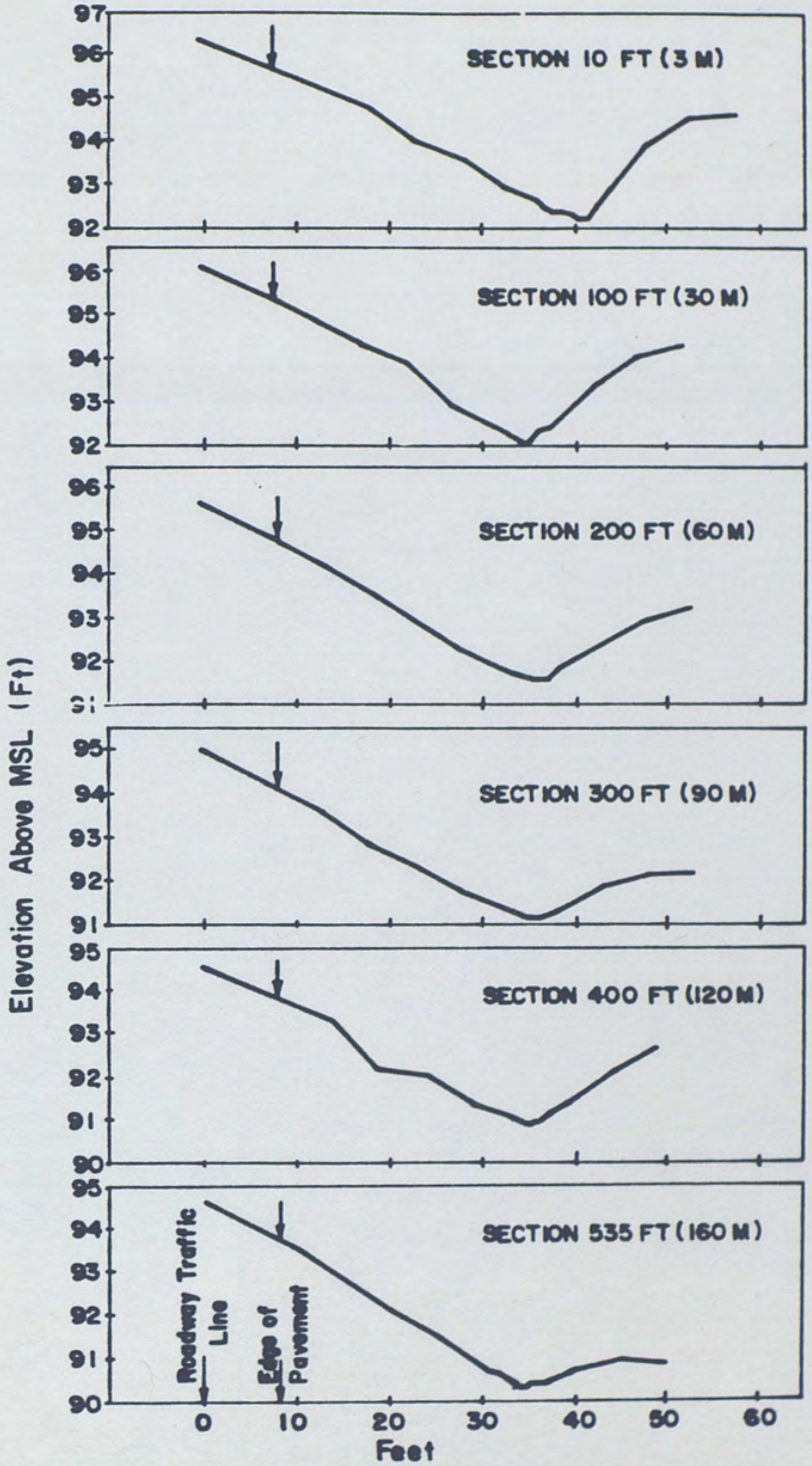

Fig. 7. Cross-sections of grassy swale along ramp A at I-4 and Epcot Interchange.

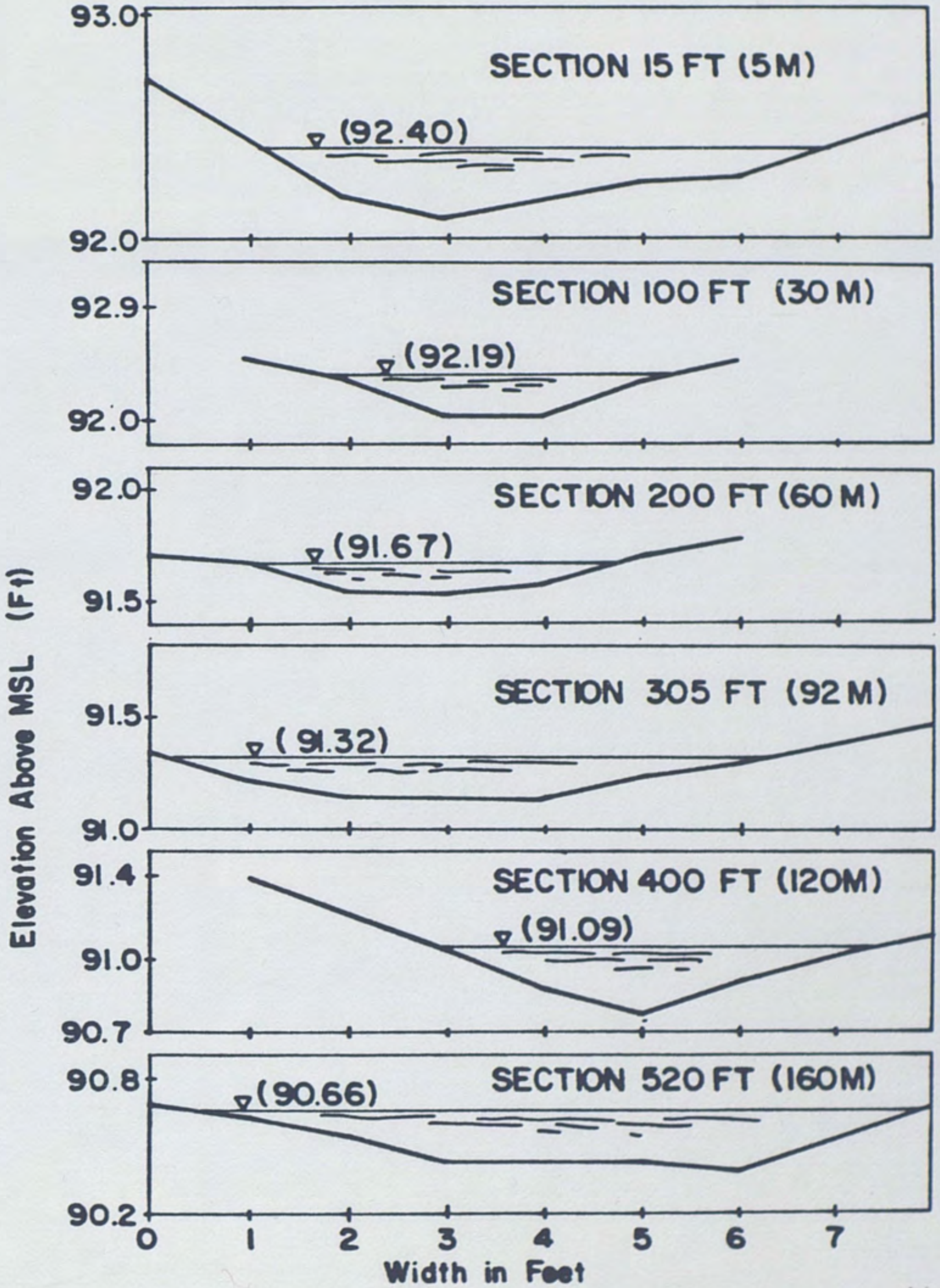

Fig. 8. Cross-sections for water flow through swales adjacent to ramp A at Epcot Interchange.

Field Experimentation

Concentrated solutions were made from selected heavy metals (Pb, Cr, Cd, Ni, Cu, Zn and Fe) and nutrients (P and N) were dosed at a fairly constant rate into the flowing water before entering the swale area. The chemical spikes were dissolved in a 120 liter nalgene container and were fed using constant head gravity flow or a peristaltic dosing pump with controlled flow discharge. At the Maitland site, the chemical solution was fed by gravity near the inlet side of the submersible pump placed in the stormwater inlet used for water intake. At the Epcot site, the chemical solution was dosed by a peristaltic pump at the inlet of the 18 inch pipe culvert (S-41) which crosses through ramp A to the start of the swale area. Every effort was made to keep the concentrations of pollutants fed into the swale areas similar to what would be expected from highway runoff. The spiked water was allowed to flow by gravity over the swale areas at selected discharge rates which were controlled by a gate valve attached to the PVC pipe near the suction side of the submersible pump.

Water flow over the swale area was monitored and initial time to reach various stations were recorded. Grab samples were collected at stated time intervals from the spiked inflow water, and at 25 ft, 75 ft, 125 ft and at the discharge exit weir located 175 ft along the swale in the Maitland area. Similarly, water samples were collected from the Epcot swale area at 10 ft, 100 ft, 200 ft and downstream the discharge from the V-notch weir located at 300 ft, 400 ft and upstream the flow to the exit weir located at 550 ft. Each swale experiment lasted for a time period ranging from 3 to 5.5 hours.

Near the end of each experiment, water depth sections were determined every twenty feet in the experimental swale areas, as shown in Figures 3, 4, 7 and 8. This procedure was necessary to accurately estimate the average cross-sectional area of the water flowing through the swale. Measurements of the inflow and outflow from the swale area and the water cross-sectional areas assisted in the determination of hydraulic and hydrologic parameters for the study areas during these experiments.

Hydraulic and Hydrologic Parameters

Experiments were conducted on January 24, February 7, February 21 and May 31, 1983 using different inflow rates to the swale area at the Maitland site. Also, two experiments were conducted on March 23 and May 16, 1983 using similar flow rates to the swale area at Epcot. The inflow rates for the Maitland area varied between 0.025 and 0.227 m^3/min (7 to 60 gallons/min) and averaged 0.189 m^3/min (50 gallons/min) for the Epcot area. The inflow rate was calibrated periodically during the experiments and adjustments of the values were made to insure constant flow. After cessation of pumping, the flow through the exit weir was monitored until flow no longer occurred to produce the shape of the fall of the hydrograph.

Flow hydrographs were developed as shown in Figures 9, 10 and 11 for the Maitland area and Figures 12 and 13 for the Epcot area. At the 2/21/83 experiment at the Maitland site, the water did not reach the end of the swale and it was totally retained on the site. Water is lost by infiltration, seepage, evaporation, transpiration and on-site storage. The hydrographs reflect clearly the water retention and the excess runoff from swale areas under various inflow rates. Hydrograph characteristics of the swale experiments are tabulated in Table 3. The average loading rates varied from 0.036 to 0.154 m^3/m^2-hr (1.42 to 6.06 in/hr) on the Maitland swale area. These rates resulted in excess runoff averaging 0.0 to 0.068 m^3/m^2-hr (0 to 2.7 in/hr). The Epcot site loading rates averaged 0.053 to 0.105 m^3/m^2-hr (2.08 to 4.13 in/hr) and the excess runoff averaged 0.039 to 0.071 m^3/m^2-hr (1.52 to 2.8 in/hr). The flow rates were calculated from the area under the hydrograph divided by the area of the swale covered with water and duration time of flow. Under the experimental conditions, there was no excess runoff for flow less than 1.42 in/hr. Excess runoff reached more than 90% of average input flow at the Epcot site when the soil was saturated with moisture.

The hydraulic characteristics of the swale experiments are listed in Table 4. The hydraulic water depth, which is defined as the cross-sectional area divided by the top width of flow, did not exceed 0.0041 m (1.6 in). The calculated water velocity varied from 0.90 and 2.98 m/min during the swale experiments under steady-state conditions of flow.

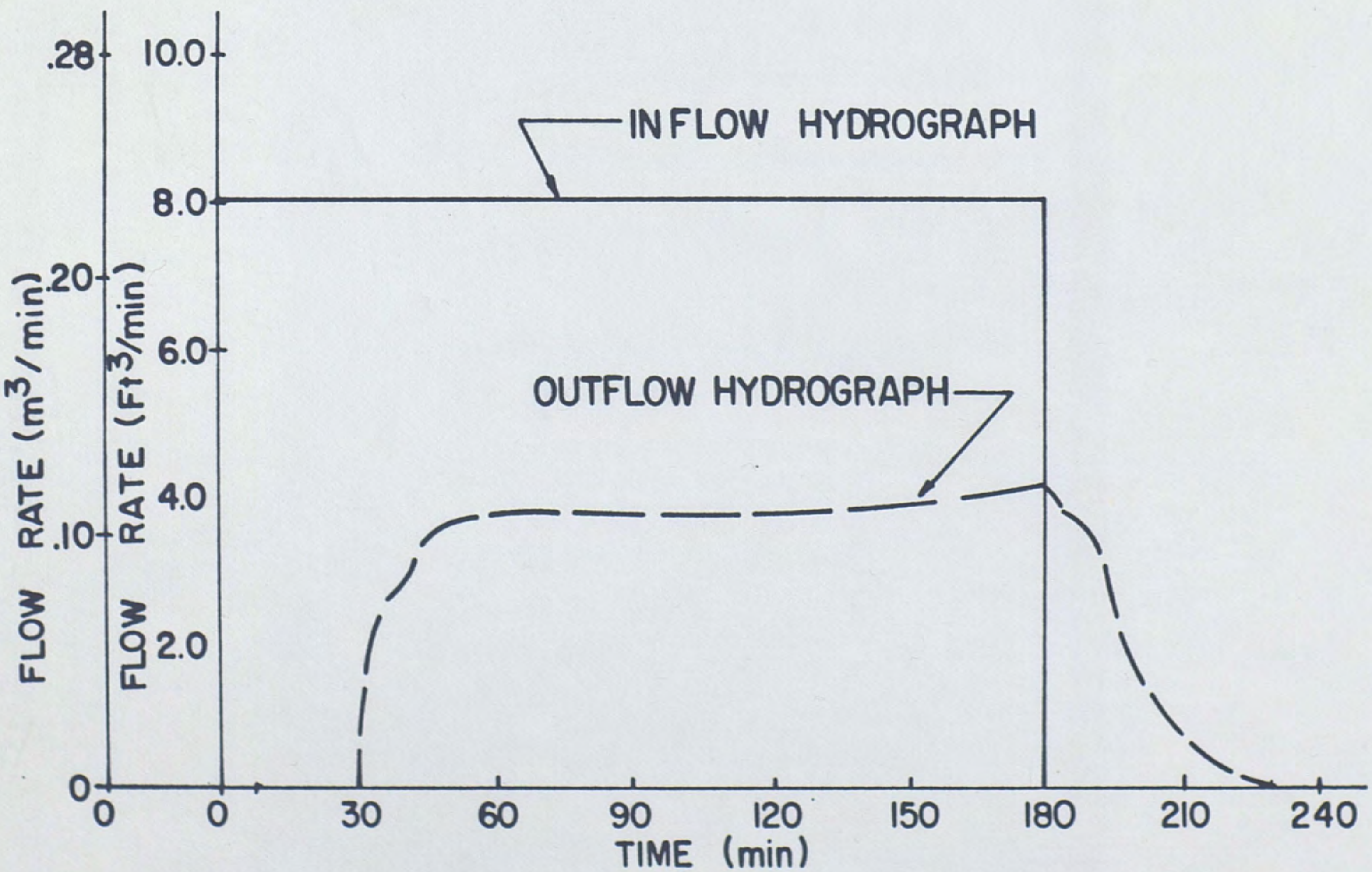

Fig. 9. Flow hydrographs for swale efficiency experiment conducted at Maitland Interchange on 1/24/83.

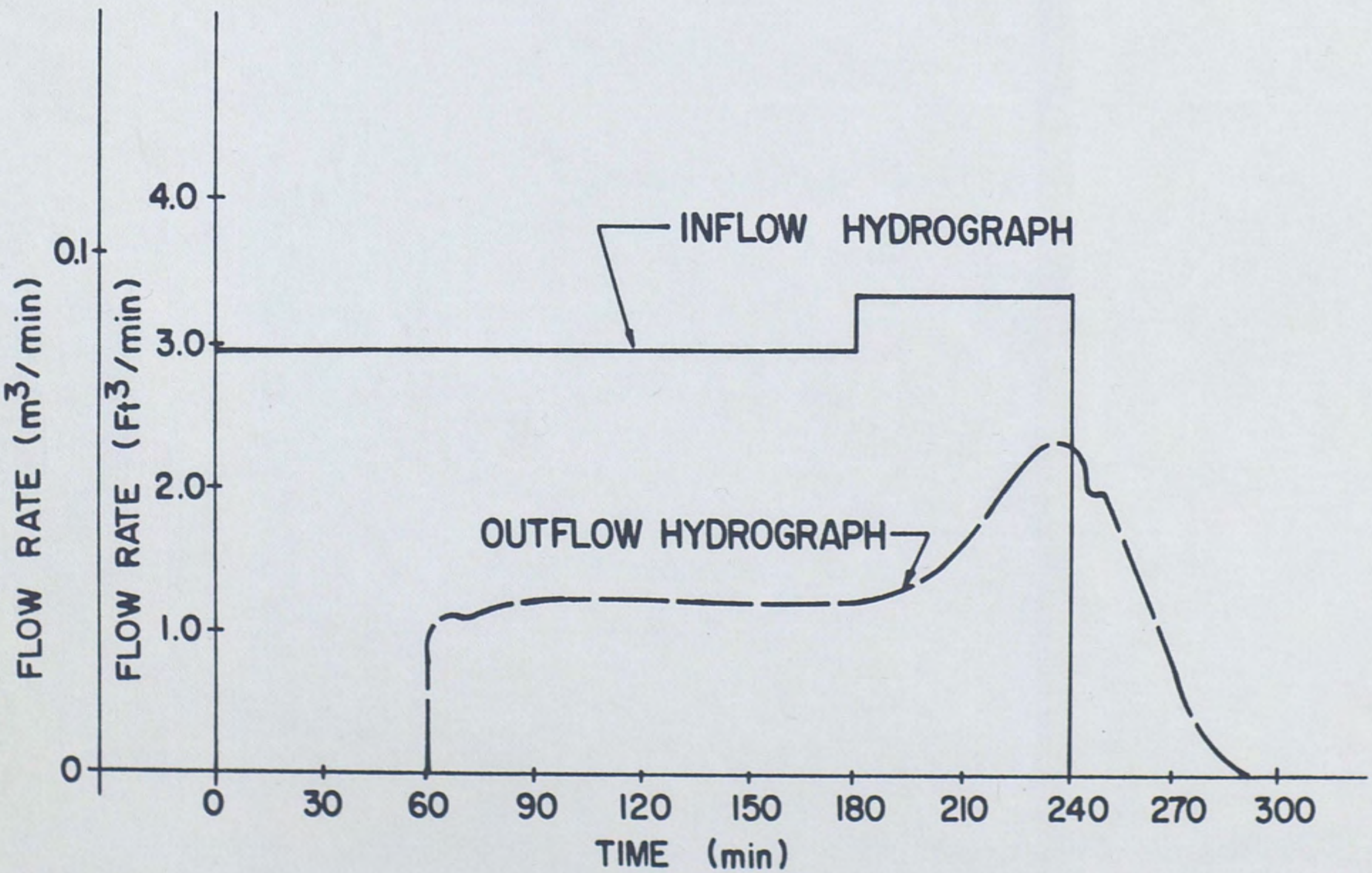

Fig. 10. Flow hydrographs for swale efficiency experiment conducted at Maitland Interchange on 2/7/83.

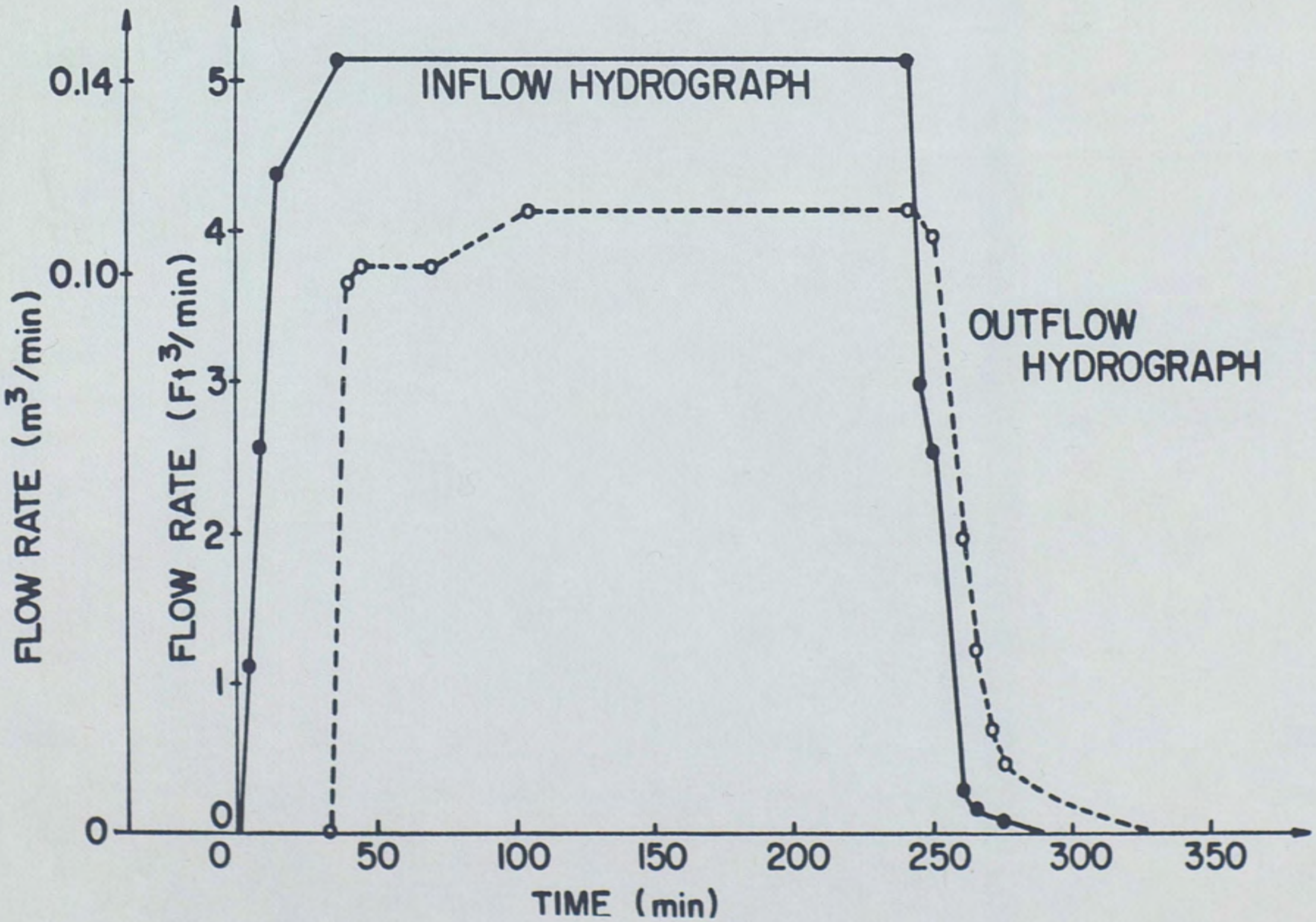

Fig. 11. Flow hydrographs for swale efficiency experiments conducted at Maitland Interchange on 5/31/83.

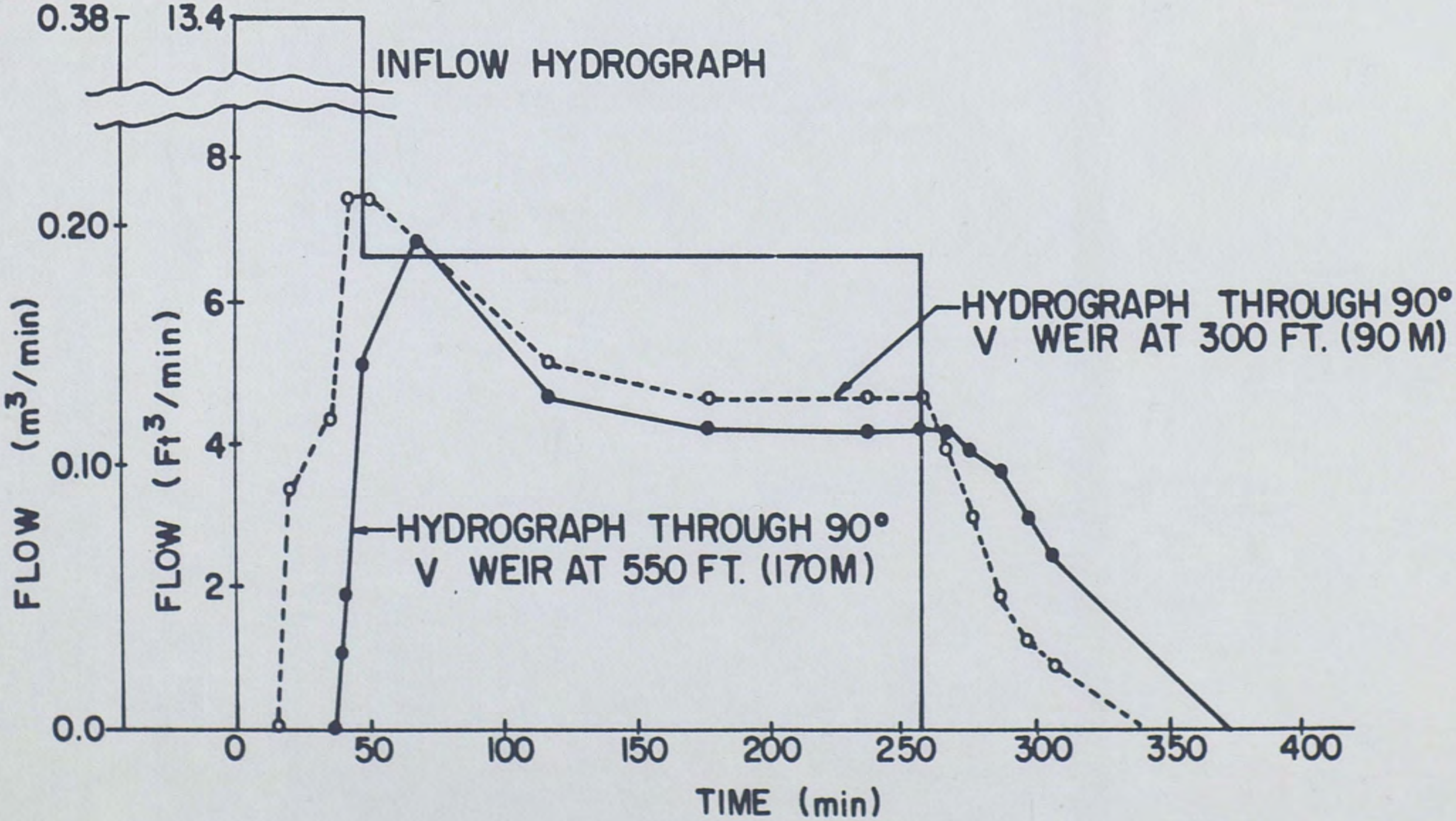

Fig. 12. Flow hydrographs for swale experiment conducted adjacent to ramp A, northwest of S-41 exit at Epcot Interchange on 3/23/83.

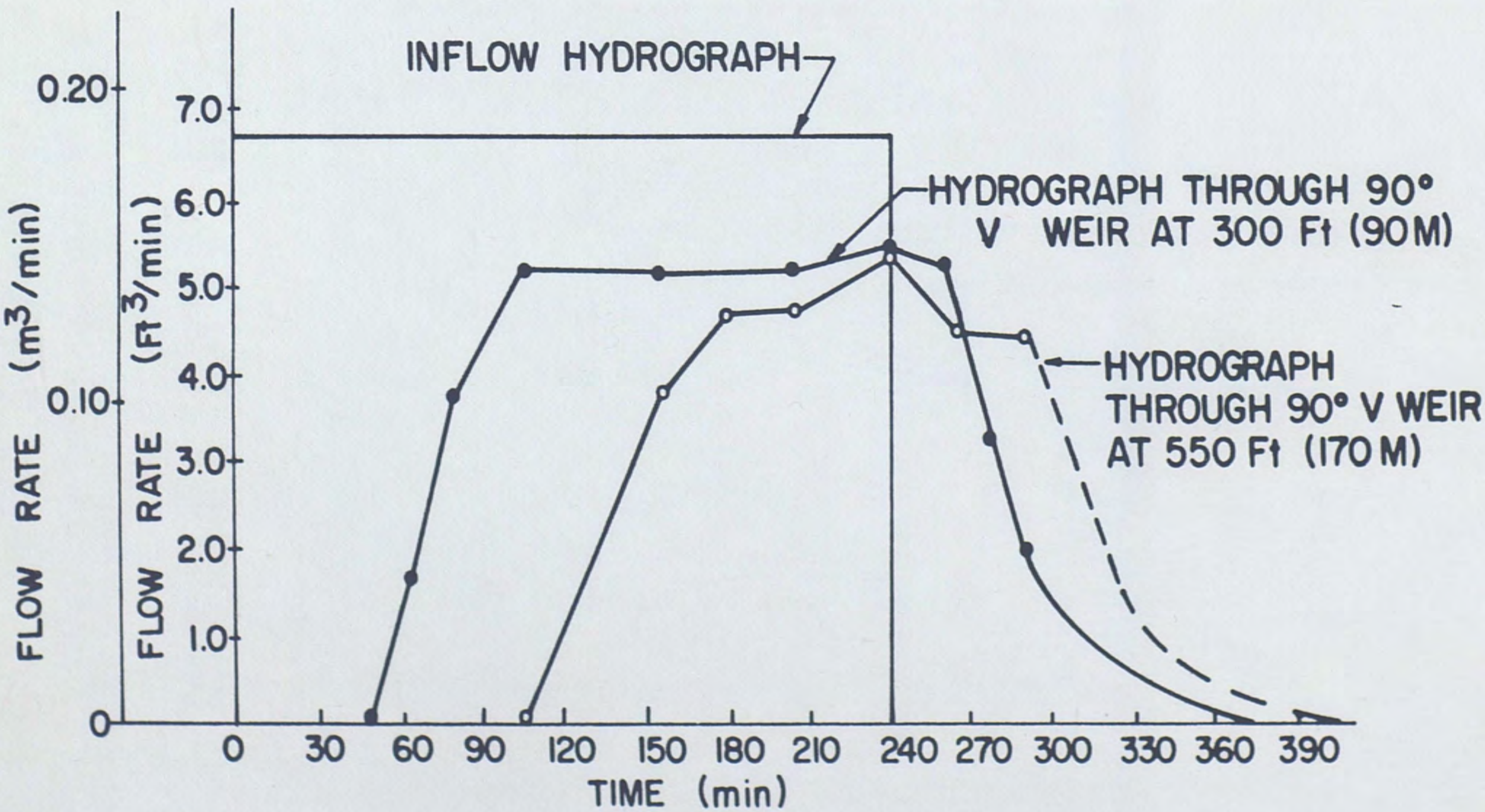

Fig. 13. Flow hydrographs for swale experiments conducted adjacent to ramp A - northwest of S-41 exit at Epcot Interchange on 5/16/83.

TABLE 3

HYDROGRAPH CHARACTERISTICS OF SWALE EXPERIMENTS

Experiment			Flow Characteristics				Loading Rates		Infiltration		Excess Runoff	
No.	Location	Date	Length (m)	Duration (hr)	In (m^3)	Out (m^3)	$\frac{m^3}{m^2\text{-hr}}$	$\frac{\text{Inch}}{\text{hr}}$	$\frac{m^3}{m^2\text{-hr}}$	$\frac{\text{Inch}}{\text{hr}}$	$\frac{m^3}{m^2\text{-hr}}$	$\frac{\text{Inch}}{\text{hr}}$
1	Maitland	1/24/83	53	3.00	40.9	17.6	0.154	6.06	0.088	3.46	0.066	2.60
2	Maitland	2/07/83	53	4.00	20.7	8.3	0.072	2.83	0.043	1.69	0.029	1.14
3	Maitland	2/21/83	49	5.50	8.14	0	0.036	1.42	0.036	1.42	0	0
4	Epcot	3/23/83	90	4.18	57.7	39.2	0.105	4.13	0.034	1.33	0.071	2.80
					39.2	35.8	0.079	3.10	0.007	0.17	0.072	2.83
5	Epcot	5/16/83	90	4.00	46.3	30.8	0.094	3.69	0.032	1.26	0.062	2.43
					30.8	23.0	0.053	2.08	0.014	0.55	0.039	1.52
6	Maitland	5/31/83	53	4.00	35.1	26	0.092	3.62	0.024	0.94	0.068	2.68

TABLE 4

HYDRAULIC CHARACTERISTICS OF SWALE EXPERIMENTS

Experiment			Swale Characteristics				Flow (m^3/min)		Flow Through Time (minutes)		
No.	Location	Date	Length (m)	Cross-Section Area (m^2)	Top Water Width (m)	Hydraulic Depth (m)	In	Out	Average Calculated Velocity (m/min)	Initial To Reach DS Point	Average Calculated
1	Maitland	1/24/83	53	0.063	1.67	0.038	0.227	0.098	2.58	31	21
2	Maitland	2/07/83	53	0.045	1.35	0.033	0.086	0.038	1.37	58	39
3	Maitland	2/21/83	49	0.014	0.85	0.017	0.026	0.000	0.90	330	--
4	Epcot	3/23/83	90 80	0.058 0.060	1.46 1.49	0.040 0.040	0.189 0.131	0.131 0.118	2.76 2.08	-- --	33 39
5	Epcot	5/16/83	90 80	0.056 0.071	1.37 1.83	0.041 0.039	0.189 0.145	0.145 0.131	2.98 1.94	48 57	30 41
6	Maitland	5/13/83	53	0.056	1.79	0.031	0.145	0.118	2.35	35	23

Laboratory Analysis

Grab samples of water flowing over the swale area were collected at different locations and times from the experimental area and transported to the Environmental Engineering Science Laboratory at the University of Central Florida for analysis. Samples were collected in polyethylene bottles and preserved at 4°C. Analyses were conducted within the time frame recommended by the U.S. Environmental Protection Agency, Methods for Chemical Analysis of Water and Wastes (1979).

Nutrients N and P analyses were run as described in Standard Methods for the Examination of Water and Wastewater. Ammonia nitrogen was determined by the phenate method, nitrate nitrogen by the brucine method, nitrite nitrogen by the diazotization technique and the phosphorus by the ascorbic acid method. Calibrations were made and solutions with known concentrations were measured along with the water samples.

Experimental Results

The results from the Maitland Interchange stormwater monitoring program and the six swale studies were analyzed for nitrogen concentrations and nitrogen mass balance.

Nitrogen Removal in Stormwater Runoff

The data from the Maitland Interchange stormwater sampling program from August 1982 to March 1983 revealed some interesting results. In Table 5, most of the nitrogen in runoff samples analyzed was organic which varied between 72 and 92% of the total nitrogen. The organic fractions averaged approximately 80% of total nitrogen in water samples collected from direct highway runoff or adjacent swale areas. The weighted average concentration of inorganic fraction is 388 μg-N/l. Nitrate nitrogen appears to be the major constituent of the inorganic nitrogen in highway runoff.

It is of interest to analyze differences in nitrogen concentrations in samples collected from highway runoff and compare them with runoff samples after flowing over adjacent swales. Station 4 receives direct highway runoff from an I-4 exit ramp, while Station 6 is located in a swale which receives runoff from Station 4. The data shown in Table 5 indicate no difference between total nitrogen in runoff samples from Station 4 and Station 6. However, the nitrogen concentrations at Stations 4 and 6 were averaged over two separate time periods, storm events in months of warmer climate (Column A) and storm events in months of colder climate (Column B), as depicted in Table 5. It appears that the organic nitrogen fraction during the warmer months is higher in swale runoff from Station 6 than the highway runoff from Station 4. The opposite was true during the cool winter months when the organic fraction in highway samples from Station 4 was higher than swale runoff samples from Station 6. Also, the ammonia and nitrite concentrations had increased in both Stations 4 and 6 in the warmer climate during the summer months. The relative percent change between Stations 4 and 6 for all nitrogen species is shown in Table 6. The total nitrogen in runoff water increased by 37% during warm summer months and decreased by 7% during the cool winter months after flowing over swale areas.

Swale Studies

The nitrogen content at each sampling station for various swale experiments was analyzed for NH_4-N, NO_2-N, NO_3-N, inorganic N, organic N, and total N. Tables 7 through 10 show the concentration at each sampling point along the swale and the corresponding percent changes of nitrogen concentration for the four Maitland swale studies. As can be seen in Table 7, during the 1/24/83 Maitland swale study, where the average surface velocity of the simulated highway runoff through the swale was 2.58 m/min, there was a decline in most of the nitrogen forms. The total N decrease was 9% between the inflow

TABLE 5

AVERAGE DISSOLVED NITROGEN CONCENTRATION IN STORMWATER
SAMPLES COLLECTED FROM MAITLAND INTERCHANGE AND I-4 DRAINAGE BASIN

Nitrogen Form	Average Concentration (μg-N/1)					
	Station #1*	Station #2**	Station #3**	Station #4**	Station #5*	Station #6*
NH_4-N	205	136	137	84	153	167
NO_2-N	19	16	4	7	7	2
NO_3-N	366	466	185	179	258	219
Inorganic N	590	490	326	289	418	388
Organic N	1934	1414	2677	1123	4764	992
Total N	2524	1912	3003	1412	5182	1380
% Organic N	77	74	89	80	92	72
# of Samples	5	15	16	17	4	16

* Runoff water samples from grassy swales

** Runoff water samples from direct highway

TABLE 6

CHANGE IN CONCENTRATIONS BETWEEN STORMWATER SAMPLING STATIONS 4 AND 6 OVER TWO DIFFERENT TIME PERIODS

Nitrogen Form	A: Average Concentration Over 8/19, 8/23, 9/8, and 10/1 Storm Events		B: Average Concentration Over 12/11, 1/21, 1/31, 2/3, 2/12, 2/14, and 2/27 Events		Average Percent Change Between 4 and 6 in Column A	Average Percent Change Between 4 and 6 in Column B
	Station 4	Station 6	Station 4	Station 6		
NH_4-N	48	57	78	231	19	196
NO_2-N	1	1	5	3	0	-40
NO_3-N	84	68	109	141	-19	29
Organic N	617	905	1439	1136	47	-21
Total N	750	1031	1631	1511	37	- 7

concentration and the weir effluent concentration. In the second Maitland study on 2/7/83, the average surface velocity through the swale was decreased to 1.37 m/min, and, consequently, the decline in concentration of total N along the swale increased to 25%. The average surface velocity through the swale during the 2/21/83 Maitland study was further decreased to 0.90 m/min and the decline of total N increased once again to 30% at a distance of 38 m in from the inflow point. The final Maitland study was performed several months later in May (5/31/83) and the surface velocity of the runoff through the swale (2.35 m/min) was similar to the velocity of flow during the first swale experiment and the decline in the nitrogen concentration averaged 2%. The data presented indicate that the slower the runoff flow through the swale, the higher the removal of nitrogen concentration.

A graphical presentation of changes in nitrogen concentrations of simulated highway runoff along a roadside swale at Maitland Interchange and I-4 is shown in Figures 14 through 17. The inorganic species (NH_4-N and NO_3-N) show a fairly uniform pattern with constant or declining concentrations as the flow travels along the swale area. However, the total nitrogen and organic nitrogen appear to be inconsistent with increasing concentrations close to the inflow

TABLE 7

TRANSPORT OF NITROGEN SPECIES THROUGH SWALE AT MAITLAND INTERCHANGE SITE ON 1/24/83

Distance Along Swale (m)	Average Concentration (μg/l)					
	NH_4-N	NO_2-N	NO_3-N	Inorganic N	Organic N	Total N
0	759	2	340	1101	202	1301
7.5	736	2	280	1018	938	1956
23	729	2	331	1062	1334	2406
38	635	2	326	963	776	1789
53	676	2	316	994	203	1190
Percent Change	-11	0	- 2	-10	- 1	- 9

TABLE 8

TRANSPORT OF NITROGEN SPECIES DURING SWALE STUDY AT MAITLAND INTERCHANGE SITE ON 2/7/83

Distance Along Swale (m)	Average Concentration (μg/l)					
	NH_4-N	NO_2-N	NO_3-N	Inorganic N	Organic N	Total N
0	1040	4	159	1203	317	1520
7.5	1024	4	232	1260	746	2006
23	833	4	102	937	910	1849
38	879	4	102	985	618	1603
53	706	4	106	816	319	1134
Percent Change	-32	0	-33	-32	1	-25

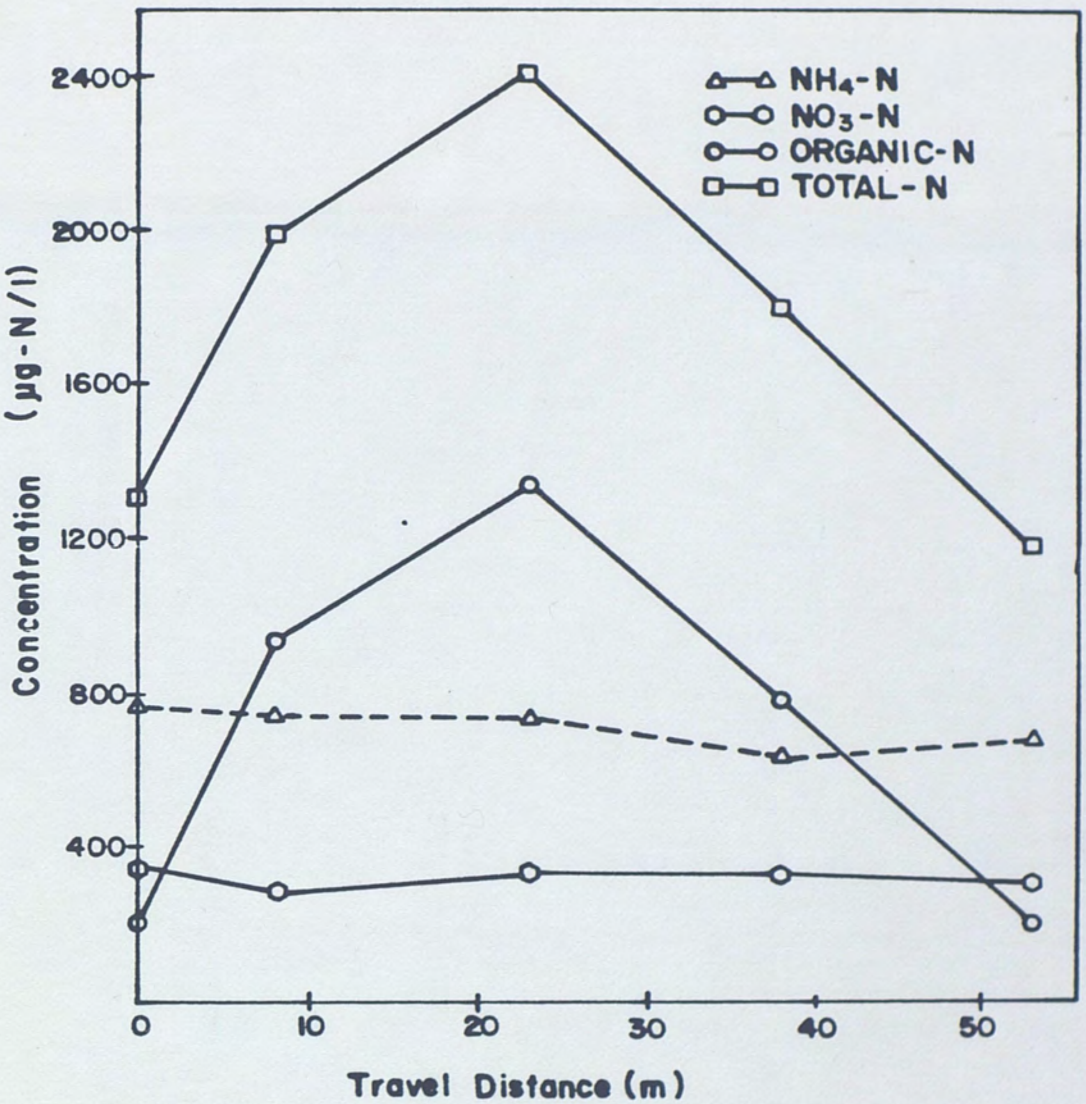

Fig. 14. Changes of nitrogen concentrations in simulated highway runoff along roadside swale at Maitland Interchange on 1/24/83.

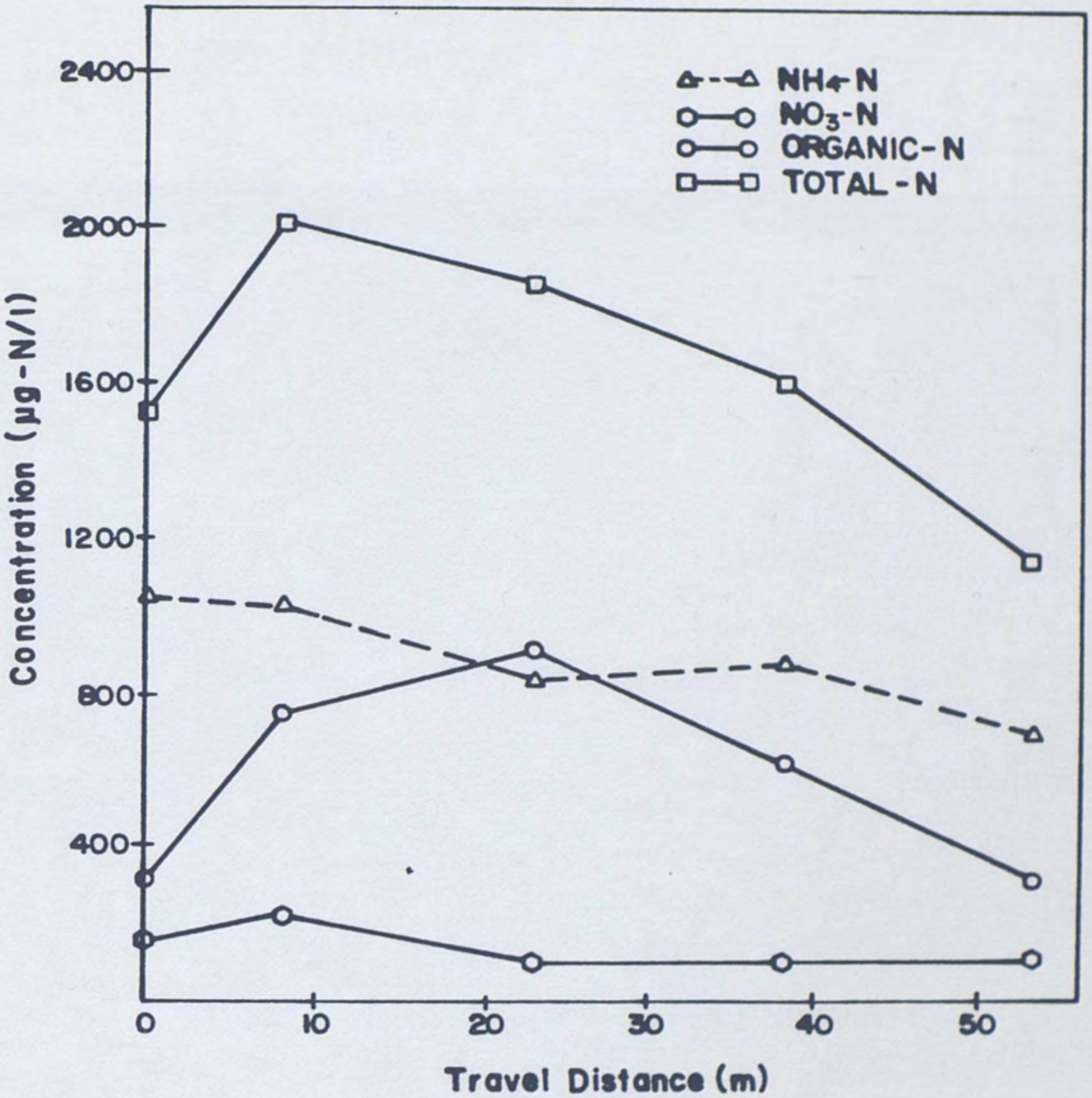

Fig. 15. Changes of nitrogen concentrations in simulated highway runoff along roadside swale at Maitland Interchange on 2/7/83.

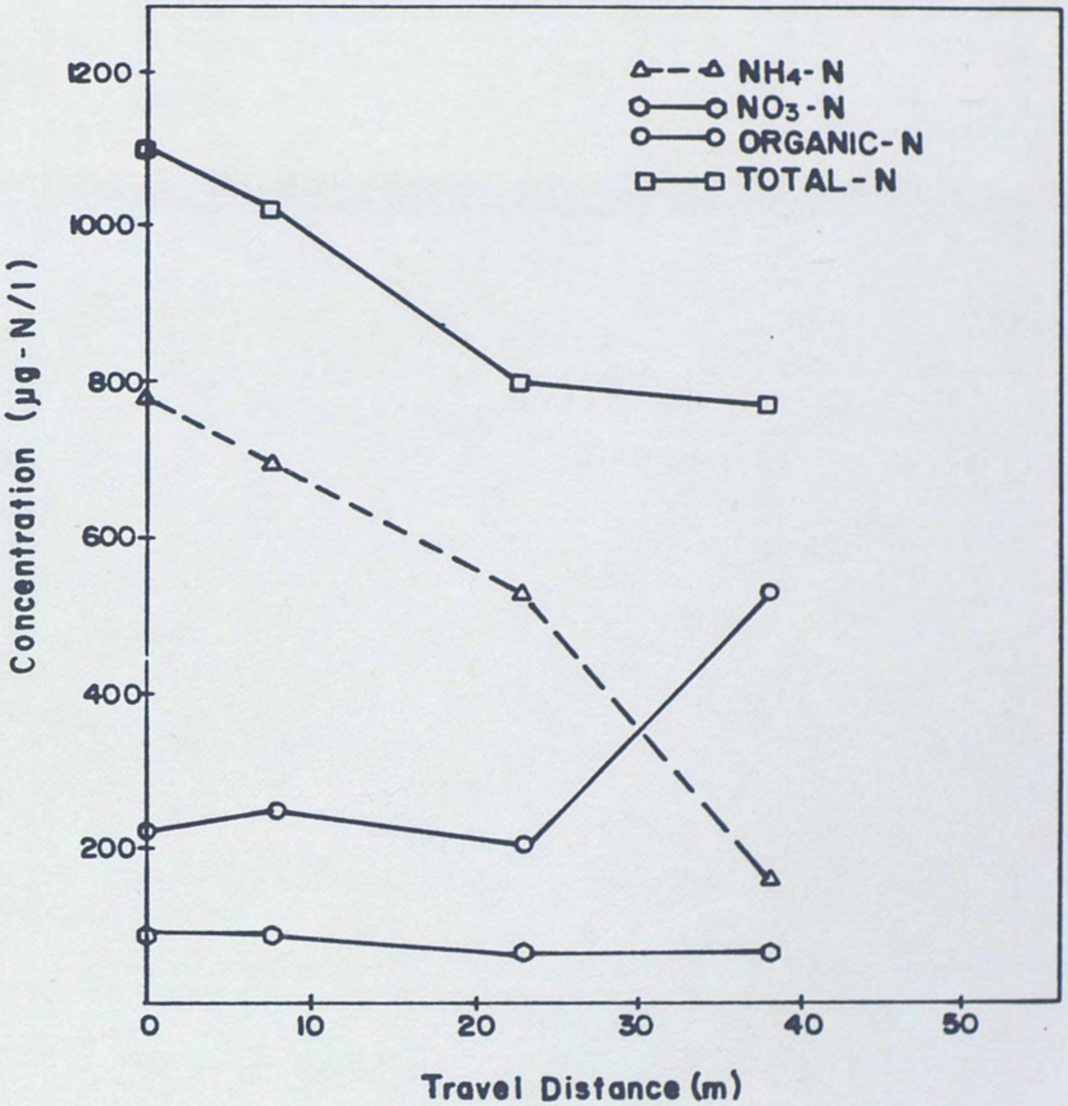

Fig. 16. Changes of nitrogen concentrations in simulated highway runoff along roadside swales at Maitland Interchange on 2/21/83.

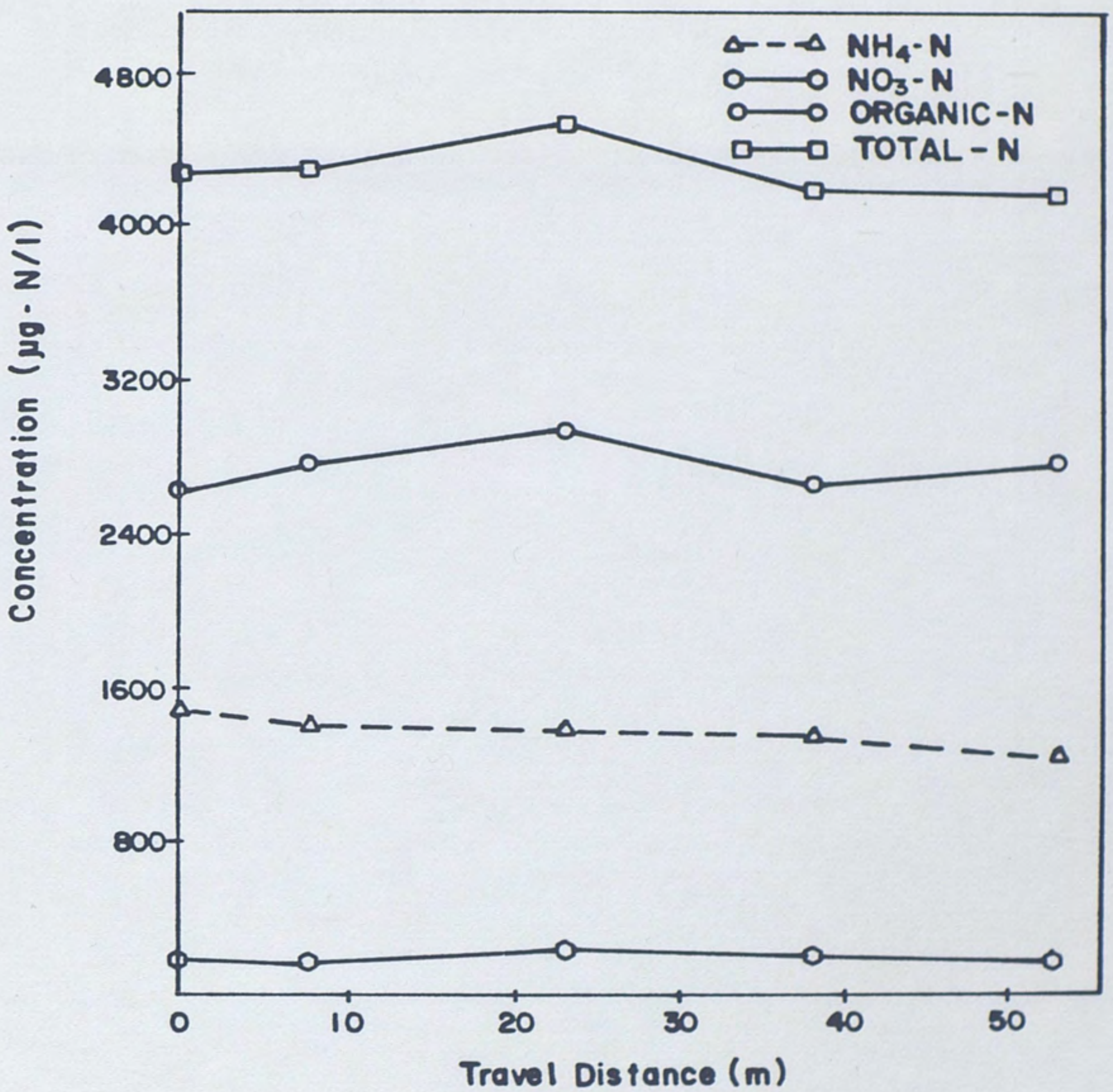

Fig. 17. Changes of nitrogen concentrations in simulated highway runoff along roadside swales at Maitland Interchange on 5/31/83.

point and then decreasing toward the end of the swale. A closer look at the specific nitrogen forms revealed that the inorganic forms of nitrogen (NH_4-N, NO_2-N and NO_3-N) were removed as a function of the surface velocity through the swale during the Maitland studies. As can be seen in Tables 7 through 10, the inorganic forms of nitrogen declined by 10% during the 1/24/83 and 2/7/83 experiments, 74% during the 3/21/83 experiment, and 13% during the 5/31/83 study. It is interesting to note that there was an increase of the total organic nitrogen species between intermediate sampling stations during the 1/24/83, 2/7/83 and 5/31/83 studies, although there was a total inorganic net decline over the entire test length. These percent changes in nitrogen are further illustrated in Figures 18 through 21 where the percent change of NH_4-N, NO_3-N, organic N and total N along the length of the swale during the four Maitland studies is graphically presented. Conversely, the organic forms of nitrogen were not removed more effectively at lower flow-through velocities. It appears that organic nitrogen was generated at intermediate stations during studies with simulated highway runoff flowing over a grassy swale area in Maitland. Grass clippings and decay products may affect the organic content in highway runoff flowing over swale areas.

At the Epcot Interchange site, two studies were performed. The 3/23/83 study was performed in a recently completed earthen swale. The 5/16/83 study was performed at the same site after the

TABLE 9

TRANSPORT OF NITROGEN SPECIES DURING SWALE STUDY AT MAITLAND INTERCHANGE SITE ON 2/21/83

Distance Along Swale (m)	Average Concentration ($\mu g/l$)					
	NH_4-N	NO_2-N	NO_3-N	Inorganic N	Organic N	Total N
0	784	3	88	875	225	1100
7.5	690	2	87	779	247	1026
23	524	2	69	595	203	798
38	155	2	71	228	527	768
53*	---	-	--	---	---	---
Percent Change	-80	-33	-19	-74	134	-30

* Flow did not reach the 53 m sampling point during the study.

TABLE 10

TRANSPORT OF NITROGEN SPECIES DURING THE SWALE STUDY AT MAITLAND INTERCHANGE SITE ON 5/31/83

Distance Along Swale (m)	Average Concentration ($\mu g/l$)					
	NH_4-N	NO_2-N	NO_3-N	Inorganic N	Organic N	Total N
0	1476	1	169	1646	2621	4267
7.5	1400	1	163	1564	2784	4312
23	1394	1	240	1635	2901	4536
38	1356	1	174	1531	2657	4188
53	1259	2	166	1427	2755	4182
Percent Change	-15	100	-2	-13	5	-2

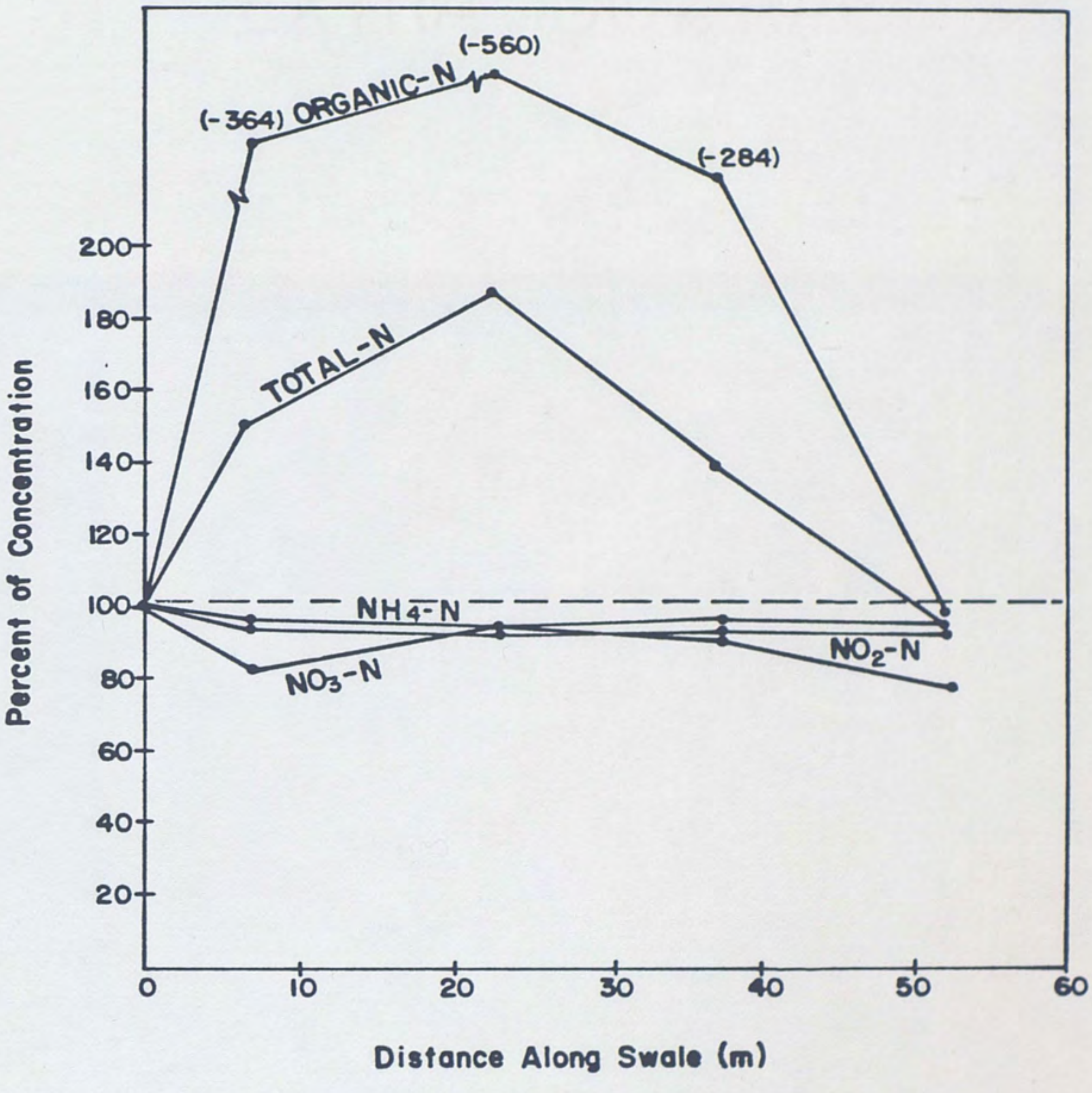

Fig. 18. Average percent changes of nitrogen species along the length of the swale during the Maitland swale study on 1/24/83.

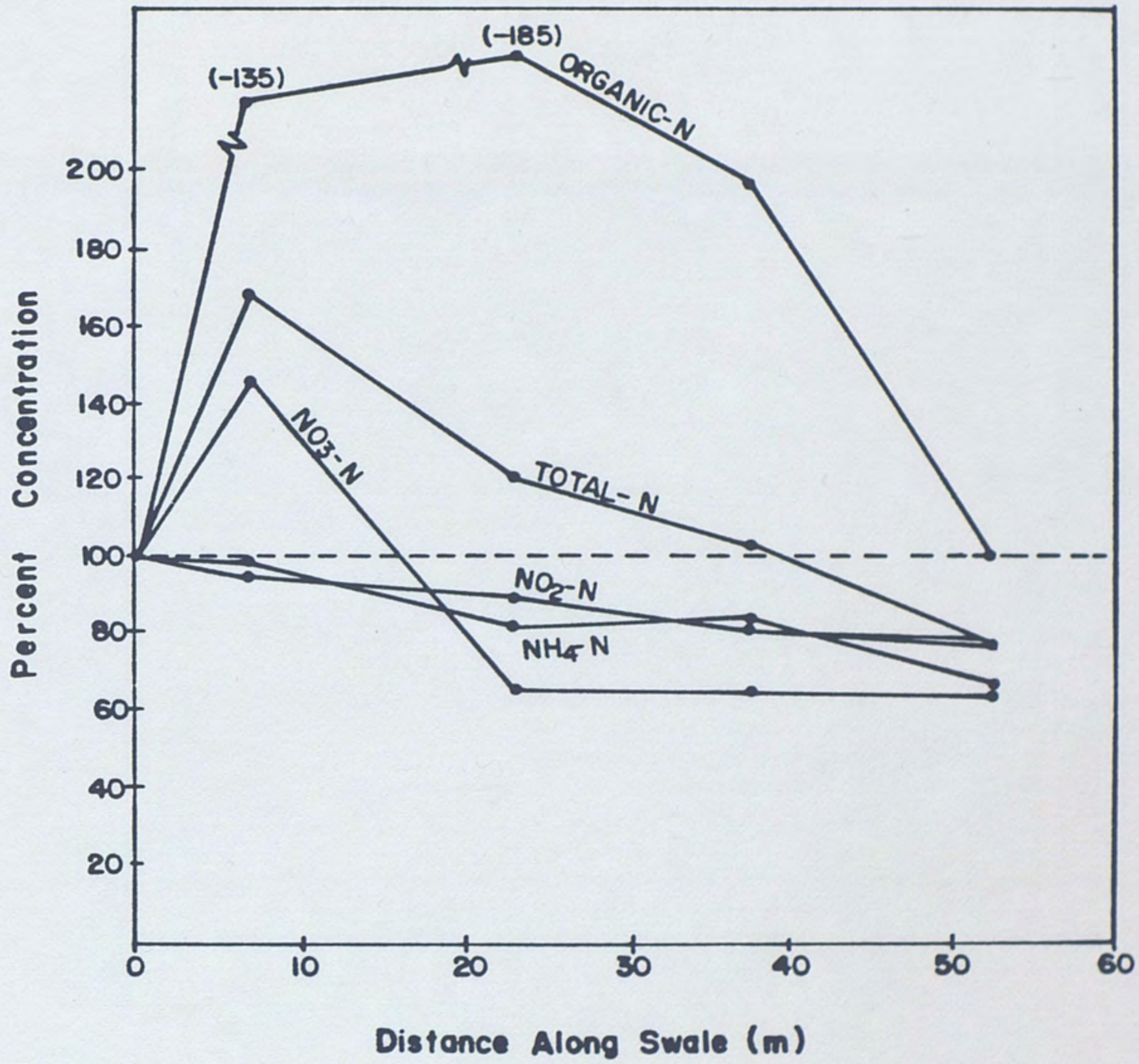

Fig. 19. Average percent changes of nitrogen species along the length of the swale during the Maitland swale study on 2/7/83.

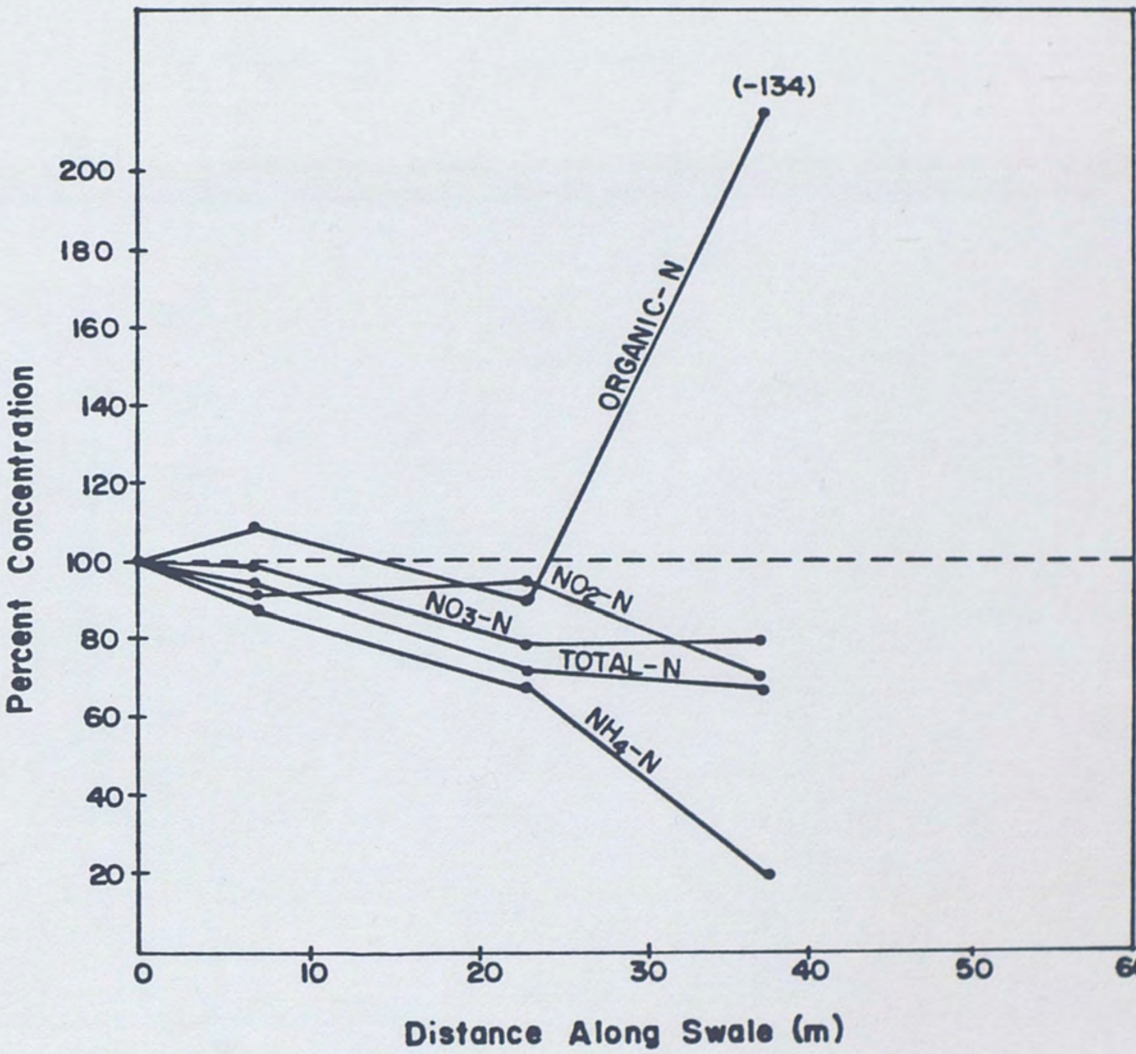

Fig. 20. Average percent changes of nitrogen species along the length of the swale during the Maitland swale study on 2/21/83.

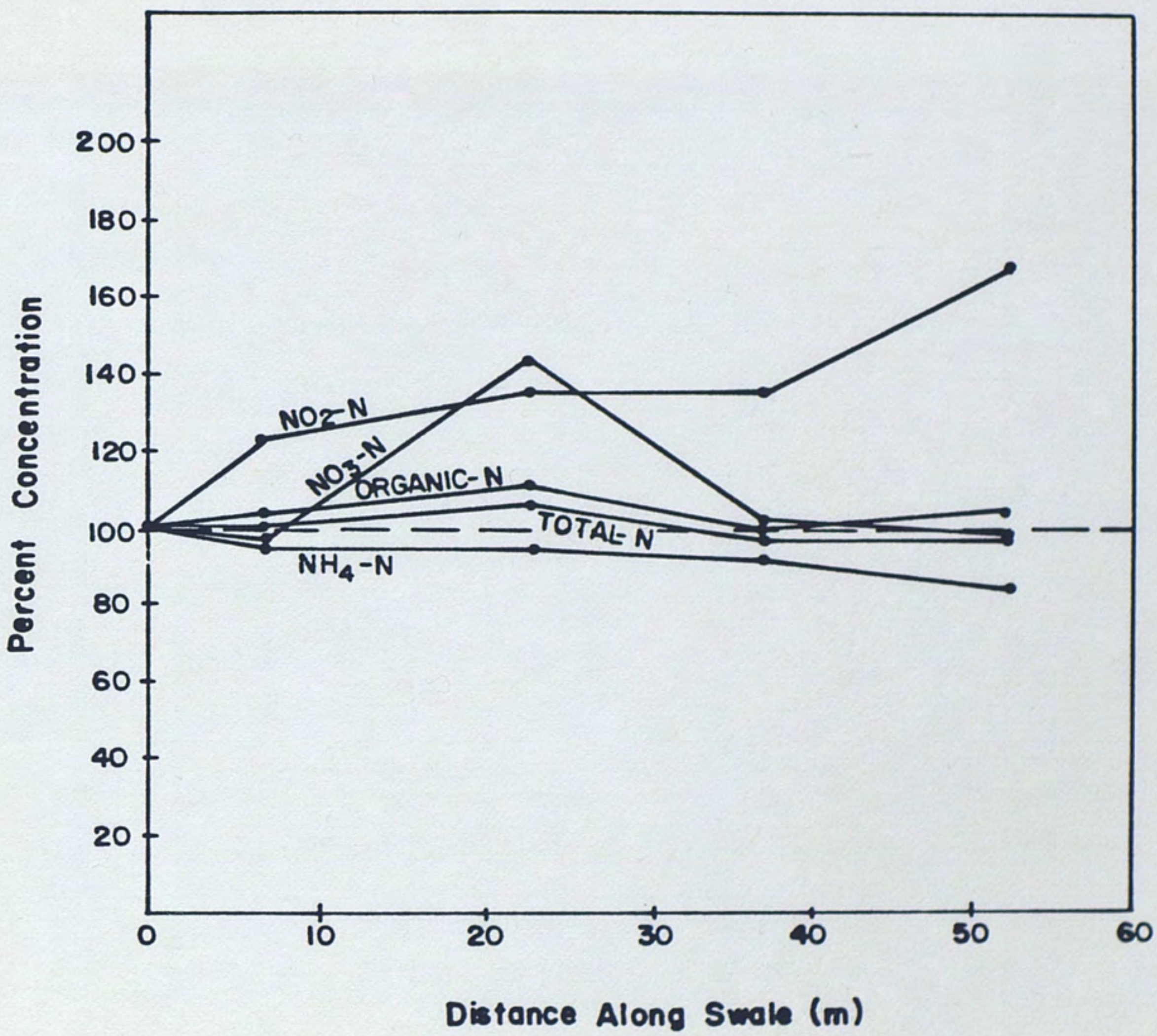

Fig. 21. Average percent changes of nitrogen species along the length of the swale during the Maitland swale study on 5/31/83.

grass cover had been placed and had grown. The flow velocity through the swale during each Epcot study was very similar; 2.44 m/min and 2.49 m/min, respectively. Two weirs were installed at the 90 m and 170 m points along the swale in both Epcot studies to obtain more accurate measurements of outflow volumes leaving two portions of the swale. The nitrogen concentrations at the sampling points along the swale during each Epcot study can be seen in Tables 11 and 12, and in Figures 22 and 23. The total N concentration declined 2% during the 3/23/83 study and increased 14% during the 5/16/83 study. The percent changes of all nitrogen forms during the two Epcot studies are further represented in Figures 24 and 25.

It appears that the difference in the removal of the organic nitrogen form, the largest single constituent of total N during all the studies, is the primary reason for the difference in total N change in the Epcot studies. Table 11 shows that organic N declined 4% during the 3/23/83 study. In contrast, Table 12 shows that organic N increased 18% during the second Epcot study on 5/16/83. Table 13 provides a summary of the percent change of nitrogen forms along the length of the swale in all six swale studies.

Nitrogen Mass Balance

The nitrogen mass balance over the entire swale, i.e., mass input ± mass change = mass output, was calculated as shown in Table 14. The percent nitrogen mass removal along the swale varied between 24% and 100% during the six swale studies. Nitrogen mass is retained in

TABLE 11

TRANSPORT OF NITROGEN SPECIES DURING THE SWALE STUDY
AT EPCOT INTERCHANGE SITE ON 3/23/83

Distance Along Swale (m)	Average Concentration (μg/l)					
	NH_4-N	NO_2-N	NO_3-N	Inorganic N	Organic N	Total N
0	49	3	216	268	1733	2001
30	45	3	200	248	1725	1973
60	67	2	207	276	1633	1909
90	35	2	192	229	1258	1487
120	55	3	210	268	1492	1759
170	68	3	230	300	1658	1959
Percent Change	39	0	6	12	-4	-2

TABLE 12

TRANSPORT OF NITROGEN SPECIES DURING THE SWALE STUDY
AT EPCOT INTERCHANGE SITE ON 5/16/83

Distance Along Swale (m)	Average Concentration (μg/l)					
	NH_4-N	NO_2-N	NO_3-N	Inorganic N	Organic N	Total N
0	538	3	72	613	1932	2545
30	589	4	95	688	2263	2940
60	575	4	88	667	2113	2779
90	559	4	95	658	2106	2714
120	573	11	210	794	2163	2956
170	531	14	79	624	2288	2912
Percent Change	-1	367	10	2	18	14

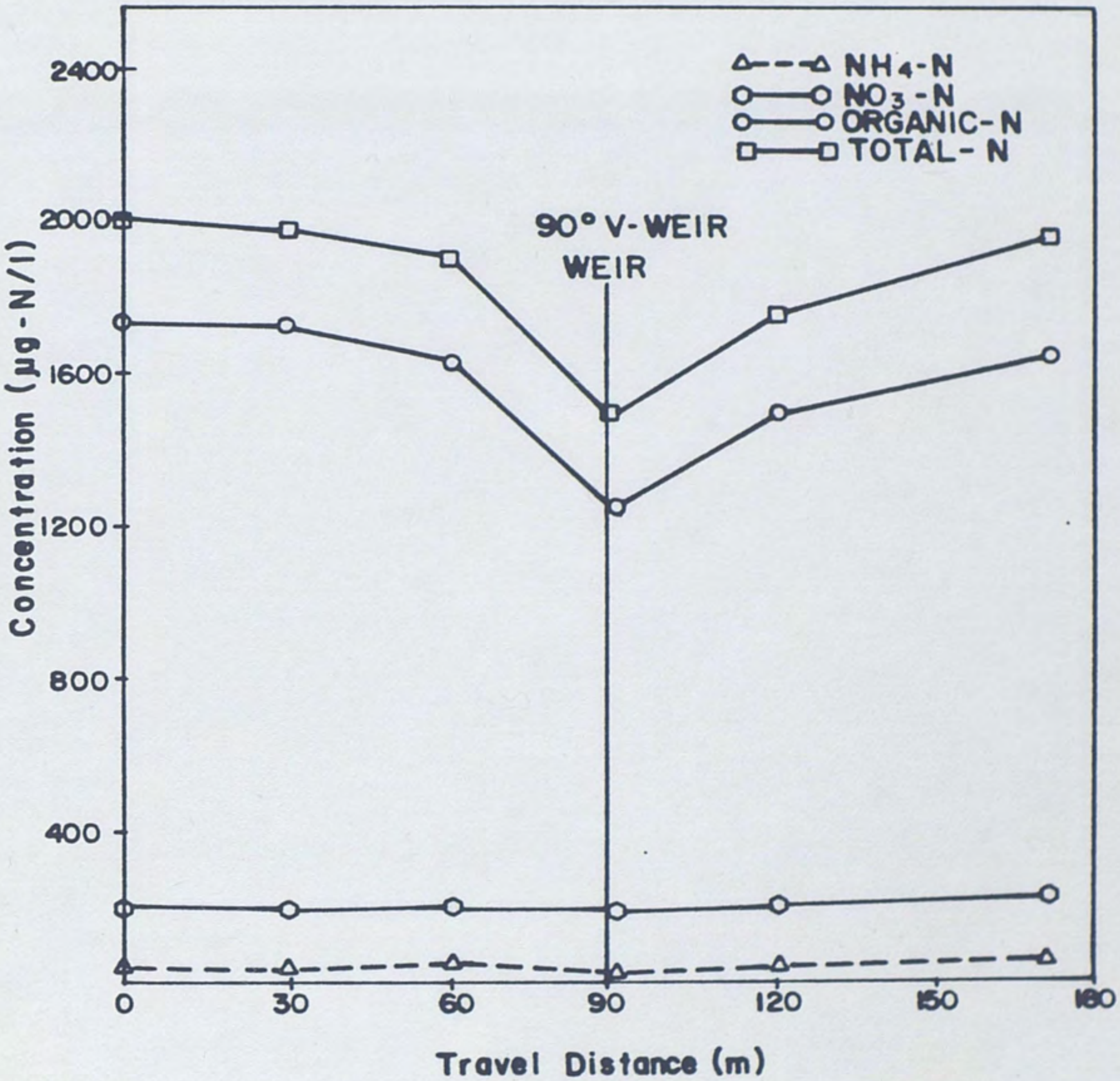

Fig. 22. Changes of nitrogen concentrations in simulated highway runoff along roadside swales at Epcot Interchange on 3/23/83.

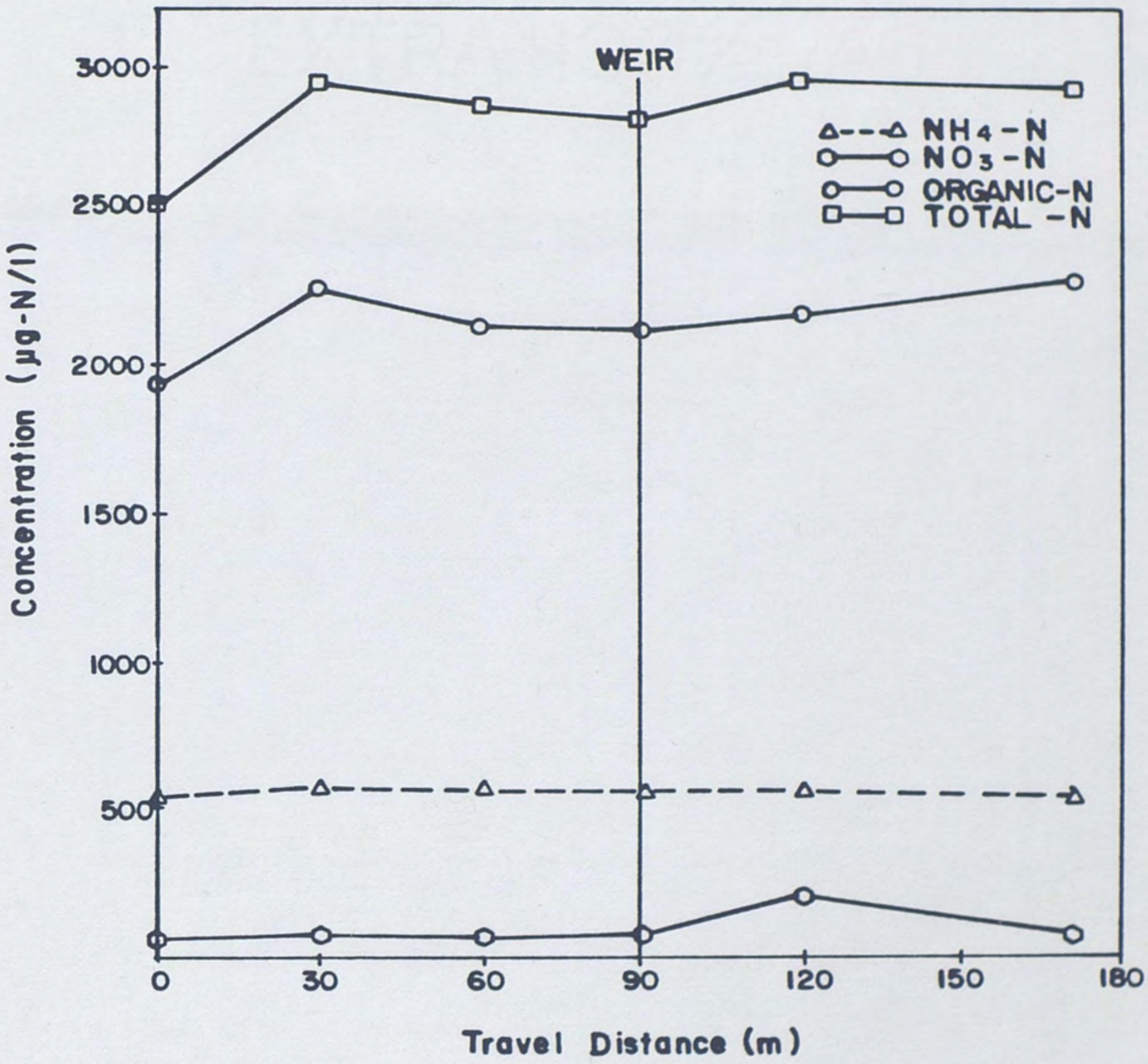

Fig. 23. Changes of nitrogen concentrations in simulated highway runoff along roadside swales at Epcot Interchange on 5/16/83.

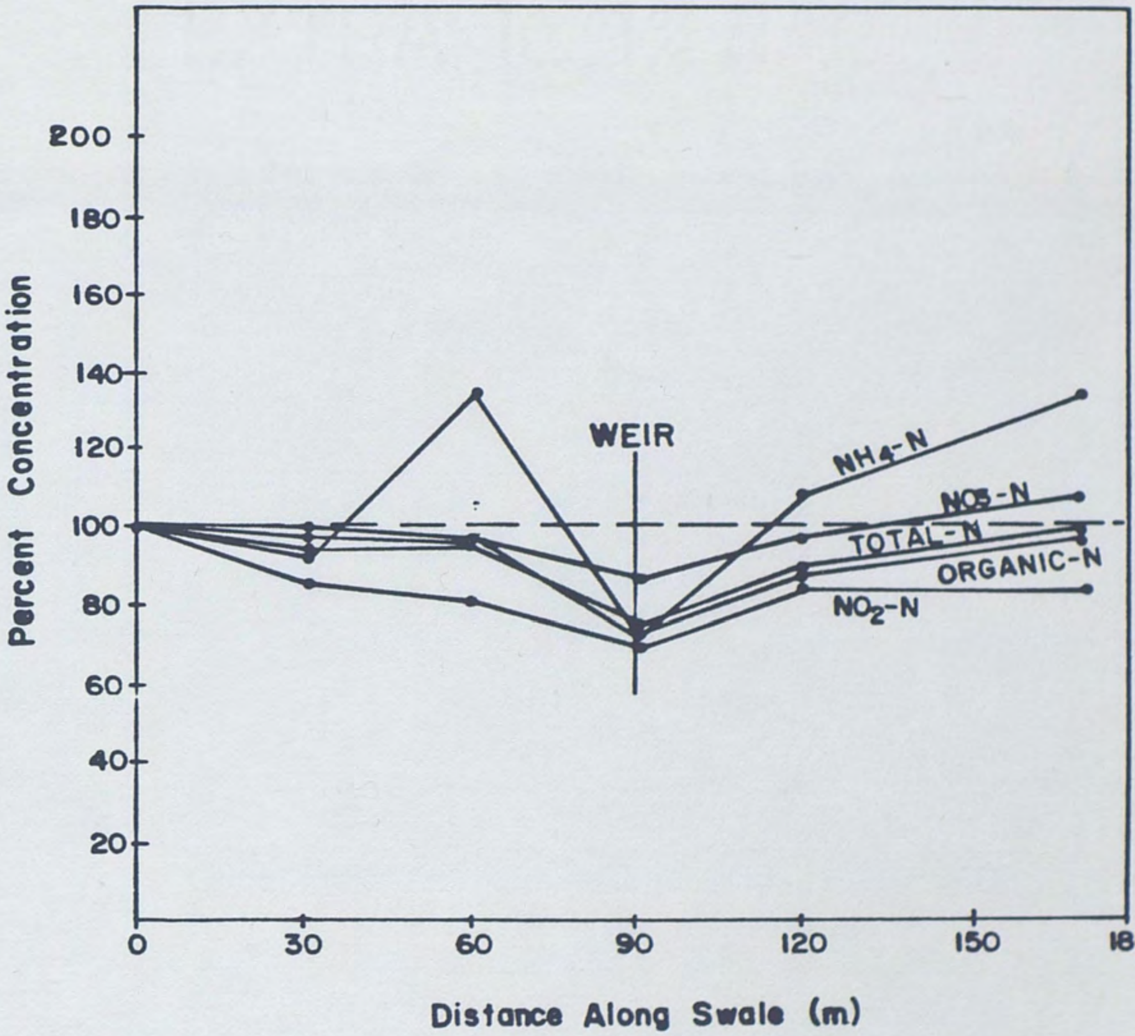

Fig. 24. Average percent changes of nitrogen species along the length of the earthen swale during the Epcot swale study on 3/23/83.

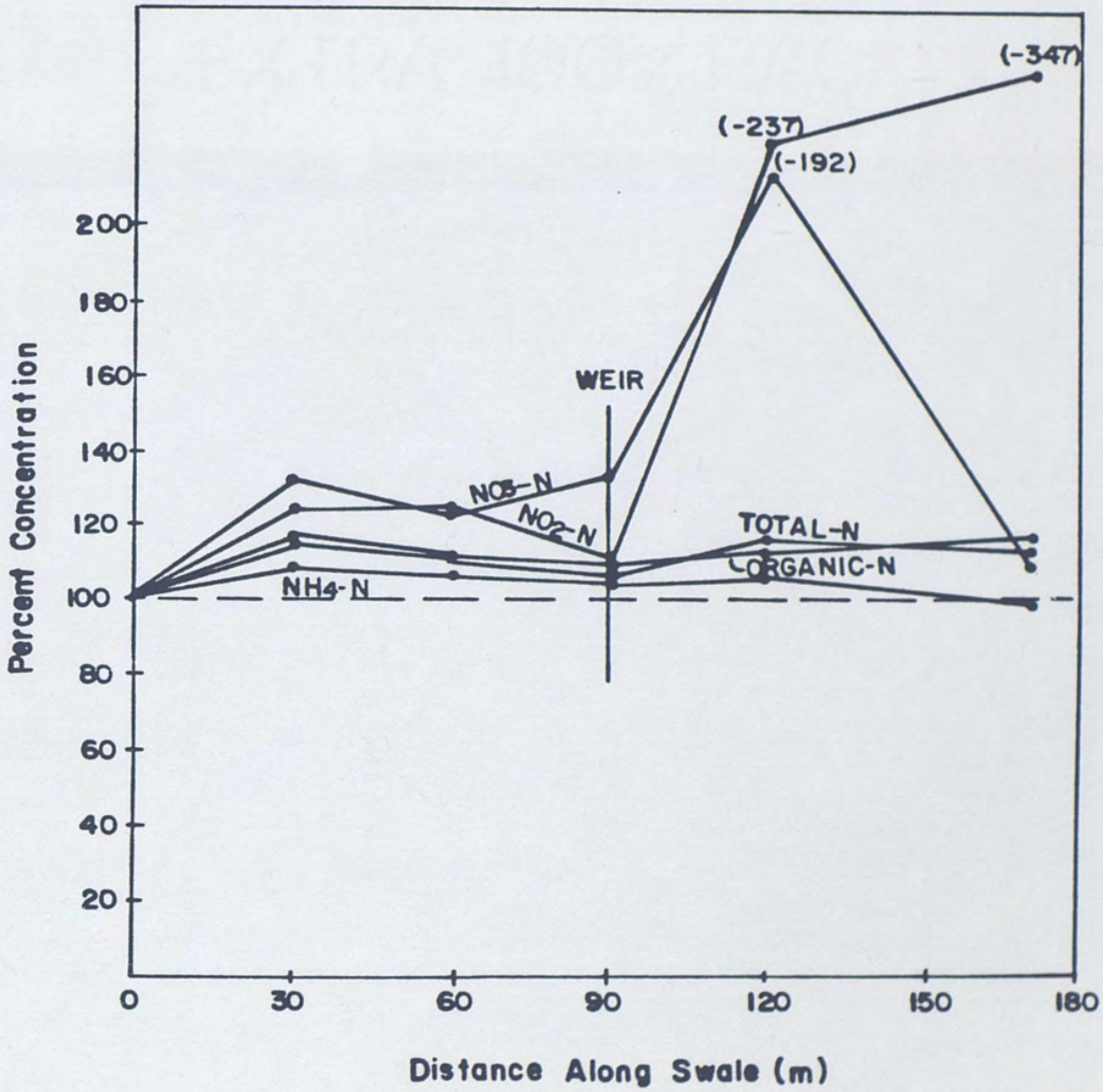

Fig. 25. Average percent changes of nitrogen species along the length of the grassy swale during the Epcot swale study on 5/16/83.

TABLE 13

COMPARISON OF PERCENT CHANGES OF NITROGEN FORMS ALONG THE LENGTH OF THE SWALE DURING THE SIX SWALE STUDIES

Swale Study	Percent Change Along Length of Swale					
	NH_4-N	NO_2-N	NO_3-N	Inorganic N	Organic C	Total N
Maitland 1/24/83	-11	0	- 7	-10	- 1	- 9
Maitland 2/7/83	-32	0	-33	-32	1	-25
Maitland 2/21/83	-80	-33	-19	-74	134	-30
Maitland 5/31/83	-15	100	- 2	-13	5	- 2
Epcot 3/23/83	39	0	6	12	- 4	- 2
Epcot 5/16/83	- 1	367	10	2	18	14

TABLE 14

TOTAL MASS OF NITROGEN THROUGH THE SWALES DURING THE SIX SWALE STUDIES AND THE CORRESPONDING % CHANGE BETWEEN OUTFLOW AND INFLOW

Swale Study	Mass of Total N (g)		Percent Change
	Inflow	Outflow	
Maitland (1/24/83)	5	2	- 61
Maitland (2/7/83)	31	9	- 73
Maitland (2/21/83)	6	0	-100
Maitland (5/31/83)	150	109	- 27
Epcot (3/23/83)	115	70	- 39
Epcot (5/16/83)	118	90	- 24

the swale area by infiltration, seepage, transpiration and soil-grass nitrogen interactions. The study at Maitland performed on 2/21/83 showed 100% nitrogen removal through the swale area since the flow did not reach the end weir. Table 15 shows the percent change in nitrogen mass flowing through the swales for various species of nitrogen. Also, the percent flow infiltration through the soil is shown and it may correlate with the percent nitrogen mass reduction. In all the studies performed, there was a decrease in the total N mass along the swale. In fact, only the NO_2-N mass in the two studies performed in May (Maitland 5/31/83, Epcot 5/16/83) increased along the swale. All other forms of nitrogen mass decreased during the studies.

If the infiltration rates during the six studies are analyzed and compared with the percent mass of total N retained, a relationship can be developed. A definite linear relationship exists between the percent runoff water loss (primarily) due to infiltration in addition to small evaporative losses) as shown in Figure 26. In other words, as the percent runoff water loss is increased, which is primarily a function of the infiltration rate, the percent mass of total N retained will also increase.

Infiltration appears to be related to the surface velocity through the swale or the residence time through the swale. Of course, infiltration is a function of many variables such as the antecedent dry period, soil porosity, etc., yet, there does appear to be an inverse relationship between surface velocity of runoff

TABLE 15

DETENTION OF NITROGEN MASS BASIS, FROM SIMULATED HIGHWAY RUNOFF THROUGH SWALES

Experiment	Average Percent Change of Nitrogen Mass					
	NH_4-N	NO_2-N	NO_3-N	Organic N	Total N	Infiltration (%)
Maitland (1/24/83)	59	57	58	64	61	57
Maitland (2/7/83)	74	67	72	63	73	60
Maitland (2/21/83)	100	100	100	100	100	100
Epcot (2/23/83)	14	48	34	41	39	38
Epcot (5/16/83)	60	121*	46	41	24	50
Maitland (5/31/83)	37	2*	27	22	27	26

* Represents an increase in mass. All other stations showed a decline in mass.

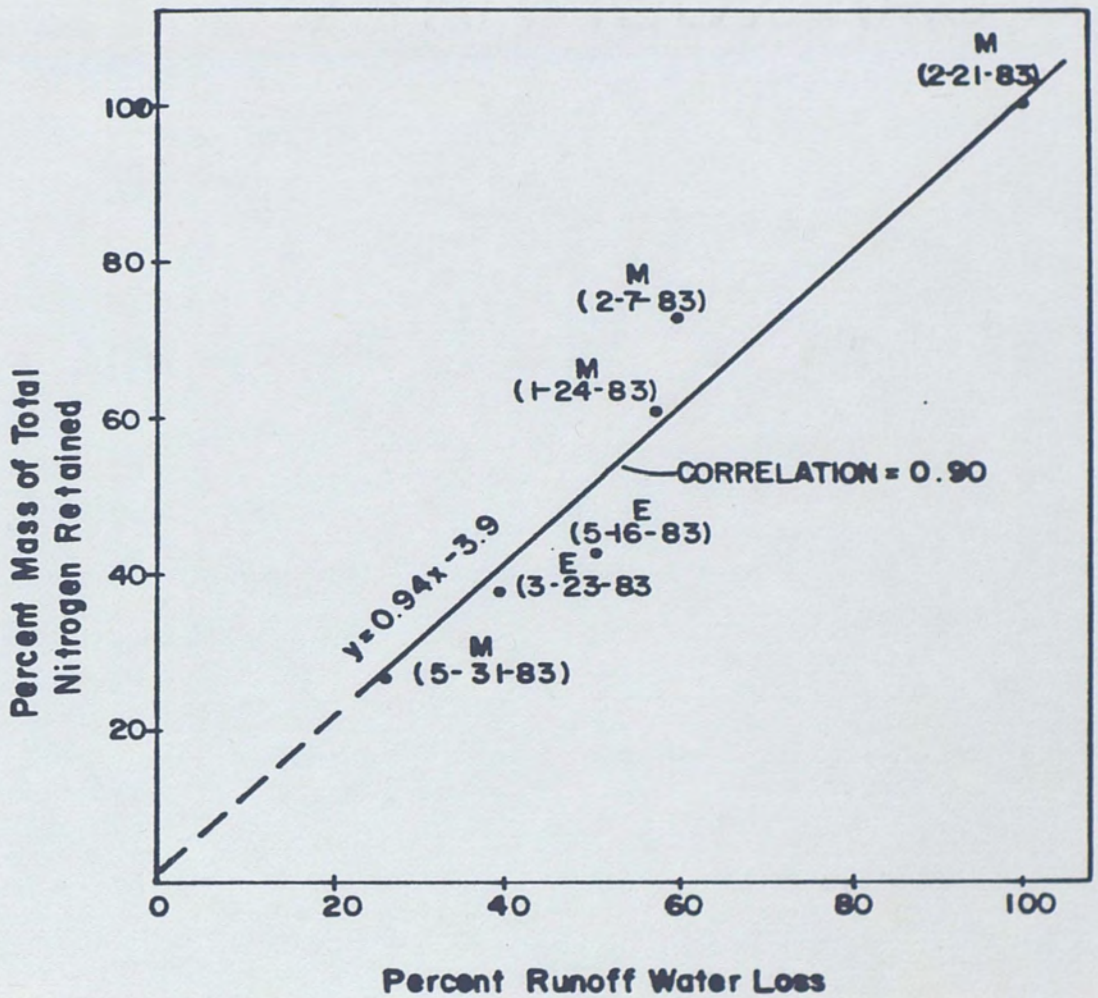

Fig. 26. Correlation between runoff water loss and nitrogen retention during the swale studies at Maitland and Epcot Interchanges.

through the swale and percent infiltrated along the swale. In other words, the lower the surface velocity of runoff through the swale, the greater the infiltration rate along the swale, if other factors are assumed to be constant.

Nitrogen mass loading into receiving water is of utmost importance to the protection of water quality. The ability of swales to retain nitrogen mass before discharge into adjacent streams is illustrated in Table 16. The average mass retained in mg-N/m^2-min is shown in the table.

Analysis of Nitrogen Concentration Removal

The data presented in Table 17 and Figure 27 shows a definite relationship between the average flow velocity permitted through the length of the swale and the percent change of total nitrogen over the entire length for the four Maitland swale studies. If the three studies performed during January and February are plotted alone, a linear relationship between the average velocity and percent total nitrogen change exists with a perfect correlation of 1.0. When all the Maitland experiments are included in the plot, the relationship shows a correlation coefficient of 0.95.

An inverse relationship appears to exist between velocity of runoff through the swale and the percent of nitrogen removed along the length of the swale. In other words, to achieve a maximum removal of total nitrogen concentration in the runoff, a minimum velocity of runoff through the swale should be allowed, given constant

TABLE 16

AVERAGE RETENTION OF NITROGEN MASS ALONG THE SWALE PER UNIT AREA PER TIME FOR THE SIX SWALE STUDIES DETERMINED FROM A MASS BALANCE OVER THE SWALE LENGTH

Experiment	Surface Velocity (m/min)	Average Mass Uptake of Nitrogen (mg removed/m^2-minute)				
		NH_4-N	NO_2-N	NO_3-N	Organic N	Total N
Maitland (1/24/83)	2.58	6	0	3	2	12
Maitland (2/7/83)	1.37	4	0	1	1	6
Maitland (2/21/83)	0.90	1	0	0	0	1
Epcot (3/23/83)	2.44	0	0	0	2.2	2.5
Epcot (5/16/83)	2.49	0.5	0	0	1	1.5
Maitland (5/31/83)	2.35	9	0	1	9	19

TABLE 17

AVERAGE CALCULATED VELOCITY THROUGH THE SWALE AND THE CORRESPONDING PERCENT TOTAL NITROGEN CHANGE FOR MAITLAND SWALE STUDIES

Maitland Swale Study Date	Average Calculated Velocity (m/min)	Percent Total Nitrogen Change	Residence Time in Swale (min)
1/24/83	2.58	- 8.7	31
2/07/83	1.37	-25.4	58
2/21/83	0.90	-30.2	330
5/31/83	2.35	- 2.0	72

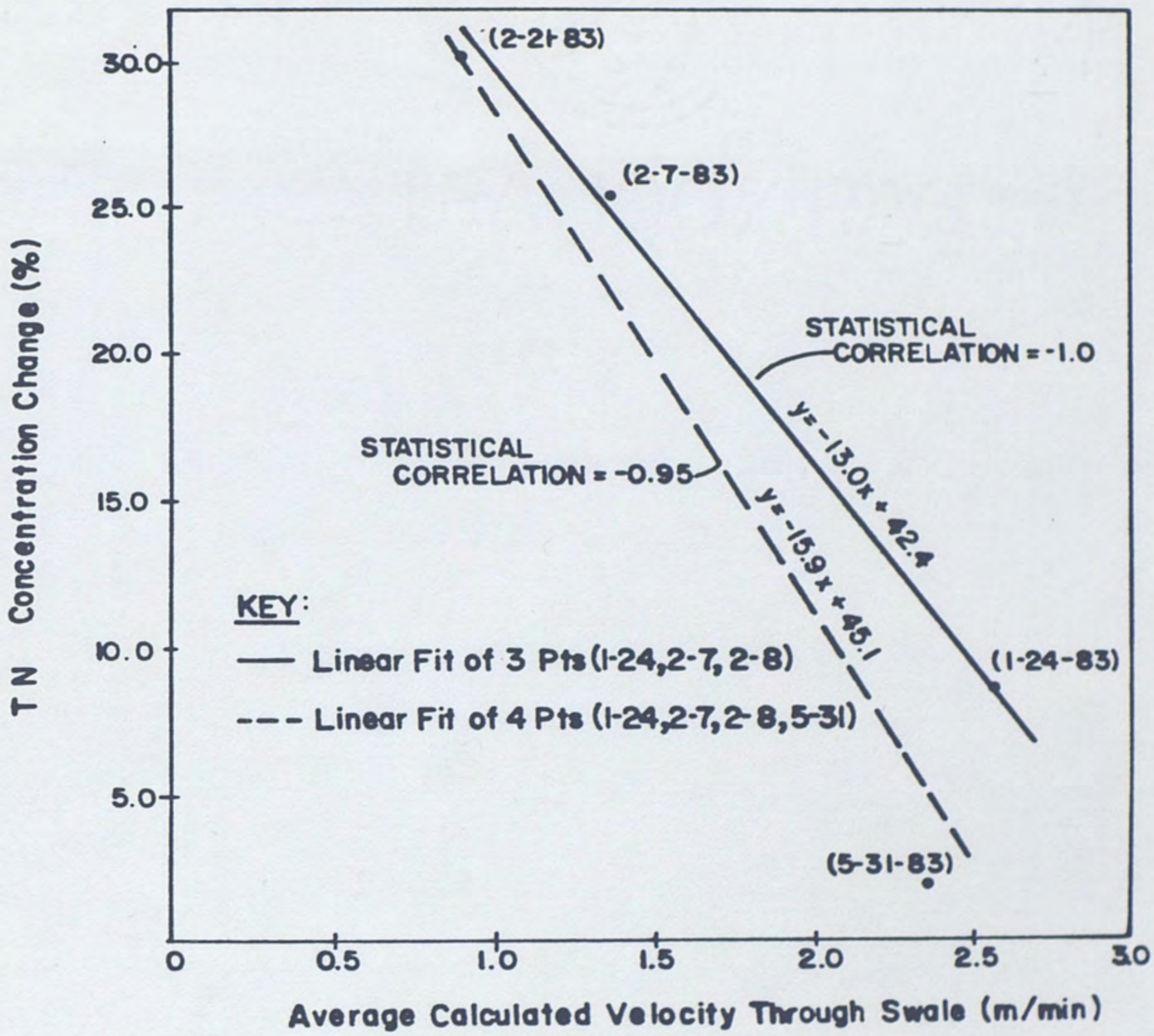

Fig. 27. Percent changes in nitrogen concentrations along the swale vs. average calculated velocity (m/min) for the four Maitland swale studies.

vegetative and climatic conditions. When vegetative and/or climatic conditions change, the inverse relationship between velocity and nitrogen removal along the swale may be reduced.

The velocity in the Maitland swale study on 5/31/83 was comparable to the study on 1/24/83 (2.35 m/min and 2.58 m/min, respectively). The percent removals of total nitrogen, however, differ slightly. The study on 5/31/83 produced a removal of 2%, while the study on 1/24/83 produced about 9% nitrogen removal. The decreased removal in the 5/31/83 Maitland study appears to be due to decreased removal of the soluble inorganic N and particularly organic N, which may be attributed to the increased nitrogen generation by grass clippings and other organic debris found in the swales during the warmer months of the year.

This decreased removal rate pattern is also evident in the Epcot swale study on 5/16/83. The Epcot swale study on 3/23/83 produced a total nitrogen removal rate of 2% while the Epcot swale study performed on 5/16/83 produced an increase of total nitrogen along the swale of 14%. This decreased nitrogen removal can also be attributed to significant decreases in the removal of the highly soluble inorganic N and the organic N concentrations along the swale. Table 18 further illustrates the differences in the removal rates of these particular nitrogen forms between the two similar studies at each site (1/24/83 and 5/31/83 at Maitland and 3/23/83 and 5/16/83 at Epcot).

TABLE 18

COMPARISON OF AVERAGE VELOCITIES, SITE CONDITIONS, AND THE PERCENT CHANGES OF NO_2-N, NO_3-N AND ORGANIC N DURING THE FOUR MAITLAND SWALE STUDIES

Swale Study	Average Velocity (m/min)	Site Conditions		Percent Change Along Swale		
		Cover	Season	NO_3-N	Organic N	Total N
Maitland 1/24/83	2.58	Grassy	Winter	- 7	- 3	- 9
Maitland 5/31/83	2.35	Thick Grass	Late Spring	- 2	5	- 2
Epcot 3/23/83	2.44	Earthen	Early Spring	6	- 4	- 2
Epcot 5/16/83	2.49	Grassy	Late Spring	10	18	14

In both cases, there is a significant decline in the removal of NO_3-N, organic N and total N concentrations in the study performed in May. In addition, NH_4-N removal declined dramatically in the Epcot study performed in May. The highly soluble forms of inorganic nitrogen are not removed as effectively during the studies performed at each site in May as they were during the studies performed in January and February. Also, the organic N concentration is not removed as efficiently, and is actually increased along the length of the swale during studies in May as the studies performed in January and February. Since the hydraulic characteristics at each site were comparable, other site conditions must be scrutinized.

An earthen, or lean grass cover, may provide more absorption sites for assimilation of the charged soluble nitrogen species. A thick grassy cover reduces the absorption sites and, thus, the potential for sorption of nitrogen species. This phenomena of sorption is similar to the lake and wetland sediment sorption of nitrogen and is well documented in various studies (Patrick 1976; Terry and Nelson 1975). The sediments act as a nutrient sink to accumulate and sorb soluble nitrogen species onto organic colloids, clays, etc. A longer contact time between the water and soil surface increases the sorption capacity, therefore, longer residence times in the swale can increase the sorption of the soluble inorganic nitrogen as shown in Table 15. A healthy grassy cover appears to have very little, if any, effect on removal of soluble nitrogen forms.

The increase in the organic N concentrations in the May studies at each site may be attributed to increased organic debris found in the swale, such as grass and weed clippings. The significant increase in mowing of the swales can leave large quantities of the grass and weed debris which will subsequently decay and deposit organics into the runoff flowing through the swale.

This organic N deposition in swales is further illustrated by the data compiled from the Maitland Interchange stormwater samples collected during 1982-1983. The average nitrogen concentrations found at Station #4 (direct highway runoff) and Station #6 (a grassy swale receiving runoff from Station #4) from individual storm events are shown in Table 5. Table 6 shows the average percent changes for storm events in the summer (8/18/82, 8/23/82, 9/8/82, 10/1/82) and for storm events of winter (12/11/82, 1/21/83, 1/31/83, 2/3/83, 2/12/83, 2/14/83, 2/27/83). There appears to be a significant difference in the removal of total nitrogen between the two time periods. This is similar to the trends seen in the swale studies performed in January, February and May, as discussed earlier.

It is interesting to note that in the Epcot studies, a weir was placed at the 90 meter point along the 170 meter study length in the swale to monitor flow rate. Examination of Figures 22 and 24 reveals that in the 3/23/83 Epcot study, there was a significant decline in the concentration of all nitrogen forms at the 90 meter sampling point immediately downstream of the weir. This suggests that ponding of the runoff caused by the wier may aid in the removal of nitrogen due to an increased contact time with the absorption sites in the soil. The decrease is not as significant in the second Epcot study on 5/16/83, as shown in Figures 23 and 25.

Analysis of Nitrogen Mass Removal

A mass balance was performed on each swale experiment to determine the overall mass removal rates. The results of the mass balance yielded a percent nitrogen mass removal along the swale and milligrams N removed per unit time per unit area for each study. As can be seen in Table 15 and in Figure 26, a linear relationship exists between average percent removal of total N mass vs. percent water loss by infiltration and other losses; a correlation coefficient of 0.90 exists between the two parameters. This relationship suggests that total nitrogen mass removal along the swale is strongly dependent on the percent infiltration.

Soil infiltration or runoff is a function of various factors such as soil type, porosity, antecedent dry period, and contact time. Contact time, or residence time through the swale, is determined by the intensity and duration of excess runoff over the swale. One can see that a slower velocity through the swale will increase filtration, assuming equal soil conditions and length in the swale during all studies at each site. If the flow does not reach the end of the swale, mass removal efficiencies are considered 100%.

If each of the Epcot swale studies is divided into two sections, the 90 meter length preceding the middle weir and the 80 meter length after the middle weir, a trend in the infiltration rate is evident. The infiltration rate in the first 90 meter section of each Epcot swale study (3/23/83 and 5/16/83) was 32% and 33%, respectively. However, the infiltration rate over the last 80 meter section of each Epcot study was 9% and 25%, respectively. This difference in each of the two studies could be attributed in part to the ponding of the runoff at the weir. The infiltration rate difference may have resulted from changes in the moisture condition or level of the water table at the swale site. During the 3/23/83 study, a period of rainfall had occurred during the preceding weeks and the soil in the swale had reached near saturation in some points along the swale. Thus, the decreased infiltration rate in the last 80 meter section during the 3/23/83 study can be attributed in a large part to soil saturation at the time. A swale built in areas of low elevation would be more susceptible to increased soil saturation and eventually standing water would occur. Since infiltration is the main determinant of nitrogen mass removal, it is obvious that more nitrogen mass was removed in the first 90 meters than in the last 80 meters during each study, as seen in Table 19. As can be seen, when the infiltration rate is low, as in section 90-170 m of the 3/23/83 study, nitrogen mass is actually increased along the section's length.

TABLE 19

INFILTRATION RATES AND NITROGEN MASS CHANGES ALONG THE LENGTH OF THE SWALE AT EPCOT INTERCHANGE

Epcot Swale Study Date	Swale Section	Infiltration Rate (%)	Nitrogen Mass Change (%)	Nitrogen Mass Change (mg/m^2-N)
3/23/83	0- 90 m	32	50	13
3/23/83	90-170 m	9	20*	3*
5/16/83	0- 90 m	33	29	9
5/16/83	90-170 m	25	20	3

* Represents an increase in nitrogen mass - all other stations show a decline in mass.

The total amount of runoff infiltrated will increase with length. Thus, the total mass of nitrogen removed, which is directly related to infiltration, will increase with the length of the swale. It is suggested, therefore, that to achieve a maximum removal of nitrogen mass, swales should be as long as the topographical con- dales built in higher areas of elevation, with the swales sloped as mildly as possible to accommodate the lowest possible flow through velocities which will increase infiltration and nitrogen mass removal.

CHAPTER IV

FATE OF NITROGEN IN DETENTION/RETENTION SYSTEMS

The primary objective of this study is to investigate and develop quantifying relationships for nutrient loadings to detention/retention ponds, removal efficiencies in these ponds and the resulting water quality. It is important to better our understanding of the sediment-water interactions in these ponds in order to reveal factors and parameters which may prove valuable for their design process. Two ponds were surveyed, instrumented and investigated: the eight year-old pond at the Maitland/I-4 Interchange and the newly constructed pond at the Epcot/I-4 Interchange. The data gathered from these two sites was ultimately compared to study common behavior and further our understanding of the design and treatment efficiency of these detention/retention ponds.

The study comprises investigations of nutrient loadings to the pond and resulting quality of both the water body and the bottom sediments. In addition, *in situ* isolation chamber experiments have been conducted to quantify the importance of exchange processes at the sediment-water interface.

Experimental Epcot Site

The detention/retention pond at the Epcot/I-4 Interchange is a 1.4 ha (3.5 Ac) basin located 0.2 ha northeast of ramp A which connects Epcot Center exit to the westward side of I-4, as shown in Figure 6.

The main stormwater input is delivered by overland flow from a grassy swale that empties into the northeast corner of the detention pond. The quality of this input was fully investigated during the stormwater monitoring program implemented later in the study. Additional sources of nutrients may result from natural atmospheric fallout and sheet flow from highways located adjacent to the pond. The land adjacent to the Epcot pond is essentially undeveloped and dominated by pine woodlands. Therefore, runoff to the pond is essentially from the roadway environment.

Water Sampling Locations at Epcot Site

Characteristics for the Epcot pond are shown in Table 20. Stormwater samples from the drainage basin of the Epcot pond were collected between June 1983 and November 1983. It was observed that the main inflow of stormwater runoff is located in the northeast corner of the pond at the stormwater sampling Station #1 (see Figure 6). Furthermore, it is important to notice that outflow from the pond is very rare and may be impossible. Therefore, the pond actually works as a retention rather than a detention pond.

TABLE 20

CHARACTERISTICS FOR EPCOT RETENTION POND

Parameter	Value
Total Drainage Area, (ha)	8.3
Approximate Surface Area, (ha)	1.4
Average Water Depth, (m)	1.1

In order to adequately determine the input, removal and accumulation of nutrients in the pond, numerous stations for field measurements and runoff sampling were chosen.

Station #1 is located west of the connector road, in a grassy swale 40 feet downstream of the exit from a 15 inch reinforced concrete culvert (S-7). This culvert drains the median section of the connector road for a grassy swale distance of 350 feet from the I-4 overpass. The flow through this station empties into the Epcot retention/detention pond which is located 40 feet downstream from the sampling point.

Station #2 is located on the east side of the connector road. It receives direct highway runoff from the exit at a 15 inch reinforced culvert (S-9) and the sampling bottle was placed at the bottom of the culvert. The culvert receives highway runoff from the connector road and is located approximately 400 feet from the center of the I-4 interchange overpass.

Station #3 receives direct runoff from the I-4 interchange overpass via 15 inch culvert. The sampling station is located on the exit side of the culvert (S-13) which empties into a nearby artificial wetland. Therefore, the concentrations found at station #3 are representative of the initial stormwater concentrations entering the wetland.

Station #4 is located at the exit point of the wetland receiving stormwater from station #3. The sampling station is located just upstream from a 12 inch pipe discharging from the wetland into a roadside swale. Therefore, the concentrations found at station #4 are representative of the effluent concentrations leaving the wetland. The overland distance from station #3 and station #4 is approximately 400 feet.

Station #5 is located approximately 1000 feet downstream from station #4 in the swale that runs along the access road parallel to I-4. Grab samples are taken just downs-ream from the exit of a 42 inch reinforced concrete culvert under ramp A (S-51) just west of the ramp B overpass. The swale receives the stormwater runoff from station #4 and other runoff upstream from station #5.

Station #6 is located approximately 300 feet downstream from station #5 just upstream for two 42 inch reinforced concrete culverts (S-42) that run under ramp A. Grab samples are collected in the swale just upstream from the culverts. The swale enters a cypress wetland at the downstream side of the culverts; therefore, the concentrations obtained at station #6 are representative of the initial concentrations entering the cypress wetland.

Station #7 is located within the cypress wetland approximately 300 feet downstream of station #6. The station is represented by a grab sample taken at the end of the wetland. The flow path overland through the wetland is in a southwesterly direction approximately parallel to I-4. The sampling site is approximately 75 feet west of ramp A, parallel to I-4.

Isolation Chambers (IC) Studies

Isolation chambers were constructed and placed at Epcot retention pond to investigate the water-sediment interactions. Two sets of experiments were conducted during 1983, as shown in Table 21.

The site locations of these experiments are shown in Figure 28. Water quality measurements and samples were taken adjacent to the chambers from the open pond at the same time samples were collected from the isolation chambers. Also, bottom sediment cores were collected from various locations along transects in the pond, as shown in Figure 29 . These cores were sectioned and analyzed for nitrogen, phosphorus and organic matter.

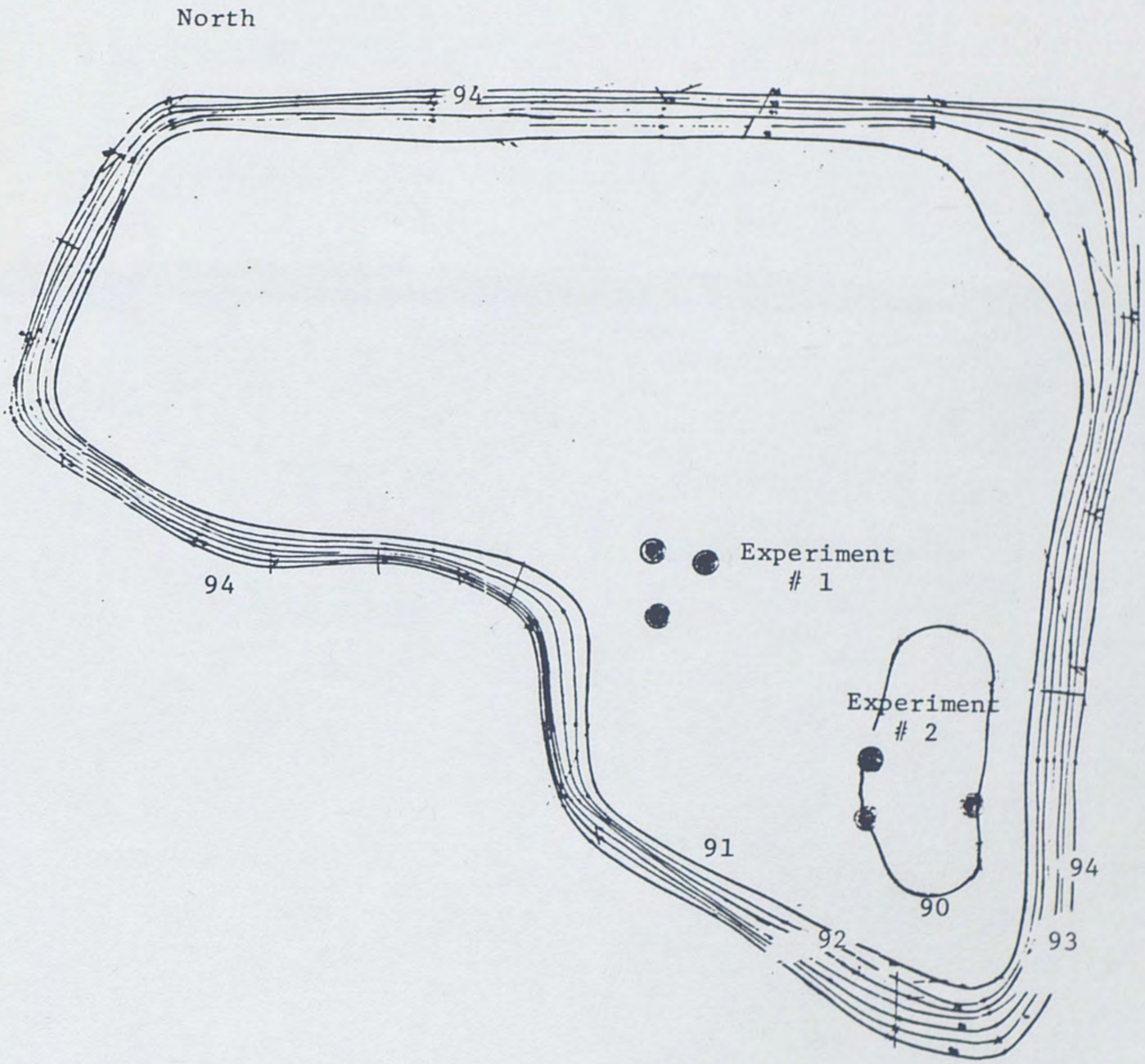

Fig. 28. Locations for isolation chamber experiments in Epcot retention pond.

North

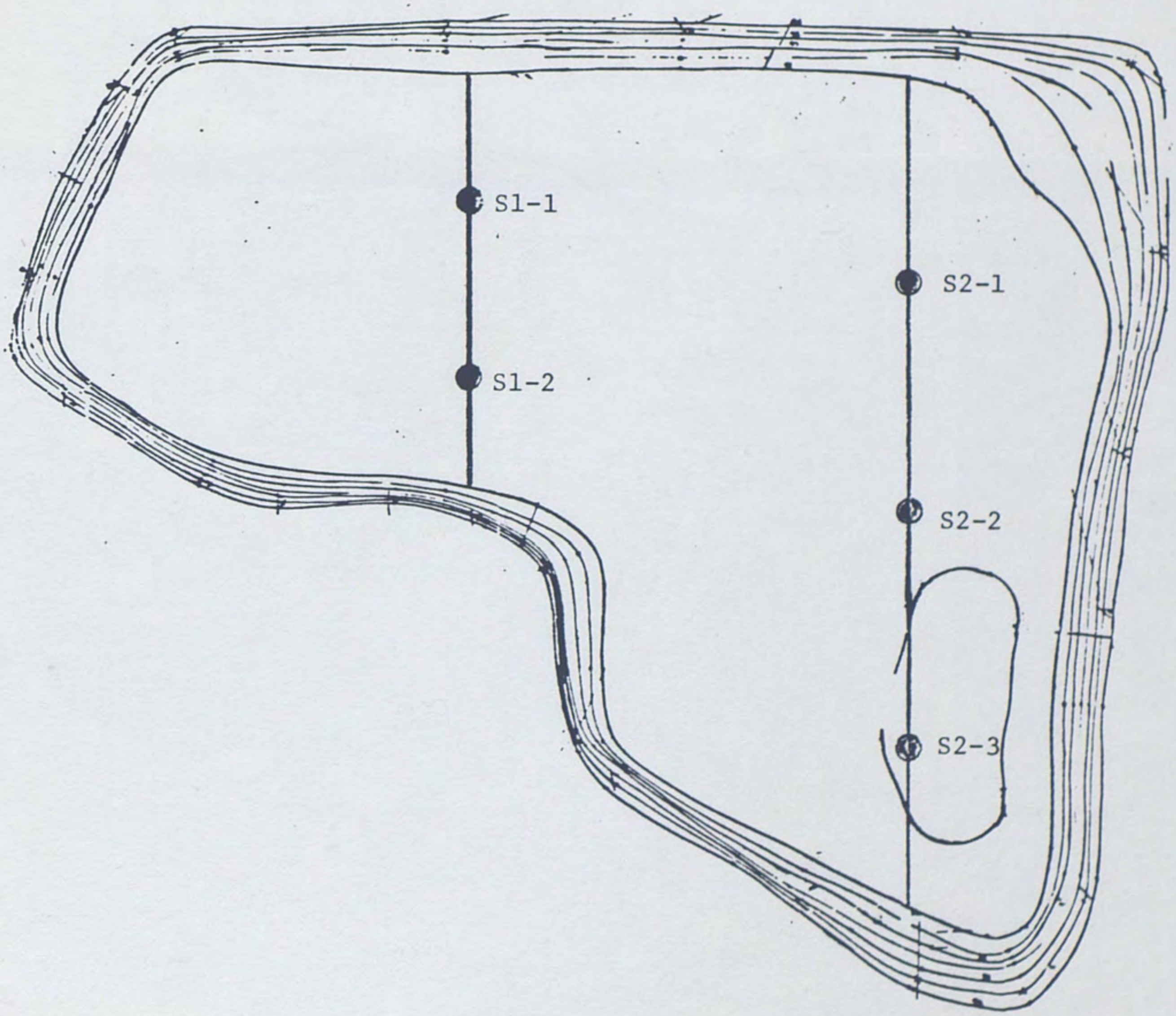

Fig. 29. Sampling stations along transects for sediment cores collected from Epcot retention pond.

TABLE 21

SUMMARY OF ISOLATION CHAMBER EXPERIMENTS
CONDUCTED AT EPCOT RETENTION POND

Experiment	Period	Conditions
First	March 25 - April 18	Aerobic
	April 21 - May 24	Anaerobic
Second	June 1 - June 30	Aerobic
	July 5 - November 7	Anaerobic

Experimental Procedures and Methodology

Sampling procedures, field experiments and measurements, and laboratory analysis were designed to determine the fate of nutrients, nitrogen and phosphorus in detention/retention ponds receiving highway runoff. Therefore, this study places special emphasis on quantifiable mass balances for nitrogen and phosphorus in the pond system, as well as experiments performed to study the rates of removal and accumulation of these nutrients.

Dissolved oxygen and temperature measurements were determined in the field using the Winkler Method and a thermometer.

Collection of Stormwater Runoff Samples

Water samples from highway runoff, as well as samples flowing through swales, were collected. The apparatus used for stormwater collection consists of a 29 cm wide, open plexiglass tray acting as an inlet in front of a 3.8 liter polyethylene bottle which was placed at a lower level than the collection tray. The collection tray is connected with a tygon tube which allowed water samples from overland and sheet flow to fill in the attached bottle. Samples were transported and analyzed for selected water quality parameters.

In Situ Isolation Chamber (IC) Apparatus

Aquatic polyethylene chambers were constructed and submerged with the open top down at the bottom sediments of the pond for the purpose of measuring nitrogen, phosphorus and heavy metal exchange rates at the sediment-water interface and nutrient transformations in the aquatic environment, as shown in Figure 30. The isolation chambers are cylindrical and each has a capacity of approximately 200 liters.

The exterior of the chambers were painted black with epoxy to insure that light did not penetrate into the chambers and activate primary production within the chamber. Each chamber had two lateral water sampling ports which could be operated by a portable peristaltic pump. Also, two release valves located on the top of the chamber were connected to tygon hoses to allow excess air and gases to be released into the air above the pond surface. The sampling ports were also used to inject nutrient and heavy metal solutions directly into selected chambers at the beginning of each experiment after establishing uniform conditions inside the tanks. Because of the fact that a certain depth (0.2 m) of the chamber was embedded into the bottom sediments, the effective volume of water retained by the IC was approximately 150 liters. The bottom surface area covered by each chamber was approximately 0.25 m^2.

Both anaerobic and aerobic conditions were created within each isolation chamber. To create aerobic conditions, an external air cylinder was connected by tygon hose to air stone diffusers located near the bottom of the chamber, as shown in Figure 30. Air flow was regulated by several valves to produce a low, continuous

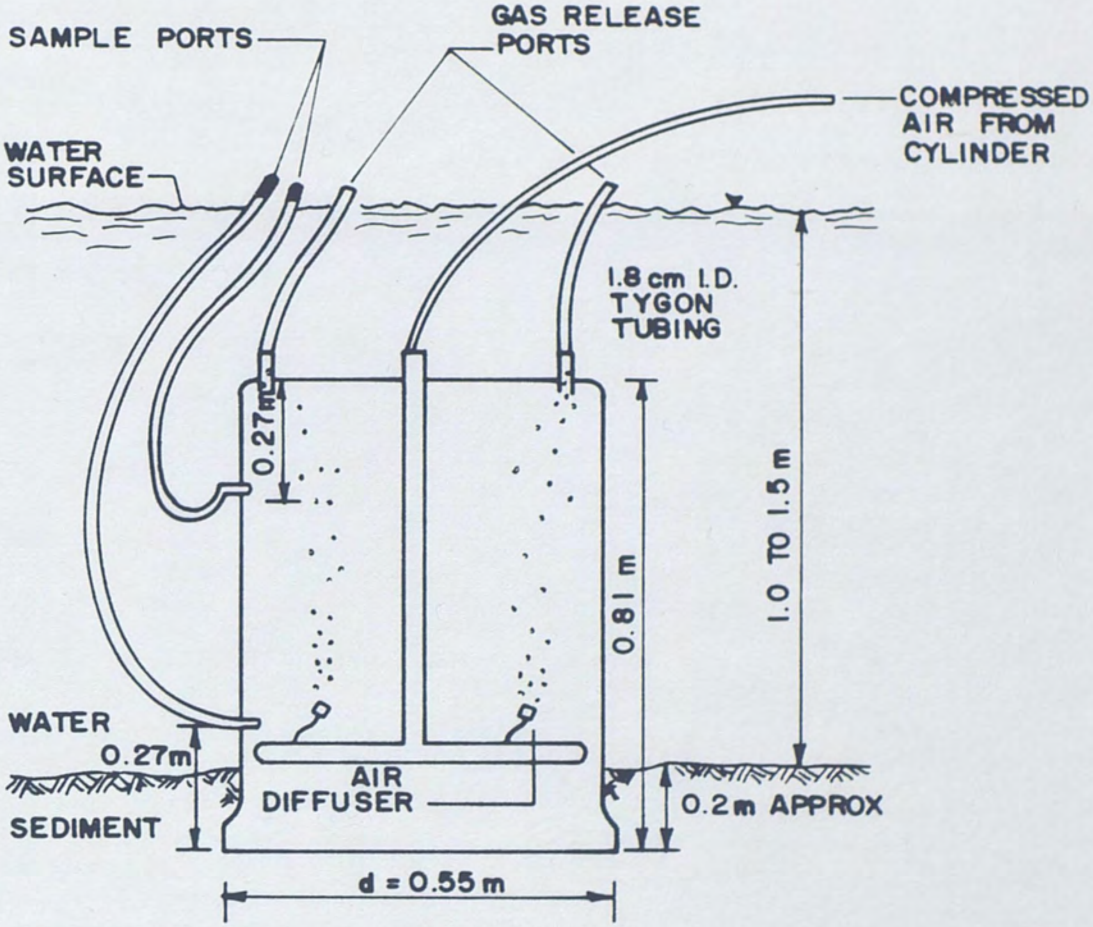

Fig. 30. Schematic diagram for isolation chamber used in the Epcot pond experiments.

flow rate into the chambers. Concrete building blocks were placed on top of the chambers to prevent floating. Water flow may be carried out from the inside of the isolation chamber to the open pond during excessive air pressure, which should be avoided. Careful attention and observation of air flow should be closely followed to avoid any loss of water from inside the isolation chambers and remedial actions should be considered. Anaerobic conditions are created after cessation of the air flow to the chambers which do not create problems associated with creating aerobic conditions.

Three chambers, as described above, were used in each experiment, two chambers (IC #1 and IC #2) were placed with water-sediment contact and a polyethylene cover was placed on the bottom sediments for IC #3 to block the exchange between water and sediment. A known amount of NH_4-N, P and heavy metals were injected into two chambers (IC #2 and IC #3). Consequently, IC #1 was used as a control during each experiment. Thus, the three chambers functioned as (1) control with sediment contact, IC #1, (2) solution of various contaminants was added to IC #2, with sediment contact, and (3) solution of various contaminants was added to IC #3, with polyethylene covering the bottom of the chamber.

Based on the analysis of samples collected from these three chambers during the study periods, in addition to water quality changes measured in pond samples, it is possible to determine:

1. Exchange rates of nutrients and heavy metals between the water phase and the bottom sediment under anaerobic and aerobic conditions.

2. The rate of oxygen depletion under aerobic conditions as well as the importance of sediments in oxygen depletion.

3. The importance of the sediment as opposed to the water phase for producing water quality changes.

4. Capacity of the sediment as a sink to absorb shock loadings.

Samples were collected one or two times a week during the earlier period of each experiment, but with less frequency (bi-weekly or monthly) during the latter stages. All samples were analyzed in the laboratory as stated earlier.

Sediment Samples

Core sediment samples were collected using plexiglass tubes of 2.5 cm inside diameter and 30 cm long. The tube is connected at the end of a 5 cm PVC pipe and a rubber stopper is attached to a nylon line to close the plexiglass tube when the line is pulled. The sediment core is collected by suction created from pushing the plexiglass tube into the sediment and simultaneously pulling the nylon line to move the stopper up the tube as the sediment enters. The sample is then quickly lifted out of the water and another rubber stopper is placed to seal off the bottom of the tube and prevent the loss of the sample.

After being transported back to the laboratory, the sediment cores were frozen and sectioned into four sub-samples for sediment depths of 0-1.0, 1.0-3.5, 3.5-6.0, 6.0-8.5 and 8.5-13 cm. The subsamples were homogenized and used to measure TKN, TP and heavy metals content as μg per gram of dry sediment. A total of 5 sediment core samples were collected, sectioned and analyzed.

Redox Potential of Water Inside the Isolation Chambers

The oxidation-reduction potential of water inside the isolation chambers is used to indicate the types of reactions most likely to occur. Therefore, continuous water flows from the upper and lower level inside the chambers were withdrawn by a peristaltic sampling pump to measure the redox potential by a Hydrolab System 8000 water quality monitor. The hydrogen electrode attached to the monitor was adjusted to pH 7. Redox potential values greater than 200 mV represent a state of oxidation and, conversely, values less than 200 mV represent a state of reduction (Mortimer 1941).

Results

Water samples were collected from the drainage basin surrounding the detention/retention pond at Epcot Interchange and from the pond to better understand the fate of highway runoff pollutants discharged to the surrounding environment. Samples of stormwater runoff, pond water, bottom sediments and water from the isolation chambers were taken from various sampling locations under different conditions. The samples were analyzed for nitrogen, phosphorus and heavy metal concentrations.

Stormwater runoff samples were collected from seven stations surrounding the retention/detention pond at Epcot Interchange site, as shown in Figure 6 during July to November 1983. Statistical analyses of the results are presented in Tables 22 and 23.

The concentrations of nitrogen and phosphorus show significant variability in the quality of the stormwater runoff; however, the average concentrations based on the total of 65 samples analyzed were 0.19 mg TP/l and 2.1 mg TN/l which showed an agreement with several previous investigations (Malmquist 1979 and 1981). The average stormwater concentrations from highway runoff, based on 12 different investigations, are 0.2-0.4 mg TP/l and 1.5-2.5 mg TN/l (Jacobsen and Yousef 1983). Three-fourths of the nitrogen in the stormwater runoff is generally of the organic form, and only one-fourth is soluble inorganic. On the contrary, approximately one-half of the total phosphorus in the runoff was dissolved orthophosphorus. Consequently, the average N/P ratio, for dissolved species, is lower, 6.2, than the N/P ratio calculated from average total concentrations, 11.2.

As shown in Table 21, sampling stations had been chosen which receive direct highway and bridge runoff, runoff after flow through swales, and runoff after flow through wetland areas. The data obtained from direct runoff, and after flowing through a swale area, from sampling stations 1 and 2, show an apparent decline in concentrations of dissolved orthophosphorus, total phosphorus, inorganic nitrogen, and total nitrogen. The average percent removals for dissolved OP, TP, IN and TN were 27, 14, 55 and 27%, respectively.

TABLE 22

STATISTICAL ANALYSIS OF WATER QUALITY PARAMETERS FROM HIGHWAY STORMWATER RUNOFF AT EPCOT INTERCHANGE DURING JULY - NOVEMBER 1983

Sampling Station Number	Site Characteristics	Number of Samples (n)	Phosphorus (μg/1)				Nitrogen (μg/1)				N/P Ratios	
			Diss OP		TP		Inorg N		TN		Diss.	Total
			$\overline{X}$	σ	$\overline{X}$	σ	$\overline{X}$	σ	$\overline{X}$	σ		
1	After flow through sample	8	200	124	356	140	599	361	1797	1455	3.0	5.0
2	Directly from highway	8	274	434	414	434	1327	1299	2455	2343	4.8	5.9
3	Highway bridge runoff	10	48	54	73	54	287	165	696	301	6.0	9.5
4	Discharge from wetland	9	27	14	230	349	741	277	6727	8047	27.4	29.2
5	After flow through swale	10	36	37	82	53	286	192	1102	545	7.9	13.4
6	After flow through swale	10	30	25	123	101	422	285	1279	679	14.1	10.4
7	After flow through wetland	10	36	36	127	118	392	271	1256	270	10.9	9.9
Overall		65	93	-	189	169	579	--	2121	1858	6.2	11.2

TABLE 23

STATISTICAL ANALYSIS OF WATER QUALITY PARAMETERS FROM HIGHWAY STORMWATER RUNOFF AT EPCOT INTERCHANGE DURING JULY - NOVEMBER 1983

Sampling Station Number	Site Characteristics	pH		Alkalinity (mg/1)		Spec. Conduc. (μhos)	
		$\overline{X}$	σ	$\overline{X}$	σ	$\overline{X}$	σ
1	After flow through swale	5.86	0.8	63	49	212	246
2	Directly from highway	6.37	0.2	36	19	93	50
3	Highway bridge runoff	6.70	0.6	43	25	93	56
4	Discharge from wetland	4.13	0.4	2	5	74	16
5	After flow through swale	6.50	0.4	88	27	170	37
6	After flow through swale	6.25	0.4	56	34	117	61
7	After flow throuth wetland	6.31	0.4	57	32	118	59

Therefore, the data suggests that swales may remove a portion of nitrogen and phosphorus received from stormwater runoff. Comparison of the data from sampling station #3, representing direct highway runoff flowing into a wetland area, and from station #4, representing the discharge from the same wetland, revealed that flow through the wetland area generated large amounts of total nitrogen and phosphorus. The average percent additions of total nitrogen and phosphorus were 215 and 867%, respectively. In addition, a comparison of the pH values of sites #3 and 4 reveals that flow through the wetland produces a significant decrease in pH. The average pH value decreased from a value of 6.70 at station #3, to 4.13 at sampling station #4 which may have caused the apparent increase in concentrations of nitrogen and phosphorus in effluent from the wetland area. This wetland area is dominated by pine trees, thus, the decrease in pH of the runoff flowing through the wetland may have been caused by the tannic acid leaching from decaying pine needles on the ground of the wetland. As shown in Figure 6 , sampling station #1 is the primary stormwater runoff input to the Epcot retention pond in addition to overland flow from the surrounding area. The average concentrations of N and P in the runoff water at station #1 were: 0.36 mg P/l for TP, 0.20 mg P/l for OP, 0.60 mg N/l for IN, and 1.8 mg N/l for TN. These concentrations were much higher than concentrations of the same contaminants measured in the retention/detention pond, which are presented in Table 26.

Total heavy metal concentrations were also measured in the runoff from the 10 storm events. Table 24 summarizes the average concentrations found at each sampling site. The average total metal concentrations extracted from runoff water collected from stations #1 were 3, 3, 10, 18, 2708, 686, 24 and 5 μg/l for Cd, Zn, Mn, Cu, Al, Fe, Pb and Cr, respectively.

A comparison between the data obtained from station #1 and station #2 indicate a general accumulation and release for selected heavy metals from the swale areas. A more detailed study on the fate of heavy metals in highway runoff flowing over roadside swales has been submitted to the Florida Department of Transportation by researchers from the University of Central Florida (Yousef et al. 1984).

A comparison of concentrations obtained from stations #3 (direct highway runoff input to a wetland area) and #4 (discharge from the wetland area) revealed that flow through the wetland significantly increased the total concentration of Cu, Al, Fe, Pb and Cr (see Table 23). This may have resulted from the decline in pH of the wetland and release of certain metals as the runoff water traveled through an acidic environment.

Field Measurements in Epcot Pond

Periodic dissolved oxygen and temperature measurements in the Epcot pond were recorded during the time period of the two isolation chamber experiments, and results are presented in Figures 31 and 32. DO in the pond varied between 5.2 and 9.4 mg/l. It can be seen from

TABLE 24

STATISTICAL ANALYSIS OF TOTAL METAL CONCENTRATIONS FROM HIGHWAY STORMWATER RUNOFF AT VARIOUS STATIONS AT EPCOT INTERCHANGE DURING JULY - NOVEMBER 1983

Sampling Station Number	Site Characteristics	Total Metal Concentrations (μg/1)															
		Cd		Zn		Mn		Cu		Al		Fe		Pb		Cr	
		$\overline{X}$	σ	$\overline{X}$	σ	$\overline{X}$	σ	$\overline{X}$	σ	$\overline{X}$	σ	$\overline{X}$	σ	$\overline{X}$	σ	$\overline{X}$	σ
1	After flow through swale	3	6	3	2	10	6	18	14	2708	2542	686	657	24	6	5	3
2	Directly from highway	1	0.5	24	34	9	10	31	23	282	342	166	144	27	15	2	1
3	Directly from highway	2	2	29	31	2	3	16	7	216	148	128	102	17	8	3	2
4	Discharge from wetland	2	2	21	37	17	11	27	19	1996	2141	1599	733	47	63	27	69
5	After flow through swale	<1.0	0.5	4	1	11	5	22	10	546	386	813	265	29	7	4	0.9
6	After flow through swale	<1.0	0.5	4	2	7	3	21	15	1910	1897	697	206	25	10	5	2
7	After flow through wetland	1	1	4	2	10	6	15	8	1448	628	708	344	23	10	5	1

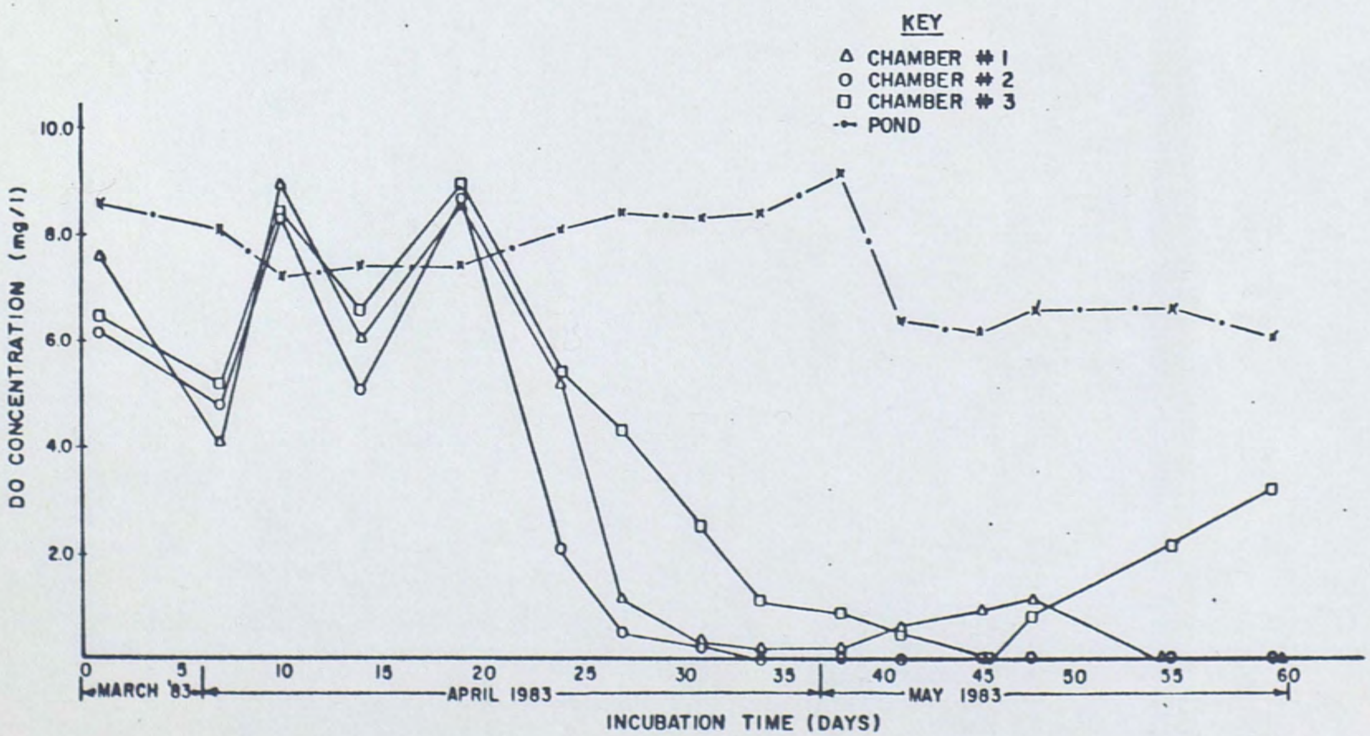

Fig. 31. Dissolved oxygen concentration in the isolation chambers and pond over the length of the first experiment.

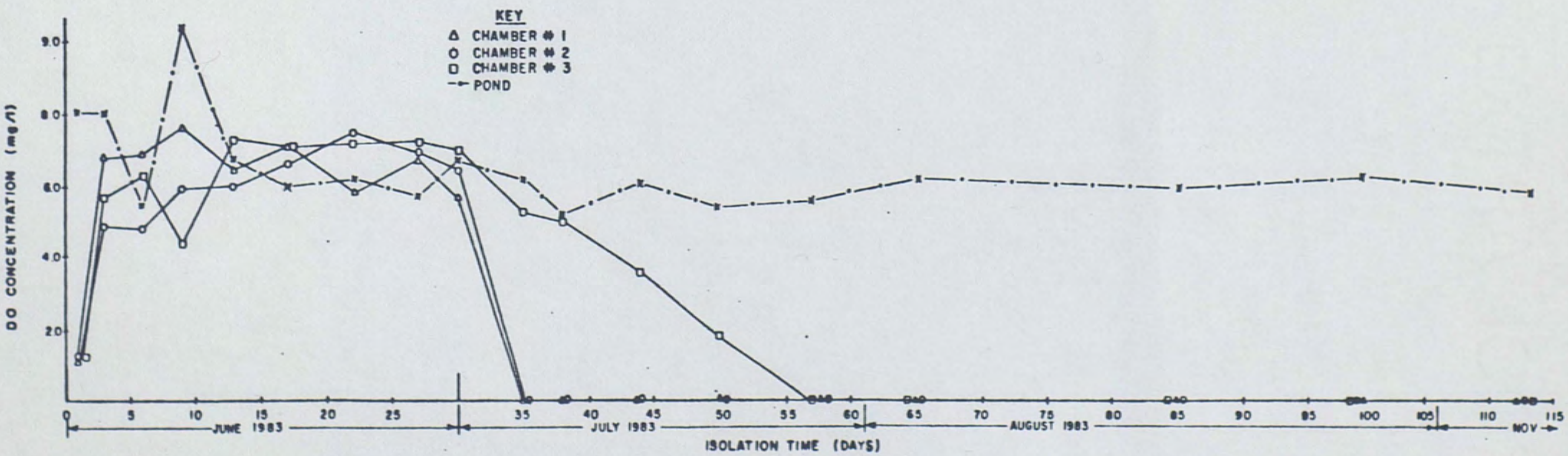

Fig. 34. Dissolved oxygen concentration in the isolation chambers and pond over the length of the second experiment.

these values that high rates of degradation and high photosynthetic activity were not observed. Levels of DO appear to be fairly constant throughout the water column, indicating that high DO concentrations existed at the water-sediment interface. The water column averaged about 1.2 meters depth during the field experimentation period. Temperatures were also measured at the top and near the bottom in the pond and no significant temperature differences were observed between the top and bottom measurements. The lowest temperature recorded, approximately 17°C, occurred in late March at the beginning of the first experiment. The highest values, about 29°C, were recorded in late July.

Water Quality Analysis in Epcot Pond

In the period from March 23 to November 7, 1983, 37 samples for water quality determination were collected in the pond. The following parameters were measured: pH, DO, dissolved orthophosphorus, total orthophosphorus, total phosphorus, TKN, ammonia, nitrite and nitrate. From data summarized in Table 25, the average concentrations of dissolved N and P in the pond are much lower than the average concentrations of runoff water entering the pond from station #1 averaged in Table 22.

The average available nitrogen to phosphorus ratio based on inorganic nitrogen and dissolved orthophosphorus concentrations is 14.4. Chiaudari and Vighu (1974) have shown that in the growth of almost all algae, neither nitrogen or phosphorus is limiting when

TABLE 25

WATER QUALITY PARAMETERS FROM EPCOT POND
(Data Summarized for the Period from March 23 to November 7, 1983)

	pH	DO (mg/1)	Phosphorus (μg/1)			Nitrogen (μg/1)				
			Diss OP	Total OP	TP	Org N	NH_4-N	NO_2-N	NO_3-N	TN
Average	6.7	7.0	13	25	84	830	103	2.1	82	1020
Standard Deviation	0.6	1.2	7	14	40	400	60	0.9	56	386
Number of Samples	37	37	36	36	36	37	36	37	37	37

the N/P ratio lies between 5:1 and 10:1. Thus, the calculated value shows that phosphorus is most likely to be the biomass limiting nutrient. Also, the low dissolved orthophosphorus values, 13 μg/l on the average, show that growth rate limitation probably exists.

In addition, the metals concentrations, as presented in Table 26, reveal that the average concentrations of all the metals except Al are comparable to the metal concentrations in stormwater runoff, station #1 in Table 24. Al concentration in the pond was significantly lower than the concentration found in the stormwater runoff input to the pond.

TABLE 26

AVERAGE TOTAL METAL CONCENTRATIONS IN WATER SAMPLES COLLECTED FROM EPCOT POND DURING THE FIRST EXPERIMENT

	Total Metals Concentration (g/l)							
	Cd	Zn	Mn	Cu	Al	Fe	Pb	Cr
Average	3	5	2	26	435	615	38	8
Standard Deviation	3	4	2	6	529	338	12	4
Number of Samples	6	6	6	6	6	6	6	6

Bottom Sediments

Five core samples of bottom sediments were collected from the Epcot pond. Each core sample was sectioned into five sub-samples representing 0-1, 1-3.5, 3.5-6.0, 6-8.5 and 8.5-13.0 cm depths. The sediments were analyzed to determine the moisture and organic content (μg/g of dry sediment) presented in Table 27. As can be seen, the moisture and organic content in the sediments decline with depth.

The theoretical calculated bulk density has been calculated according to the following expression:

$$\rho = \frac{100\ (\rho_i)}{100 + (W + OG) \times (\rho_i - 1)}$$

TABLE 27

SUMMARY OF AVERAGED VALUES FROM SEDIMENT ANALYSIS AT EPCOT POND

	Sample Depth (cm)	Avg Depth (cm)	Moisture Content (%)	Organic Content (%)	Dry Weight of Sample (g)	Nutrients (g/g)	
						TN	TP
Average of 5 Samples	0 - 1.0	0.5	59.1	18.4	2.11	598	1559
	1.0- 3.5	2.25	32.7	7.2	13.83	230	981
	3.5- 6.0	4.75	26.8	5.5	16.54	151	810
	6.0- 8.5	7.25	26.3	5.1	17.56	162	874
	8.5-13.0	10.75	25.5	5.0	19.27	137	859

where:

ρ = bulk density of the sediment (g/cm^3)

ρ_i = density of inorganic components (g/cm^3)

W = moisture content (percent)

OG = organic content (percent)

It has been assumed that ρ_i = 2.5 g/cm^3 (sand and silt particles) and that the density of the organic matter is 1.0 g/cm^3 (Jacobsen and Yousef 1983).

This theoretical curve is shown in Figure 33. The apparent density of the sediment is calculated as follows:

$$\rho \text{ APP} = W^1 + \frac{M}{V}$$

where:

ρ APP = apparent bulk density of the sediment (g/cm^3)

M = dry weight of the sub-sample (g)

V = volume of the sub-sample in the tube (cm^3)

W^1 = moisture content in the fraction of core

The values of apparent density (ρ APP) calculated based on data in Table 27 are shown in Figure 33. A comparison between these apparent and theoretical densities indicates less than 10% expansion of the sediment cores during sampling and freezing .

From results shown in Table 27, it is apparent that nutrient concentrations for TP and TN decrease with sediment depth. TN at

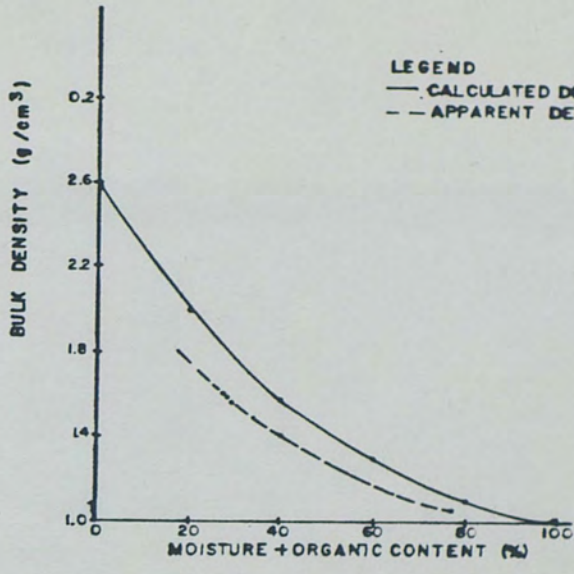

Fig. 33. Sediment bulk densities as a function of moisture and organic content.

the surface was 598 μg N/g dry weight sediment and 137 μg N/g at a depth of 8.5-13 cm.

Nitrogen in Sediments

Based on the analysis of 5 sediment core samples, the averaged total concentrations of nitrogen and phosphorus have been presented in Table 27. The concentrations of N at different sediment depths are shown in Figure 34. It can be seen that the concentration of nitrogen, on the average, is fairly constant in the three deepest layers. Therefore, it is assumed that these three depths represent the background level, 0.15 mg N/g dry weight.

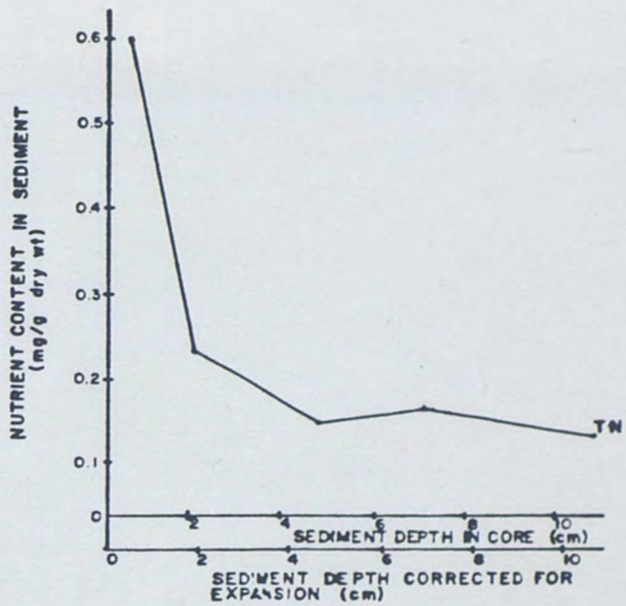

Fig. 34 . Average total nitrogen content versus sediment depth from the Epcot pond.

Isolation Chamber Experiments

Isolation chamber experiments, as described earlier, consist of two experiments. Each experiment included an aerobic and anaerobic phase inside the three isolation chambers used.

The data obtained from the two isolation chamber experiments were gathered and tabulated in Tables 28 and 29. Table 28 shows the concentration of nitrogen forms during the first experiment at

TABLE 28

THE CONCENTRATIONS OF NITROGEN FORMS IN WATER FROM THE FIRST EPCOT RETENTION POND EXPERIMENT BETWEEN 3/25/83 AND 5/24/83

Location	Nitrogen Form	Concentration at Sampling Date (μg-N/1)															
		3/25	4/1	4/1*	4/4	4/8	4/13	4/18	4/21	4/25	4/28	5/2	5/5	5/9	5/12	5/19	5/24
Isolation Chamber # 1	NH_4-N	112	127		104	84	50	10	49	---	42	53	41	21	5	87	161
	NO_2-N	1.6	2.3		2.4	3.2	4.6	1.9	2.0	3.7	1.8	0	0	0.9	1.0	0.8	1.0
	NO_3-N	260	102		94	92	141	136	136	53	107	108	70	114	96	35	36
	Organic N	1108	543		453	428	558	440	379	456	493	544	916	407	378	718	492
	Total N	1482	774		653	607	754	588	566	513	644	705	1027	543	480	841	690
Isolation Chamber # 2	NH_4-N	46	143	2306	353	84	53	13	79	---	292	286	326	409	457	336	381
	NO_2-N	0.9	3.9	4.1	3.1	19.7	2.5	2.0	2.4	2.6	2.5	0.1	0.5	3.1	3.1	1.5	1.4
	NO_3-N	238	202	890	185	161	167	102	87	107	191	125	125	95	91	57	40
	Organic N	808	470	705	474	428	634	550	506	625	378	513	574	503	421	705	519
	Total N	1093	819	3905	1015	693	857	667	674	735	864	924	1026	1010	972	1100	941
Isolation Chamber # 3	NH_4-N	266	97	1982	252	104	56	40	49	---	34	8	8	29	13	32	40
	NO_2-N	0.7	2.0	2.2	3.1	16.9	2.6	2.9	2.5	2.4	1.7	0.2	0.8	1.0	1.1	0.7	0.8
	NO_3-N	144	98	797	234	215	101	185	195	292	305	296	221	189	274	280	245
	Organic N	954	425	511	530	442	676	602	458	467	512	453	442	399	392	469	309
	Total N	1099	622	3292	1019	778	836	830	705	761	853	757	672	618	680	782	595
Pond	NH_4-N	253	74		94	73	50	44	61	---	35	93	37	117	83	61	36
	NO_2-N	0.6	2.9		3.0	2.2	2.5	2.8	2.7	2.2	2.5	2.1	1.0	0.3	1.0	1.4	0.9
	NO_3-N	68	139		152	321	94	85	79	134	96	99	79	45	86	33	15
	Organic N	55	607		497	524	682	609	783	486	1074	1156	954	682	547	845	853
	Total N	377	823		746	920	829	741	926	622	1208	1350	1071	844	717	940	905

* Concentrations in samples after addition of contaminant.

TABLE 29

THE CONCENTRATION OF NITROGEN FORMS IN WATER FROM THE SECOND EPCOT RETENTION POND EXPERIMENT BETWEEN 6/1/83 AND 11/7/83

Location	Nitrogen Form	Concentration at Sampling Date (μg/N-1)																			
		6/1	6/3	6/6	6/6*	6/9	6/13	6/17	6/22	6/27	6/30	7/5	7/8	7/14	7/20	7/27	8/4	8/24	9/7	9/21	11/7
Isolation	NH_4-N	1688	1085	1146		298	306	316	123	255	300	555	858	1464	1378	1633	1664	1782	1909	1549	2270
Chamber	NO_2-N	5.7	10.9	11.0		134	6.7	8.3	2.6	3.9	8.3	4.2	6.9	3.0	4.6	9.4	1.5	9.0	7.1	7.1	7.3
# 1	NO_3-N	330	240	155		512	1211	1345	1590	1667	990	102	207	179	197	100	320	140	209	231	350
	Organic N	235	872	664		1240	667	928	2026	2290	967	769	1180	1930	1326	1523	531	2041	815	2851	2832
	Total N	2259	2208	1976		2184	2191	2597	3442	4216	2265	1430	2252	3576	2906	3265	2517	3972	2940	4638	5459
Isolation	NH_4-N	1831	1583	1347	2319	2315	69	67	91	180	76	457	686	1068	1142	1462	1493	1583	1725	1506	2673
Chamber	NO_2-N	5.6	17.7	7.7	8.2	76	98.4	1.8	2.9	3.9	3.6	4.8	2.8	3.0	0.6	4.8	1.3	4.8	4.2	4.7	0
# 2	NO_3-N	315	336	174	1729	1683	2062	1922	1291	844	378	133	100	227	77	60	93	95	183	139	289
	Organic N	567	747	689	3201	2210	1107	838	859	895	942	572	649	640	1030	902	1290	1878	1951	9501	2078
	Total N	2719	2684	2298	7257	6284	3336	2829	2244	1923	1400	1167	1438	1938	2250	2429	2877	3561	3863	11231	5040
Isolation	NH_4-N	1104	543	182	1842	143	128	62	36	63	51	78	32	18	13	25	161	202	61	120	133
Chamber	NO_2-N	6.9	11.1	64.0	43	164	2.9	2.3	1.8	1.7	1.4	1.3	0.9	0.5	2.4	1.3	8.4	26.4	0.8	0.0	0
# 3	NO_3-N	237	174	162	1343	1843	2169	2377	1614	1661	1742	1823	1787	1371	1233	1245	1024	1016	222	19	100
	Organic N	265	713	870	805	1203	935	1058	903	819	786	669	760	717	645	665	--	567	369	493	342
	Total N	1613	1441	1278	4033	3353	3235	3499	2555	2545	2580	2571	2580	2107	1893	1936	--	1811	653	632	575
Pond	NH_4-N	242	118	121		111	162	107	158	146	181	73	194	148	167	63	151	<10	38	31	50
	NO_2-N	2.2	2.3	2.9		2.7	3.3	1-7	1.8	2.4	3.3	2.0	2.6	1.5	1.3	3.1	3.0	0	1	0.6	1.2
	NO_3-N	200	52	62		32	74	65	112	96	56	77	79	79	54	20	86	56	56	9	127
	Organic N	775	889	887		1145	1037	1250	985	883	758	706	643	870	998	1206	788	685	779	976	2676
	Total N	1220	1061	1073		1291	1276	1424	1257	1129	998	858	919	1099	1220	1292	1028	746	900	1017	2851

* Concentrations in samples after addition of contaminant solutions.

Epcot pond (March 25 - May 24, 1983). The concentrations of nitrogen species: NH_4-N, NO_3-N, ON and TN during the second isolation experiment (June 1 - November 7, 1983) are presented in Table 29. The nitrogen concentrations in the three isolation chambers, as well as in the pond itself, are plotted over each experimental run time, as shown in Figures 35 through 42. Each figure represents a different N species concentration change over time (days) in the three chambers and the pond. Figures 35 through 38 are plots of nitrogen concentrations during the length of the first experiment. As can be seen in Figures 35 through 38, the NH_4-N and heavy metal solutions were injected into IC #2 and IC #3 on day 7 (April 1, 1983), distinguished by a marked increase in NH_4-N, NO_3-N and TN. Thereafter, each of these nitrogen forms declined over time under aerobic conditions until they reached minimum concentrations near day 25 (April 18, 1983), when the oxygen supply to the chambers was discontinued. After cessation of air, a release of NH_4-N in IC #2 began on day 25 and continued until reaching a peak near day 50, as can be seen from Figure 35. A much smaller release of NH_4-N was observed in IC #1 (control) under anaerobic conditions. In IC #3, no apparent release of NH_4-N was observed. Also, the TN concentrations increased slightly in IC #2 due to the significant release of NH_4-N. The TN concentration in IC #1 and IC #3 remained fairly constant under anaerobic conditions.

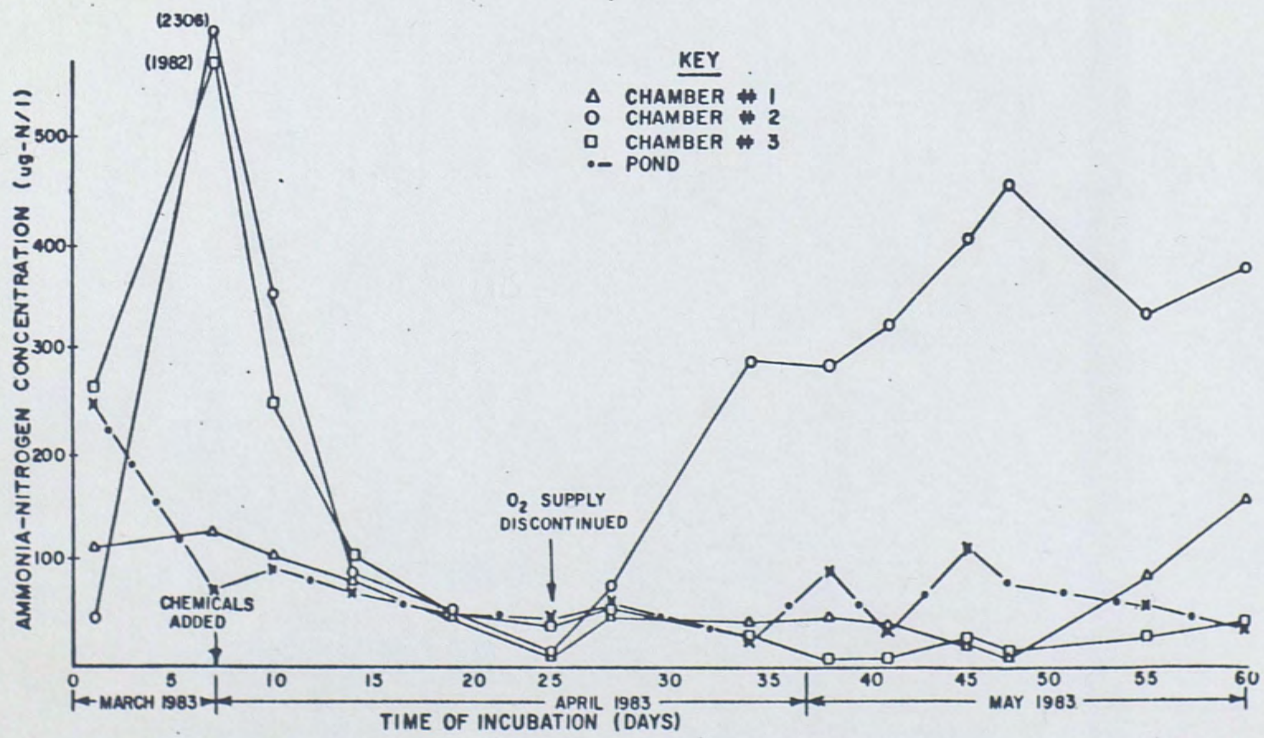

Fig. 35. NH_4-N concentrations in water samples from isolation chambers during the first experiment at Epcot pond.

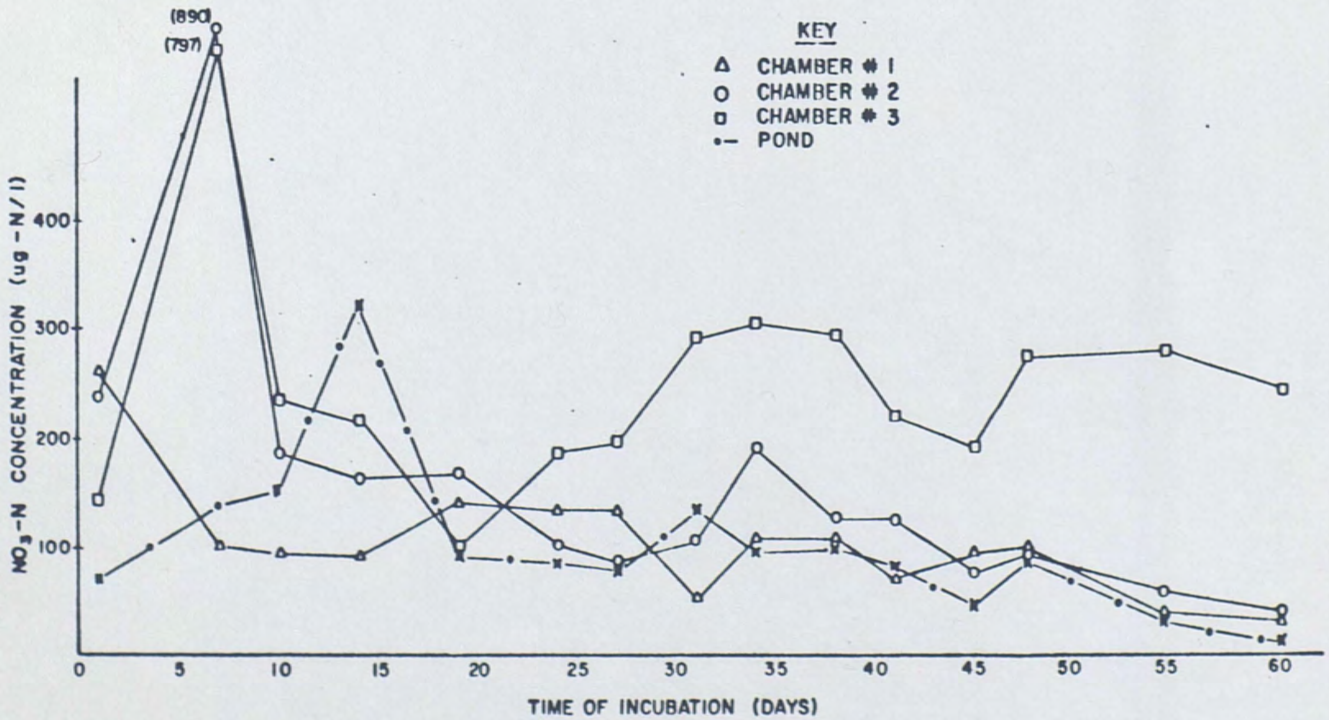

Fig. 36. Nitrate-nitrogen concentrations in water samples from isolation chambers during the first experiment at Epcot pond.

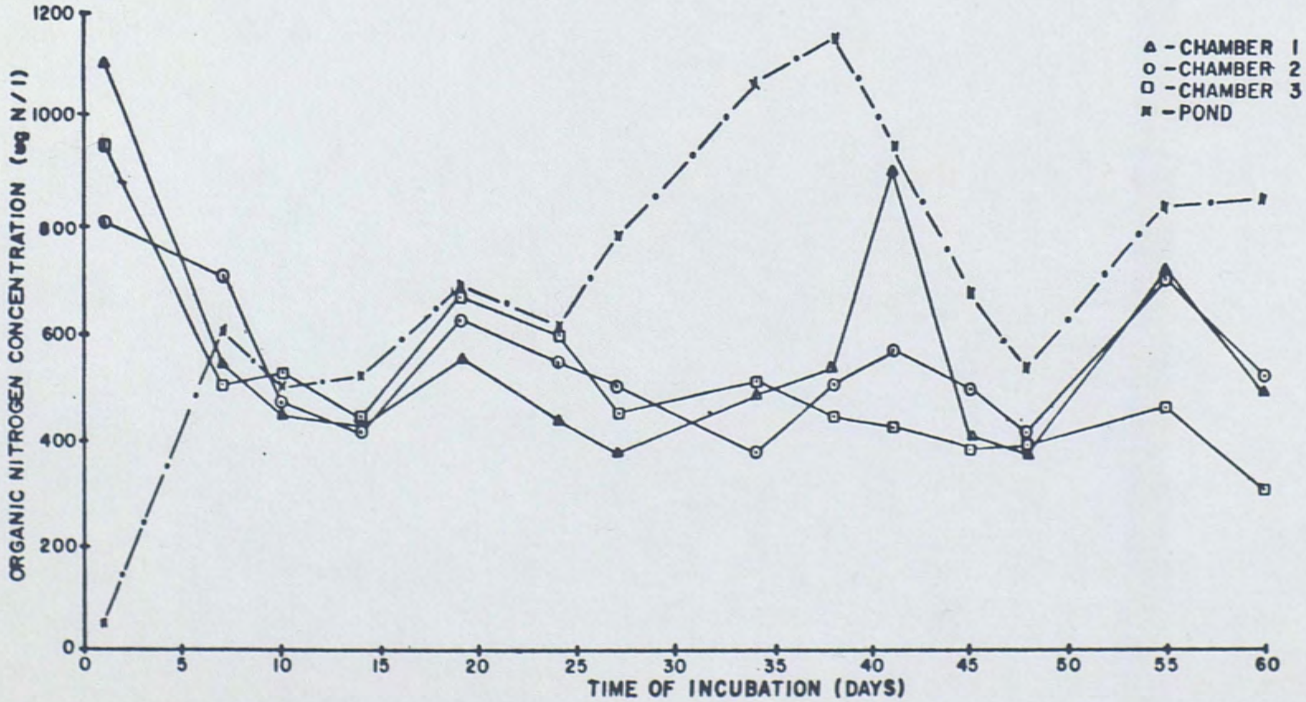

Fig. 37. Organic nitrogen concentration with time in water samples from isolation chamber during the first experiment at Epcot pond.

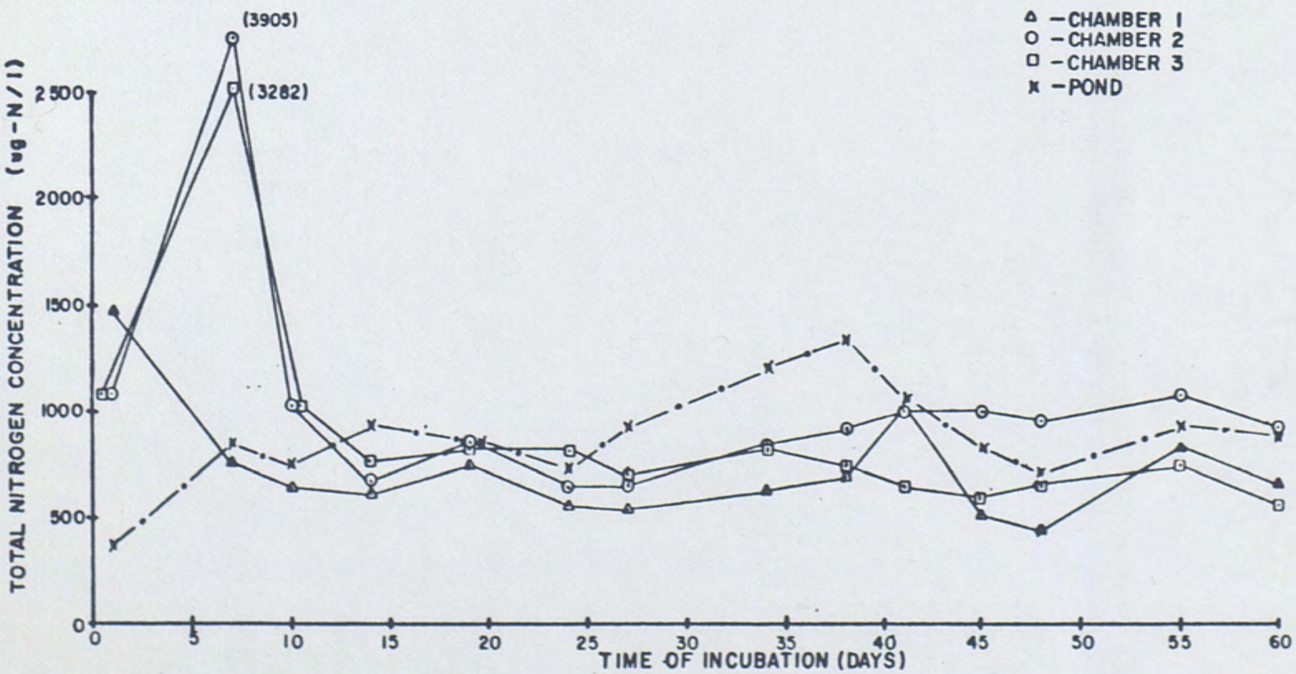

Fig. 38. Total nitrogen concentration with time in water samples from isolation chamber during the first experiment at Epcot pond.

Figures 39 through 42 are plots of N concentrations during the length of the second experiment. The NH_4-N and heavy metal solutions were injected on day 7 (June 6, 1983), producing a significant increase in the concentrations of NH_4-N, NO_3-N and TN in chambers #2 and #3. As was the trend in the first experiment, each form of nitrogen declined with time under aerobic conditions. Minimum values were generally reached after 35 days (July 5, 1983), except for NH_4-N in chamber #2, which reached a minimum on day 30 (June 30) when the O_2 supply was discontinued. As observed in the first experiment, there was an immediate release of NH_4-N after cessation of O_2 and establishing anaerobic conditions during the second experiment (see Figure 39). NH_4-N release began on day 30 and continued during the entire experiment run time until a period of 130 days, up to November 7. A similar release rate of NH_4-N was also observed in chamber #1 under anaerobic conditions. This release was comparable to the release in chamber #2 for much of the anaerobic period; however, from days 113-160, the release rate of NH_4-N in IC #1 was less than the release rate in IC #2. No release of NH_4-N was observed in IC #3, which confirms the observation from the first experiment. The TN concentrations in IC #1 and #2 increased under anaerobic conditions due to the high rate of NH_4-N release. In IC #3, however, TN decreased very slowly under anaerobic conditions.

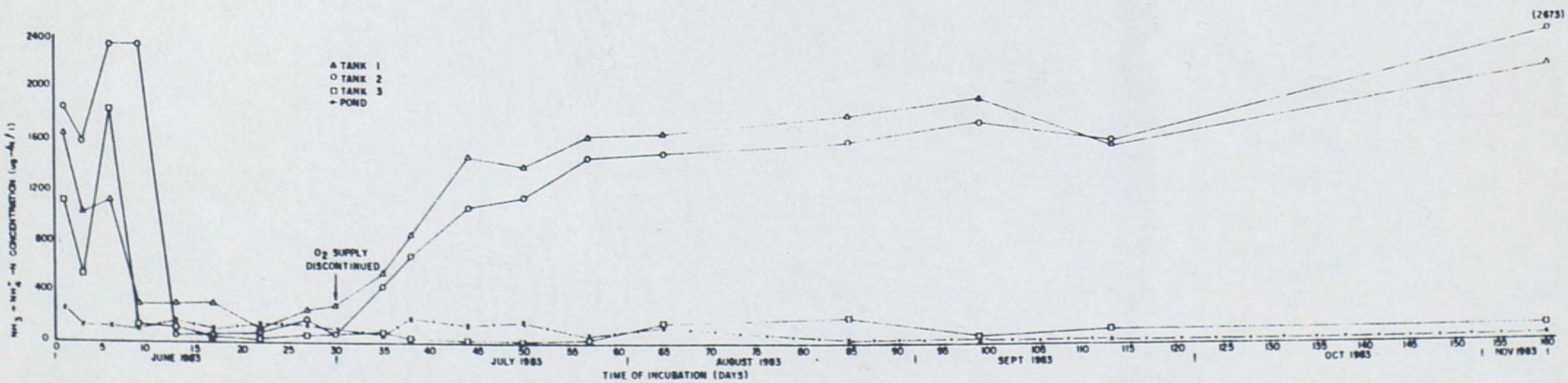

Fig. 39. NH_4-N concentration in water samples from isolation chambers during the second experiment at Epcot pond.

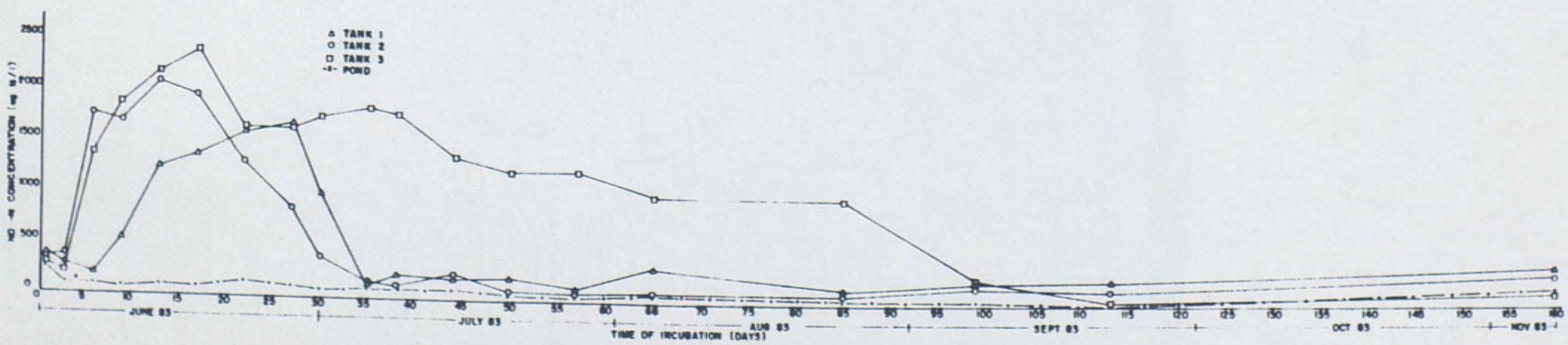

Fig. 40. Nitrate-nitrogen concentration in water samples from isolation chambers during the second experiment at Epcot pond.

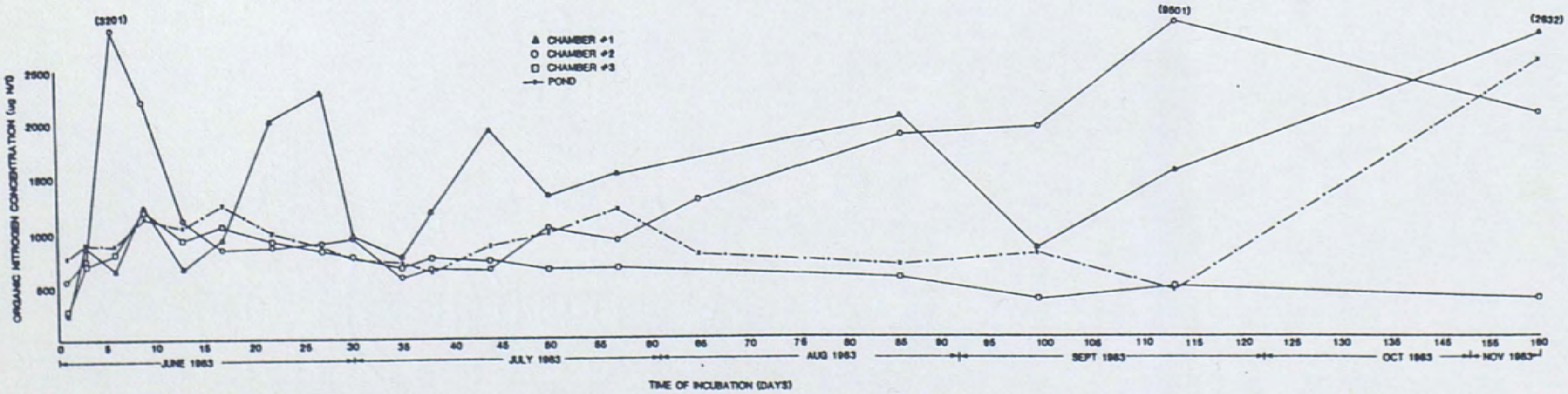

Fig. 41 Changes of organic nitrogen concentration with time in water samples from isolation chambers during the second experiment at Epcot pond.

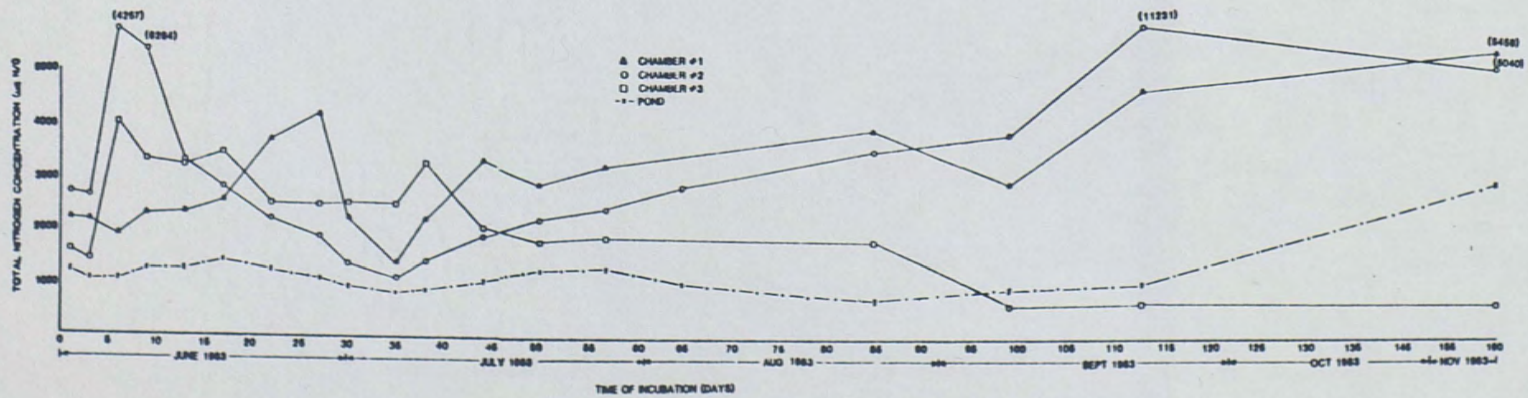

Fig. 42. Changes in total nitrogen concentration with time in water samples from isolation chambers during the second experiment at Epcot pond.

Analysis of Results

The scope of this study was to examine the fate of nutrients in detention/retention ponds receiving highway runoff as a basis for development of design criteria to improve the water quality. Therefore, samples of stormwater runoff, pond water, and sediment from Epcot interchange sites were collected. In addition, in situ isolation chamber experiments were conducted. Results obtained have been presented and a detailed and comprehensive analysis of those results follows.

Water and Sediment Quality in Epcot Pond

One of the main objectives of this study was to determine the actual sediment and water quality of the pond as a basis for evaluating the loading effect relationship. Therefore, during the experimentation period from March to November 1983, an attempt has been made to quantify the fate of contaminants, particularly nutrients, discharged in highway runoff to the retention pond. Data presented in Tables 25 and 26 show important water quality parameters. The average dissolved nutrient concentrations in the pond water based on a total of 37 samples were: 13 $\pm$ 7 μg P/l for dissolved orthophosphorus, 84 $\pm$ 40 μg P/l for total phosphorus, 830 $\pm$ 400 μg N/l for organic nitrogen and 1020 $\pm$ 386 μg N/l for total nitrogen. The total inorganic nitrogen averaged 190 μg N/l.

The nutrients measured in water samples from the retention pond averaged much lower concentrations than water samples collected from highway runoff presented in Table 22. Station #1, which represented the major highway runoff water source to the pond, showed concentrations of 200 ± 124 μg P/l for OP, 356 ± 140 μg P/l for TP, 599 ± 361 μg N/l for IN and 1797 ± 1455 μg N/l for TN.

Based on the analysis of sediment core samples from the Epcot pond, the relationship between the organic content and the nitrogen content of the sediments is shown in Figure 43. The relationship obtained is based on several homogenized samples from 0-1.0 cm depth, from 1.0-3.5 cm, from 3.5-6.0 cm, 6-8.5 cm and 8.5-13 cm depth. There is strong evidence in documentation of a direct correlation between total nitrogen content and organic content in lake sediments (Hankanson 1981). In addition, Jacobsen and Yousef (1983) found a strong relationship between nitrogen and organic content in surficial sediments of the west detention/retention pond at the Maitland/I-4 Interchange. Furthermore, this relationship may be used to determine the level of bioproduction as defined by Jacobsen and Yousef (1983). The value of BPI (Bioproduction Index) which is defined as the nitrogen content on the regression line for an organic content equal to 10%, and the value of BPN (Bioproduction Number) is defined as the slope of the regression line. Based on Figure 43, the following regression line was determined:

$$TN = 0.05 \times OC + 0.62, \qquad r = 0.97$$

where:

TN = nitrogen content in the sediment (mg N/g dry weight)

OC = organic content (percent dry weight)

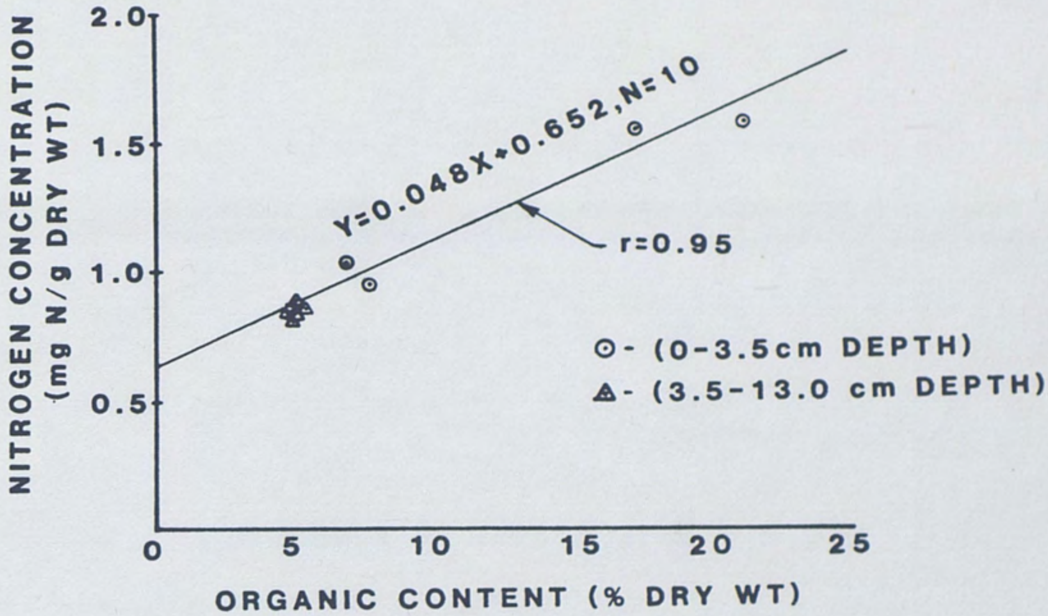

Fig. 43. Relationship between nitrogen concentration and organic content in bottom sediments from the Epcot pond.

From the preceding equation, the slope is 0.05 which represents the BPN. Also, if we substitute the OC in the equation with 10%, the TN would be 1.12, which represents the BPI. The values of BPN and BPI generally reflect the trophic state of sediments in lakes. Hankanson (1981) suggested: BPN > 0.5 for eutrophic state, $0.3 <$ BPN < 0.5 for mesotrophic state and BPN < 0.3 for oligotrophic state.

Thus, it can be seen that the Epcot pond BPN value is in an oligotrophic state and the quality of sediments is high. It appears that the water and sediment quality in the pond is at a high level when evaluated based on nutrient, oxygen and metal concentrations and trophic state indicators.

Exchange of Nitrogen Between Sediments and Overlying Water

The isolation chamber experiments were utilized to measure the removal and release of nitrogen under aerobic and anaerobic conditions by isolating a known volume of pond water in chamber #1. Similarly, the addition of approximately 3 mg/l ammonia into IC #2 and #3 may represent a shock nitrogen loading, and it is desired to evaluate its impact on nitrogen species under aerobic and anaerobic conditions. In addition, water samples were collected from the open Epcot pond adjacent to the location of the isolation chambers.

The nitrogen analysis of the Epcot pond samples collected adjacent to the chambers during the first and second experiments indicate that the total nitrogen concentrations remained fairly constant and oscillated slightly around a mean value of 868 ± 233µg N/l and 1193 ± 438 µg N/l, respectively. Similarly, the organic fraction averaged 736 ± 216 and 998 ± 441 µg N/l and the inorganic fraction averaged 167 ± 92 and 207 ± 110 µg N/l during the first and second experiments, respectively. The overall average concentration of total nitrogen is 1020 ± 386 µg N/l.

The data from isolation chamber #1 under aerobic conditions during the first and second experiments indicate that the average total nitrogen content remained fairly constant at 405 ± 51 and 1643 ± 527 mg N/m^2 of bottom sediment, respectively, as presented in Tables 30 and 31. Similarly, the organic and inorganic fractions averaged 291 ± 37 and 752 ± 391 mg N/m^2 sediment, respectively, in the first experiment and 115 ± 18 and 868 ± 186 mg N/m^2 sediment, respectively, in the second experiment. The data indicated that the organic fraction of nitrogen concentrations was stable during periods in IC #1.

However, when ammonia was added to IC #2 and IC #3, a noticeable decline over time during the aerobic period was observed in TN content during the first and second experiment, as shown in Tables 30 and 31. The decline in TN content as a function of time under anaerobic conditions in each chamber, as shown in Figure 46, can be simulated by a first order equation:

$$N = N_o e^{-k_n t}$$

TABLE 30

ESTIMATION OF TOTAL NITROGEN IN WATER ENCLOSED BY ISOLATION CHAMBERS UNDER AEROBIC CONDITIONS DURING THE FIRST EXPERIMENT AT EPCOT POND

Isolation Chamber	TN Content in IC (mg N/m^2 sediment)				
	4/1/83	4/4/83	4/8/83	4/13/83	4/18/83
1	464	392	364	452	353
2	2343	609	416	514	400
3	1975	611	467	502	502

TABLE 31

ESTIMATION OF TOTAL NITROGEN IN WATER ENCLOSED BY ISOLATION CHAMBERS UNDER AEROBIC CONDITIONS DURING THE SECOND EXPERIMENT AT EPCOT POND

Isolation Chamber	TN Content in IC (mg N/m^2 sediment)						
	6/6/83	6/9/83	6/13/83	6/17/83	6/22/83	6/27/83	6/30/83
1	1186	1310	1315	1558	2245	2530	1359
2	4354	3770	2002	1697	1346	1154	840
3	2420	2012	1952	2099	1533	1527	1548

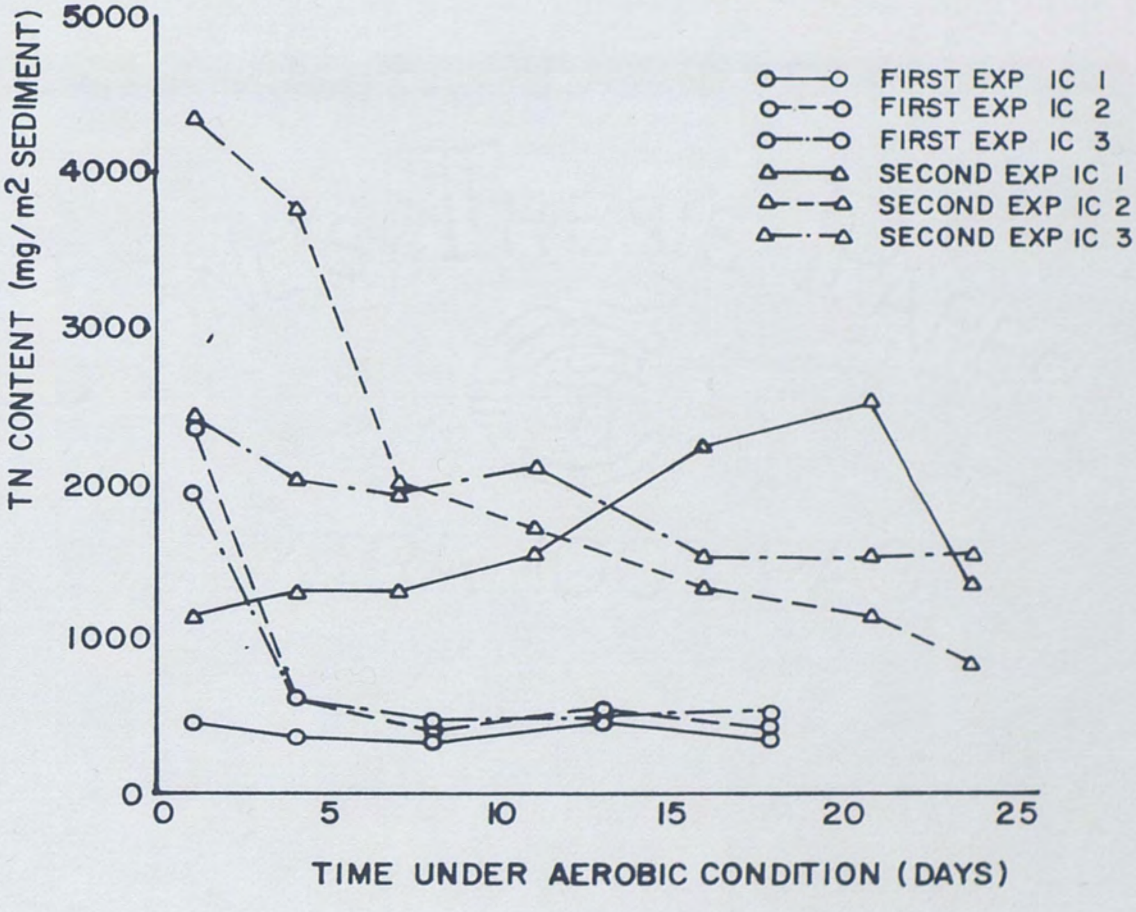

Fig. 44. Changes in TN content over time under aerobic conditions.

where:

N = TN content at time "t" (mg/m^2 sediment)

N_o = initial TN content (mg/m^2 sediment)

t = time under aerobic conditions (days)

k_n = TN removal rate constant ($days^{-1}$)

The equations for approximating the decline of TN content and their corresponding correlations to the plots shown in Figure 46 are summarized in Table 32. It can be seen that the TN removal in IC #2 and #3 can be accurately predicted by the first order reactions since the correlations are very good. The change of TN in IC #1, however, is not approximated as accurately by the first order reactions, as evidenced by poorer correlations. A comparison of the TN removal rate constants, k_n, reveals that the constants calculated in IC #2 during the first and second experiments, -0.076 $days^{-1}$ and -0.065 $days^{-1}$, respectively, are much greater than the removal constants in IC #1, -0.09 day^{-1} and 0.02 day^{-1}. The only difference between the experimental conditions was the increased concentrations of nitrogen in IC #2 due to the addition of ammonia at the commencement of each experiment; therefore, the rate of TN removal is a function of the TN concentration in the surrounding water. Similarly, a comparison of removal rate constants found in IC #2 and #3 can explain the role of sediments in the removal of TN from the overlying water column under aerobic conditions. The

TABLE 32

SUMMARY OF THE FIRST ORDER RELATIONSHIPS BETWEEN TN CONTENT AND TIME UNDER AEROBIC CONDITIONS IN THE ISOLATION CHAMBERS DURING EACH EXPERIMENT

Experiment	IC #	Independent Variable	Dependent Variable	Relationship	n	r
First	1	TN	Time	$TN = 435\ e^{-0.009t}$	5	0.48
	2	TN	Time	$TN = 1318\ e^{-0.079t}$	5	0.74
	3	TN	Time	$TN = 1167\ e^{-0.062t}$	5	0.69
Second	1	TN	Time	$TN = 1223\ e^{+0.02\ t}$	7	0.62
	2	TN	Time	$TN = 4221\ e^{-0.065t}$	7	0.97
	3	TN	Time	$TN = 2311\ e^{-0.019t}$	7	0.90

removal rate constants, 0.079 day^{-1} and 0.065 day^{-1} obtained in IC #2, correspond to a half life of 9 and 11 days, respectively, while the removal rate constants determined in IC #3, 0.062 day^{-1} and 0.019 day^{-1}, correspond to a half life of 11 and 36 days, respectively. In each experiment, the removal rate constant in IC #2 was higher than the constant in IC #3. This difference may be quantified by a comparison of the average removal rates of TN found in each chamber, as seen in Table 33. These values were obtained by determining the total amount of TN (mg/m^2 sediment) removed during the period of aerobic conditions for each experiment, and then dividing by the total number of days where aerobic conditions to determine the mg of TN removed per m^2 of sediment per day. The removal rate of TN during the first and second experiment were then averaged to obtain the average rate of TN removal. From the average rate of TN removal values, the calculated and removal in IC #2 (98.1 mg N/m^2-day) minus the TN removal in IC #3 (40.0 mg N/m^2-day) is equal to 58 mg N/m^2-day and is representative of the capacity of the sediments at Epcot pond to remove TN from the overlying water column under aerobic conditions after addition of a shock load. The processes involved in this removal are not fully understood, but probably both immediate adsorption to the sediments and subsequent nitrification in the oxidized top layer is involved. Figure 45 shows a comparison of NH_4-N and NO_3-N content in IC #1 during the first and second experiment. As can be seen in Figure 47, NH_4-N appears to be converted to NO_3-N on day

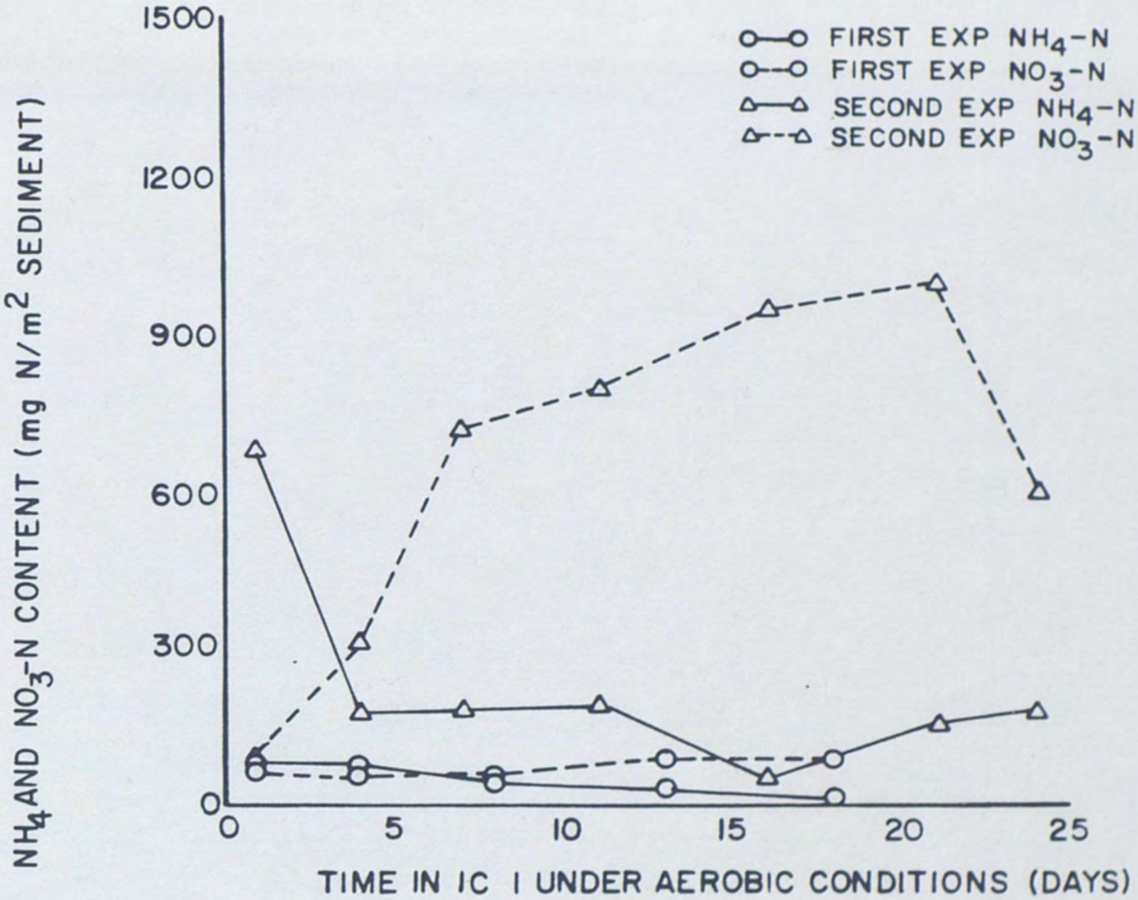

Fig. 45. Changes in the content of ammonia nitrogen and nitrate nitrogen under aerobic conditions during the first and second experiments.

TABLE 33

THE AVERAGE RATE OF TN REMOVAL UNDER AEROBIC CONDITIONS IN EACH ISOLATION CHAMBER DURING THE TWO EPCOT POND EXPERIMENTS

Isolation Chamber Number	Rate of TN Removal (mg N/m^2 sediment-day)		
	First Experiment	Second Experiment	Average
1	3.6	31.4*	13.9*
2	57.3	138.9	98.1
3	44.7	35.3	40.0

* Showed an increase in nitrogen with time.

8 and day 4 under aerobic conditions during the first and second experiment, respectively. The rate of NH_4-N decline in IC #1 was calculated using a first order reaction ($N_t = N_o e^{-K_N t}$) identical to the form used to calculate the TN decline discussed previously. The calculated rates of NH_4-N decline in IC #1 during the first and second experiments were 6.1 mg N/m^2-day and 8.7 mg N/m^2-day. Conversely, the rate of increase of NO_3-N in IC #1 during the first and second experiments were 1.6 mg N/m^2-day and 39.3 mg N/m^2-day. Thus, it appears that a relationship does exist between the decline of NH_4-N and the increase of NO_3-N under aerobic conditions.

If the TN content is divided into inorganic and organic fractions, further analysis of the removal rate results can be performed. From the results shown in Tables 34 and 35, it can be seen that in IC #1, the organic N (ON) and inorganic N (IN) content remained fairly constant under aerobic conditions during the first and second experiment. IC #1 contained pond water with existing nitrogen concentrations at the time of the experiment. However, when additional ammonia nitrogen was added in IC #2, both the IN and ON content declined under aerobic conditions during the first and second experiments. The IN content declined at a much greater rate than the ON content during each experiment. In IC #3, the ON content remained constant, while the IN content declined significantly. A comparison between the removal rates of IN and ON content in IC #2 and IC #3 provides an estimate of the role of sediments in the removal of each nitrogen species. By calculating the difference between the removal rates of IN and ON (mg N/m^2-day) in IC #2 and #3, the capacity for removal by the sediments can be determined. The removal rates were determined using first order reactions similar to the ones used to approximate TN removal rates. The relationships between IN and ON content and time determined in each isolation chamber are presented in Table 36.

Data analyzed in Table 36 showed relatively good correlations between the inorganic species of nitrogen removed with time during aerobic conditions. The correlation coefficients were 0.9 and 0.82 for IC #2 and IC #3 during the first experiment and 0.97 and

TABLE 34

ESTIMATES OF INORGANIC N (IN) AND ORGANIC N (ON) CONTENT IN WATER ENCLOSED BY ISOLATION CHAMBERS UNDER AEROBIC CONDITIONS DURING THE FIRST EXPERIMENT AT EPCOT POND

IC	Content in IC (mg N/m^2 sediment)									
	4/1/83		4/4/83		4/8/83		4/13/83		4/18/83	
	IN	ON	IN	ON	IN	ON	IN	ON	IN	ON
1	139	326	120	272	107	257	118	335	89	264
2	1920	423	325	284	159	257	134	380	70	330
3	1669	307	293	318	202	265	96	406	137	362

TABLE 35

ESTIMATES OF INORGANIC N (IN) AND ORGANIC N (ON) CONTENT IN WATER ENCLOSED BY ISOLATION CHAMBERS UNDER AEROBIC CONDITIONS DURING THE SECOND EXPERIMENT AT EPCOT POND

IC	Content in IC (mg N/m^2 sediment)													
	6/6/83		6/9/83		6/13/83		6/17/83		6/22/83		6/27/83		6/30/83	
	IN	ON	IN	ON	IN	ON	IN	ON	IN	ON	IN	ON	IN	ON
1	787	398	566	744	914	400	1001	557	871	1216	1156	1374	779	580
2	2434	1921	2444	1326	1337	664	1195	503	831	515	617	537	275	565
3	1937	483	1290	722	1380	561	1465	635	991	542	1036	491	1076	472

TABLE 36

SUMMARY OF FIRST ORDER RELATIONSHIPS BETWEEN ON AND IN CONTENT AND TIME UNDER AEROBIC CONDITIONS IN THE ISOLATION CHAMBERS DURING EACH EXPERIMENT

Experiment	IC #	Independent Variable	Dependent Variable	Relationship	n	r
First	1	IN	Time	$IN = 136e^{-0.021t}$	5	0.86
		ON	Time	$ON = 300e^{-0.004t}$	5	0.23
	2	IN	Time	$IN = 1073e^{-0.167t}$	5	0.90
		ON	Time	$ON = 336e^{-0.002t}$	5	0.10
	3	IN	Time	$ON = 858e^{-0.134t}$	5	0.82
		ON	Time	$IN = 290e^{+0.014t}$	5	0.59
Second	1	IN	Time	$IN = 736e^{+0.012t}$	7	0.46
		ON	Time	$ON = 456e^{+0.032t}$	7	0.57
	2	IN	Time	$IN = 2948e^{-0.087t}$	7	0.97
		ON	Time	$ON = 1337e^{-0.048t}$	7	0.78
	3	IN	Time	$IN = 1652e^{-0.021t}$	7	0.75
		ON	Time	$ON = 612e^{-0.009t}$	7	0.48

0.75 for IC #2 and IC #3 during the second experiment. IC #2 which has a direct contact with the bottom sediments in the pond show the highest correlation coefficients in all the experiments conducted. Also, the four inorganic removal rate coefficients were the highest in IC #2 in all experiments, varying between 0.167 day^{-1} during the first experiment and 0.087 day^{-1} during the second experiment. On the contrary, the organic fraction was removed at a much slower rate than the inorganic fraction or increased slightly during all experiments conducted.

The average removal rate (mg N/m^2-day) was then determined by dividing the total IN or ON removal by the number of days under aerobic conditions. The calculated average inorganic and organic forms of nitrogen were 18.7 mg N/m^2-day and 5.2 mg N/m^2-day during the first experiment and 80.4 mg N/m^2-day and 33.1 mg N/m^2-day during the second experiment, respectively. Thus, it appears that during each experiment, the sediments removed inorganic N at a much greater rate than organic N.

Similarly, the removal rates of IN and ON in IC #1 during each experiment under aerobic conditions were calculated to be: 2.40 mg N/m^2-day and 1.2 N/m^2-day during the first experiment and -10.2 mg N/m^2-day and -21.9 mg N/m^2-day during the second experiment, respectively. Therefore, there appeared to be little, if any, removal of IN and ON in IC #1 under aerobic conditions and with lower nitrogen concentrations in the enclosed water column.

The removal rates calculated for inorganic N in IC #2 during the first and second experiments, -0.167 day^{-1} and -0.087 day^{-1}, respectively, correspond to half lives of 4 and 8 days. Thus, it can be concluded that the removal of nitrogen to the sediments is efficient for soluble inorganic nitrogen if the nitrogen concentration in the overlying water column is sufficiently high.

Under anaerobic conditions, release of ammonia from the sediment took place during the two experiments. Similar observations are well known from lake systems due to biochemical and physiochemical activity (Austin and Lee 1973). In addition, the release of ammonia in a detention/retention pond under anaerobic conditions has been observed (Jacobsen and Yousef 1983).

The release of ammonia observed in the isolation chambers under anaerobic conditions can be quantified by a semi-logarithmic plot of NH_4-N content versus time under anaerobic conditions. The release of NH_4-N over time under anaerobic conditions was found to be most closely approximated by first order equations. As can be seen in Table 47, the first order equations have reasonable correlation to the data as shown by the correlation coefficients (r). As can be seen, during each experiment, the release rate of NH_4-N in IC #2 was significantly greater than the release rate of NH_4-N in IC #1. In addition, there was apparently little or no release rate of NH_4-N in IC #3 under anaerobic conditions as would be expected since there is no sediment contact within the chamber.

TABLE 37

THE EQUATIONS OF AMMONIA RELEASE RATES AND THE CORRESPONDING CALCULATED RELEASE RATES FOR THE ISOLATION CHAMBER EXPERIMENTS AT EPCOT POND

Isolation Chamber	Anaerobic Period (days)		Equation for Release Rate of NH_4				Release Rate (mg NH_4/m^2/day)		Average Release Rate (mg NH_4/m^2/day)
	Exp. 1	Exp. 2	Exp. 1	r	Exp. 2	r	1st Exp.	2nd Exp.	
1	37	130	$NH_4=11e^{0.049t}$	0.75	$NH_4=475e^{0.011t}$	0.69	1.8	15.3	8.6
2	37	130	$NH_4=40e^{0.063t}$	0.70	$NH_4=295e^{0.017t}$	0.67	11.1	20.7	15.9
3	29	130					0.0	0.0	0.0

The average release rates represent the release rates averaged over the first and second experiments. The average release rates for IC #1 and #2 were determined to be 4.5 and 7.2 mg NH_4/m^2-day, respectively. Therefore, it can be shown that the sediments release ammonia at rates near 8-9 mg/m^2-day under anaerobic conditions at Epcot pond with lower nitrogen concentrations in the water phase, and near 15-16 mg/m^2-day when nitrogen concentrations are increased in the water phase. No release of NH_4-N was found in IC #3 (without sediment contact) during the first or second experiment. Therefore, sediments may be the proper media for nitrogen transformation where effective nitrification/denitrification process can take place.

Sediment Oxygen Uptake

As discussed previously, the existence of an oxidized layer at the sediment surface is a fundamental factor for removal of nitrogen, phosphorus and metals from the overlying water phase. The sediment oxygen demand (SOD) is an important parameter when determining the conditions at the sediment-water interface. SOD was calculated based on measurement of DO in the isolation chambers using the conversion from aerobic to anaerobic conditions (see Table 38 for values

TABLE 38

OXYGEN CONTENT IN THE THREE ISOLATION CHAMBERS LOCATED IN THE EPCOT POND DURING EXPERIMENTS 1 AND 2

Exp.	Isolation Chamber	O_2 Content in Chamber (g O_2/m^2-sediment)				
		4/13/83	4/18/83	4/21/83	4/25/83	4/28/83
1st	1	5.1	3.1	0.7	0.2	0.1
	2	5.2	1.3	0.3	0.1	0.0
	3	5.4	3.2	2.6	1.5	0.7
	Isolation Chamber	6/30/83	7/05/83	7/08/83	7/14/83	7/20/83
2nd	1	3.4	0	0	0	0
	2	3.9	0	0	0	0
	3	4.2	3.2	3.0	2.2	1.1

of O_2 content in the first and second experiments). O_2 depletion was best approximated by first order reactions, as shown in Table 39. It is evident that decline of O_2 was first order in all the chambers except in IC #1 and #2 during the second experiment (where the O_2 decline was so rapid that only two samples could be obtained and, thus, a relationship cannot be determined). The rate of O_2 removal is a function of existing concentrations; however, an average overall removal rate can be calculated arithmetically by dividing the total removal by the period of incubation, as shown in Table 40. The average rate of O_2 depletion in IC #1, 0.51

TABLE 39

RELATIONSHIPS BETWEEN O_2 CONTENT (mg/m^2 sediment) WITH TIME DURING TRANSITION FROM AEROBIC TO ANAEROBIC CONDITIONS IN THE ISOLATION CHAMBERS

Experiment	IC #	Independent Variable	Dependent Variable	Relationship	n	r
First	1	O_2	Time	$O_2 = 9.40e^{-0.28t}$	5	0.98
	2	O_2	Time	$O_2 = 7.81e^{-0.34t}$	5	0.995
	3	O_2	Time	$O_2 = 6.93e^{-0.13t}$	5	0.97
Second	1	O_2	Time	*	2	--
	2	O_2	Time	*	2	--
	3	O_2	Time	$O_2 = 10.10e^{-0.11t}$	5	0.92

*Insufficient data to determine a valid relationship.

TABLE 40

THE OVERALL AVERAGE RATE OF OXYGEN REMOVAL IN EACH ISOLATION CHAMBER DURING THE FIRST AND SECOND EXPERIMENTS

Isolation Chamber	Avg. Rate of O_2 Removal ($g\ O_2/m^2$-day)		Average ($g\ O_2/m^2$-day)	Remarks
	First Exp.	Second Exp.		
1	0.33	0.68	0.51	Sediment Contact
2	0.43	0.78	0.61	Sediment Contact
3	0.31	0.15	0.23	No Sediment Contact

g O_2/m^2-day, is slightly lower than the value determined in IC #2, 0.61 g O_2/m^2-day. The difference in O_2 depletion rates in IC #2 and #3 is 0.38 g O_2/m^2-day and is representative of the sediment oxygen uptake at an average temperature of 25.8°C.

The value of the s-diment oxygen demand obtained in the Epcot pond is a relatively low value as compared with measurements from lakes. Yousef, et al (1980) have measured a DO uptake rate of 0.9 g/m^2-day in Lake Eola; this value, however, includes processes in the water phase. Edeberg and Hofsten (1973) report values from 1.8-2.6 g O_2/m^2-day in eutrophic lakes measured between 5 and 18°C. In addition, Jacobsen and Yousef (1983) report a DO uptake rate of 0.9 g/m^2-day in a detention/retention pond receiving highway runoff at Maitland Interchange site.

The transport of oxygen to the bottom sediments is a fundamental requirement to insure the high removal efficiency from the water phase to the sediment (Jacobsen and Yousef 1983). Thus, it appears that a shallow pond and a sediment with a large percentage of inorganic matter are important design and quality parameters to maximize the effect of oxygen transport. Consequently, Epcot pond, which is a newly constructed pond with a relatively low organic content in the bottom sediment and, in addition, has a sandy, silty sediment which allows for oxygen diffusion, can be expected to perform more efficiently than a pond with a highly organic bottom and fine decayed debris.

Reduction of Nitrogen with Depth in the Sediments

As shown in Figure 34, the concentration of nitrogen measured as mg/g dry weight decreases with sediment depth and remains fairly constant below about 4 cm. This reduction follows an exponential equation:

$$TN = 0.61 \times e^{-.13z} \qquad r = -0.94$$

where:

TN = mg N/g dry wt accumulated in the sediments

z = sediment depth (cm)

The nitrogen reduction constant, $k = 0.13\ cm^{-1}$, is much less than a similar calculated rate for nitrogen, $0.91\ cm^{-1}$, found in a 7 year old detention/retention pond receiving highway runoff

(Jacobsen and Yousef 1983). In addition, the N content in the surficial sediments and the subsequent sediment depths at Epcot pond (0.61 mg/g dry wt) is much less than the N content of 2.4 mg N/g dry wt found in the sediments of the older pond. This reduction is the result of an external force, not a characteristic parameter for the matter itself (Jacobsen and Yousef 1983). Probably benthic organisms found in the upper part of the sediment are mixing the sediment to some extent (Christopher 1980), thus affecting the exchange of matter across the water-sediment interface and diffusion through the sediments. This mixing, however, takes place at a fairly low level as shown by the following simple steady-state diffusion model for the sediments:

$$D \frac{d^2c}{dz^2} + W \frac{dc}{dz} = 0$$

where:

D = diffusion coefficients (cm^2/year)

c = concentration at given sediment depth, TN (mg/g)

z = sediment depth (cm)

W = accumulation rate for the sediment (cm/year)

From visual observations of sediment cores, the value for W can be estimated by 0.10 cm/yr. With the value for the nitrogen reduction constant, k = 0.013, the diffusion coefficient of nitrogen through the sediments, D, is about 0.06 cm^2/yr. This value is less than a similar value determined for the sediments of the 7 year old detention/retention pond of 0.10 cm^2/yr and less than 0.2 cm^2/yr, indicating a portable limit between low and moderate mixing of the surface sediments. This result shows that no intensive sediment-water exchange, i.e., release of matter from the sediments due to activity of bottom organisms can be expected. The low calculated value of D shows slow diffusion through the sediments.

CHAPTER V

SUMMARY, CONCLUSIONS AND RECOMMENDATIONS

Roadside swales are commonly used along highways to infiltrate runoff water from intermittent storm events, and convey rainfall excess to selected locations for retention, detention, storage or discharge into adjacent water bodies. Their effectiveness for pollution control needs to be investigated since information on the fate of highway runoff pollutants flowing over swales is scarce or non-existent. Therefore, the Florida Department of Transportation (FDOT) and the Federal Highway Administration (FHWA) sponsored a research project to study "Best Management Practices for Highway Runoff" by researchers at the Department of Civil Engineering and Environmental Sciences, University of Central Florida. Two sites were carefully instrumented for the study at the Maitland Interchange/I-4 and Epcot Interchange/I-4, near Orlando, Florida. One phase of the study was to evalute the effectiveness of roadside swales in the removal of highway pollutants.

Effectiveness of Swales

Runoff samples were collected from highways and adjacent roadside swales at Maitland Interchange for analysis of nutrients (N and P) content in order to investigate the removal efficiencies of pollutants by swales. Also, simulated highway runoff was allowed to flow over designated areas of swales located close to the Maitland and Epcot Interchanges at a constant rate of flow. Water samples from inlet, outlet and intermediate stations were collected for analysis to establish quantitative and qualitative data on the effectiveness of swales for pollution control.

Swales are designed to transmit, treat and store stormwater runoff. Contaminants in runoff such as nutrients may be reduced and retained on the site. However, regeneration and relocation of loosely bound contaminants may occur at intermediate locations. It is reasonable to assume that particulate contaminants are filtered out by the grassy cover on swales and settle down to the bottom sediments. These contaminants are exposed to various dry and wet periods and slowly decay, release nutrients and metals to overlying water, relocate by erosion or permanently attach to soil. Dissolved contaminants will interact with surrounding water, soil and biota.

Nitrogen Removal by Roadside Swales

From the results obtained in swale experiments, it appears that ionic nitrogen species (NH_4^+, NO_2^-, NO_3^-) may be retained on the swale site by deposition, sorption and biological uptake processes. These processes can reduce the nutrient concentrations in highway runoff flowing over swales. Also, it appears that the removal of soluble inorganic nitrogen species occurs more rapidly in bare earth swales than in grass-lined channels. Additionally, thinner grass seems to be more efficient in decreasing nitrogen than a thicker grass cover. It is believed that thick grass cover may affect available sorption sites and increase organic debris (grass clippings, mower debris, litter). The organic debris is then subjected to decay processes and relocation. This was evident by the decline in the removal efficiency of soluble NO_3 and NH_4 forms of nitrogen and organic nitrogen in thick grassy swales (Epcot study, 5/16/83 and Maitland study, 5/31/83). Also, the decrease in the removal of organic N concentration may be attributed to an increase in organic deposition in the swale due to organic debris that exists during periods of rapid grass growth.

Occasional increases in nitrogen concentration were observed at intermediate stations during swale experiments, particularly close to inflow point. This appears possible due to theinitial flow effects on resuspension and resolubilization of loosely bound contaminants. The swale experiments showed better removal efficiencies at slow flow rates than high velocities. The removal of nitrogen in swales on a concentration basis (measured in this study as μg/liter), is inversely related to the velocity of the runoff through the swale, i.e., the residence time of the runoff in the swale. There seems to be very little removal of nitrogen concentrations when the excess runoff is above 3.00 inches per hour. Therefore, it is apparent that if swales are designed to produce low inflow rates and velocities, some nitrogen concentration removal could be expected, with the amount of removal being a function of site conditions such as swale cover and soil characteristics.

The removal of nitrogen species on a mass basis, is directly related to infiltration losses through swales. Therefore, retention of as much water as possible on the swale area will reduce the nitrogen loadings to adjacent receiving waters.

Effectiveness of Detention/Retention Systems

Water quality is a very important part of stormwater detention. Typical highway runoff pollutants, nitrogen and phosphorus, can cause significant biological growth and depress dissolved oxygen in receiving waters. Thus, the need exists for effective treatment to minimize water quality impact upon receiving water through the use of various treatment facilities: retention/detention, wetlands and marshlands. Currently, design of these systems is limited to hydraulic criteria and little is known about quality improvement and their removal efficiencies of nutrients.

The study conducted at Epcot retention pond investigated the fate and removal efficiency of nitrogen in retention/detention ponds receiving highway runoff. This pond is very new, about 1 year old. A similar study was carried out in a retention pond at Maitland Interchange, which is approximately 8 years old. The data gathered from each study was compared to evaluate relationships and/or differences in the treatment efficiency of these ponds.

Epcot investigations developed nutrient inputs to the pond, the water quality of the pond, and accumulation of nutrients in the bottom sediments. In situ isolation chamber experiments were also conducted to quantify the importance of nutrient exchange processes at the sediment-water interface under aerobic and anaerobic environments. The role of bottom sediments in maintaining nutrient balance within the water body was evaluated.

The water quality analysis at Epcot pond shows that the total nitrogen to total phosphorus ratio is 14.4; thus, phosphorus appears to be the limiting nutrient. The measured values of dissolved species of nitrogen and phosphorus are less than or equal to 0.1 mg/l. The average TP and TN concentrations in the Epcot pond water are 24 and 57 percent of the average concentration detected in the primary highway stormwater input, respectively. These data suggest that the pond is effective in removal of nutrients from highway stormwater runoff.

Two in situ experiments were carried out in isolation chambers which demonstrated that bottom sediments were important as a nutrient sink for nitrogen under aerobic conditions at the sediment-water interface which resulted in efficient removal of IN and TN. However, when anaerobic conditions existed in the overlying water, no removal of TN was observed and the NH_4^+-N was released. Therefore, it is desirable to maintain an aerobic environment in the pond water to sustain lower nitrogen levels and prevent release of nitrogen from the bottom sediments. Also, maintaining an aerobic environment at the water-sediment interface followed by an anaerobic environment in the deeper layers of the bottom sediment may enhance the processes of nitrification-denitrification resulting in a net loss of the nitrogen mass received by the detention/retention pond.

The nitrogen content in the bottom sediment cores collected from the Epcot pond declined with increasing sediment depth. Also, by comparison between the sediments from Epcot and Maitland ponds, it appears that the nitrogen content increases as the pond ages since its excavation. For example, the surficial nitrogen content was found to be 0.6 mg-N/g dry weight of sediments from Epcot pond and 2.4 mg-N/g dry weight of sediments from Maitland pond. Maitland pond is approximately 8 years old, while Epcot pond is about 1 year old. Epcot pond is also more acidic in nature (with an average pH of 6 , which is lower than the pH of Maitland pond). Neutral pH values or higher may impact the eventual biological growth, deposition, and decomposition in the bottom sediments. Similarly, the TN content was found to be 0.23 mg-N/g and 0.81 mg-N/g in sediments from Epcot and Maitland ponds, respectively, at an average depth of approximately 2.0 cm. It appears that the sediments in the older Maitland pond have accumulated nearly four times as much nitrogen than the sediments in the newly constructed Epcot pond.

Nitrogen Contribution from Drainage Basins

A five-month stormwater sampling program was conducted to determine nutrient inputs to Epcot pond, removal efficiencies of swale areas receiving highway runoff, and removal efficiencies of artificial and natural wetlands receiving highway runoff.

The results from the stormwater sampling program conducted at Epcot interchange revealed that (1) swale areas removed IN and TN in highway runoff by 55 and 27%, respectively, (2) the pine-dominated artificial wetland area is not currently removing contaminants from stormwater runoff, and, in fact, large amounts of N, P and heavy metals are released due to existing low pH values in runoff after flowing through the wetland due to decaying vegetative matter and humic substances.

Conclusions

Studies conducted at Maitland Interchange and Epcot Interchange sites on the effectiveness of swales and detention/retention systems for nitrogen removal resulted in the following conclusions:

1. Concentrations of nitrogen in highway runoff flowing over roadside swales may decline, increase, or remain constant depending on nitrogen species, flow characteristics, and swale environment. However, significant nitrogen mass removals can be achieved if infiltration losses are considered. If no water reaches the downstream discharge, then the removal efficiency is actually 100% when all nutrients are retained on the site.

2. Swales built on high ground with good drainage and high infiltration rates showed good mass removal efficiencies (57% for IN and 54% for TN) for nitrogen in highway runoff.

3. An inverse relationship exists between removal efficiency of total nitrogen concentration and the average velocity of runoff in swales, i.e., the slower the velocity of the runoff through a swale, the greater the removal of total nitrogen concentration.

4. Occasional increases in nitrogen concentration are observed at intermediate sampling stations along the swale during the experiments, particularly close to the inflow point, which may be attributed to resuspension of the loosely bound nitrogen forms.

5. From the results of the stormwater sampling programs conducted at Maitland Interchange site, it can be concluded that swale areas efficiently removed nitrogen from the runoff of storm events occurring in colder months but did not remove nitrogen from runoff of storm events occurring in warmer months. This difference in nitrogen removal can be attributed to increased organic nitrogen content in swales such as grass clippings from increased mowing of the swales. All sampling dates for Epcot study were taken during warm months of the year (July through October).

6. There appears to be very little removal of nitrogen concentrations when the inflow runoff is above 3.00 miles per hour, thus, if the rate is lowered by swale design, nitrogen concentration removal can be expected. The degree of removal is a function of site conditions such as swale cover and soil characteristics.

7. Retention/detention ponds are effective in the removal of nitrogen from highway runoff as evidenced by a decrease in the TN

concentration in the primary stormwater inputs to Epcot and Maitland ponds of 43 and 79%, respectively. In addition, the N removal rate constants determined in the isolation chamber experiments under aerobic conditions at Epcot and Maitland ponds were 0.071 day and 0.054, respectively. These removal constants correspond to half-lives of 10 and 13 days, respectively.

8. The removal rates of IN and TN were found to follow first order kinetics; therefore, the removal rate is a function of the initial N concentration in the water column. The removal rate constants calculated from the total nitrogen in the isolation chamber experiments are summarized in Table 41 below.

TABLE 41

FIRST ORDER REMOVAL RATE CONSTANTS DETERMINED AT EPCOT POND

Experiment	IC	Kn ($days^{-1}$)	1/2 Life (days)
1	1	0	-
	2	0.076	9
	3	0.062	11
2	1	0.020	35
	2	0.065	11
	3	0.017	36

9. Aerobic conditions must be maintained in retention ponds in order to facilitate the removal of IN and TN from the overlying water column to the bottom sediments. IN and TN were removed at

rates of 19 mg/m^2-day and 98 mg/m^2-day, respectively, at Epcot pond, during the first experiment, and 80 mg/m^2-day and 146 mg/m^2-day, respectively during the second experiment.

10. Anaerobic conditions in the water body are undesirable in retention ponds since nitrogen is released in the form of NH_4-N from the bottom sediments to the overlying water body. The release rates of NH_4-N at Epcot and Maitland ponds under anaerobic conditions were determined to be 16 mg/m^2-day and 30 mg/m^2-day, respectively.

11. The bottom sediments act as an important sink by accumulating nitrogen from the water column. The TN content in the surficial sediments in Epcot and Maitland ponds was calculated to be 2.5 mg-N/g sediment and 0.6 mg/g, respectively. Similarly, the TN content at an average depth in the sediments of 2.0 cm in Epcot and Maitland ponds was found to be 0.2 and 0.8 mg/g sediment, respectively. Therefore, the sediments in the 8 year old Maitland retention pond have accumulated approximately four times as much nitrogen as the newly constructed Epcot pond.

12. Water samples collected from artificial wetland at Epcot Interchange site contained higher concentrations than those detected in stormwater runoff entering the wetland probably due to lower pH values resulting from decay of vegetative and humic substances within the wetland.

Recommendations

The swale efficiency is increased by increasing contact time and infiltration rates, therefore, it is recommended to:

1. Reduce longitudinal slopes as much as possible.

2. Increase surface area by increasing wetted perimeter to cross-section area ratio using flat side slopes whenever possible.

3. Swales should not be built in areas where portions remain wet most of the time as a result of low ground.

4. Maximum on-site retention by storage of runoff water in swales built on upland may be achieved by earthen cross barriers (swale blocks) at selected length intervals along the swale.

5. Plant a thin, cover crop for erosion control and follow effective maintenance procedures, particularly for removal of grass clippings, loose debris and litter.

6. Consider slow-growing grass species with low maintenance requirements and avoid thick and rapidly growing grass wherever possible.

7. Maintenance of a permanent oxidized sediment-water interface is a big factor for nitrogen removal in detention/retention ponds. Therefore, sufficient oxygen concentrations and aerobic conditions at the water sediment interface can be achieved by designing shallow detention/retention ponds of similar depths to those studied. Maintenance procedures may be considered to remove excessive debris deposits.

REFERENCES

Anderson, D.E. "Evaluation of Swale Design." Thesis, University of Central Florida, Orlando, Florida, 1982.

Austin, E.R., and Lee, G.F. "Nitrogen Release from Lake Sediments." Journal Water Pollution Control Federation 45 (May 1973): 870-879.

Berthouex, P., and Rudd, D. Strategies of Pollution Control. New York: John Wiley and Sons, Inc., 1977.

Bondurant, J.A., and Chen, C. "Some Aspects of Sedimentation Pond Design." In Proceedings of the International Symposium on Urban Rainfall Runoff and Sediment Control, University of Kentucky, Lexington, Kentucky, 1975.

Burke, W. "Fertilizer and Other Chemical Losses in Drainage Water from a Blanket Bog." Irish Journal of Agricultural Resources 14 (May 1975): 163.

Chancellor, G.L. "Stormwater Management." Thesis, University of Central Florida, Orlando, Florida, 1975.

Chen, C. "Design of Sediment Retention Basins." In Proceedings of the International Symposium on Urban Hydrology and Sediment Control, University of Kentucky, Lexington, Kentucky, 1974.

Chen, C. "Evaluation and Control of Soil Erosion in Urbanizing Watershed." In Proceedings of the International Symposium on Urban Rainfall Runoff and Sediment Control, University of Kentucky, Lexington, Kentucky, 1974.

Chiaudari, G., and Vighu, G. "The N:P Ratio and Tests with Selenastrum to Predict Eutrophication in Lakes." Water Research 8 (July 1974): 89.

Crisp, D.T. "Input and Output of Minerals for an Area of Pennine Moorland: The Importance of Precipitation, Drainage, Peat Erosion, and Animals." Journal of Applied Ecology 3 (1966): 327.

Edeberg, N., and Hofsten, B.V. "Oxygen Uptake of Bottom Sediments Studies and in the Laboratory." Water Research 7 (1973): 1285-1294.

Federal Highway Administration. Constituents of Highway Runoff, Vol. 1: State-of-the-Art Report. FHWA/RD-81/042, Washington, D.C.: Government Printing Office, February 1981.

Ferrara, R.A., and Witowski, P. "Stormwater Quality Characteristics in Detention Basins." ASCE Journal of Environmental Division 14 (June 1981): 119-131.

Hakansson, L. "Lake Sediments in Aquatic Pollution Control Programs, Principles, Processes and Practical Examples." Swedish Environmental Protection Agency, #SNV PM 1398, Stockholm, 1981.

Hvitved-Jacobsen, T.; Yousef, Y.A.; Wanielista, M.P.; and Pearce, D.B. "Fate of Phosphorus and Nitrogen in Ponds Receiving Highway Runoff." In Proceedings of the International Symposium on Highway Pollution, London, United Kingdom, September 6-9, 1983.

Klopatek, J.M. "The Role of Emergent Macrophytes in Mineral Cycling in a Freshwater Marsh." In Mineral Cycling in Southeastern Ecosystems. ERDA Symposium Series (CONF-740513), 357 (1975).

Klopatek, J.M. "Nutrient Dynamics of Freshwater Riverine Marshes and the Role of Emergent Macrophytes." In Freshwater Wetlands, Econological Processes and Management Potential. New York: Academic Press, 1978.

Kluesener, J.W., and Lee, G.F. "Nutrient Loading from a Separate Storm Sewer in Madison, Wisconsin." Journal Water Pollution Control Federation 46 (May 1974): 932.

Lee, G.F., and Hasler, A.P. "Effects of Marshes on Water Quality." In Coupling of Land and Water Systems. New York: Springa-Verlag, 1975.

Malmquist, P.A. "Atmospheric Fallout and Street Cleaning - Effects on Urban Stormwater and Snow." Progress Water Technology 10 (May 1979): 495-505.

Malmquist, P.A. "Stormwater Pollution." Swedish Environmental Protection Agency, #SNV Rm 1109, Stockholm, 1981.

Mattraw, H.G., Jr., and Miller, R.A. "Stormwater Quality Processes for Three Land-Use Areas in Broward County, Florida." USGS Water Resources, # 81-23, Tallahassee, FL, 1981.

McPerson, B.V.; Waller, B.G.; and Mattraw, H.C. "Nitrogen and Phosphorus Uptake in the Everglades Conservation Areas, Florida, with a Special Reference to the Effects of Backpumping Runoff." USGS. Water Resources, #78-62, Tallahassee, FL, 1978.

Mortimer, C.H. "The Exchange of Dissolved Substances Between Mud and Water in Lakes." Journal Ecology 29 (Jan. 1941): 280-329.

Mortimer, C.H. "The Exchange of Dissolved Substances Between Mud and Water in Lakes." Journal Ecology 30 (Aug. 1942): 147-201.

Nichols, Dale S. "Capacity of Natural Wetlands to Remove Nutrients from Wastewater." Journal Water Pollution Control Federation 55 (May 1983): 495-505.

Patrick, W.H., Jr., and Reddy, K.R. Nitrate Removal from Water at the Water-Mud Interface in Wetlands. U.S. Environmental Protection Agency, Ecological Research Series, Report No. EPA-600/3-76/042, Office of Research and Development, Washington, D.C.: U.S. Government Printing Office, May 1976.

Prentki, R.T. "Nutrient Movements in Lakeshore Marshes." In Freshwater Wetlands, Ecological Processes and Management. New York: Academic Press, 1978.

Pitt, R.E., and Amy, G. Toxic Materials Analysis of Street Surface Contaminants. Environmental Protection Agency, Report No. 440/9-73, Office of Research and Development, Washington, D.C.: U.S. Government Printing Office, August 1973.

Poertner, H.G. Practices in Detention of Urban Stormwater Runoff. American Public Works Association, Special Report No. 43, Chicago, Illinois, APWA, 1974.

Randall, G.R. "Stormwater Detention Facilities." In Proceedings of the Conference on Stormwater Detention Facilities - Planning, Design, Operation and Maintenance, New York, American Society of Civil Engineers, 1982.

Sartor, J.D.; Boyd, G.B.; and Acardy, F.J. "Water Pollution Aspects of Street Surface Contaminants." Journal Water Pollution Control Federation 46 (March 1974): 458-467.

Shaheen, Donald G. Contributions of Urban Roadway Usage to Water Pollution. U.S. Environmental Protection Agency, Report No. 600/2-75-004, Washington, D.C.: U.S. Government Printing Office, April 1975.

Snyder, B. "Biological Nitrogen Fixation and Denitrification in Lake Okeechobee, Florida." Thesis, University of Central Florida, Orlando, Florida, 1981.

Standard Methods for the Examination of Water and Wastewater, 14th Edition. Water Pollution Control Federation, American Water Works Association, 1976, pp. 402-438.

Terry, R.E., and Nelson, D.W. "Factors Influencing Nitrate Transformations in Sediments." Journal of Environmental Quality 4 (March 1974): 549.

Tilton, D.L., and Kallec, R.H. "The Utilization of a Freshwater Wetland for Nutrient Removal from Secondarily Treated Wastewater Effluent." Journal of Environmental Quality 8 (Sept. 1979): 328.

Wanielista, M.P.; Yousef, Y.A.; Golding, B.D.; and Cassagnol, C.L. Stormwater Management Manual. University of Central Florida, Orlando, FL. Florida Department of Environmental Regulation, Tallahassee, FL, 1981.

Ward, A.J.; Haan, C.T.; and Barfield, B.J. "Simulation of the Sedimentology of Sediment Detention Basin." In Proceedings of the International Symposium on Urban Hydrology and Sediment Control, University of Kentucky, Lexington, Kentucky, 1977.

Whipple, W., and Hunter, J.V. "Settleability of Urban Runoff Pollution." Journal Water Pollution Control Federation 53 (December 1981): 1726-1731.

Winkelman, D.A. "Microbial Biochemical Potential of Two Detention/Retention Marshes in the Kissimmee River Valley Watershed." Thesis, University of Central Florida, Orlando, Florida, 1981.

Yousef, Y.A.; Harper, H.H.; and Jellerson, D. "Inactivation of Lake Sediment Release of Phosphorus." Engineering and Industrial Experiment Station, University of Central Florida, Orlando, Florida, 1980.

Yousef, Y.A.; Harper, H.H.; Pearce, D.B.; and Tolbert, R.D. "Fate of Stormwater Pollutants in Roadside Swales." Florida Department of Transportation/University of Central Florida, 1984.

Yousef, Y.A.; Harper, H.H.; and Wanielista, M.P. "Impact of Stormwater Runoff on Lake Eola Water Quality." In Proceedings of the Second International Conference on Urban Drainage, University of Illinois, Urbana, Illinois, 1981.