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16. Abstract

A research project to investigate the best management practices for highway runoff is being conducted. One phase of the project is to study the removal of highway contaminants by roadside swales. Their hydraulic efficiencies are based on their ability to infiltrate and percolate stormwater. Pollutant removal efficiencies, based on quality considerations, have not been determined. Therefore, field experiments were designed to study pollutant concentrations and mass balances during flow through swales under controlled environments.

From the results obtained, it appears that ionic species of metals, nitrogen and phosphorus species may be retained on the swale site by sorption, precipitation, co-precipitation and biological uptake processes. These processes can reduce pollutant concentrations in highway runoff flowing over swales.

Occasional increases in highway contaminants were observed at intermediate stations during swale experiments particularly close to the inflow point. This may result from the initial flow resuspension and resolubilization of loosely bound contaminants. The removal of heavy metals, nitrogen and phosphorus species on a mass basis was directly related to infiltration losses through swales. Therefore, retention of as much water as possible on the swale area will reduce the highway contaminant loadings to adjacent receiving waters. Recommendations for the construction of roadside swales are presented.

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FINAL REPORT

on

BEST MANAGEMENT PRACTICES - REMOVAL OF HIGHWAY CONTAMINANTS BY ROADSIDE SWALES

by

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ABBREVIATIONS AND CONVERSION FACTORS

[Factors for converting inch-pound units to International System of units (S1) and abbreviation of units]

Multiply	By	To Obtain
inch (in)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
acre	0.4047	hectare (ha)
cubic foot per second (ft^3/s)	0.02832	cubic meter per second(m^3/s)
square foot (ft^2)	0.09290	square meter (m^2)
pound (lb)	0.4536	kilogram (kg)
cubic foot (ft^3)	0.02832	cubic meter (m^3)

NOTICE

The contents of this report reflect the views of the authors who are responsible for the facts and the accuracy of the data presented herein. The contents do not necessarily reflect the official views of policies of the Federal Highway Administration. This report does not constitute a standard, specification, or regulation.

Neither the State of Florida nor the United States Government endorse products or manufacturers. Trade or manufacturers names appears herein only because they are considered essential to the object of this report.

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CHAPTER I

INTRODUCTION AND OBJECTIVES

HIGHWAY CONTAMINANTS

Studies conducted by the Federal Highway Administration, the U.S. Environmental Protection Agency, the U.S. Geological Survey and various State Departments of Transportation have shown that roadways, including bridges, may act as depositories for significant quantities of contaminants (Sartor et al., 1972; Shaheen, 1975; Hardee et al., 1978; Irwin et al., 1979; and Gupta, 1978) which may be washed off and transported to adjacent receiving waters during storm events. Studies as early as 1957 documented elevated levels of lead in soils near highways which were felt to be potentially toxic to agricultural crops in the vicinity (Prince, 1957). Since that time, there has been a proliferation of studies showing evidence of high concentrations of heavy metals not only in soils, but in the atmosphere and in receiving bodies of water near highways.

Highway contaminants include deposited heavy metals such as lead, zinc, copper and iron. Motor vehicles also deposit quantities of grease, oil and other petroleum products. In addition to metals and petroleum products, quantities of sand, dirt and dust are transported from place to place while the surface is itself abraded. Asphalt surfaces deteriorate much more rapidly than concrete, yielding oils and inert sand (Gupta et al., 1981). A rougher surface is capable of collecting high contaminant loading which is then available to contribute to stormwater runoff. Billions of gallons of stormwater runoff are collected each year within the highway rights-of-way from more than 1.5 million land-miles of existing roadways across our nation.

Stormwater runoff is eventually discharged to a multiplicity of receiving waters; varying greatly in type, size, quantity, characteristics and usage. The effects of highway pollutants on receiving waters have never been clearly defined and efforts to assess their impacts have been scarce and limited in both detail and depth.

Gupta, 1978, summarized research studies outlined by the Federal Highway Administration (FHWA) to identify and quantify the effects of runoff and to develop measures for protection of the environment from any adverse effects. The objectives of the program were as follows:

1. Identify and quantify the constituents of highway runoff.
2. Identify the sources of these pollutants and migration paths from the highway to receiving waters.
3. Analyze the effects of these pollutants on receiving waters.
4. Develop the necessary abatement/treatment methodology for objectionable constituents.

The first phase of the program resulted in the development of simplified predictive models to calculate the quantity and quality of rainfall-generated highway runoff. Models for the rainfall-runoff relationships, pollutant build-up, pollutant wash-off and constituent loadings were developed. These models require refinement through continued research. Also, pollutant loadings in lbs/ac/in of runoff for various highway runoff constituents were developed. These constituents included total solids (TS), total dissolved solids (TDS), suspended solids (SS), volatile suspended solids (VSS), metals such as lead (Pb), zinc (Zn), iron (Fe), copper (Cu), chromium (Cr), and nickel (Ni), and other constituents such as five-day biochemical oxygen demand

(BOD_5) and chemical oxygen demand (COD), nutrients, bacteriological analysis and pesticides (Gupta et al., 1981).

Pollutants emitted by automobiles are deposited on the highway system or transported by advective and diffusive airborne mechanisms. Other transport mechanisms include stormwater washoff and/or splashing of contaminated stormwater from roadway surface by vehicles. Once washed off the highway surface, contaminants may be carried with the runoff water to receiving water bodies or they may infiltrate into the soil. The extent of infiltration will depend on existing soil moisture conditions, intensity and duration of rainfall, soil type, vegetation cover, and the topography of adjacent lands. Pollutants associated with particulate matter will probably be attenuated by the surface soil layers, while dissolved forms may move downward into the soil.

MANAGEMENT OF HIGHWAY RUNOFF

In recognition of the potentially toxic and degrading properties of highway runoff waters, a number of management practices have been proposed to either prevent the discharge of these waters into receiving bodies or to treat them prior to release. Some of the management practices in current use are detention basins, retention basins, and swales (percolation areas). Little is known about the degree of nutrient and heavy metal removal achieved by these practices or about the processes which cause this removal. Information in these areas would likely allow the development of alternative management practices or modifications of existing practices to make them more cost-effective and efficient in terms of pollutant removal.

Previous studies have reported elevated levels of lead in soils indicating that the soil is an important "sink" for this element (Yousef et al., 1983). Other research efforts examining the inputs and outputs of lead in highway drainage basins have also speculated that the soil is important for removal of lead from runoff waters (Sylvester and DeWalle, 1972; and Rolfe and Jennett, 1973). It is likely that other heavy metals are similarly removed from runoff water by the soil. Wanielista and Gennaro (1977) determined relatively higher concentrations of lead, chromium and zinc in the soils adjacent to highways than the background concentrations of these metals. In light of this ability of soils to retain or immobilize heavy metals such as lead, zinc and chromium, it is likely that management practices which promote maximum exposure of highway runoff to soils will result in reduced loadings of heavy metals to receiving waters. However, more data is needed to determine the extent of the soil's capacity to retain heavy metals and to identify the properties of the soil which are important to this capacity.

Roadside swales generally consist of a network of channels designed to infiltrate runoff water from intermittent storm events or convey rainfall excess to desired locations for retention, detention, storage, or discharge into receiving streams. They are commonly used along highways and their effectiveness for pollution control is not fully understood.

ROADSIDE SWALES

Florida Statutes (1983) define swales as manmade trenches. Swale systems utilized to control stormwater quantity may be effective for pollution control if proper considerations of volume, routing, infiltration, and pollutant removal efficiencies are adequately assessed in the design phase. It must be realized that soil conditions such as antecedent moisture content, infiltration rate, existing foliage, soil characteristics, types of undesirable debris, swale maintenance procedures, land slope, swale geometry and hydrologic characteristics of the site are important parameters which affect the efficiency of swales. In a watershed with high moisture content for its soil, limited percolation capacity, and an excess of decaying vegetation clogging soil pores, the effectiveness of swales for infiltration of runoff and pollutant mass removal should be expected to be reduced when compared with swales placed in areas with more optimal conditions.

Existing swale design methodologies are generally based on hydrologic principles, since qualitative data on fate of runoff pollutants through swales are scarce or nonexistent. Infiltration and soil storage appear to be the most important parameters currently used in evaluating the swale efficiency. It has been reported that a well maintained grassy swale percolates at higher rates than areas which are unkept, (Anderson, 1982). He also reported that grassy cover generally enhances infiltration through swales and he summarized available information and design methodologies for swales.

OBJECTIVES

Swales are designed to be used for management of highway runoff. Their hydraulic efficiencies are based on their ability to convey, infiltrate and percolate stormwater. However, the water quality and fate of pollutants transported through the swales are not known. Pollutant removal efficiencies, based on quality considerations, have not been determined. Therefore a series of experiments was designed to study pollutant concentrations and mass balances during flow through swales under a controlled environment. A continuous flow of simulated highway runoff was allowed to flow over the experimental area of a swale at the Maitland Interchange and I-4 site and at the EPCOT Interchange site. These experiments were designed to assist in answering some of the following questions:

1. Are swales efficient in phosphorus and nitrogen removal from stormwater runoff? If they are, to what extent and for how long do they retain those nutrients?
2. Are swales efficient in removal of heavy metals from highway runoff? If they are, to what extent and for how long do they retain those metals? Also, are there different affinities for different metals and under what conditions are these metals released?
3. Is it possible to develop design considerations based on pollutant removal efficiencies?

CHAPTER II

EXPERIMENTAL PROCEDURES

A survey of various grassy swales in the Central Florida area showed that they are built with longitudinal slopes between 0.06 percent and 3.14 percent and lateral slopes between 8 percent and 29 percent (Anderson, 1982). Very few 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.

EXPERIMENTAL SITES

In order to investigate the efficiency of roadside swales for removal of pollutants in highway runoff such as nutrients (N&P) and heavy metals, a series of experiments was conducted at two locations in the Central Florida area. The study areas include: (a) an older established swale near the Maitland/I-4 interchange; and (b) a newly constructed swale which receives stormwater runoff from the main exit road to EPCOT center. The location of the two sites selected for this study in Orange County, Florida, is shown in Figure 2-1.

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 m grassy median, as they approach the interchange, which widens to 13.5 m

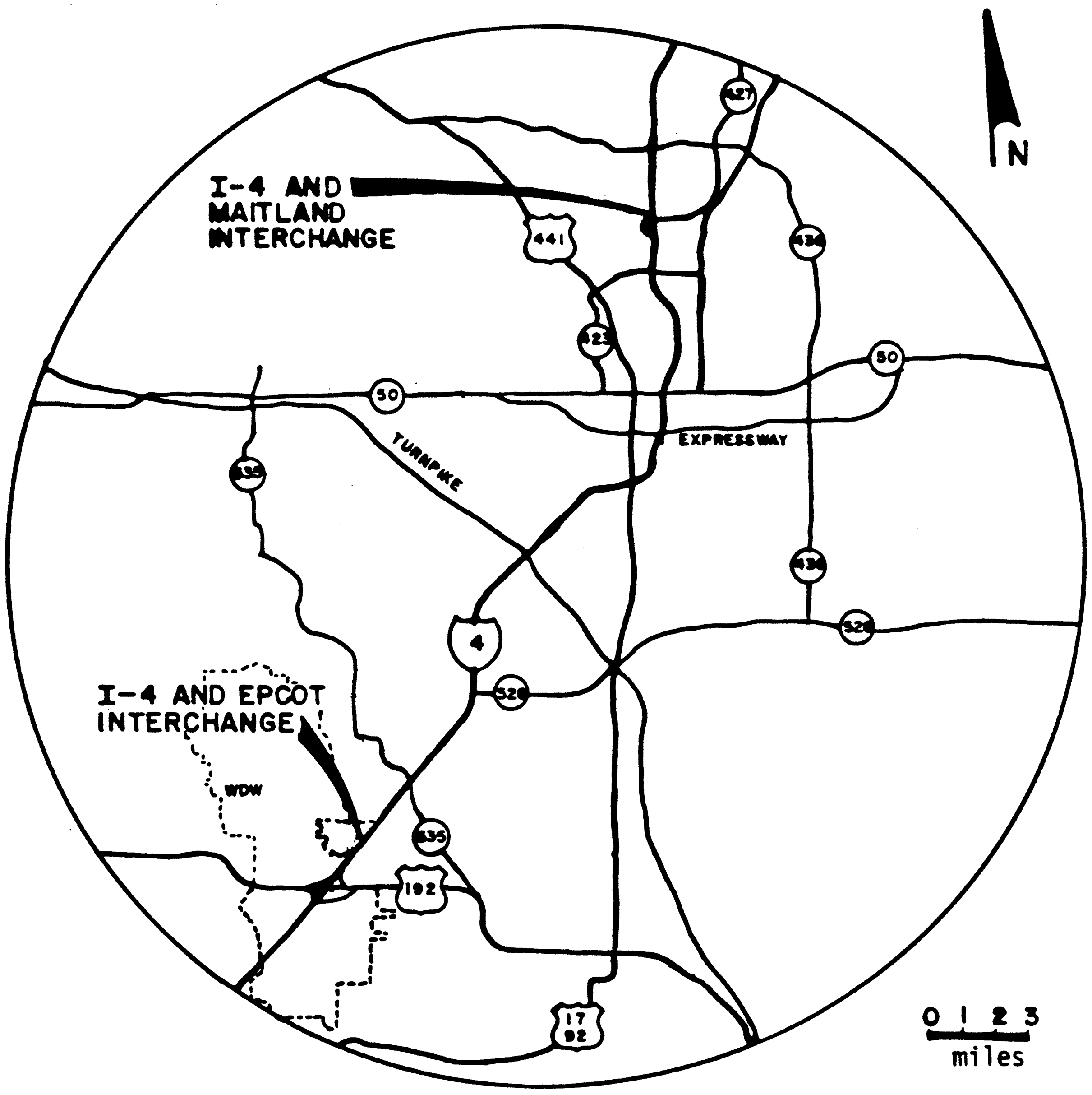

Figure 2-1: Study sites selected in Orange County, Fla. for best management of highway runoff

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 also having a 16 m roadway and a 16 m horizontal clearance. The average daily traffic volume (ADT) on Maitland Boulevard approximates 15,000.

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 45,000 ADT in each direction, east and west bound.

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

TABLE 2-1

Total Design Drainage Areas for Ponds
Located at Maitland Interchange

Pond	Location of Pond	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

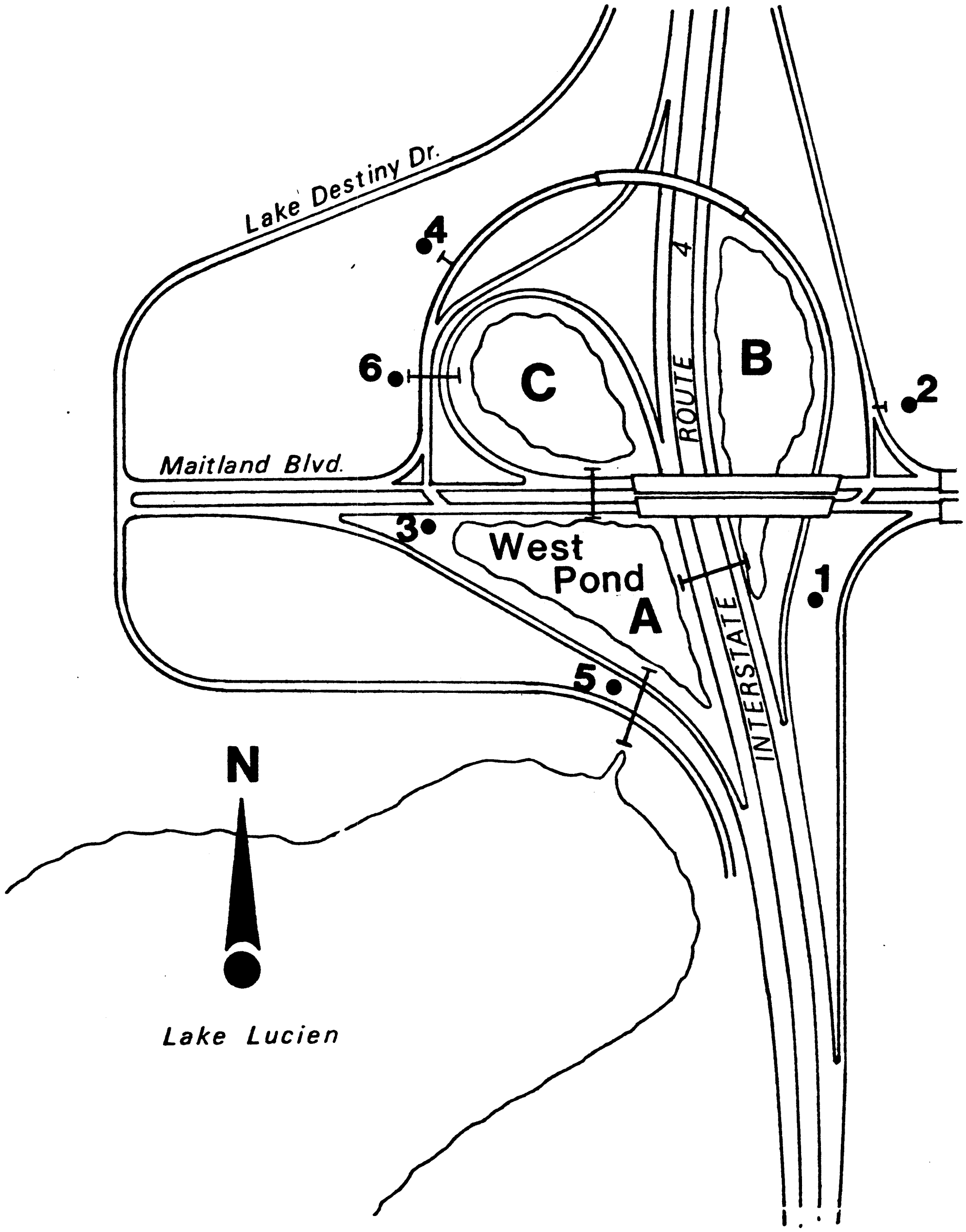

Figure 2-2: **SAMPLING SITE FOR HIGHWAY RUNOFF AT MAITLAND INTERCHANGE AND INTERSTATE 4**

over Interstate 4 is conveyed directly off the roadway surface through stormwater inlets to culverts that ultimately discharge directly into Pond A. The ponds are interconnected so that the water from the two northernmost ponds flows into the southwest pond (referred to hereafter as the west pond) when it reaches 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 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 reinforced concrete pipe (RCP), as shown in Figure 2-3.

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. The water was spiked with a concentrated solution of heavy metal (Pb, Cd, Zn, Cr, Cu, Ni and Fe) and nutrients (P and N) 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

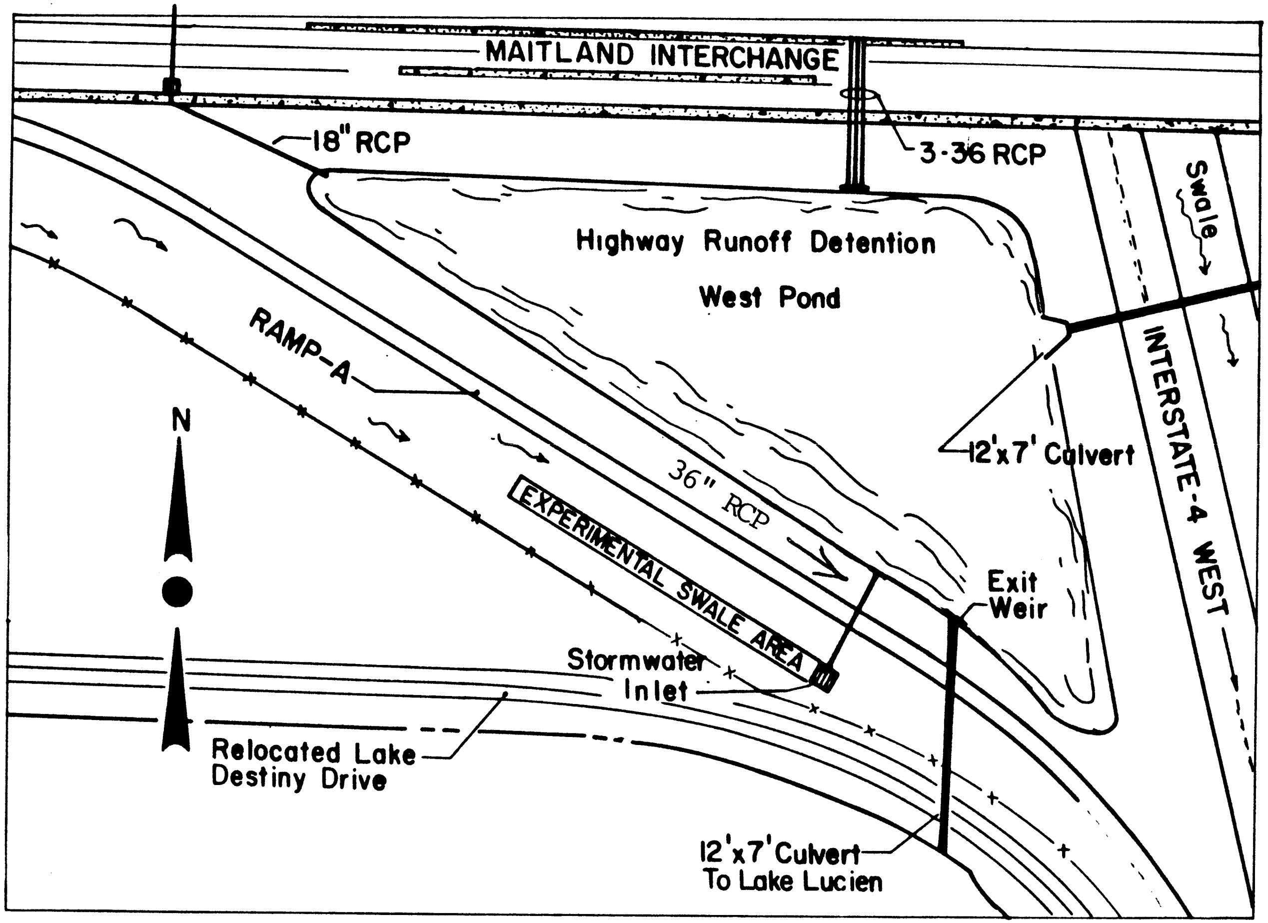

Figure 2-3: LOCATION OF EXPERIMENTAL SWALE ALONG RAMP-A AT MAITLAND INTERCHANGE AND I-4

the swale and return back to the inlet. The swale area was covered with grasses, predominately bahia. Cross sectional areas were taken at 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 2-4 and 2-5. Figure 2-4 represents swale cross sections for the study area at Maitland. However, Figure 2-5 shows enlarged scale of the water flow sections along the swale area. A sharp-crested 90° V notch weir, shown in Figures 2-6 and 2-7, was placed before the outlet 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 was included in the stormwater sampling program, with various numbers of samples taken at each station. The sampling locations are depicted in Figure 2-2. Collection of these samples assisted in differentiating between runoff quality from highways before and after flowing over adjacent swales.

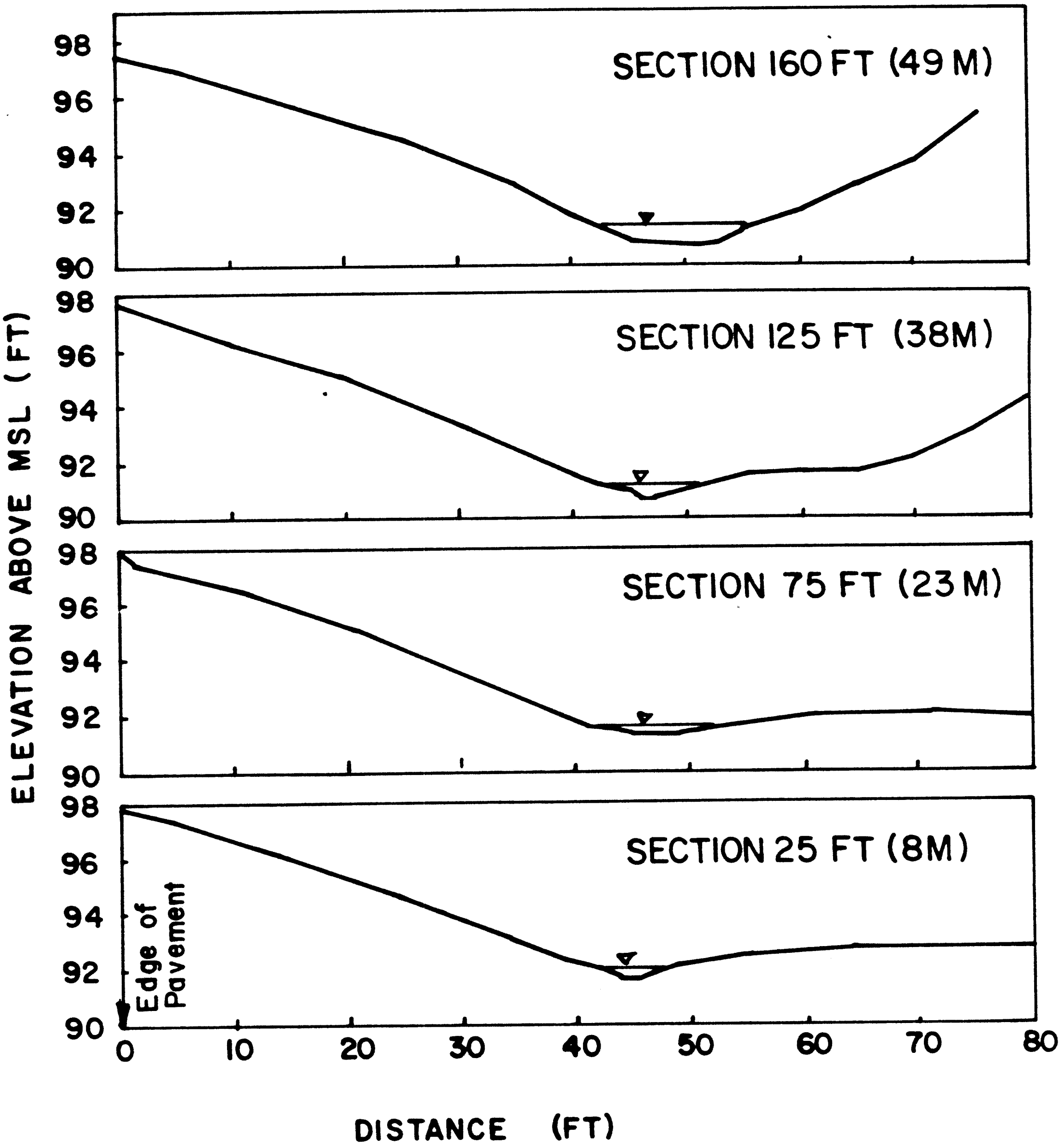

Figure 2-4: CROSS SECTIONS OF GRASSY SWALE ALONG RAMP AT MAITLAND INTERCHANGE AND I-4

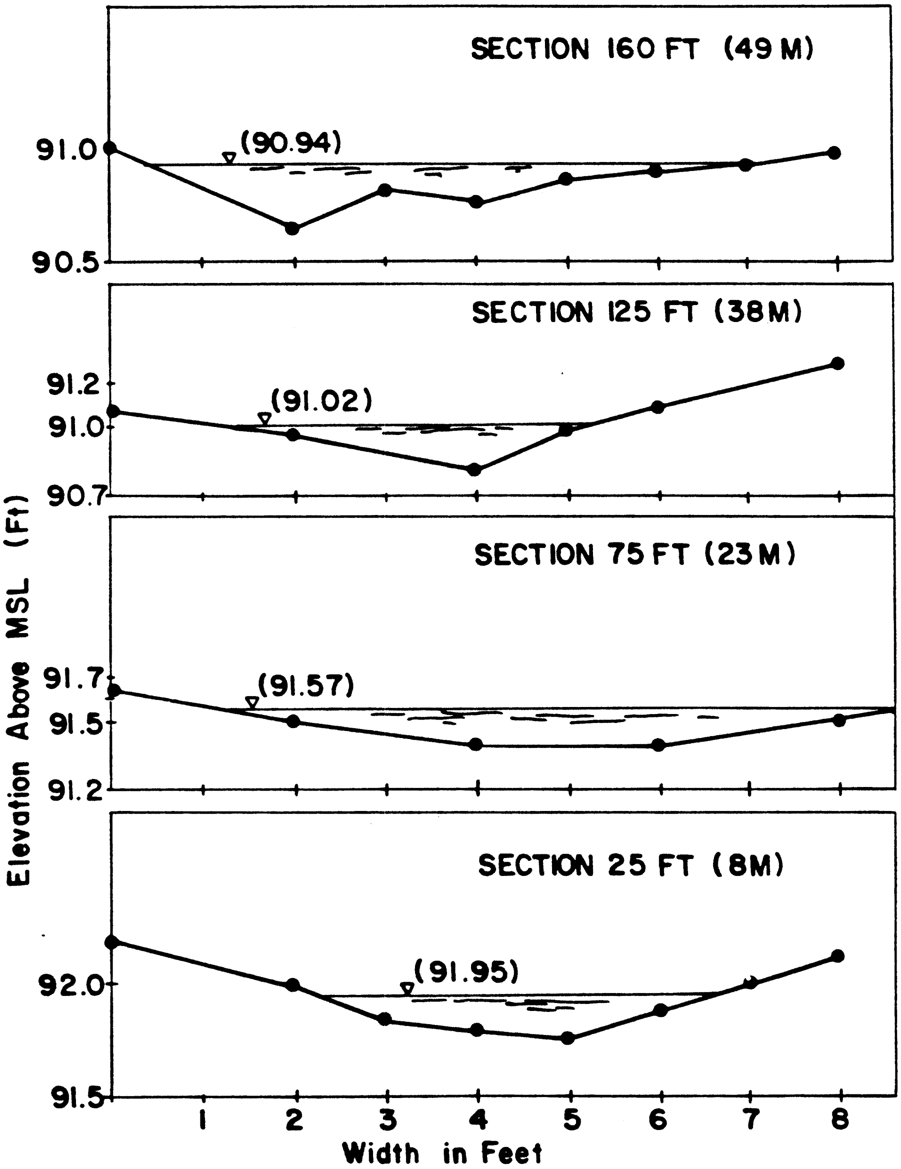

Figure 2-5: CROSS SECTIONS FOR WATER FLOW THROUGH SWALE ADJACENT TO MAITLAND INTERCHANGE AND I-4.

FIGURE 2-6: WATER SAMPLING THROUGH AN END 90° V-NOTCH WEIR DURING EXPERIMENT AT MAITLAND SITE.

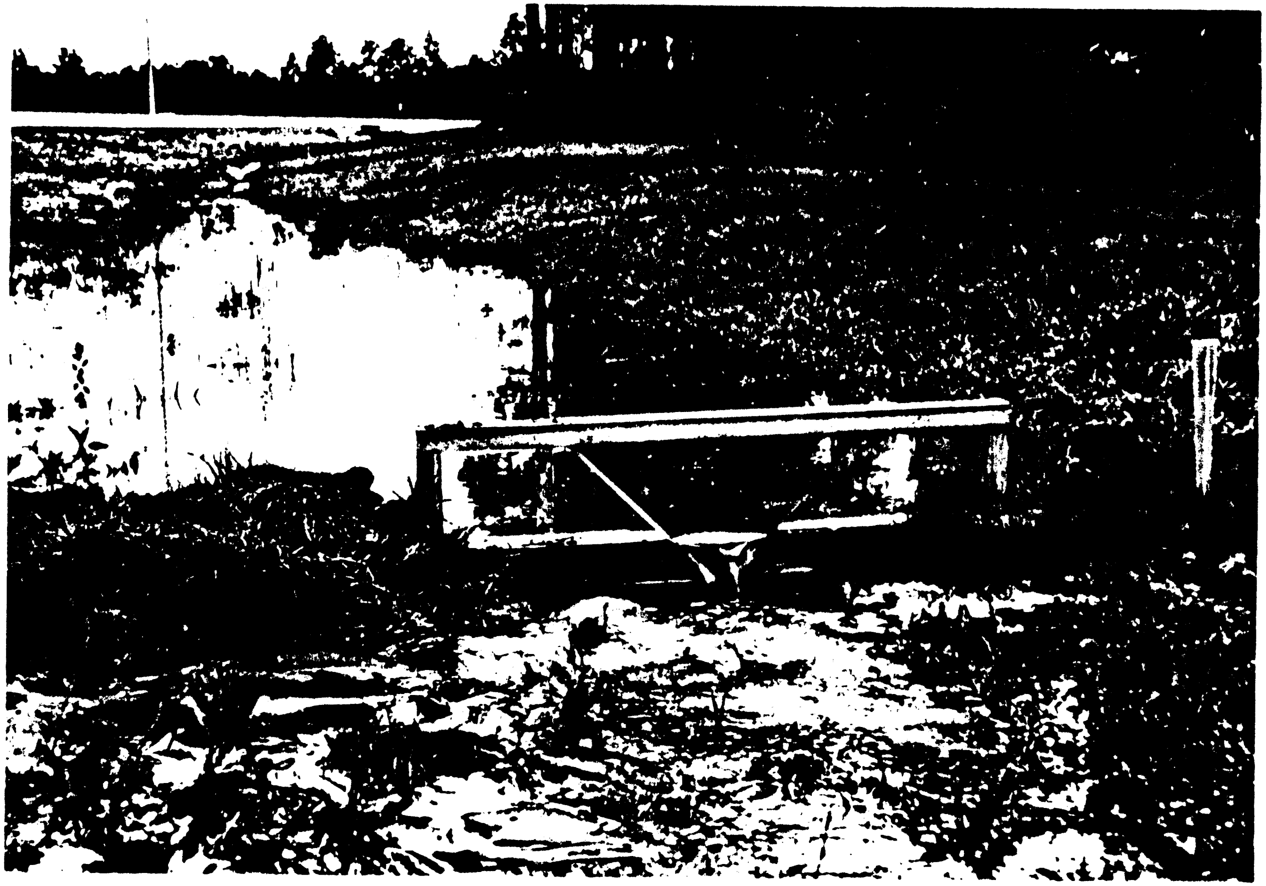

FIGURE 2-7: A 90° V-NOTCH WEIR LOCATED 300 FEET FROM FLOW INLET DURING SWALE EXPERIMENTS AT EPCOT SITE

EPCOT

EPCOT interchange was constructed during 1982-83 as a 0.8 mile multilane 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 2-8. The swale area selected for the study was a newly constructed swale along ramp A which connected the EPCOT Center Exit to the westbound lanes of I-4. A detailed map of the swale area is shown in Figure 2-9. Cross sectional profiles of the swale area were taken at 15 ft (5 m), 100 ft (30 m), 200 ft (60 m), 305 ft (92 m), 400 ft (120 m) and 525 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 2-10 and 2-11. 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-41) for flow measurements. A photograph of the weir installed at 300 ft (90 m) is shown in Figure 2-7. 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 percent in the first experiment and 80 percent 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 inch RCP connector culvert to the swale area. Hence it flowed by gravity over the swale area as shown in Figures 2-7 and 2-9.

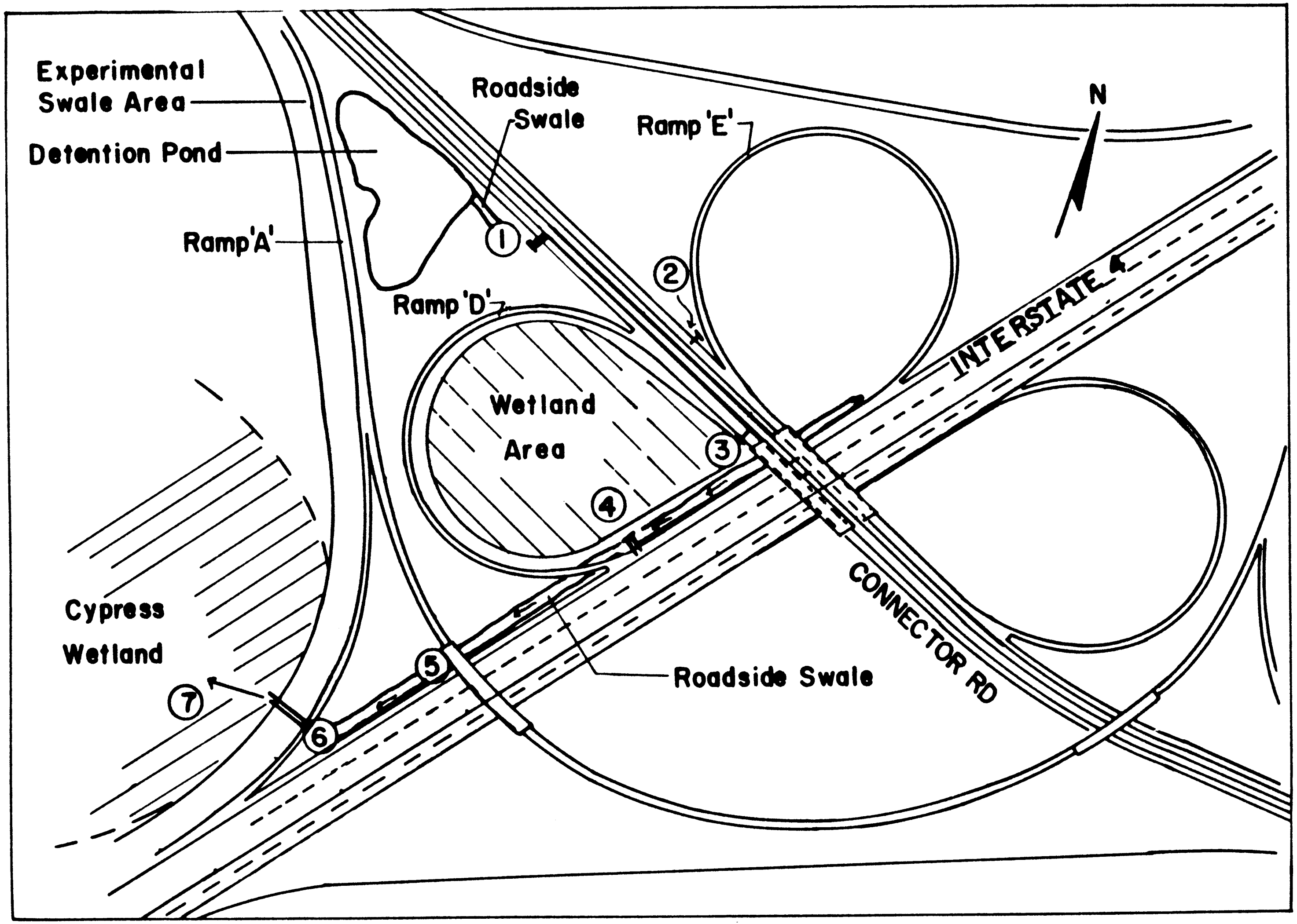

Figure 2–8: HIGHWAY RUNOFF STUDY AREA AT I-4 AND EPCOT INTERCHANGE

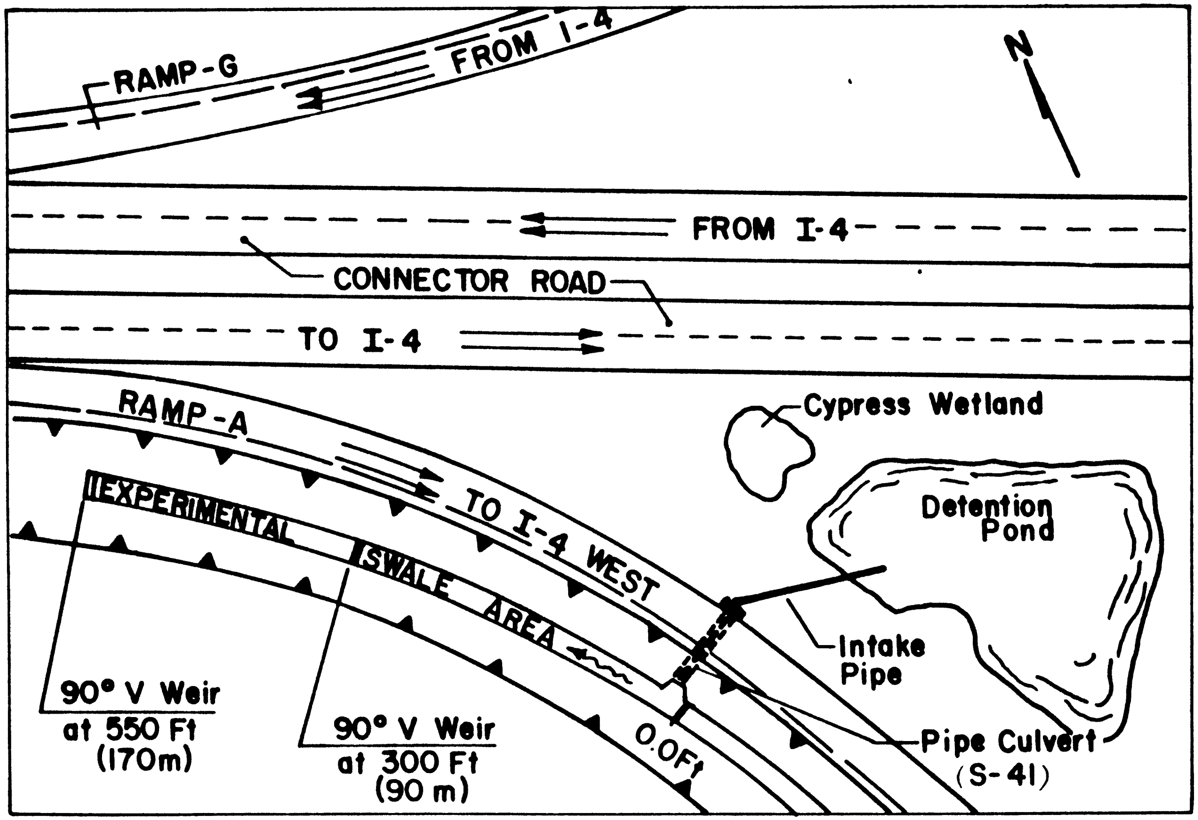

Figure 2-9: LOCATION OF EXPERIMENTAL SWALE ALONG RAMP-A AT I-4 AND EPCOT INTERCHANGE.

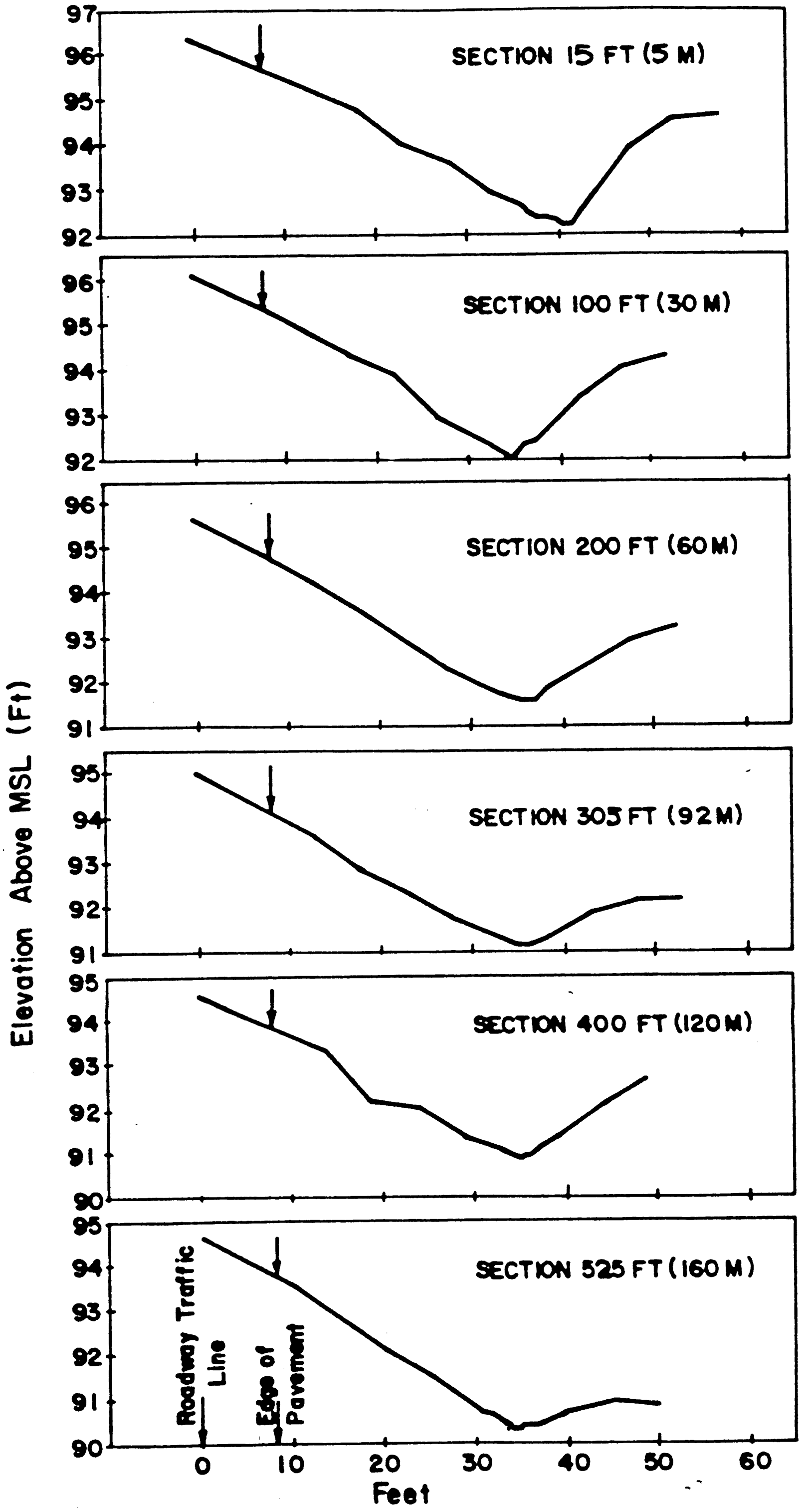

Figure 2-10: CROSS SECTIONS OF GRASSY SWALE ALONG RAMP A AT I-4 AND EPCOT INTERCHANGE

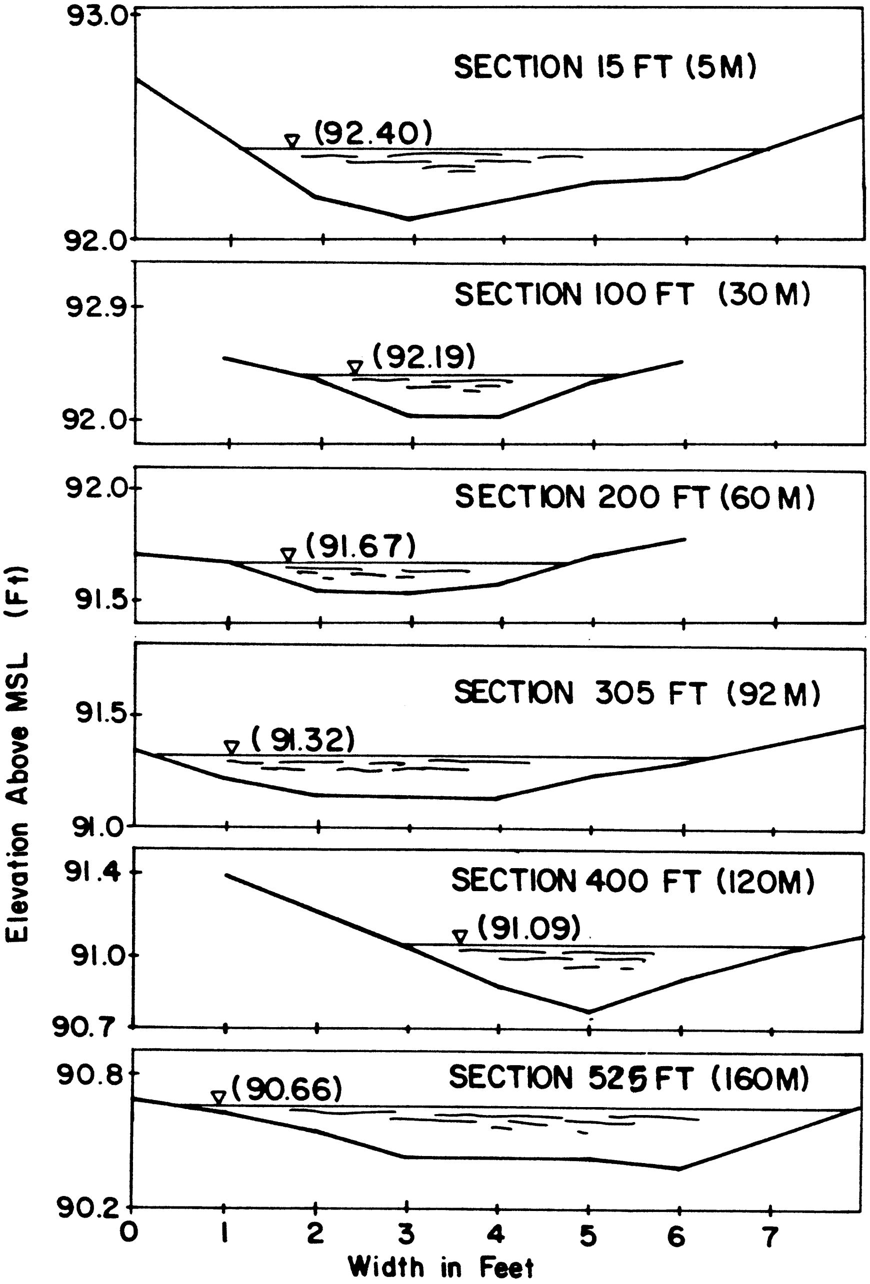

Figure 2-11: CROSS SECTIONS FOR WATER FLOW THROUGH SWALE ADJACENT TO RAMP A AT EPCOT INTERCHANGE

FIELD EXPERIMENTATION

Concentrated solutions made from selected heavy metals and nutrients were dosed at a 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. 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 was 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, 305 ft, 400 ft and 520 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 2-5 and 2-11. 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.

HYDROLOGIC AND HYDRAULIC 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 valves 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 2-12, 2-13, and 2-14 for the Maitland area and Figures 2-15 and 2-16 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 2-2. 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

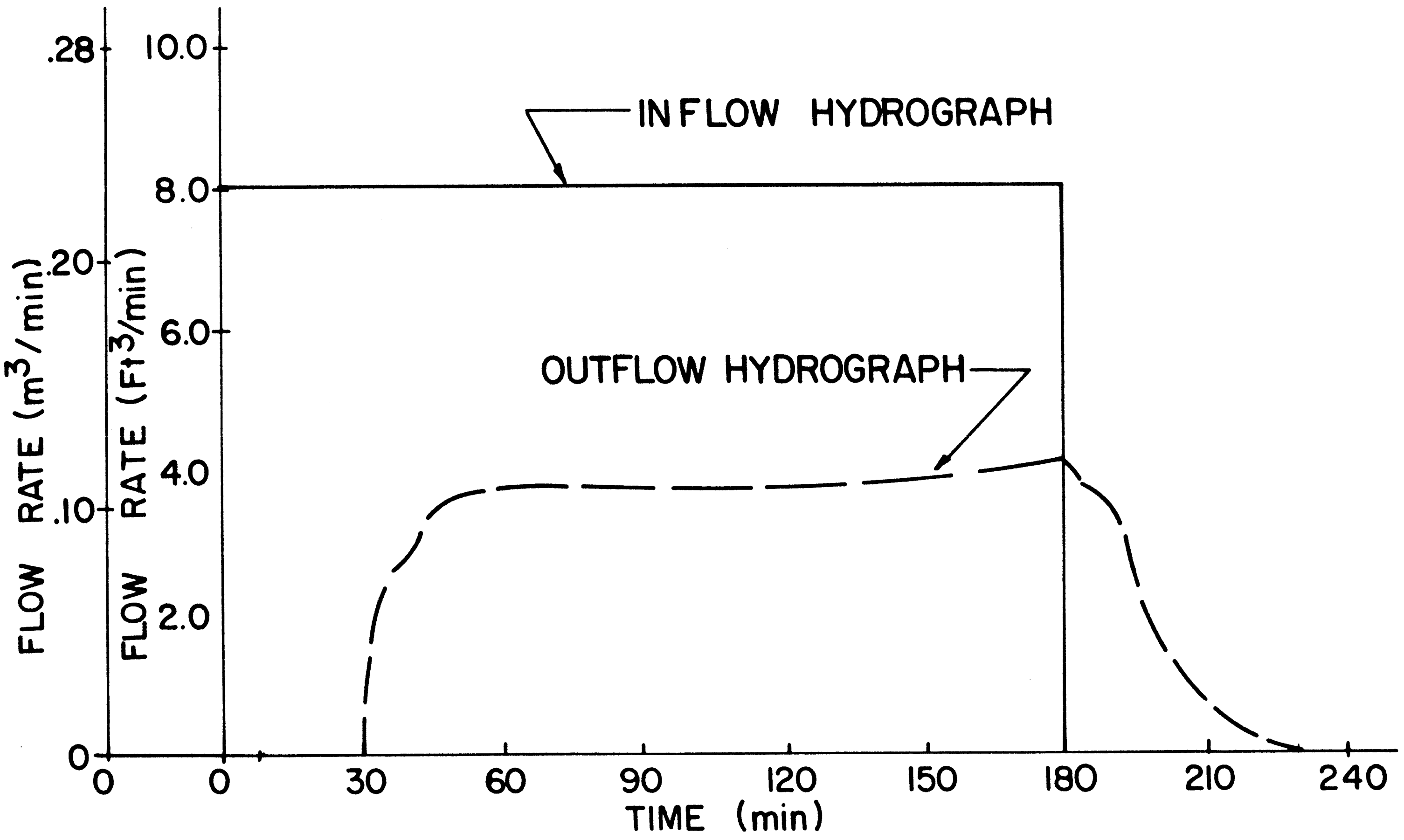

Figure 2-12: FLOW HYDROGRAPHS FOR SWALE EFFICIENCY EXPERIMENTS CONDUCTED AT MAITLAND INTERCHANGE ON 01/24/83

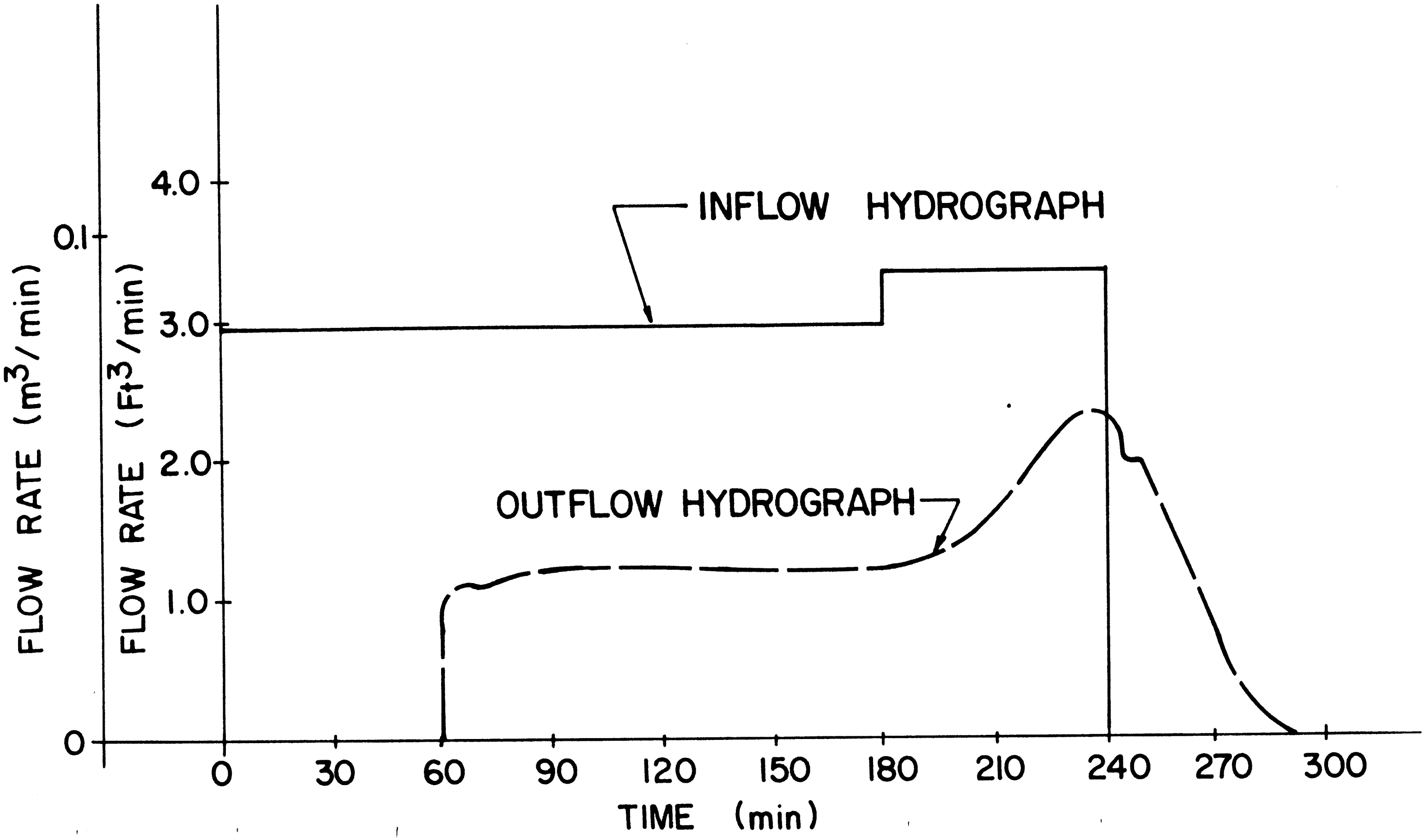

Figure 2-13: FLOW HYDROGRAPHS FOR SWALE EFFICIENCY EXPERIMENTS CONDUCTED AT MAITLAND INTERCHANGE ON 2/7/83

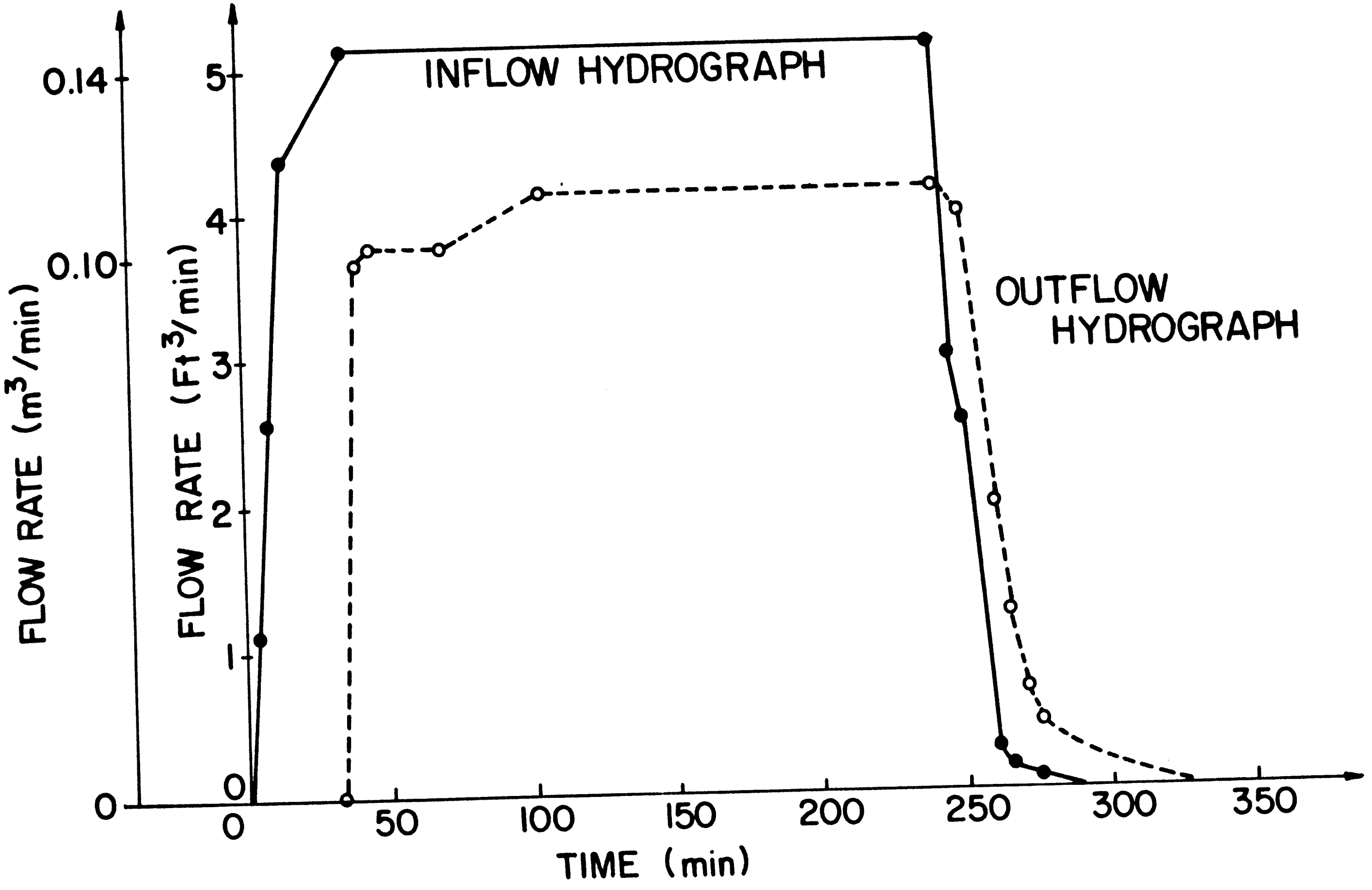

Figure 2-14: FLOW HYDROGRAPHS FOR SWALE EFFICIENCY EXPERIMENTS CONDUCTED AT MAITLAND INTERCHANGE ON MAY 31, 1983

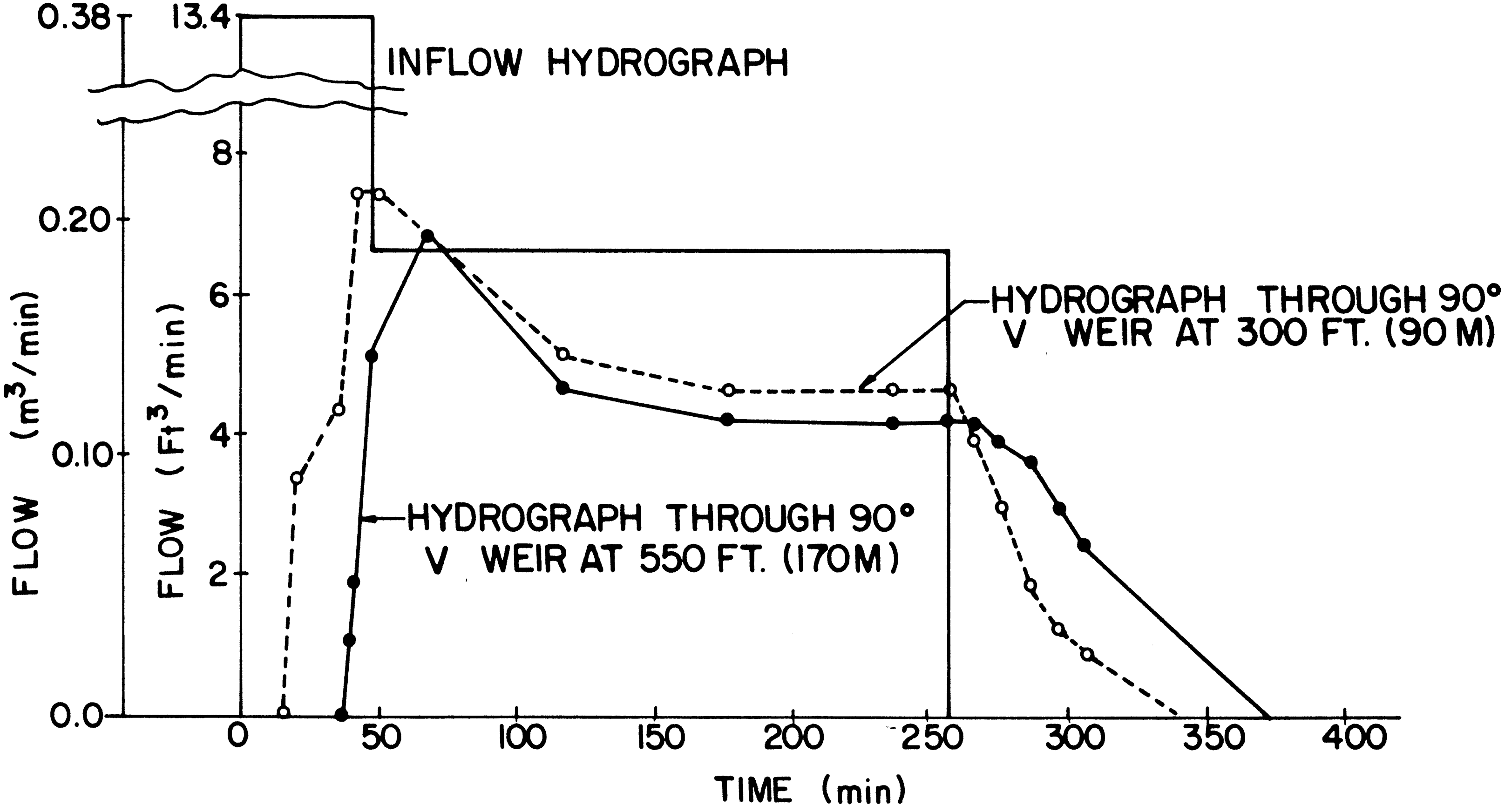

Figure 2-15: FLOW HYDROGRAPHS FOR SWALE EXPERIMENTS CONDUCTED ADJACENT TO RAMP-A, NORTHWEST OF S-41 EXIT AT EPCOT INTERCHANGE ON 3/23/83

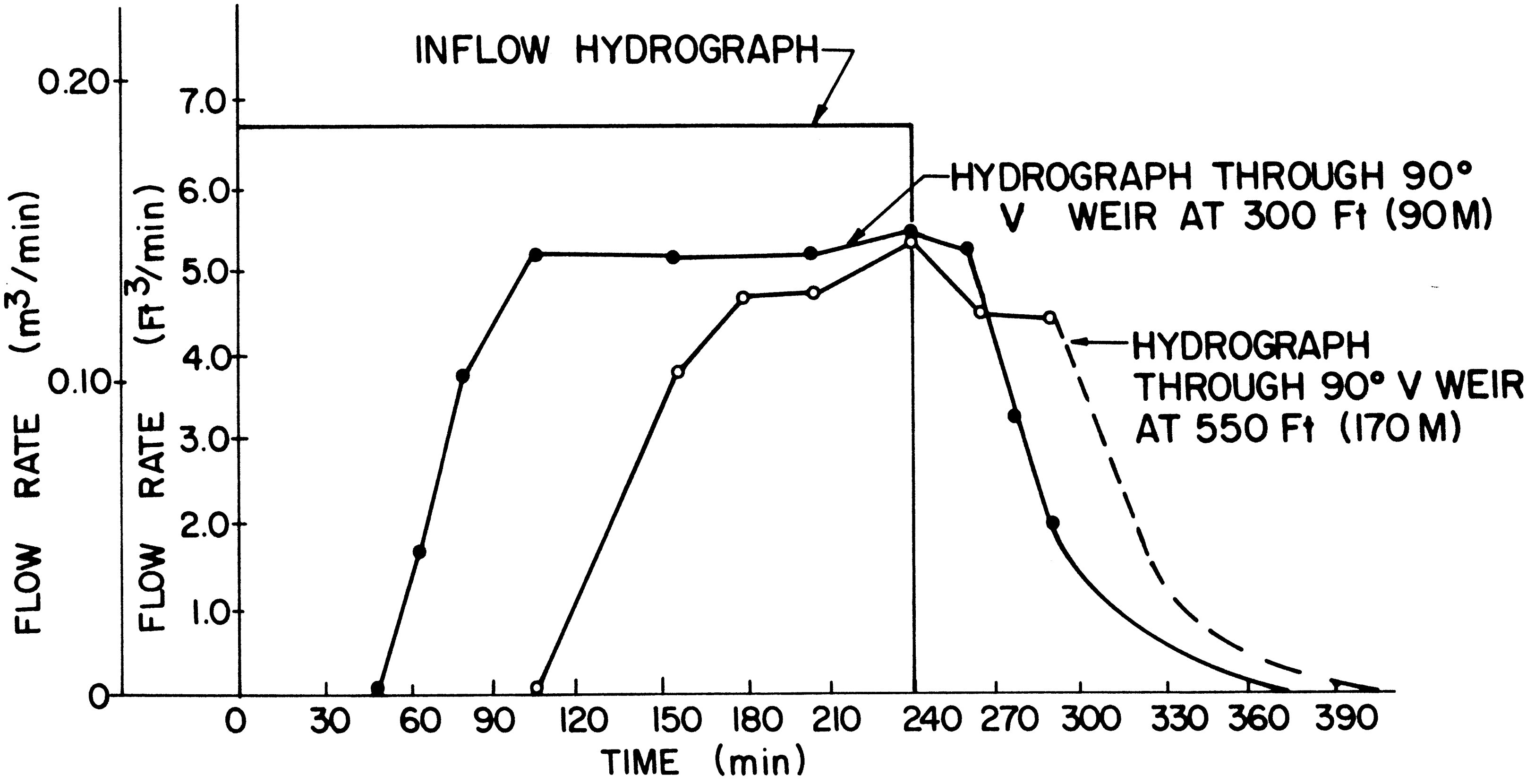

Figure 2-16: FLOW HYDROGRAPHS FOR SWALE EXPERIMENTS CONDUCTED ADJACENT TO RAMP–A, NORTHWEST OF S-41 EXIT AT EPCOT INTERCHANGE ON 5 /16/83

TABLE 2-2

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 80	4.18 4.00	57.7 39.2	39.2 35.8	0.105 0.079	4.13 3.10	0.034 0.007	1.33 0.17	0.071 0.072	2.80 2.83
5.	EPCOT	5/16/83	90 80	4.00	46.3 30.8	30.8 23.0	0.094 0.053	3.69 2.08	0.032 0.014	1.26 0.55	0.062 0.039	2.43 1.52
6.	Maitland	5/31/83	53		35.1	26	0.092	3.62	0.024	0.94	0.068	2.68

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 (0.036 m^3/m^2-hr). Excess runoff reached more than 90 percent 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 2-3. The hydraulic water depth, which is defined as the cross sectional area divided by the top width of flow, did not exceed 0.041 m (1.6 in). The calculated water velocity varied from 0.90 to 2.98 m/min during the swale experiments under steady state conditions of flow.

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).

Water quality analyses were conducted on filtered aliquots of water samples for the following parameters: pH, specific conductance, alkalinity, ammonia, nitrite, nitrate, total kjeldahl nitrogen (TKN),

TABLE 2-3

Hydraulic Characteristics of Swale Experiments

Experiment			Swale Characteristics				Flow (m^3/min)			Flow Through Time (min)	
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	0.058	1.46	0.040	0.189	0.131	2.76	--	33
			80	0.060	1.49	0.040	0.131	0.118	2.08	--	39
5.	EPCOT	5/16/83	90	0.056	1.37	0.041	0.189	0.145	2.98	48	30
			80	0.071	1.83	0.039	0.145	0.131	1.94	57	41
6.	Maitland	5/31/83	53	0.056	1.79	0.031	0.145	0.118	2.35	35	23

dissolved and total orthophosphorus, total phosphorus, and heavy metal such as cadmium, zinc, copper, iron, lead, nickel and chromium.

Nutrients N and P analyses were run as described in Standard Methods for 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.

Water samples were also prepared for the heavy metal determination in accordance with the procedures outlined in Standard Methods. Samples were preserved with 2 ml of concentrated HNO_3 for every 100 milliliters of sample and digested down to about 10 ml. Then each sample was brought to a final volume of 20 ml using distilled and deionized water. A blank was also prepared using an equivalent amount of glass distilled, deionized water and nitric acid to that used in the procedure for the raw water samples. The blank readings were subtracted from each sample to obtain its proper concentration. The analysis was done for all the elements simultaneously using the multi-element cassette on a Spectrometrics Inc. Spectrospan III Plasma Emission Spectrophotometer.

CHAPTER III

HEAVY METALS IN ROADSIDE SWALES

INTRODUCTION

Within the past decade a substantial amount of research has accumulated relating to air and water pollution caused by the operation of motor vehicles. This concern is based mainly on the potential aquatic toxicity of heavy metals such as lead, zinc, and chromium. Heavy metals have been proposed by several researchers as the major toxicant present in highway runoff samples (Shaheen, 1975; Winters and Gidley, 1980). Many heavy metals are known to be toxic in high concentrations to a wide variety of aquatic plants and animals (Wilber and Hunter, 1977).

Heavy Metals in Highway Runoff

Heavy metals in highway runoff have been reported to originate not only from the operation of motor vehicles and direct fallout, but also from the highway pavement materials as well. Sources of metals include gasoline (Pb), exhaust emissions (Pb, Ni), crankcase and lubricating oils (Pb, Ni, Zn), grease (Zn, Pb), tire wear (Cd, Zn), wear on moving bearings (Cu, Pb), decorative and protective coatings (Al, Cd, Cu, Zn, Ni, Fe), brake lining wear (Cu, Cr, Ni), moving engine parts (Fe, Mn, Cr, Co), and asphalt paving wear (Ni, V) (Hopke et al., 1980; Novotny and Chesters, 1981; and Portele et al., 1982). Christensen and Guinn (1979) indicated an average deposition of 0.0030 g zinc/vehicle km predominately from tire wear and 0.0049 g lead/vehicle km from lead in gasoline.

In a study on the contribution of urban roadway use to water pollution, Shaheen (1975) stated that the majority of solids deposited on roadways are vehicle dependent with over 95 percent of the material being inorganic in nature. He found that less than 5 percent of the solids deposited on the roadway originated from the vehicles themselves and that vehicles act predominantly as a transport mechanism for pollutants picked up at other locations. Several investigators have found close correlations between metal concentrations and total solids concentrations in highway runoff (Clark et al., 1981; and Portele et al., 1982).

Heavy metals contributed by direct precipitation on highway areas have been studied with conflicting results. Wilber and Hunter (1977) found that concentrations of heavy metals in precipitation were insignificant in relation to the concentrations found in highway runoff. However, Larsen (1975), stated that bulk precipitation in western Washington contributed significant quantities of trace metals. Bourcier and Hindin (1979) conducted a clever investigation in which they measured deposition rates of heavy metals inside a viaduct to represent deposition due solely to operation of vehicles and compared these values to deposition rates at a control station outside the viaduct, in an area unaffected by highway traffic. A summary of their findings is presented in Table 3-1. They concluded that the relatively low loadings of metals at the control station compared to that for the viaduct indicate that the major source of heavy metal deposition is vehicle related. Each of the total metals found in the runoff existed primarily in particulate form with only a small percentage in the dissolved-colloidal fraction. They also concluded that a residual amount of metals will remain on the road surface after each storm.

TABLE 3-1

Deposition Rates of Metals in Dust Samples at Viaduct Stations and Control Stations (Bourcier and Hindin, 1979)

Heavy Metal	No. of Obs.	Concentration ($mg/m^2.day$) Mean	Standard Deviation	Range
Viaduct Stations:	18			
Fe		317	258	61.3 - 1048
Pb		2.92	2.85	0.377 - 10.2
Cr		0.513	0.495	0.038 - 1.58
Zn		42.2	52.1	1.08 - 163
Control Stations	6			
Fe		6.78	9.63	0.380 - 25.6
Pb		0.366	0.720	0.004 - 1.83
Cr		0.222	0.256	0.002 - 0.646
Zn		0.798	0.542	0.082 - 1.38

Among the toxic metals, Pb, Zn and Cu are the most abundant in highway runoff and have received the most study. Wilber and Hunter (1977) found that the major contributors of heavy metals in stormwater were lead, zinc, and copper. Together these accounted for approximately 90 to 98 percent of the total metals observed, with lead and zinc accounting for 89 percent. Nickel and chromium were found in considerably smaller quantities. It was also reported that peak concentrations of heavy metals were observed shortly after the initiation of runoff, usually within the first thirty minutes. They also reported a tendency for solids to settle out in the stormsewers during the latter stages of the storm flow.

The significance of average daily traffic, percent imperviousness, and dustfall to metal concentrations in highway runoff was examined by Gupta et al., 1981. Both simple and multiple correlations were performed between the site characteristics and heavy metal concentrations. For the simple correlations heavy metal concentrations were found to correlate best with percent imperviousness although significant correlations were found with only Pb, Zn, and Cu. However when multiple correlations were considered the correlations improved considerably, with the combination of all three site characteristics producing the best correlations. A summary of their results is presented in Table 3-2.

TABLE 3-2

Correlation Coefficients for Metal Concentrations and Site Characteristics (Gupta et al., 1981)

	Simple Correlation Coefficients			Multiple Correlation Coefficients			
Heavy Metal	ADT	% IMP	DF	ADT and % IMP	ADT and DF	DF and % IMP	ADT, % IMP and DF
Pb	-0.174	0.942	0.125	0.950	0.913	0.945	0.978
Zn	-0.540	0.949	0.255	0.949	0.950	0.950	0.996
Fe	-0.315	0.605	-0.008	0.669	0.955	0.621	0.988
Cu	0.095	0.954	0.387	0.965	0.928	0.972	0.994
Cd	0.036	0.894	0.302	0.898	0.831	0.901	0.913
Cr	-0.474	0.779	-0.279	0.892	0.707	0.904	0.907
Hg	0.960	0.027	0.889	0.963	0.962	0.905	0.987

ADT = Average Daily Traffic
% IMP = Percent Imperviousness
DF = Dustfall

The quantities of heavy metals in highway runoff and the forms in which they exist depend to a large degree on the physical and chemical behavior of the specific elements. Lead has been shown to exist on the highway in a predominantly insoluble particulate form which is largely inorganic in nature (Laxen and Harrison, 1977). It was also found that most of the lead emitted from automobile exhaust occurs in two distinct particle sizes: < 1 um and 5-50 um. Habibi (1973) found in wind tunnel experiments that between 50 and 65 percent of Pb particles greater than 9 um are deposited within 7 m of the exhaust pipe. Wang et al., (1982), in a study conducted in Washington State found that approximately 92 percent of the total lead percent was associated with particles greater than 20 um in size. In most highway runoff investigations it is found that Zn, Cu and Cd are all considerably more soluble than Pb. Shaheen (1975) observed that dissolved Zn was almost always higher than dissolved Pb in roadway runoff, despite the fact that Pb was approximately eight times more abundant in materials deposited on the highway.

Actual concentrations and loading rates of heavy metals in highway runoff have been presented by numerous researchers from many areas of the country. Most of the data is highly site specific and reflects variations in dustfall, average daily traffic, land use in the area of the highway, as well as many other contributing parameters. However, one of the most extensive studies of metals in highway runoff was conducted by Gupta et al., (1981). Samples were collected at seven highway sites in five states (Pennsylvania, Tennessee, Louisiana, Wisconsin and Colorado), representing a wide range of ADT values, geographic locations, drainage area, design characteristics, pavement characteristics and receiving water characteristics. A summary of data for 159 storm events is listed in Table 3-3.

Previous Studies on Removal of Heavy Metals in Roadside Swales

Although it is a routine practice to drain highway runoff through various engineered channels, there is little documentation of the effect which this practice has on water quality, especially heavy metals. The only attempt to investigate this phenomenon in a previous

TABLE 3-3

Summary of Heavy Metals in Highway Runoff
at Seven Locations during 1976-77 (Gupta et al., 1981)

Heavy Metal	Pollutant Concentration (mg/l)		Pollutant Loadings (kg/ha/event)		Pollutant Loading (Kg/ha/cm runoff)	
	Average	Range	Average	Range	Average	Range
Pb	0.96	0.02 - 13.1	0.065	0.00 - 0.54	0.097	0.002 - 1.31
Zn	0.41	0.01 - 3.4	0.025	0.00 - 0.13	0.041	0.0009 - 0.34
Fe	10.30	0.10 - 45.0	0.56	0.00 - 3.9	1.03	0.01 - 4.50
Cu	0.103	0.01 - 0.88	0.0063	0.00 - 0.033	0.010	0.0009 - 0.088
Cd	0.040	0.01 - 0.40	0.0019	0.00 - 0.016	0.004	0.0009 - 0.040
Cr	0.040	0.01 - 0.14	0.031	0.00 - 0.033	0.004	0.0009 - 0.014
Hg	0.003	0.001 - 0.067	0.0007	0.00 - 0.0024	0.0003	0.0128 - 6.70

study was one performed by Wang et al., (1982). This study was limited in scope and designed to measure heavy metal concentrations in stormwater runoff draining through a paved channel, a mud-bottomed ditch and channels vegetated with grasses. No general conclusions were reached; however, it was reported that a 60 m grass channel of small slope reduced highway runoff dissolved Pb concentrations by 80 percent or more. Removal efficiencies for dissolved Cu averaged 60 percent and for dissolved Zn averaged 70 percent in the same channel. Bare earthen 15 meter long channels and paved channels did not produce significant reductions in heavy metal concentrations. However, no mechanisms were suggested for the observed results. A summary of average removal efficiencies for various heavy metals from the four experiment sites used by Wang for various travel distances through the grassed swale is given in Table 3-4. Removal efficiencies were averaged for each sampling distance over different sampling times from various sites.

MECHANISMS OF HEAVY METAL IMMOBILIZATION

It has been reported that most of the dissolved heavy metals entering or being transported by natural water systems are, under normal physiochemical conditions, rapidly removed from the water phase and concentrated in the sediment phase (Swanson and Johnson, 1980; Wilber and Hunter, 1977; Guthrie and Cherry, 1979; and Hem and Durum, 1973). Nightingale (1975) found concentration factors of 32.0, 8.4 and 4.4 for lead, zinc and copper, respectively, in the sediments of a two-year-old storm drainage retention basin when compared to background concentrations in control basins.

TABLE 3-4

Average Removal Efficiencies for Dissolved Heavy Metals after Passage Through Various Lengths of Grassy Channels (from Wang, 1982)

Distance (m)	No. of Samples	Average Percent of Metal Removed (%)					
		Cd	Cu	Fe	Mn	Pb	Zn
2.5	2	BDL*	< 0	6	10	3	8
5.0	1	46	34	68	73	72	60
10.0	1	BDL	29	40	49	59	17
15-21	11	68	37	53	56	60	38
25	2	BDL	62	70	81	85	76
30-40	3	87	52	41	58	60	44
40-50	6	80	39	73	61	72	70
50-60	2	35	20	54	75	67	64
60-70	7	100	60	75	60	85	69
70-80	2	BDL	58	71	92	81	72

*BDL = Below Detectable Limit

From studies based on the movement of heavy metals in the Amazon and Yukon rivers, Gibbs (1973) found that, despite their dissimilar originations, natural and man-made heavy metal species form groups of heavy metal associations in aquatic solid substances that can be characterized by the following bonding processes: (1) precipitation of heavy metals, (2) cation exchange and adsorption, (3) coprecipitation of hydrous Fe/Mn oxides, (4) association with organic molecules, and (5) incorporation into crystalline or detrital minerals.

The efficiency of the various influencing factors on entrapment and enrichment of heavy metals in aquatic sediments depends on the chemical characteristics, salinity, pH and redox potential as well as various hydrodynamic processes. The general effects of these influential factors are summarized by Forstner and Wittman (1979).

Remobilization of Heavy Metals from Sediments

Remobilization of heavy metals from suspended material and sediments could be potentially hazardous for aquatic ecosystems. Remobilization of metals is mainly caused by four types of chemical changes in waters (Forstner and Wittman, 1979).

1. Elevated salt concentrations which allow alkali and alkaline earth cations to compete with the metal ions sorbed onto solid particles.
2. Changes in redox potential, usually in conjunction with a decrease in the oxygen content, which can cause iron and manganese hydroxides to be destroyed and release part of the incorporated metal load.
3. Lowering of the hydrogen ion concentration which could lead to a dissolution of carbonates and hydroxides as well as to increased desorption of metal cations due to competition with hydrogen ions.
4. Increased use of natural and synthetic complexing agents, which can form soluble metal complexes, sometimes of high stability, with heavy metals that are otherwise adsorbed to solid particles.

In addition to these chemical processes, there are other biochemical processes, by means of which the heavy metals are either transferred from the sediments to animal or plant species or, by way of decomposition products, into the latter. These biochemical processes are not well defined.

EXPERIMENTAL RESULTS

In order to investigate the efficiency of roadside swales for removal of heavy metals a series of experiments was conducted at two locations in the Central Florida area: (1) an older established swale area near the Maitland/I-4 interchange, and (2) a newly constructed swale area at the EPCOT interchange.

Maitland Interchange

Three swale experiments were conducted at the Maitland Interchange during which the removal of dissolved heavy metals was investigated in a flow through situation at three different flow velocities ranging from 2.58 m/min to 0.90 m/min. A summary of the data for the first experiment averaged over a time period of 180 minutes at an average velocity of 2.58 m/min is presented in Table 3-5 and in Figure 3-1. Dissolved zinc was removed at the fastest rate with 77 percent being removed over a distance of 53 meters. As seen in Figure 3-1, the removal rate for zinc was approximately linear, and the shape of the curve suggests that additional removal could have been obtained if the swale test length had been increased. Excellent removal was also noted for dissolved iron, although the concentration increased slightly during the first 10 - 15 meters. Although lead was removed to only 40 percent of its original concentration after a travel distance of approximately 20 m, a significant increase in lead concentration was observed in the 53 m sample. Removals of dissolved copper, aluminum, and chromium were much less with an average of approximately 80 percent remaining after 53 m. Measured concentrations of nickel and cadmium were too low to accurately comment on removal efficiencies for these

TABLE 3-5

Average Removal Efficiencies for Heavy Metals in Maitland Swale Study Conducted 1/24/83

Heavy Metal	Distance Along Swale (m)				
	0	7.5	23	38	53
ZINC:					
Avg. Conc. (ug/l)	22	19	13	10	5
Avg. Conc. (Mole/l)	2.0×10^{-7}	1.7×10^{-7}	1.2×10^{-7}	0.89×10^{-7}	0.45×10^{-7}
% Remaining	100	86	59	45	23
COPPER:					
Avg. Conc. (ug/l)	8	8	7	6	7
Avg. Conc. (Mole/l)	1.3×10^{-7}	1.3×10^{-7}	1.1×10^{-7}	0.94×10^{-7}	1.1×10^{-7}
% Remaining	100	100	88	75	88
ALUMINUM:					
Avg. Conc. (ug/l)	131	136	131	127	106
Avg. Conc. (Mole/l)	6.3×10^{-7}	50.0×10^{-7}	49.0×10^{-7}	47.0×10^{-7}	39.0×10^{-7}
% Remaining	100	104	100	97	81
IRON:					
Avg. Conc. (ug/l)	35	36	32	24	17
Avg. Conc. (Mole/l)	6.3×10^{-7}	6.4×10^{-7}	5.7×10^{-7}	4.3×10^{-7}	3.0×10^{-7}
% Remaining	100	103	91	69	49
LEAD:					
Avg. Conc. (ug/l)	12	15	5	6	15
Avg. Conc. (Mole/l)	0.58×10^{-7}	0.72×10^{-7}	0.24×10^{-7}	0.29×10^{-7}	0.72×10^{-7}
% Remaining	100	125	42	50	125
CHROMIUM:					
Avg. Conc. (ug/l)	12	11	9	9	12
Avg. Conc. (Mole/l)	2.3×10^{-7}	2.1×10^{-7}	1.7×10^{-7}	1.7×10^{-7}	2.3×10^{-7}
% Remaining	100	92	75	75	100

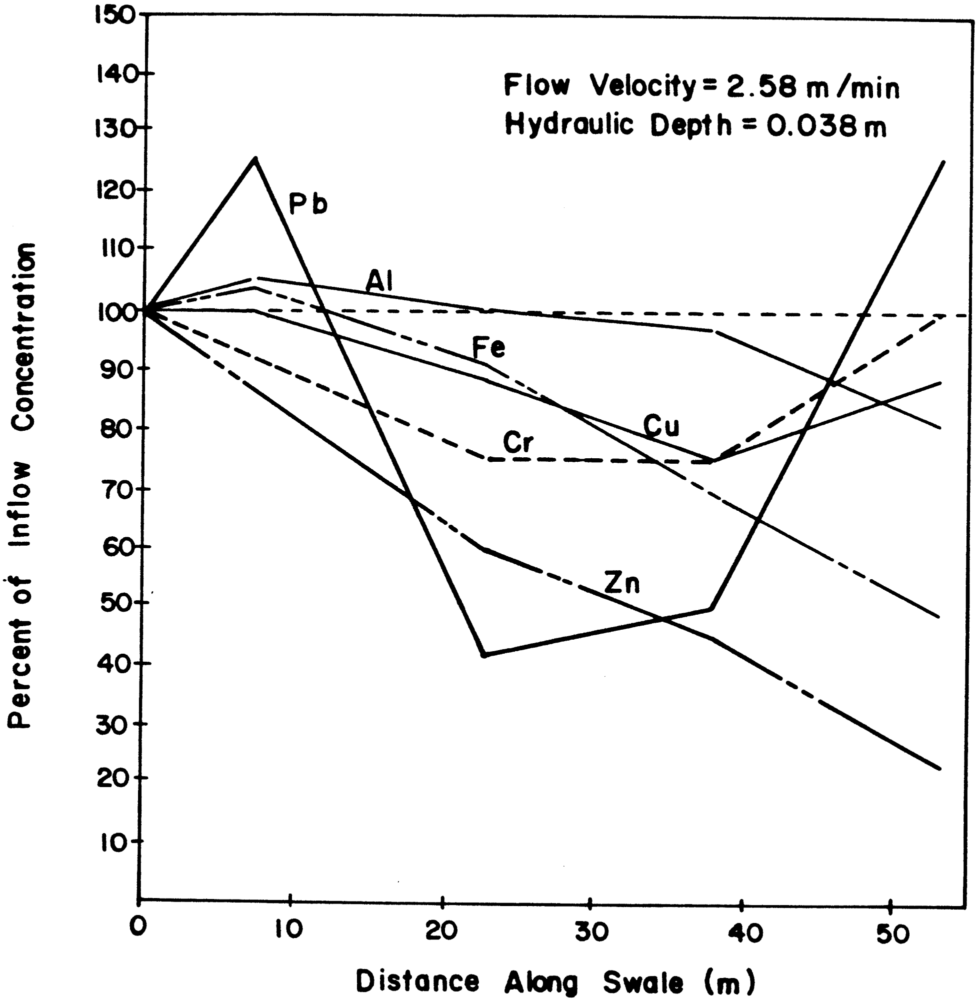

Figure 3-1: CHANGES IN CONCENTRATION OF VARIOUS HEAVY METALS DURING FLOW THROUGH A GRASSY SWALE AT MAITLAND ON 1/24/83.

two metals. The average pH maintained during the experiment was approximately 6.5.

A summary of average removals for the second Maitland experiment, conducted at a slower average velocity of 1.37 m/min, is given in Table 3-6 and in Figure 3-2. Dissolved zinc was again removed at the fastest rate with a calculated removal of 92 percent over the test distance of 53 m. Approximately 70 percent of the iron and 60 percent of the copper was also removed. Lesser removals averaging between 25 and 40 percent were noted for aluminum, lead, and chromium. It is interesting to note that lead, aluminum, and iron did not exhibit increases in the first 10 m at this slower velocity as they did in the first experiment which was conducted at a velocity almost twice as great. Final removal efficiencies for each of the metals listed in Table 3-6 exceeded those of the previous experiment. Measured concentrations of nickel and cadmium were again determined to be too low to accurately comment on removal efficiencies. The average pH of the runoff water during this experiment was approximately 6.5.

Another experiment conducted at Maitland Interchange was designed to test the effect of a very low velocity on removal efficiencies. A velocity of only 0.90 m/min was maintained throughout the experiment. As seen in Table 3-7 and in Figure 3-3, dissolved forms of zinc and iron were again removed most rapidly. Approximately 90 percent of the dissolved zinc and 70 percent of the dissolved iron were removed during overland flow at this velocity over a distance of 38 m. Aluminum was also removed to a large degree with an average final removal of approximately 75 percent. Concentrations of dissolved nickel, copper, lead, and cadmium were very low in the simulated stormwater flowing through

TABLE 3-6

Average Removal Efficiencies for Heavy Metals in Maitland Swale Study Conducted 2/07/83

Heavy Metal	Distance Along Swale (m)				
	0	7.5	23	38	53
ZINC:					
Avg. Conc. (ug/l)	26	23	9	6	2
Avg. Conc. (Mole/l)	2.0×10^{-7}	3.5×10^{-7}	1.4×10^{-7}	0.92×10^{-7}	0.31×10^{-7}
% Remaining	100	88	35	23	8
COPPER:					
Avg. Conc. (ug/l)	7	7	6	4	3
Avg. Conc. (Mole/l)	1.1×10^{-7}	1.1×10^{-7}	0.94×10^{-7}	0.63×10^{-7}	0.47×10^{-7}
% Remaining	100	100	86	57	43
ALUMINUM:					
Avg. Conc. (ug/l)	123	92	93	91	75
Avg. Conc. (Mole/l)	6.3×10^{-7}	34.0×10^{-7}	34.0×10^{-7}	34.0×10^{-7}	28.0×10^{-7}
% Remaining	100	75	76	74	61
IRON:					
Avg. Conc. (ug/l)	469	342	190	168	143
Avg. Conc. (Mole/l)	84.0×10^{-7}	6.4×10^{-7}	34.0×10^{-7}	30.0×10^{-7}	26.0×10^{-7}
% Remaining	100	73	41	36	30
LEAD:					
Avg. Conc. (ug/l)	8	8	5	8	6
Avg. Conc. (Mole/l)	0.39×10^{-7}	0.39×10^{-7}	0.24×10^{-7}	0.39×10^{-7}	0.29×10^{-7}
% Remaining	100	100	63	100	75
CHROMIUM:					
Avg. Conc. (ug/l)	6	6	2	4	4
Avg. Conc. (Mole/l)	1.2×10^{-7}	1.2×10^{-7}	0.38×10^{-7}	0.77×10^{-7}	0.77×10^{-7}
% Remaining	100	100	33	67	67

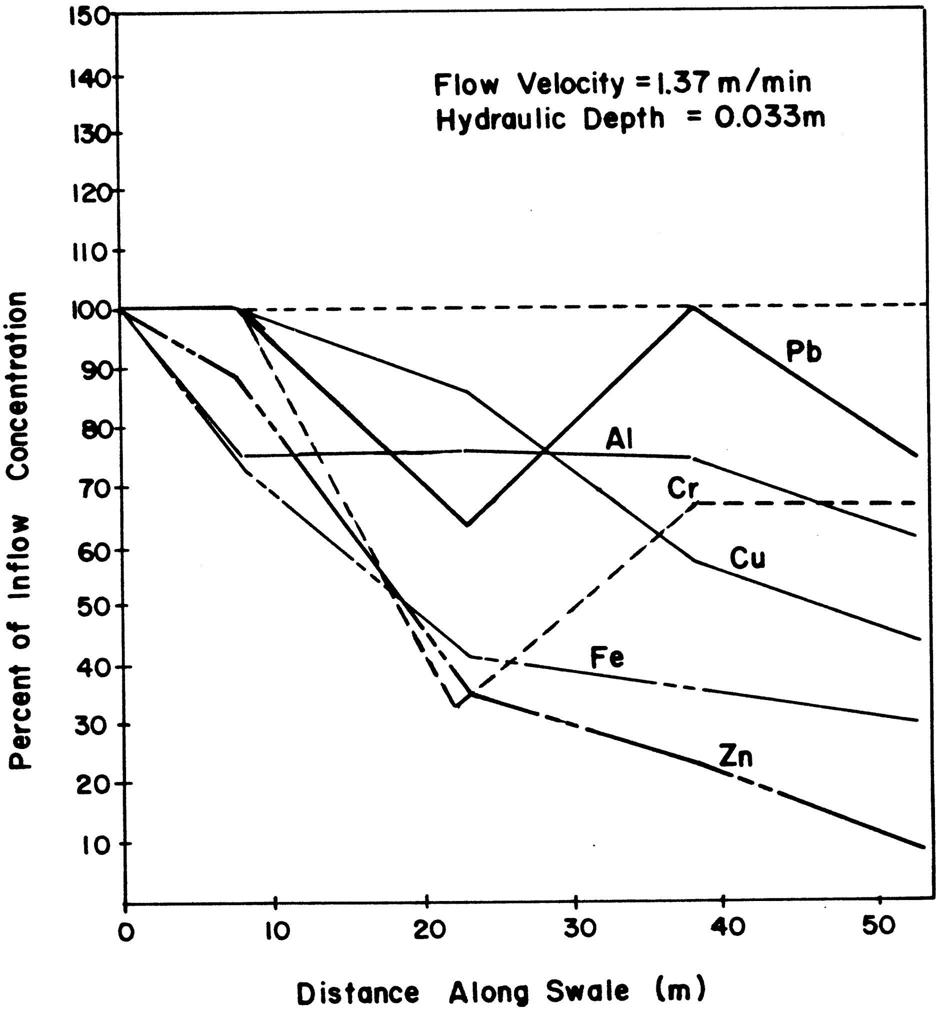

Figure 3-2: CHANGES IN CONCENTRATION OF VARIOUS HEAVY METALS DURING FLOW THROUGH A GRASSY SWALE AT MAITLAND ON 2/7/83.

TABLE 3-7

Average Removal Efficiencies for Heavy Metals in Maitland Swale Study Conducted 2/21/83

Heavy Metal	Distance Along Swale (m) 0	7.5	23	38
ZINC:				
Avg. Conc. (ug/l)	18	6	5	2
Avg. Conc. (Mole/l)	2.8×10^{-7}	0.92×10^{-7}	0.76×10^{-7}	0.31×10^{-7}
% Remaining	100	33	28	11
COPPER:				
Avg. Conc. (ug/l)	2	2	4	4
Avg. Conc. (Mole/l)	0.31×10^{-7}	0.31×10^{-7}	0.62×10^{-7}	0.62×10^{-7}
% Remaining	100	100	200	200
ALUMINUM:				
Avg. Conc. (ug/l)	37	8	8	10
Avg. Conc. (Mole/l)	14.0×10^{-7}	3.0×10^{-7}	3.0×10^{-7}	3.7×10^{-7}
% Remaining	100	22	22	27
IRON:				
Avg. Conc. (ug/l)	277	86	84	83
Avg. Conc. (Mole/l)	50.0×10^{-7}	15.0×10^{-7}	15.0×10^{-7}	15.0×10^{-7}
% Remaining	100	31	30	30
LEAD:				
Avg. Conc. (ug/l)	6	4	5	6
Avg. Conc. (Mole/l)	0.29×10^{-7}	0.19×10^{-7}	0.24×10^{-7}	0.29×10^{-7}
% Remaining	100	67	83	100

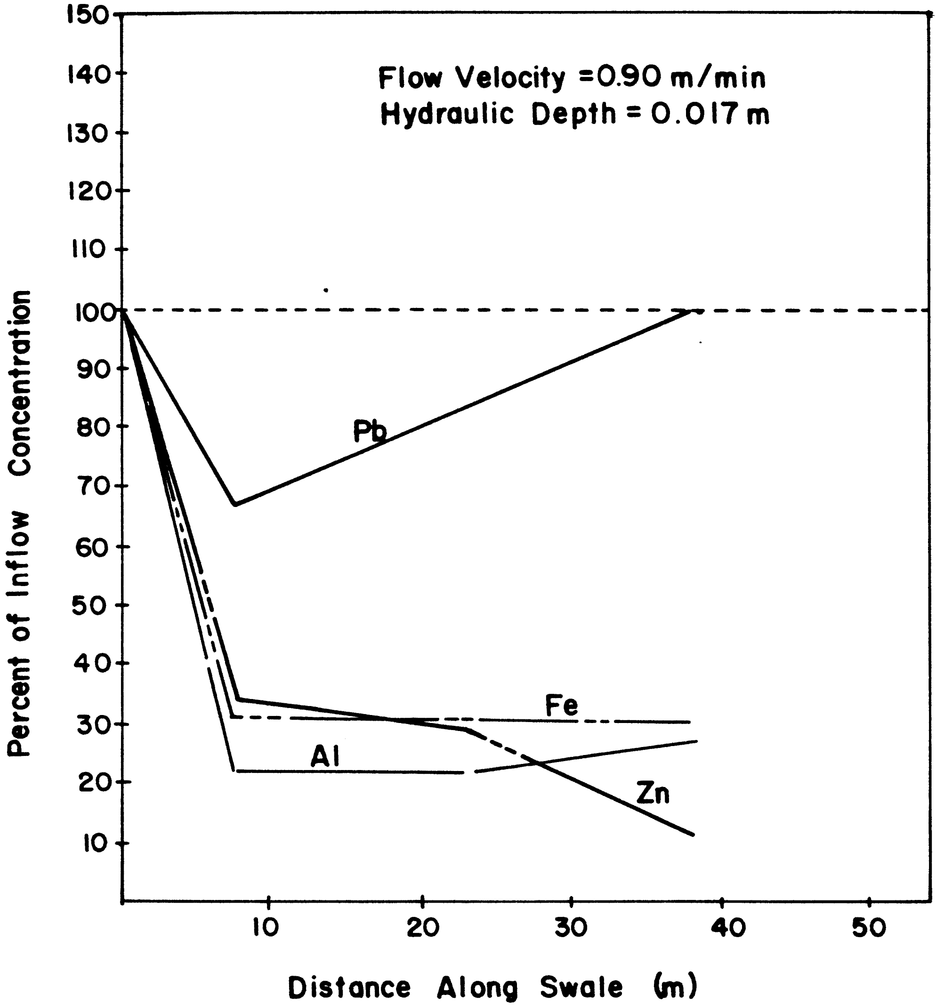

Figure 3-3: CHANGES IN CONCENTRATION OF VARIOUS HEAVY METALS DURING FLOW THROUGH A GRASSY SWALE AT MAITLAND ON 2/21/83.

the swale. At these concentrations accurate analysis becomes difficult, and the significance of measurement errors increases considerably. As a result, these metals are not indicated in Figure 3-3.

EPCOT Center

Two additional swale investigations were conducted in a sandy, newly constructed, swale area near EPCOT Center. The area was originally a wetland forest area before construction, and as a result natural groundwater in the area is presumably high in organic content as evidenced by the dark color of the water. Normal seepage flow through the swale at the start of both experiments was also highly colored and although no organic speciation determinations were performed, this background flow was presumed to be rich in humic organic materials. The first experiment, conducted on March 23, 1983, was performed before a grass cover had formed on the swale to investigate the effect of bare soil in removing dissolved heavy metals. Grass cover at this time was estimated at approximately 20 percent. The second experiment, conducted on May 16, 1983, was performed when the same swale area had become approximately 80 percent covered with vegetation to determine if the presence of plants would affect heavy metal removal.

A summary of removal efficiencies for the investigation conducted on March 23, 1983 is given in Table 3-8 and shown in Figure 3-4. Average flow velocity for this experiment was 2.44 m/min. The average pH value of the runoff flow was slightly less than 6.0. As seen in the previous experiments at the Maitland site, zinc was removed from solution rapidly with a total removal of 61 percent over 170 m. It should be noted, however, that even though the swale length was tripled

TABLE 3-8

Average Removal Efficiencies for Heavy Metals in EPCOT Swale Study Conducted 3/23/83

Heavy Metal	Distance Along Swale (m)					
	0	30	60	90	120	
CADMIUM:						
	11	9	8	8	8	7
	0.98 x 10^{-7}	0.80 x 10^{-7}	0.71 x 10^{-7}	0.71 x 10^{-7}	0.17 x 10^{-7}	0.62 x 10^{-7}
	100	82	73	73	73	64
ZINC:						
Avg. Conc. (ug/l)	256	188	158	140	122	99
Avg. Conc. (Mole/l)	38.0 x 10^{-7}	28.0 x 10^{-7}	24.0 x 10^{-7}	21.0 x 10^{-7}	18.0 x 10^{-7}	15.0 x 10^{-7}
% Remaining	100	73	62	55	48	39
COPPER:						
Avg. Conc. (ug/l)	22	25	22	22	20	19
Avg. Conc. (Mole/l)	3.5 x 10^{-7}	4.0 x 10^{-7}	3.5 x 10^{-7}	3.5 x 10^{-7}	3.2 x 10^{-7}	3.0 x 10^{-7}
% Remaining	100	114	100	100	91	86
IRON:						
Avg. Conc. (ug/l)	489	454	403	410	458	518
Avg. Conc. (Mole/l)	88.0 x 10^{-7}	82.0 x 10^{-7}	73.0 x 10^{-7}	74.0 x 10^{-7}	82.0 x 10^{-7}	90.0 x 10^{-7}
% Remaining	100	93	82	84	94	106
LEAD:						
Avg. Conc. (ug/l)	91	46	53	44	44	32
Avg. Conc. (Mole/l)	4.4 x 10^{-7}	2.2 x 10^{-7}	2.5 x 10^{-7}	2.1 x 10^{-7}	2.1 x 10^{-7}	1.5 x 10^{-7}
% Remaining	100	51	58	48	48	35
NICKEL:						
Avg. Conc. (ug/l)	112	92	80	74	62	52
Avg. Conc. (Mole/l)	19.0 x 10^{-7}	16.0 x 10^{-7}	14.0 x 10^{-7}	13.0 x 10^{-7}	11.0 x 10^{-7}	88.0 x 10^{-7}
% Remaining	100	82	71	66	55	46
CHROMIUM:						
Avg. Conc. (ug/l)	11	8	82	8	10	8
Avg. Conc. (Mole/l)	2.1 x 10^{-7}	1.5 x 10^{-7}	1.5 x 10^{-7}	1.5 x 10^{-7}	1.9 x 10^{-7}	1.5 x 10^{-7}
% Remaining	100	73	73	73	91	73

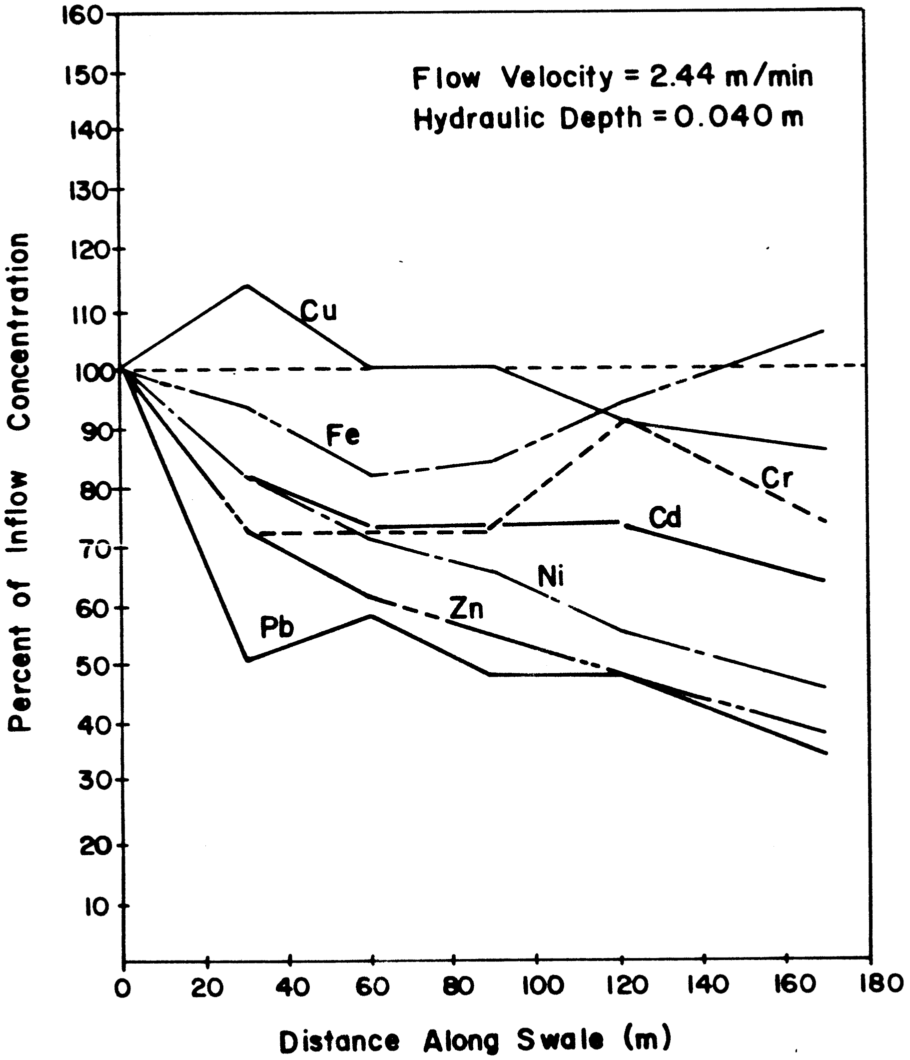

Figure 3-4: CHANGES IN CONCENTRATION OF VARIOUS HEAVY METALS DURING FLOW THROUGH AN EARTHEN SWALE AT EPCOT ON 3/23/83.

in the EPCOT experiments and the flow rates were calculated to be approximately equal to those measured during the first Maitland experiment, the removal of dissolved zinc was less than at the Maitland site. Removal of dissolved nickel and lead closely followed that observed for zinc. Approximately 54 percent of the dissolved nickel and 65 percent of the dissolved lead were removed in the 170 m test area. It appears that cadmium was also removed to a small degree with 64 percent still remaining at the end. Removal of copper, iron, and chromium by passage through the earthen swale seemed to be minimal, and as seen in Figure 3-4, concentrations of iron may have actually increased after a distance of 90 m.

A summary of removal efficiencies for the second EPCOT investigation conducted on May 16, 1983, after the establishment of a grass cover, is listed in Table 3-9 and shown in Figure 3-5. As seen in all previous experiments, dissolved zinc was removed from solution at a rapid rate. A total removal of 65 percent was calculated which is quite close to the removal of 61 percent obtained in the earthen channel. Dissolved lead was also removed from solution particularly at the 120 m and 170 m sites with only 59 percent remaining at the end. Removal of nickel was also found to occur and, as seen in Figure 3-5, exhibited removal efficiencies approximately midway between the removal of zinc and that of lead. Concentrations of iron, copper and chromium, however, exhibited initial increases and remained near or above their original concentrations throughout the test area. It should be noted that in the first EPCOT experiment, conducted in the earthen channel at the same site, significant removals of both copper and chromium were

TABLE 3-9

Average Removal Efficiencies for Heavy Metals in EPCOT Swale Study Conducted 5/16/83

Heavy Metal	Distance Along Swale (m)					
	0	30	60	90	120	
CADMIUM:						
	2	2	1	1	1	1
	0.18×10^{-7}	0.18×10^{-7}	0.089×10^{-7}	0.089×10^{-7}	0.089×10^{-7}	0.089×10^{-7}
	100	100	50	50	50	50
ZINC:						
Avg. Conc. (ug/l)	23	19	15	14	11	8
Avg. Conc. (Mole/l)	3.5×10^{-7}	2.9×10^{-7}	2.3×10^{-7}	2.1×10^{-7}	1.7×10^{-7}	1.1×10^{-7}
% Remaining	100	83	65	61	48	35
COPPER:						
Avg. Conc. (ug/l)	29	34	38	36	36	28
Avg. Conc. (Mole/l)	4.6×10^{-7}	5.1×10^{-7}	5.7×10^{-7}	5.4×10^{-7}	5.4×10^{-7}	4.0×10^{-7}
% Remaining	100	117	131	124	124	97
IRON:						
Avg. Conc. (ug/l)	91	125	127	112	94	114
Avg. Conc. (Mole/l)	16.0×10^{-7}	23.0×10^{-7}	23.0×10^{-7}	20.0×10^{-7}	17.0×10^{-7}	21.0×10^{-7}
% Remaining	100	137	140	123	103	125
LEAD:						
Avg. Conc. (ug/l)	44	40	42	39	32	26
Avg. Conc. (Mole/l)	2.1×10^{-7}	1.9×10^{-7}	2.0×10^{-7}	1.8×10^{-7}	1.5×10^{-7}	1.3×10^{-7}
% Remaining	100	91	95	89	73	59
NICKEL:						
Avg. Conc. (ug/l)	28	26	21	20	17	16
Avg. Conc. (Mole/l)	4.8×10^{-7}	4.4×10^{-7}	3.6×10^{-7}	3.4×10^{-7}	2.9×10^{-7}	0.17×10^{-7}
% Remaining	100	93	75	71	61	57
CHROMIUM:						
Avg. Conc. (ug/l)	8	10	11	11	10	9
Avg. Conc. (Mole/l)	0.15×10^{-7}	0.19×10^{-7}	0.21×10^{-7}	0.21×10^{-7}	0.19×10^{-7}	0.17×10^{-7}
% Remaining	100	125	138	138	125	113

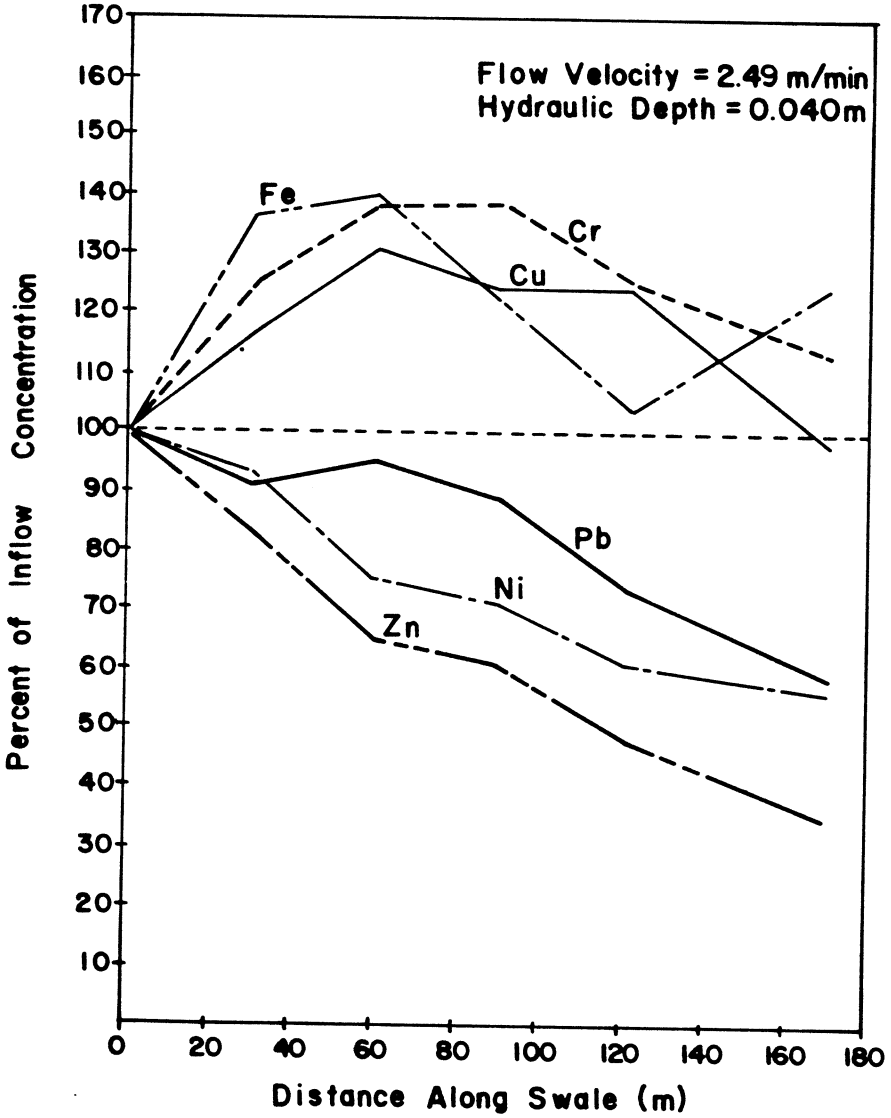

Figure 3-5: CHANGES IN CONCENTRATION OF VARIOUS HEAVY METALS DURING FLOW THROUGH A GRASSY SWALE AT EPCOT ON 5/16/83.

found. Concentrations of cadmium throughout the experiment were too low to form accurate conclusions.

Stormwater Sampling Program

Stormwater from both highway areas and swale areas receiving highway runoff were collected from six different stations surrounding Maitland Interchange, as shown in Figure 2-2, (page 10), for a period of approximately eight months during 1982-1983. Stormwater was collected from a total of seventeen storm events, although a collectable amount of runoff did not occur at some locations on specific dates. A summary of average concentrations of both total and dissolved heavy metals is presented in Table 3-10.

In order to make a comparison and analysis of the various collection locations it was necessary to group the data collectively into two categories according to the source of stormwater runoff: (1) locations 1, 5, and 6 grouped together as "swale flow." With the exception of locations 4 and 6, it is not possible to certify the highway location which contributed to the swale runoff samples. Grouping of the stormwater data into "highway runoff" and "swale flow" does allow a comparison between average highway stormwater quality and average swale flow quality. A comparison of average concentrations for highway and swale areas is given in Table 3-11. It can be seen that the concentrations of both total and dissolved forms of every heavy metal analyzed are lower in the swale flow than in the highway flow. It also appears that the percentage of dissolved metal versus suspended forms increases in the swale flow indicating that some of the particulate metal forms have been removed by passage through the swale. As reported in previous swale experiments, zinc concentrations were decreased to the

TABLE 3-10

Summary of Average Heavy Metal Concentrations in Stormwater Runoff Collected During 1982-1983 at Maitland Interchange (ug/l)

Heavy Metal	#1 Grassy Swale Receiving S. W.* from I-4	#2 Direct Runoff from Maitland Blvd.	#3 Direct Road and Bridge Runoff to Pond	#4 Direct Runoff from I-4 Exit Ramp	#5 Swale Next to I-4 Entrance Ramp	#6 Swale Receiving S.W.* From I-4
CADMIUM:						
Diss.	0.8	1.4	1.1	0.8	0.6	0.9
Total	1.0	1.6	1.9	0.9	0.6	1.0
ZINC:						
Diss.	11	192	50	42	29	14
Total	37	334	347	154	37	17
COPPER:						
Diss.	16	18	32	31	23	24
Total	23	30	60	41	29	26
ALUMINUM:						
Diss.	70	68	73	58	57	46
Total	314	1142	1411	512	374	183
IRON:						
Diss.	30	87	48	27	35	26
Total	717	948	1176	454	180	98
LEAD:						
Diss.	25	48	43	19	16	16
Total	86	356	723	197	31	22
NICKEL:						
Diss.	1.8	5.3	3.2	1.9	2.0	1.8
Total	3.6	7.5	28	24	2.3	2.5
CHROMIUM:						
Diss.	4.2	3.6	3.3	2.8	2.5	2.0
Total	9.0	7.5	10	6.7	6.0	2.9
AVERAGE pH	5.49	5.71	5.89	5.67	5.42	5.60
NUMBER OF SAMPLES:	5	14	16	17	4	16

*S.W. = stormwater

TABLE 3-11

Comparison of Highway Runoff and Swale Flow at Maitland Interchange During 1982-1983

Heavy Metal	Highway Runoff Avg. Conc. (ug/l)	Highway Runoff Percent Dissolved(%)	Swale Flow Avg. Conc. (ug/l)	Swale Flow Percent Dissolved(%)	Change in Concentration Through Swale %	T-Test Probability for Unequal Means
CADMIUM:						
Diss.	1.1	79	0.9	90	-18	81.2
Total	1.4		1.0		-29	98.2
ZINC:						
Diss.	89	35	16	64	-82	99.9
Total	255		25		-90	99.9
COPPER:						
Diss.	27	61	22	85	-19	42.9
Total	44		26		-41	90.1
ALUMINUM:						
Diss.	66	7	53	23	-20	86.5
Total	970		233		-76	99.9
IRON:						
Diss.	52	6	29	12	-44	87.3
Total	830		240		-71	99.7
LEAD:						
Diss.	36	9	18	50	-50	99.7
Total	417		36		-91	99.9
NICKEL:						
Diss.	3.4	17	1.8	75	-47	97.2
Total	21		2.4		-88	78.2
CHROMIUM:						
Diss.	3.2	39	2.8	61	-13	54.3
Total	8.2		4.6		-44	99.3
AVERAGE pH	5.75		5.54		+62	99.2
NUMBER OF SAMPLES:	47		25			

greatest degree by passage through a swale area, with an average of 82 percent reduction in dissolved and 90 percent reduction in total species. Concentrations of total lead and nickel were reduced approximately 90 percent with a 50 percent reduction in dissolved forms. Iron and aluminum were removed less efficiently with a 75 percent reduction in total forms and a 20-40 percent removal of dissolved species. Cadmium, copper, and chromium were removed to the smallest degree with 30-40 percent removal of total forms and only a small removal of dissolved species.

A more direct analysis of the effectiveness of swales in removing heavy metals can be obtained by comparing locations 4 and 6. Location 4 received direct stormwater runoff from an exit ramp to I-4, and location 6 received the runoff from location 4 after travel through approximately 60 m of grassy swale. A comparison of the runoff collected at these two locations in both total and dissolved forms of all metals tested was achieved by passage through the swale. As seen in the previous comparison between highway runoff and swale flow, the particulate fractions of the metal species were removed more effectively. Approximately 90 percent of the total zinc, lead, and nickel was removed over this distance, 78 percent of the total iron, approximately 60 percent of the total aluminum and chromium, but only 37 percent of the total copper. Dissolved species of most metals, with the exception of zinc, were removed to a much lesser degree. Copper, aluminum, iron, lead, nickel and chromium exhibited removals of approximately 25 percent or less with iron being reduced by only 4 percent. It should be noted that with the exception of cadmium, the average reductions in total metal concentrations which occurred by passage through the swale area between Stations 6 and 4 closely match the differences between highway runoff and swale flow listed in Table 3-12.

DISCUSSION OF RESULTS

Maitland Experiments

Upon review of the data for the Maitland experiments in which heavy metals were measured it becomes evident that not all metals were removed at the same rate. Zinc and iron were consistently removed from solution more rapidly than copper, aluminum, lead, and chromium. Obviously some factors must have been present which affected zinc and iron to a larger degree than other metals. One of the first possible explanations of this result is to examine the solubility of each of these metals in the test water. The larger removals found for zinc and iron could possibly be explained by determining that the initial concentrations of each of these metals had exceeded the allowable solubility product. To test this hypothesis a survey was conducted of known inorganic equilibria for iron, zinc, copper, lead, cadmium, and nickel, and reactions which were thought to be important under these particular conditions were listed. Solubility diagrams including all important soluble and insoluble species were then constructed for each of the six heavy metals. These diagrams are reproduced in Figures 3-6 to 3-11.

A review of soluble concentrations measured during the swale study for iron, zinc, copper, lead, cadmium, and nickel reveals that only iron was initially present in concentrations sufficiently high to cause formation of a precipitate at the experimentally measured pH of approximately 6.5. As seen in Figure 3-6, the initial concentration of iron,

TABLE 3-12

Comparison of Runoff Collected at Stations 4 and 6 at Maitland Interchange During 1982-1983

	Station 4 Highway Runoff		Station 6 Swale Flow		Change in Concentration Through Swale	T-Test
Heavy Metal	Avg. Conc. (ug/l)	Percent Dissolved(%)	Avg. Conc. (ug/l)	Percent Dissolved(%)	%	Probability for Unequal Means
CADMIUM:						
Diss.	0.8	89	0.9	90	+13	79.8
Total	0.9		1.0		+11	64.2
ZINC:						
Diss.	42	27	14	82		99.7
Total	154		17		-67	99.1
					-89	
COPPER:						
Diss.	31	76	24	92		36.2
Total	41		26		-23	60.6
					-37	
ALUMINUM:						
Diss.	58	11	46	25		72.5
Total	512		183		-21	98.5
					-64	
IRON:						
Diss.	27	6	26	27		1.8
Total	454		98		-4	87.9
					-78	
LEAD:						
Diss.	19	10	16	73		57.8
Total	197		22		-16	81.4
					-89	
NICKEL:						
Diss.	1.9	8	1.8	72		30.6
Total	24		2.5		-5	68.1
					-90	
CHROMIUM:						
Diss.	2.8	42	2.0	69		54.5
Total	6.7		2.9		-29	99.9
					-57	
AVERAGE pH	5.67					
			560			68.7
NUMBER OF SAMPLES:	17				+17	
			16			

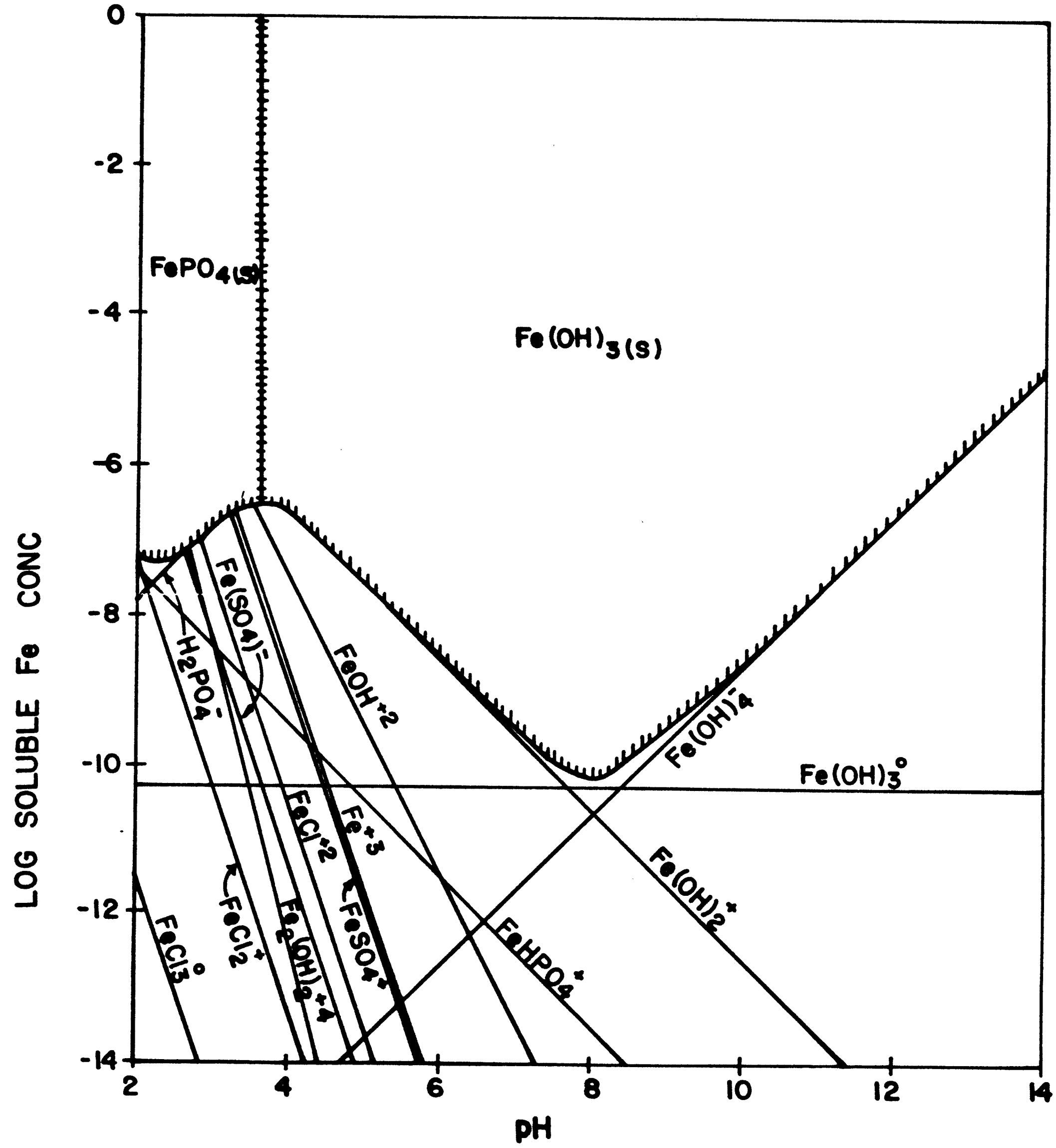

Figure 3-6: SOLUBILITY OF INORGANIC IRON IONS IN NATURAL SURFACE WATERS

(Assumptions: $C_T = 5 \times 10^{-4}$M; $[Cl^-] = 5 \times 10^{-4}$M; and $[SO_4^{-2}] = 1 \times 10^{-4}$M; and $[PO_4^{-3}] = 1 \times 10^{-5}$M).

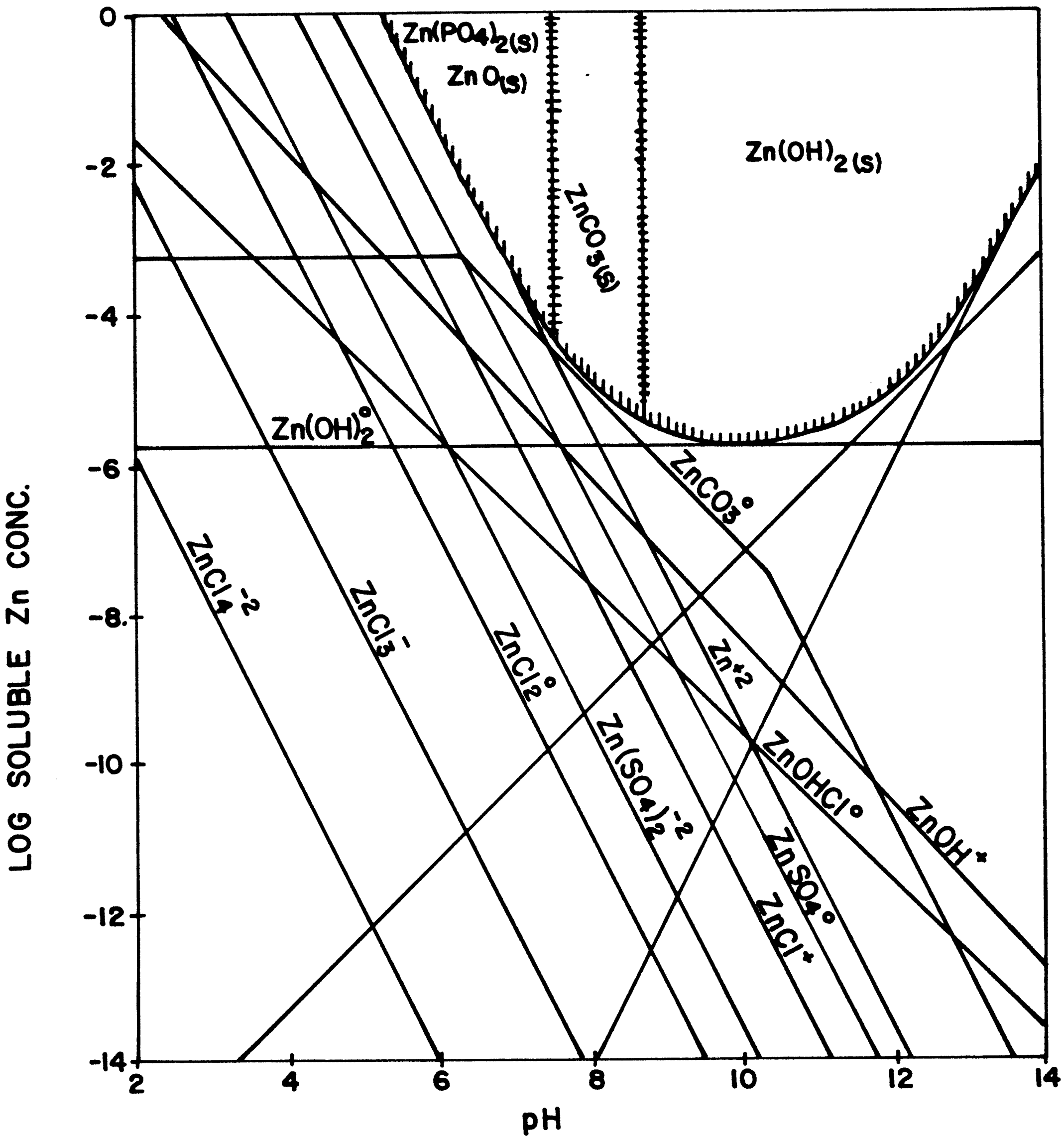

Figure 3-7: SOLUBILITY OF INORGANIC ZINC IONS IN NATURAL SURFACE WATERS

(Assumptions: $C_T = 5 \times 10^{-4}$ M; $[Cl^-] = 5 \times 10^{-4}$ M; $[SO_4^{-2}] = 1 \times 10^{-4}$ M; and $[PO_4^{-3}] = 1 \times 10^{-5}$ M).

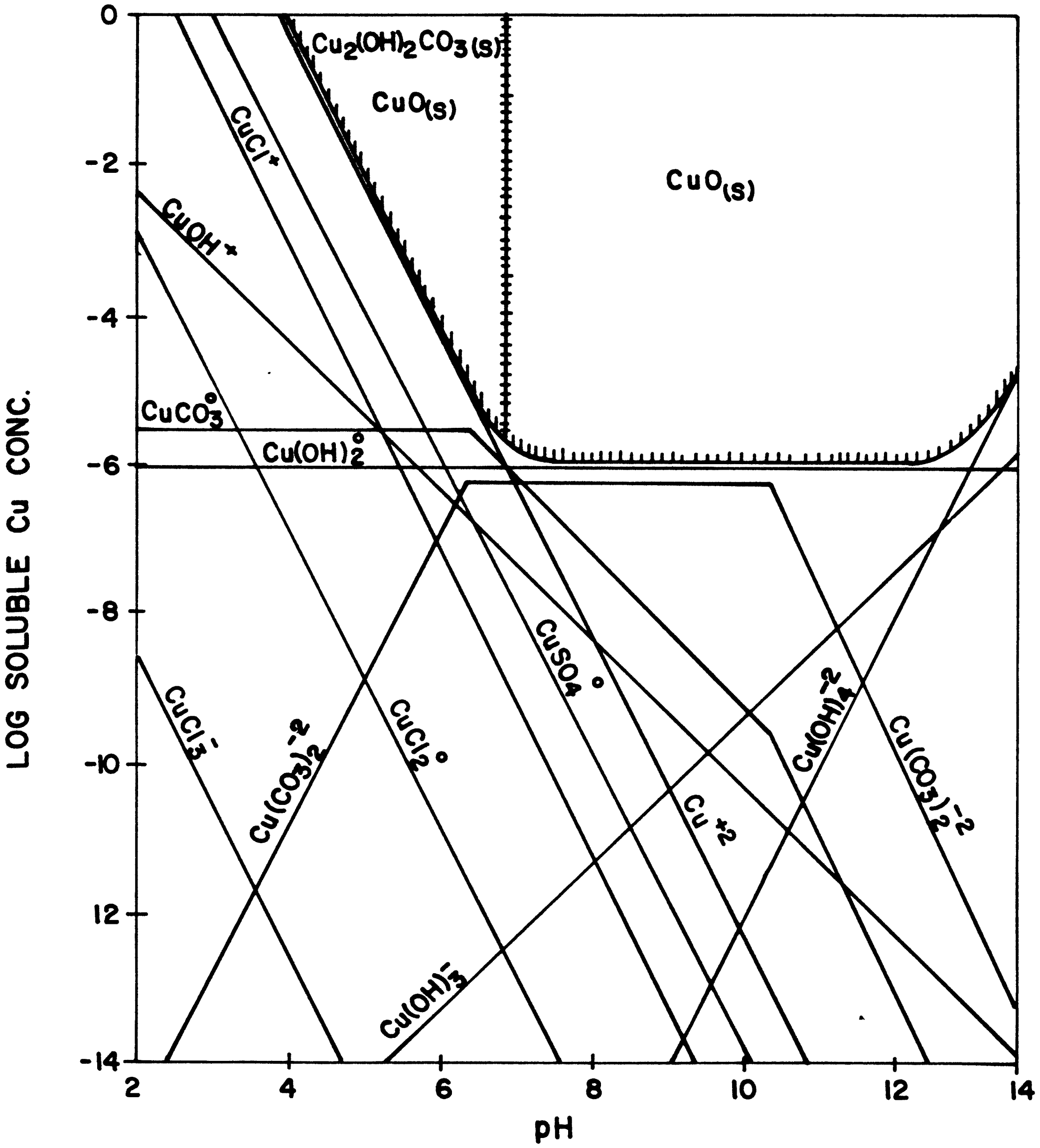

Figure 3-8: SOLUBILITY OF INORGANIC COPPER IONS IN NATURAL SURFACE WATERS

(Assumptions: $C_T = 5 \times 10^{-4}$M ; $[Cl^-] = 5 \times 10^{-4}$M ; $[SO_4^{-2}] = 1 \times 10^{-4}$M; and $[PO_4^{-3}] = 1 \times 10^{-5}$M).

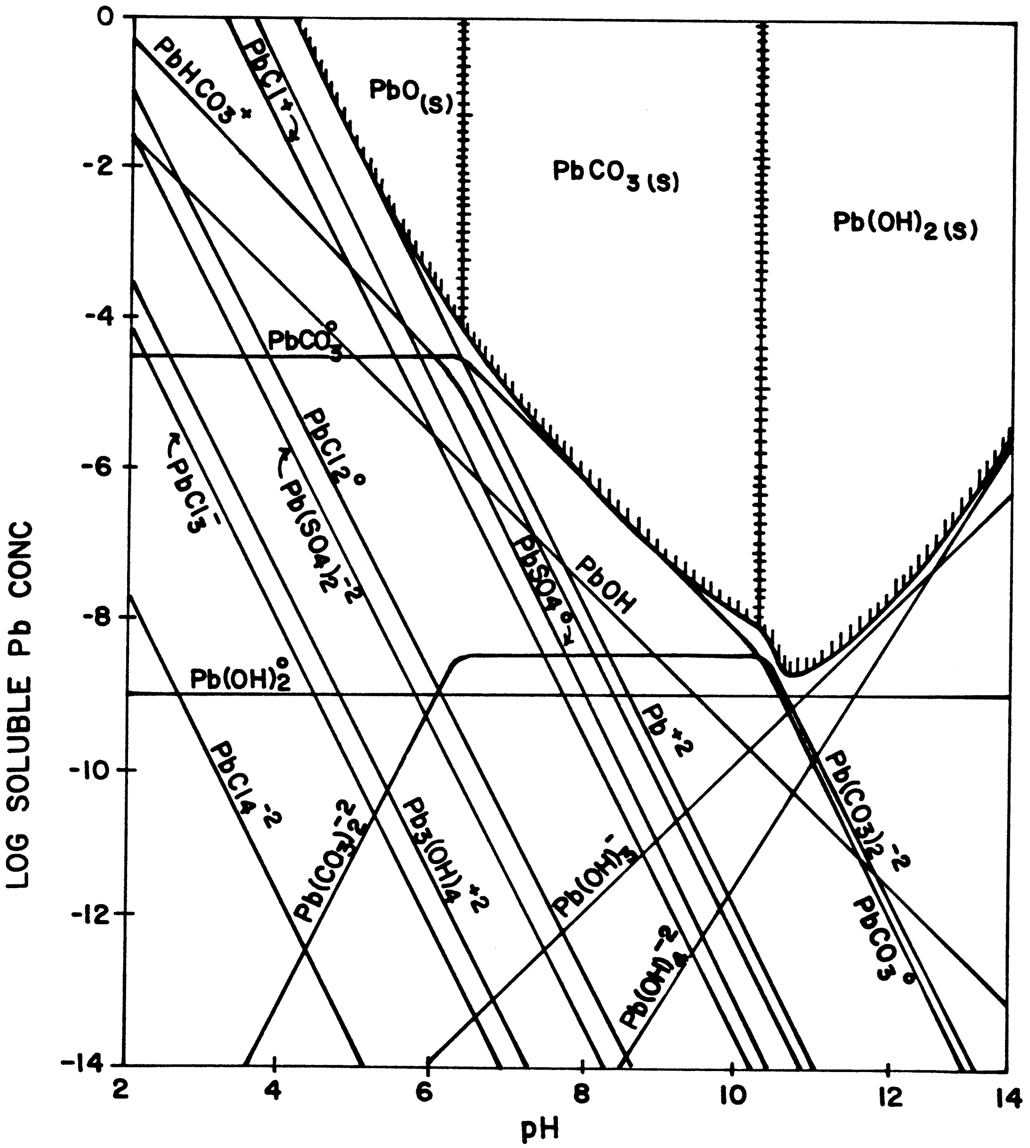

Figure 3-9: SOLUBILITY OF INORGANIC LEAD IONS IN NATURAL SURFACE WATERS

(Assumptions: $C_T = 5 \times 10^{-4}$M ; [Cl^-] = 5×10^{-4}M ; and [SO_4^{-2}] = 1×10^{-4}M; and [PO_4^{-3}] = 1×10^{-5} M).

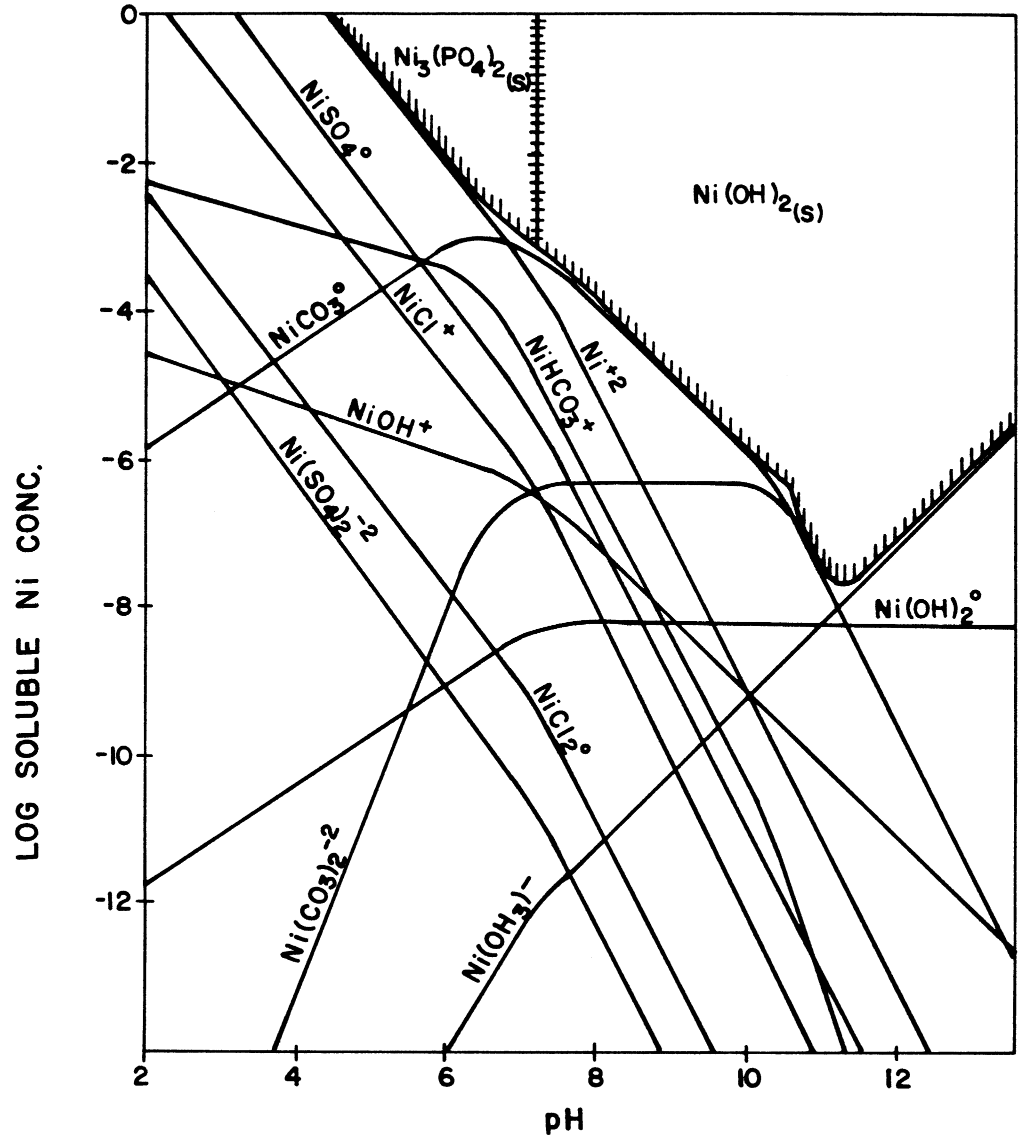

Figure 3-10:SOLUBILITY OF INORGANIC NICKEL IONS IN NATURAL SURFACE WATERS

(Assumptions: $C_T = 5 \times 10^{-4}$ M ; $[Cl^-] = 5 \times 10^{-4}$ M; $[SO_4^{-2}] = 1 \times 10^{-4}$ M; and $[PO_4^{-3}] = 1 \times 10^{-5}$ M).

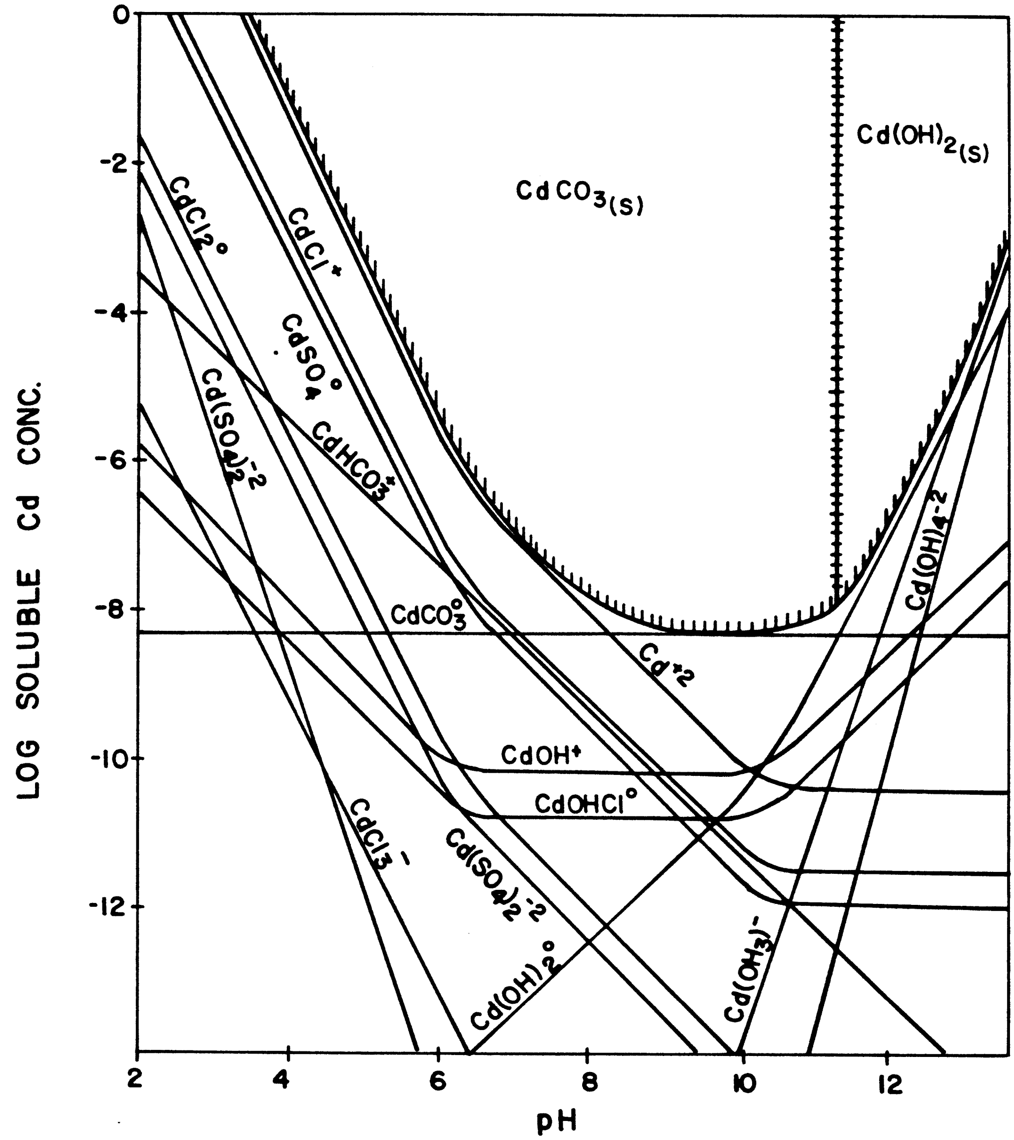

Figure 3-11: SOLUBILITY OF INORGANIC CADMIUM IONS IN NATURAL SURFACE WATERS

(Assumptions: $C_T = 5 \times 10^{-4}$ M; $[Cl^-] = 5 \times 10^{-4}$ M; $[SO_4^{-2}] = 1 \times 10^{-4}$ M; and $[PO_4^{-3}] = 1 \times 10^{-5}$ M).

6×10^{-7} M, was well into the solid portion of the iron solubility diagram. It is therefore probable to assume that the iron removal was, at least in part, due to formation and settling of a precipitate. However, zinc, which was present in concentrations in the range of 10^{-7} M and 10^{-8} M, was clearly in the soluble region of the diagram, Figure 3-7, yet excellent removal was obtained. Another factor other than precipitation must have affected heavy metal removal in these swale investigations.

It is a well known fact that negatively charged particles in the soil and organic litter layer have the ability to adsorb cations from solution in an exchange reaction. If this phenomenon were to be responsible for heavy metal removal in swales, then the charge and size of the cationic species would determine to a large degree the magnitude of removal observed. This relatively simple assumption is complicated by the fact that both inorganic as well as organic species of metals can exist in solution. The net effective charge of organo-metal species is difficult to determine since any charge, if present, is dispersed over a relatively large surface area. However, investigations by Wiseman (1983) using Anodic Stripping Voltametric (ASV) analysis technique have shown that heavy metal complexes in actual stormwater collected at the Maitland Interchange as well as in the retention pond water itself, which was used as a water source for these experiments, are almost totally inorganic in nature. The assumption will, therefore, be made that the chemistry of removal of heavy metals at this site is based on inorganic reactions.

Assuming that inorganic species are the major heavy metal forms present, differences in the dominant species for a particular metal should help to explain the variability in removals. As seen in Figure 3-7, the dominant species of zinc present at the experimentally measured pH of approximately 6.5 is the Zn^{+2} ion. Since this ion is relatively small, the net effective charge should be relatively large, and zinc removal as predicted by the assumed model should be possible. This explanation corresponds well with the observed results. If the adsorption model is valid, then an increase in removal should be observed for zinc at lower velocities of flow since contact time is increased. If removals are compared in the three experiments at a distance of 38 meters, since the last Maitland experiment did not reach the 53 m point, it is evident that removal efficiencies increase from 55 to 77 to 89 percent as the flow rate is decreased. It appears, therefore, that the assumption of removal by adsorption of Zn^{+2} ions onto soil and organic particles is appropriate for zinc removal at this site.

As discussed previously, removal of iron in these experiments can be attributed at least partially to the formation of a solid $Fe(OH)_3$ precipitate. However, as indicated in Figure 3-6, the major soluble species of iron at pH of 6.5 is the $Fe(OH)_2^+$ complex. Although this complex is physically larger and possesses a lower and more diffuse effective charge than Fe^{+3}, it still exhibits a certain amount of cationic character, and it is possible this ion could also be removed by adsorption. Removal of iron, therefore, can be explained by both precipitation as well as adsorption, although the relative importance of each cannot be determined from these investigations.

Unlike zinc and iron, copper was not removed from solution at a significant rate. Maximum removal of dissolved copper observed during these investigations was approximately 50 percent. An examination of Figure 3-8 will show that at a pH of approximately 6.5, the major inorganic copper ions in solution are Cu^{++} as well as $CuCO_3^{o}$. Assuming the adsorption model to be valid, then only Cu^{++} is available for adsorption while $CuCO_3^{o}$, which has a net charge of zero, will not be available. This explains the lower removal efficiency observed for copper. If this assumption is correct, then copper could be more easily removed in waters with lower pH values where Cu^{++} is the major species.

As seen in figures 3-1 to 3-3, average concentrations of lead measured at various distances during swale flow in each Maitland experiment appear to be somewhat erratic. A great deal of this inconsistency can be explained by experimental error since measurement of lead at the low concentrations found in these experiments is subject to extreme variability. However, it appears that little or no removal of lead was achieved. Examination of Figure 3-9 reveals that at a pH of 6.5 the dominant lead species would be $PbCO_3^{o}$ and Pb^{+2}. $PbCO_3^{o}$ species carries no charge and would not be expected to be affected by adsorption processes. Adsorption of Pb^{+2} may occur at lower pH values than experimental pH. The adsorption model appears to hold for lead also.

As stated previously, the concentrations of cadmium and nickel measured during these investigations were quite low and in most cases approached the limits of detection for our measurement technique. Therefore, no reliable data is available from which the behavior of these two metals in a swale environment can be analyzed. However, assuming that these two elements would follow the general trends observed for other metals, some predictions as to their expected behavior can be made. A solubility diagram for nickel is presented in Figure 3-10 and for cadmium in Figure 3-11. As seen in Table 3-11, the average dissolved concentrations of these two metals in highway runoff are approximately 2 ug/l (1.8×10^{-8} moles/l) for cadmium and 4 ug/l (6.8×10^{-8}) for nickel. At the above concentration of cadmium at a pH of approximately 6.5, the major soluble cadmium species would be the Cd^{+2} ion. If this ion exhibits behavior similar to the Zn^{+2} ion in the absence of significant organic concentrations then removal of cadmium would be expected to occur. At higher concentrations of cadmium at the same pH, the soluble concentrations would begin to approach the solid-soluble boundary and precipitation as $CdCO_{3(s)}$ may also occur. In any case dissolved cadmium can be expected to be removed from highway runoff by passage through a swale. As seen in Figure 3-10, the dominant dissolved forms of nickel at a pH of 6.5 would be both the Ni^{+2} and $NiCO_3^{o}$. This situation is very similar to that observed in the case of copper. Removal of nickel would be expected to occur but not to the extent exhibited by zinc. It is expected that the removal of nickel should increase at lower pH values where Ni^{+2} becomes the dominant species.

It can be concluded, therefore, that heavy metal removal in this series of investigations was predominantly an adsorption phenomenon. The ability of a heavy metal to be removed was found to be closely related to the dominant inorganic species present.

EPCOT Site

As stated previously, the presence of organic acids was indicated at this location by the dark stain which characterized the normal seepage flow into the swale. Although no specific organic determinations were performed, this background flow was presumed to be rich in humic and fulvic organic compounds. Therefore, at this site organic as well as inorganic complexing must be considered. The measured pH for the first EPCOT experiment was approximately 6.0 while the pH for the second investigation was approximately 7.0.

As seen in previous experiments at the Maitland site, zinc was removed from solution rapidly, although not to the extent as was previously observed. No significant organic complexes are known to form between humic or fulvic acids and zinc at a neutral pH. Therefore, according to Figure 3-7, the dominant zinc species will still be Zn^{+2} for a pH value less than 7.5. Removal of zinc is still presumed to be predominantly an adsorption phenomenon, yet a lower removal was noted in the EPCOT experiments in spite of the fact that the calculated velocity for each EPCOT experiment was very close to the calculated velocity in the first Maitland experiment.

Contrary to the results observed in previous experiments at Maitland, significant removal of iron did not occur at EPCOT even though the molar concentration of approximately 10^{-6} exceeds the solubility product for iron at a pH of 6.0. It has been reported by numerous researchers that iron can form quite stable complexes with humic substances. Evidently the formation of these complexes increased the solubility of iron and no removal was possible. In the second EPCOT experiment, conducted on May 16, 1983 with a grass cover, the concentrations of dissolved iron actually increased by passing through the swale. Evidently iron which had been previously deposited in the swale was present in a loosely bound form, and was able to be removed by the force of the flow.

As observed in the previous Maitland experiments, copper was not removed to any significant amount and, in fact, may have exhibited increased solubility in the EPCOT experiments, particularly in the second experiment with the grass cover. Copper can form significant organic complexes with both fulvate and humate even at low organic concentrations. As seen in Figure 3-8, $Cu(OH)_2{}^o$ is the dominant inorganic copper species between 7.0 and 13.0 with Cu^{+2} becoming an important species at pH values less than 7.0. However, at pH values less than neutral, organic complexation becomes more significant. Copper, therefore, seems to exist in a number of organic and uncharged inorganic species over a wide range of pH values. Neither organic complexes of copper nor the inorganic forms are suitable for adsorption, and removal is not expected to occur. As noted previously for iron, copper concentrations also increased by passing through the grassed swale presumably due to mobilization or solubilization of previously deposited species.

It is quite interesting to note that, contrary to the results obtained at Maitland, lead was removed quite well in the EPCOT experiments. No significant organic complexes of lead are known to occur under the test conditions. Therefore, the removal mechanism must be based on a change in inorganic speciation. As seen in Figure 3-9, the speciation of lead changes quite rapidly at a pH of approximately 6.2 from $PbCO_3{}^o$ to PB^{+2}. Below a pH of 6.2, the Pb^{+2} ion quickly becomes dominant. Above a pH of 6.2, however, Pb^{+2} as well as $PbCO_3{}^o$ becomes dominant. In the first EPCOT experiment, the average laboratory measured pH was approximately 6.0, and the dominant lead species would be expected to be Pb^{+2}. If this is the case the removal of lead should follow closely the removal exhibited by zinc. As seen in Figure 3-4, the removal curves for both zinc and lead are very similar. Thus, it appears that the decrease in pH from 6.5 at the Maitland site to 6.0 at the EPCOT site placed a portion of the dissolved lead in a form suitable for adsorption, and removal efficiencies were increased. In the second EPCOT experiment the pH increased to approximately 7.0. At this pH the dominant lead species should be Pb^{+2} as well as $PbCO_3^o$. Since $PbCO_3^o$ is not expected to be removed to a large degree, the overall removal at this pH should be less. As seen in Figure 3-7, the change in concentration of lead during passage through the swale test area at this pH was approximately one-half that observed at a pH of 6.0.

Unlike the Maitland experiments in which concentrations of dissolved nickel were too low for accurate analyses, concentrations at EPCOT were greater and mechanisms of nickel removal can be discussed. As seen in Figure 3-10, the inorganic speciation of nickel is very similar to that observed for lead. Below a pH of approximately 6.5 the dominant species is Ni^{+2}. Between 6.5 and 7.5 both Ni^{+2} and $NiCO_3^o$ are important with $NiCO_3^o$ becoming dominant above 7.5. No significant organic complexes of nickel are known to occur in this pH range. In the first EPCOT experiment, conducted at a pH of approximately 6.0, the major form of nickel was Ni^{+2} and removal should be expected to occur readily. As seen in Figure 3-4, the removal of nickel closely matched that found for zinc and lead. In the second EPCOT experiment, both Ni^{+2} and $NiCO_3^o$ forms are important. The removal of nickel is expected to follow a pattern similar to that of lead with reduced removals. As seen in Figure 3-5, this assumption fits the observed data well.

Of the five swale experiments in which heavy metals were measured, the concentration of cadmium was sufficient to allow for an accurate analysis in only one experiment. As seen in Figure 3-4, a decrease in the concentration of cadmium was observed after passage through the swale area. Cadmium exists predominantly in the Cd^{+2} form at pH values less than 8.0 (Figure 3-11) and good removal should be expected to occur. However, a cadmium-fulvate complex begins to form at concentrations of fulvate greater than 10 mg/l. The presence of these soluble organic forms should decrease the total removal of cadmium somewhat. As seen in Figure 3-4, cadmium was removed from solution but not the extent that was seen by nickel, zinc and lead.

In summary, the addition of an organic complexing agent has caused significant changes in removal efficiencies for various heavy metals, especially copper, cadmium, and iron. Although the solubility of zinc has remained relatively unchanged, iron and copper have apparently formed into soluble organic complexes which have resulted in a loss of removal efficiencies. The removal of both nickel and lead increased in one experiment, presumably due to a slight decrease in pH to 6.0 and decreased in the other experiment by increasing the pH to 7.0. Cadmium was removed to a lesser extent by passage through the swale area presumably due to the presence of cadmium-fulvate complexes. Bare earth was found to be more efficient in removing heavy metals from solution than a vegetated swale. It appears that under certain conditions heavy metals such as iron, chromium, and copper, and in some cases lead, can be mobilized from the swale by erosion resulting from the force of the water flow.

The experiments conducted at both Maitland and EPCOT have served to demonstrate the possible dominant dissolved species of heavy metals under conditions experienced in stormwater management systems. Cadmium, iron and copper were shown to exhibit significant organic complexes with fulvic and humic acids, while the other metal ions remained predominantly in the inorganic state even at high concentrations of organic complexing agents. A summary of anticipated heavy metal species for both inorganic complexing and organic complexing conditions at various pH ranges is given in Table 3-13. The breakpoints in pH values were chosen to be 6.5 and 7.5 since several of the metals investigated exhibit changes in their speciation in this range.

Effect of Cover Crop on Dissolved Metal Removal

It is possible by using the data collected in the two EPCOT investigations to comment on the effect of a grass cover on removal of dissolved heavy metals. A comparison of Figure 3-4, which represents removals obtained on an earthen channel, with Figure 3-5 representing a grassed channel reveals that the ultimate removal of many metals was greater in the earthen condition than in a grassed condition. Some heavy metals, iron, chromium, and copper, actually increased in concentration by passage through the swale. This decrease in removal efficiency found in the second experiment occurred in spite of an increase in average pH from approximately 6.0 to 7.0. It is generally recognized that the rate of sorption processes increases with increases in pH, and as a result, the second experiment should have exhibited increased removal rather than a decrease in removal efficiency. Since the flow rates were similar in both EPCOT experiments, the variable

TABLE 3-13

Summary of Inorganic and Organic Speciation of Heavy Metals at Various pH Ranges in Natural Aquatic Systems

Heavy Metals	Inorganic Complexation			Inorganic + Organic Complexation		
	pH<6.5	6.5<Ph<7.5	pH>7.5	pH<6.5	6.5<pH<7.5	pH>7.5
Cd	Cd^{+2}	Cd^{+2}	Cd^{+2} $CdCO_3^0$ $Cd(OH)_2^0$	Cd^{+2} Cd-fulvate	Cd^{+2} Cd-fulvate	Cd^{+2} $CdCO_3^0$ $CdOH)_2^0$
Zn	Zn^{+2}	Zn^{+2} $ZnCO_3^0$	$ZnCO3^0$ $Zn(OH)_2^0$ $Zn(OH)_4^{-2}$	Zn^{+2}	Zn^{+2} $ZnCO_3^0$	$ZnCO_3^0$ $Zn(OH)_2^0$ $Zn(OH)^{-2}$
Fe	$Fe(OH)2^+$	$Fe(OH)_2^+$ $Fe(OH)_3^0$	$Fe(OH)_3^0$ $Fe(OH)_4^-$		numerous species	
Pb	Pb^{+2}	Pb^{+2} $PbCO_3^0$	$PbCO3^0$ $Pb(CO_3)_2^{-2}$ $Pb(OH)_3^-$ $Pb(OH)_4^{-2}$	Pb^{+2}	Pb^{+2} $PbCO_3^0$	$PbCO_3^0$ $Pb(CO_3)_2^{-2}$ $Pb(OH)_3^-$ $Pb(CO_3)_4^{-2}$
Cu	Cu^{+2}	Cu^{+2} $CuCO_3^0$ $Cu(OH)_2^0$	$Cu(OH)2^0$	Cu^{+2} Cu-fulvate Cu-humate	Cu^{+2} $CuCO_3^0$ $Cu(OH)_2^0$ Cu-fulvate	$Cu(OH_2^0$
Ni	Ni^{+2}	Ni^{+2} $NiCO_3^0$	$NiCO3^0$ $Ni(CO_3)_2^{-2}$ $Ni(OH)_3$	Ni^{+2}	Ni^{+2} $NiCO_3^0$	$NiCO_3^0$ $Ni(CO_3)_2^{-2}$ $Ni(OH)_3$

which appears to have been responsible for the differences in removal efficiencies is cover crop.

This theory, although somewhat surprising, fits precisely in line with the theory of heavy metal removal in swales. It is assumed that heavy metals are removed by adsorption onto exchange sites in the soil and on organic particles. The better removal efficiencies obtained on bare soil merely reflect the fact that more exchange sites are open and available than when a grass cover is present. The release of heavy metals into the swale flow is probably due to decomposition products of plant matter which had trapped heavy metals either by uptake or surface adsorption. It appears the bonding mechanism between heavy metals and plants is weaker than the mechanism between heavy metals and soil.

Relationship Between Removal Efficiency and Flow Velocity

The analysis of any possible relationship between the removal of dissolved metal species during flow through a swale and the flow velocity is more difficult than a similar analysis for nutrients and other ions. This complication is due to the fact that heavy metals do not exist as a single ionic species. As indicated previously, metals may exist as the free metal ion, an inorganic complex and in some cases as an organic complex. Each of these ionic forms possesses a different affinity for adsorption onto soil particles which is believed to be the major mechanism of heavy metal removal in swales. As a result, heavy metals such as copper, iron and cadmium, which may exist in the presence of organic complexing agents as uncharged or diffusely charged large organic complexes, possess little or no affinity for adsorption onto soil particles. In such a case the flow velocity would have little or no effect on the removal process since the metal species are not in a form suitable for removal.

If the inorganic complexation alone is considered, there are also numerous inorganic species which must be considered. As seen in Figures 3-6 through 3-11, as well in Table 3-13, each of the heavy metals examined in these experiments exists in several inorganic species in the pH range of 5-8 which is typical of highway runoff. As discussed previously each various form exhibits a unique affinity for adsorption. In addition many of the heavy metals change in species or exist in multiple species in the pH range of 6.5-7.5. It is possible that heavy metals entering a swale system as one inorganic species may change in dominant speciation several times during travel through the swale with each form possessing a different adsorption affinity. In these cases the dominating removal mechanism would be the pH and not the flow velocity.

Although the prediction of heavy metal removal is somewhat complex, some generalizations can be made. If a metal species is present in a form which is capable of adsorption onto soil particles and it remains in that form during travel through the swale, then the rate of adsorption and removal of metal ions would be expected to increase as the flow rate decreases due to increased contact time with the soil particles. This phenomenon was especially evident for zinc and iron in the Maitland experiments. Zinc is present as the Zn^{+2} ion and iron is present as $Fe(OH)_2^+$ at pH values less than 7.5 and presumably did not change in speciation during travel through the swale. Both zinc and iron exhibited strong correlations between velocity of flow and ultimate removal efficiency with a correlation coefficient of 0.997 for zinc and 0.991 for iron.

In summary it can be concluded that velocity of flow is not as important to removal of heavy metals as the presence of organic complexes and the average pH value. However, given a metal species in a stable inorganic complex which does not change speciation during travel through the swale, then removal efficiency could be expected to exhibit a strong inverse correlation with velocity of flow. In addition, high velocities might cause scour of bottom sediments and transport debris which may reduce removal efficiency by swales.

Total Mass Removals of Dissolved Heavy Metals

All of the results presented previously have dealt with removal of heavy metals in terms of a reduction in concentration in a continuous flow situation. This is a useful approach since many State of Florida and Federal water quality regulations are based on allowable concentrations of pollutants entering waterways rather than on a more difficult total mass approach. These regulations were presumably developed in this fashion to make application and enforcement an easier task. However, this attempt to simplify can sometimes backfire since a small discharge which may violate certain parameters of the regulations may be far less damaging on a long-term basis than a continuous input which meets the regulations. Certain regulations, particularly those involving effluent waste streams, have been based on allowable daily mass inputs to a water body. In any case, a discussion of total mass removals during travel through swales is indicated for heavy metals.

If the removal of heavy metals in roadside swales is considered on a total mass basis, the removal efficiencies change considerably. Not only must the removal due to a change in concentration through the swale be considered, but also the removal of metal ions by infiltration into the ground. Therefore, if infiltration occurs to a large degree, then the total mass removal will far exceed the removal indicated by changes in concentration alone. This approach can be carried to an extreme in a case such as the third Maitland experiment which investigated the effects of a very low flow velocity on decreasing concentrations. As seen in Figure 3-3, concentrations of iron, aluminum and zinc were reduced considerably, yet on a total mass basis the total removal was actually 100 percent since the relatively rapid infiltration, combined with the low flow velocity, produced a situation in which no flow reached the outfall.

The removal efficiency of 100 percent mentioned above is obviously an extreme case but is not common under normal conditions. The most likely situation will be one in which the actual removal will be between the efficiencies listed for dissolved species with no infiltration and total removal. In order to better illustrate how consideration of infiltration can affect removal efficiency, a mass balance was calculated for each of the five swale experiments in which heavy metals were measured. It is assumed in these calculations that the removal of heavy metals with distance during flow through a swale followed a linear curve and that average input and output concentrations could be estimated by the average of input and output concentrations measured throughout the experiment. A total mass balance for the three Maitland swale studies is presented in Table 3-14, and for the two EPCOT experiments in Table 3-15.

TABLE 3-14

Total Heavy Metal Mass Balance for Maitland Swale Investigations

Experiment	Percent Infiltration	Heavy Metal	Total Mass Input (mg)	Total Mass Output (mg)	Change In Total Mass (%)	Mass Removal Rate (ug/m^2-min)	Change In Concentration (%)
Maitland	57	Zn	900	88	-90	51	-77
(1/24/83)		Cu	327	123	-62	13	-13
(Vel = 2.58 m/min)		Al	5358	1866	-65	217	-19
		Fe	1432	299	-79	71	-51
		Pb	491	264	-46	14	+25
		Cr	491	211	-57	18	0
Maitland	60	Zn	538	17	-97	30	-92
(2/7/83)		Cu	145	25	-83	7	-57
(Vel = 1.37 m/min)		Al	2546	623	-76	112	-39
		Fe	9708	1187	-88	496	-70
		Pb	166	50	-70	6.8	-25
		Cr	124	33	-73	5.3	-33
Maitland	100	Zn	147	0	-100	11	-89
(2/21/83)		Cu	16	0	-100	1.2	+100
(Vel = 0.90 m/min)		Al	301	0	-100	22	-73
		Fe	2555	0	-100	186	-70
		Pb	49	0	-100	3.6	0

TABLE 3-15

Total Heavy Metal Mass Balance for EPCOT Swale Investigations

Experiment	Distance (m)	Percent Infiltration	Heavy Metal	Total Mass Input (mg)	Total Mass Output (mg)	Change In Total Mass (%)	Mass Removal Rate (ug/m²-min)	Change In Concentration (%)
EPCOT (3/23/83)	0-90	32	Cd	635	314	-51	9.7	-27
			Zn	14,771	5,488	-63	282	-45
			Cu	1,269	862	-32	12	0
			Fe	28,215	16,072	-43	368	-16
			Pb	5,250	1,723	-67	107	-52
			Ni	6,462	2,430	-62	122	-34
			Cr	635	392	-38	7.4	-27
	90-170	9	Cd	314	251	-20	2.1	-13
			Zn	5,488	3,544	-35	65	-29
			Cu	862	680	-21	6.1	-14
			Fe	16,072	18,544	+15	-83	+13
			Pb	1,723	1,146	-33	19	-27
			Ni	2,430	1,861	-23	19	-30
			Cr	392	286	-27	3.6	0
EPCOT (5/15/83)	0-90	33	Cd	93	31	-67	2.1	-50
			Zn	1,065	431	-60	21	-39
			Cu	1,342	1,109	-17	7.9	+24
			Fe	4,213	3,450	-18	26	+23
			Pb	2,037	1,201	-41	28	-11
			Ni	1,296	616	-52	23	-29
			Cr	370	339	-8	1.1	+38
	90-170	25	Cd	31	23	-26	0.2	0
			Zn	431	184	-57	7.6	-43
			Cu	1109	644	-42	13	-22
			Fe	3450	2622	-24	24	+2
			Pb	1201	598	-50	17	-33
			Ni	616	368	-40	7.1	-20
			Cr	339	207	-39	3.8	-18

As seen in Table 3-14, consideration of total mass removal produced increased removal efficiencies when compared to removal of dissolved species only. This increase in removal was particularly evident in the case of species such as copper, aluminum, lead, and chromium which did not exhibit significant removal of dissolved forms.

It is particularly interesting to note that in the first Maitland experiment lead exhibited an increase in concentration of 25 percent during passage through the swale but was reduced in total mass by 46 percent. Chromium, which was not reduced in concentration, was reduced in total mass by 57 percent. In the third Maitland experiment concentrations of copper doubled during passage through the swale yet on a mass removal basis, copper was totally removed since no flow reached the end of the swale.

The total mass balance calculations for the EPCOT swale investigations are listed in Table 3-15. In this table the swale test area was divided into two sections: the 0-90 m section which can be classified as a sandy upland area and the 90-170 m area which is lower in elevation and maintains a constant level of standing water. The soil in this area is a mixture of sand, silt and organic materials. As seen in Table 3-15, all of the heavy metals measured, with the exception of copper and iron, were removed much more efficiently both on a concentration reduction and total mass approach in the upper reaches of the swale than in the lower wet portions. This is most probably explained by the fact that in perpetually wet regions a combination of biological, chemical, and physical reactions occur continuously which can cause changes in the speciation and bonding mechanisms which hold heavy metals in the soil. Although heavy metals may be removed and retained by these soils under certain conditions, they may also be rereleased under other conditions. As seen in Table 3-15, this is particularly common with copper and iron which increased in concentration during passage through this area. These results indicate that swales placed in low areas which remain in a wet condition much of the time are much less efficient in removal of heavy metals than higher areas, and in some cases may actually act as a source by releasing previously deposited metals.

Mass removal rates (in terms of ug/m^2-min) for each of the Maitland and EPCOT experiments are also listed in Tables 3-14 and 3-15. These rates are calculated by dividing the total mass removed during travel through the swale by the wetted area and experiment duration. These removal rates are useful in explaining removal mechanisms and in identifying contributing factors. However, it is important to note that these rates are highly site and condition specific and should not be used in design applications. In order to identify factors which are important in regulating removal of heavy metals during travel through swales, correlation coefficients were computed between the mass removal rate and various parameters such as total mass input, velocity of flow and percent infiltration. Of these three parameters, only total mass input was found to have a significant linear relationship with the mass removal rate. Calculated correlations are given in Table 3-16. With the exception of copper and iron at the EPCOT site all heavy metals exhibited a strong correlation between mass removal rate and total mass input. This relationship is intuitively correct since the removal model is basically one of adsorption onto soil particles. This adsorption process should be expected to increase as the driving force in the form of total mass input increases. The fact that significant correlations were not observed between mass removal rates and other parameters such as flow velocity or infiltration rates does not indicate that these parameters are not important in regulating removal of heavy metals. It does indicate, however, that the total mass input was the dominant driving force for removal rates in these experiments and has masked the importance of the other regulating factors.

TABLE 3-16

Correlation Coefficients Between Mass Removal Rates and Various Parameters for the Maitland and EPCOT Experiments

Site Input	Heavy Metal	Correlation Coefficient (r) Between Mass Removal Rate and Total Mass
Maitland	Zn	0.999
	Cu	0.996
	Al	1.000
	Fe	0.990
	Pb	0.999
EPCOT	Cd	0.938
	Zn	0.988
	Cu	0.396
	Fe	0.711
	Pb	0.994
	Ni	0.973
	Cr	0.857

Comparison Between Steady State Flow and Actual Conditions

Although the removal of only dissolved species of heavy metals has been discussed previously, heavy metals in highway runoff often have significant particulate fractions as well. As seen in Table 3-11, zinc, aluminum, iron, lead, nickel and chromium all exist in highway runoff in predominantly particulate form with zinc, lead and nickel all approaching 90 percent particulate. With the exception of copper and nickel, flow through the swale was able to produce a significant decrease in the particulate fraction with t-test probabilities in excess of 98 percent. Swales, therefore, appear to be quite efficient in removing particulate forms of heavy metals provided that these particulate forms remain on the site and are not leached or transported by subsequent water flow.

If the combined data listed in Table 3-11, page 59, for highway runoff and swale flow along with the data for Stations 4 and 6 in Table 3-12 are compared with the results from the swale experiments, it is apparent that the measured reductions over the 8-month sampling period are not as great as those observed in discrete continuous flow situations. This difference could be due to the sampling technique used to collect the stormwater samples. However, it is also possible, and perhaps likely, that certain species are being released and remobilized in the swale area with time. This may be particularly important for metal species which change from the free metal state to a neutral ion such as $PbCO_3$. Copper, lead, zinc and nickel all exchange forms from Me^{+2} to a neutral ion in the pH range of 6.0 - 7.5. Changes in the soil pH between storm events could resolubilize certain species. It is also possible that the shearing action of the water flow could remove attached species, but the importance and significance of this phenomenon are not known. In any case, the actual obtained removal efficiencies for dissolved species over a long period would probably be less than that obtained in continuous flow experiments.

CHAPTER IV

PHOSPHORUS IN ROADSIDE SWALES

INTRODUCTION

Phosphorus is one of the essential elements for the productivity of natural water systems and is generally associated with accelerated eutrophication problems. Over fertilization of waters from point and non-point sources results in excessive production of aquatic weeds and algae which can be aesthetically objectionable, create odors, cause dissolved oxygen problems, interfere with recreational uses, and create mats of floating material at shorelines. Attempts have been made to control lake and stream pollution by reducing phosphorus loadings.

There is very little or no data in the literature evaluating the efficiency of swales to remove nutrients before their discharge to the receiving water body. However, there is a vast amount of information written about the fate of phosphorus in aquatic systems. This chapter reviews phosphorus dynamics and transformations that occur in aquatic systems. Also, the phosphorus content of highway runoff is determined and compared to the phosphorus content of highway runoff after flowing over a swale area. In addition, data collected from several controlled swale experiments conducted at the Maitland Interchange and EPCOT sites are presented and evaluated.

Phosphorus in Highway Runoff

The amount of pollutants on a highway surface is dependent on traffic volume, vehicle emissions and friction wear, local land use conditions, and highway maintenance procedures (Gupta et al., 1981; Asplund et al., 1980). The major constituent of street surface contaminants has been found to be inorganic, mineral-like matter, similar to common sand and silt. In addition, Sartor et al., (1974) determined that one-third to one-half of the total phosphorus was held within the fine solids (< 43 u) which only accounted for 6 percent of the total solids by weight. They divided solids into three groups: < 43 u; 43 u - 246 u; > 246 u; and the total phosphorus fraction by weight was 56.2, 36.0, and 7.8 percent respectively.

Shaheen (1975) has shown that while a majority of the solids are related to traffic volumes, less than 5 percent originate from the vehicles themselves. This implies that vehicles must function as both a source and transport mechanism for pollutants. The pollutants deposited by vehicles consist of direct and indirect or "acquired" pollutants. Direct pollutants result from wear and tear on the vehicle as well as exhaust emissions. Acquired pollutants are solids and aerosol films that vehicles collect from industrial sites, parking lots, construction sites, farming areas, etc., and transport for later deposition onto roadway surfaces during storms. Most of the phosphorus deposited on the highway is attributed to oil and gasoline leakage and combustion productions (Asplund et al., 1980). Gas and oils contain additives such as antioxidant Zn-dithiophosphate (Lagerwerff and Specht, 1970). Indirect sources of phosphorus depend on where the vehicle has been and what it has picked up, and will be highly variable from one vehicle to another.

Rimer et al., (1978) collected runoff water samples and determined the total phosphorus concentrations for the seven land types listed in Table 4-1. The data indicate that the highest phosphorus concentration level was found in the runoff from low activity rural land. This finding bears a direct relationship to the agricultural practices (fertilization) associated with that particular land use classification. More phosphorus is applied to agricultural land than to other land types. Some of this phosphorus will end up on the roadways due to vehicular transportation and wind blown dust. The data also indicate, except for the first and last land types, that the level of nonpoint source phosphorus increases with increasing impervious area and vehicular activity. ·Of course the low activity rural land type must be deleted from this trend because of fertilization activities.

TABLE 4-1

Average Peak Concentrations of Phosphorus in Runoff Water Samples From Various Land Use Drainage Areas

Land Cover Types	# Of Storms Sampled	Percent Impervious Area	TP mg/l
Low Activity Rural	5	2.7	1.40
High Activity Rural	5	5.1	0.76
Low Activity Commercial	9	12	0.83
Low Activity Residential	9	16	0.84
High Activity Residential	8	32	1.03
High Activity Commercial	8	35	1.08
Central Business District	11	80	0.66

Gupta et al., (1981) investigated the highway runoff of the six highway locations and the phosphorus data from this study are presented in Table 4-2. The sample sites represent direct runoff from highway surfaces except for one Milwaukee site that was selected for grass cover. This site was chosen in an attempt to characterize runoff from the highway after flowing over a grassy area. The overall average total phosphorus concentration ranged from 0.50 to 1.92 mg/l. The grassy site had an average total phosphorus concentration of 0.81 mg/l which is higher than some of the sites and lower than others. This indicates that the possible effects of the grassy area were minimal. The phosphorus loading data is given below in Table 4-3. The overall average total phosphate loading ranged from 0.24 to 0.124 lbs/-acre/event.

TABLE 4-2

Concentrations of Nutrients in Highway Runoff
(Based on Gupta et al, 1981)

	Total Phosphate, mg/l					
	Overall 1976-77 Monitoring Period		Non-Winter Periods[a]		Winter Periods[b]	
	Avg.	Range	Avg.	Range	Avg.	Range
Milwaukee-Hwy. 794	0.50	0.12-1.81	0.31	0.12-0.63	0.90	0.28-1.81
Milwaukee-Hwy. 45	0.52	0.10-1.27	0.48	0.10-1.27	0.59	0.28-1.23
Milwaukee-Grassy Site	0.81	0.31-1.51	0.90	0.33-1.51	0.64	0.31-1.11
Harrisburg	0.34	0.05-0.86	0.29	0.05-0.56	0.39	0.12-0.86
Nashville	1.92	0.77-3.55	1.89	0.77-3.55	1.97	0.78-3.50
Denver	0.92	0.48-2.36	0.92	0.48-2.36		c

The quality of runoff from a rural highway bridge near Tallahassee, Florida, was investigated by Irwin and Losey (1978). The mean total phosphorus concentration was determined to be 0.15 mg/l and ranged from 0.01 to 0.30 mg/l. Beaven and McPherson (1978) studied the runoff quality of a highway interchange, I-95 and S.R. 836, in Miami, Florida, and reported that the total phosphorus concentration of the runoff averaged 0.08 mg/l.

These studies indicate that highway runoff contains concentrations of phosphorus that are high enough to cause pollution problems to the receiving water body. In many cases, the runoff water is channeled through swale areas before discharge to the receiving water body. Little et al., (1983) from the University of Washington, attempted to

TABLE 4-3

Loadings of Nutrients in Highway Runoff
(Based on Gupta et al., 1981)

	Total Phosphate, lbs/acre/event					
	Overall 1976-77 Monitoring Period		Non-Winter Periods[a]		Winter Periods[b]	
	Avg.	Range	Avg.	Range	Avg.	Range
Milwaukee-Hwy 794	0.041	0.003-0.11	0.04	0.003-0.10	0.042	0.005-0.11
Milwaukee-Hwy 45	0.024	0.001-0.125	0.02	0.001-0.06	0.026	0.001-0.13
Milwaukee-Grassy Site	0.032	0.00003-0.13	0.02	0.0004-0.08	0.050	0.00008-0.13
Harrisburg	0.036	0.0002-0.169	0.02	0.0002-0.17	0.050	0.002-0.15
Nashville	0.124	0.0045-0.36	0.11	0.005-0.29	0.150	0.06-0.36
Denver	0.026	0.004-0.089	0.03	0.004-0.09		c

[a] Represents monitoring periods from April through October, 1976-77. Actual numbers may vary between sites.

[b] Represents monitoring periods from November through March, 1976-77. Actual numbers may vary between sites.

[c] No storm events monitored during winter at Denver due to lack of sufficient precipitation.

Metric Units: to convert lbs/ac/event to kg/ha/event, multiply by 1.12.

determine the amount of nutrient removed from runoff flowing through vegetated channels (swales). Although no specific data is given, they report that total phosphorus and soluble reactive phosphorus were reduced by at least 20 percent during flow through a 73 meter vegetated channel. However, some of the samples showed higher soluble reactive phosphorus levels in the channel than in the input flow, while total phosphorus was still reduced by the channel. Possible reasons given for the decline in nutrient removal indicated by these samples include reduced vegetative growth in the fall and possible release of phosphorus due to decomposition of dead plant material.

Khalid et al., (1981), demonstrated that overland flow of sewage could be effective in removing phosphorus as a result of grass infiltration. However, he concluded that while initial phosphorus removal and retention were high, after long periods of flooding of the soil the phosphorus sorption capacity markedly decreased. This decrease was followed by an increase in phosphorus mobility.

Characterization of Phosphorus

Phosphorus tends to be the limiting nutrient in aquatic productivity due to its usually low concentrations in the environment and its biological demand (Dillon and Rigler, 1974; Jones and Bachmann, 1976; Schindler, 1978). It plays a major role in nearly all phases of metabolism and is required in the synthesis of nucleotides, phosphatides, sugar phosphates, and other phosphorylated intermediate compounds. Of vital importance are the energy transformation reactions involving the two enzymes adenosine diphosphate (ADP) and adenosine triphosphate (ATP) and the phosphorylation reactions of photosynthesis. When energy is increased to a system, more of the phosphorus is bound up in organic storage, leaving the pool of available raw nutrient in short supply, and limiting production (Odum, 1983). One principle step of lake restoration projects in recent years has been to control and limit the input loading of phosphorus as well as the internal cycling of phosphorus.

Soluble orthophosphates that enter a water body are readily assimilated by plants and other aquatic organisms, forming particulate organic phosphorus compounds. At death and during decomposition of the sediment, most phosphorus is in organic form and must undergo bacterial degradation before it is again available as orthophosphorus (Wetzel, 1975). Mortimer (1942) found that the sediment layer acts as a sink and is able to release phosphates into the water column under favorable conditions.

Among the phosphorus containing compounds found in natural waters, inorganic soluble phosphates are the most significant (Lung et al., 1976). Inorganic phosphates occur largely as orthophosphate (PO_4^{-3}), or as condensed phosphates, such as metaphosphate (PO_3^-), triphosphate ($P_3O_{10}^{-5}$), and pyrophosphate ($P_2O_7^{-4}$). The condensed phosphates are generally hydrolyzed in aqueous solutions to the more common ortho form (Hutchinson, 1957). Separation of the total phosphorus into inorganic and organic fractions indicates that a large majority of the total phosphorus is in the organic phase. Of the total organic phosphorus, up to 70 percent and more is within the sestonic (particulate) organic material. Soluble organic phosphorus includes a major fraction of colloidal phosphates. Therefore, it is evident that in many water bodies only a small fraction (< 5%) of the total phosphorus present is available as soluble orthophosphorus (Stumm and Stumm-Zollinger, 1972). It is important to note that the orthophosphate ion (PO_4^{-3}) is very chemically reactive and that there is also a great biological demand for it. Free and unattached phosphates are continually cycled and recycled.

Phosphorus Transformations in Natural Water Systems

Soluble and particulate phosphorus enter the system by point and non-point discharges such as industrial and municipal waste, stormwater runoff, direct precipitation, wildlife waste, and urban drainage.

Particulate phosphorus will tend to settle in the aquatic environment; however, fine colloidal particles may not be settleable. The rate of settling is affected by particle size, specific gravity, and surface configuration (McKee et al., 1970). Larger and heavier particulates will settle near the discharge point while more flocculent material will settle slower and will be carried toward deeper parts of the water body by density currents. There is strong evidence that phosphates are sorbed to clay particles (Hesse, 1973). Amorphous iron and aluminum hydrated oxides with hydroxyl groups in surface positions react readily with orthophosphate by exchange adsorption and replacing of orthophosphate for hydroxyl (Williams et al., 1971). Sridham and Lee (1974), studied sediments of Lower Green Bay, Wisconsin, and determined that phosphorus associated with iron and aluminum particles is more tightly bound when compared to more flocculent, clay-type material.

Soluble orthophosphorus in addition to a small fraction of the polyphosphates and organically bound phosphorus are available to algae, plant and animal life in aquatic systems. Soluble phosphorus quickly disappears when added to lake waters. This is due largely to the sorption of the phosphate ion onto the lake sediment and to precipitation and coprecipitation. The dissolved phosphorus that does remain in ionic balance with the surrounding water may be converted into algal cell material by converting soluble phosphates into organic phosphorus compounds. At death, some of the organic phosphorus is released to the surrounding water, and the remaining phosphorus of the dead cell settles to the sediment to be recycled. Golterman (1973), indicates that the amount of time for phosphorus to cycle between soluble form, cell incorporation, and subsequent lysis is about ten days.

Submerged, emergent, and floating-leaved macrophytes are important in the phosphorus cycle. The roots and leaves function in the absorption of nutrients which are later recycled through decomposition (Landers and Lottes, 1983). Hayes and Phillips (1958), indicated that phosphorus is absorbed by the littoral vegetation as well as the phytoplankton. It was observed that phosphorus apparently was released slowly by plants, but more rapidly from algae. With the decay of annual macrophytes at the end of summer, release of phosphorus from decaying vegetation can be expected. Carignan and Kalff (1982), have also discovered that living submerged macrophytes are able to release phosphorus to the water in a very soluble and available form. They determined that between 3.4 and 8.9 percent of the phosphorus present in loosely held epiphytes was contributed by the macrophytes.

Role of Bottom Sediment

The chemical properties of a water saturated soil are similar to those for bottom sediments. Waterlogged soil will be found in swales that are continuously wet or contain water most of the time. The waterlogged soil exhibits bottom sediment properties because of the fact that it is oxygen deficient. The oxidized surface of the soil is reduced and this results in the liberation of ammonia, hydrogen sulfide, iron (II), organic compounds, silicates, and phosphates. Iron (II) ions diffuse upward through to the soil surface where they are oxidized and form a thin layer of iron (III) hydroxide and iron (III) humate complexes. These compounds accumulate as a flocculent layer and strongly adsorb ions such as phosphates. When these compounds are destroyed by reduction of the iron, phosphates and other ions are released into the water.

REMOVAL OF PHOSPHORUS BY ROADSIDE SWALES

Swale Studies

Six swale experiments were conducted between January 24, 1983 and May 31, 1983 with four studies performed at Maitland and two at EPCOT. Water was allowed to flow over the swale area for periods of 3-5 hours. Water samples were collected from various selected stations, transported to the laboratory, filtered through 0.45 u millipore, and analyzed for the dissolved orthophosphorus (OP) and total phosphorus (TP). The data were corrected for time lapse from one station to the following and averaged to obtain phosphorus concentrations along the length of the swale. The average phosphorus concentrations measured at the Maitland and EPCOT sites are presented in Tables 4-4 and 4-5, respectively. One should note that the experiment conducted on February 21, 1983 is lacking data at the 53 meter sampling location, since the flow was very low and did not reach the weir installed at the end of the study length of the swale.

TABLE 4-4

Average Concentrations of Dissolved Phosphorus in Simulated Highway Runoff Flowing Over Roadside Swale at Maitland Interchange and I-4

Date	Form	Average Phosphorus Concentration (ug-P/l)				
		0 m	7.5 m	23 m	38 m	53 m
1/24/83	TP	625	700	682	576	538
	OP	582	616	594	508	510
2/07/83	TP	587	680	452	365	307
	OP	478	610	298	265	253
2/21/83	TP	208	42	83	173	-
	OP	201	29	62	140	-
5/31/83	TP	239	236	250	245	222
	OP	211	202	216	213	210

TABLE 4-5

Average Concentrations of Dissolved Phosphorus in Simulated Highway Runoff Flowing Over Roadside Swale Along Ramp A at EPCOT Interchange

	Average Phosphorus Concentration (ug-P/l)						
Date	Form	0 m		7.5 m	23 m	38 m	53 m
3/23/83	TP	1078	1070	1008	988	1000	983
	OP	1077	1025	1000	968	960	960
5/16/83	TP	120	103	123	127	113	178
	OP	83	67	67	60	50	100

The data presented in Tables 4-4 and 4-5 indicate that phosphorus concentration did not always decrease between one station to the next. There are several instances where phosphorus increased along sections of the swale. In fact, dissolved phosphorus increased occasionally between some stations and locations further downstream. This is readily revealed by Figures 4-1 to 4-3. Maitland data is presented in Figures 4-1 and 4-2, and EPCOT data is given in Figure 4-3.

The first two Maitland studies exhibit an initial increase of phosphorus concentration during the first 7.5 m section of swale. Total phosphorus (TP) increased from 625 to 700 ug-P/l during the first Maitland experiment and from 587 to 680 ug-P/l during the second. For these two studies, total phosphorus concentration decreased along the rest of the swale length between 7.5 and 53 meters. The final concentrations measured were found to be 538 and 307 ug-P/l respectively. During the first two experiments at Maitland, a uniform rate of phosphorus decrease was observed between 7.5 and 53 meters along the swale length, as shown by the steady slope of the lines in Figures 4-1 and 4-2. The third Maitland experiment (2/21/83) indicates a different sequence of events with a rapid decrease in phosphorus during the first

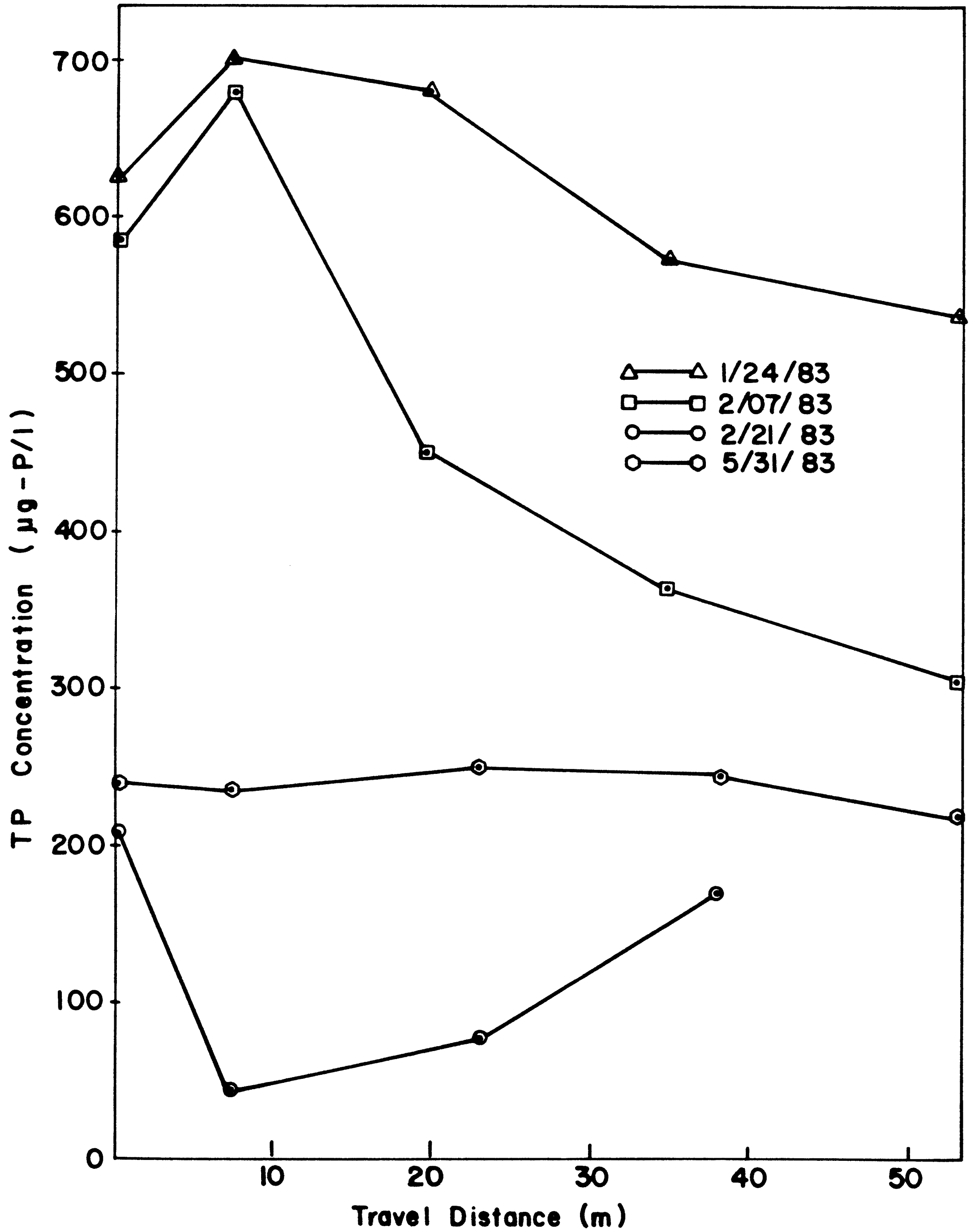

Figure 4-1: TRANSPORT OF TOTAL DISSOLVED PHOSPHORUS ALONG ROADSIDE SWALE RECEIVING SIMULATED HIGHWAY RUNOFF AT MAITLAND INTERCHANGE SITE.

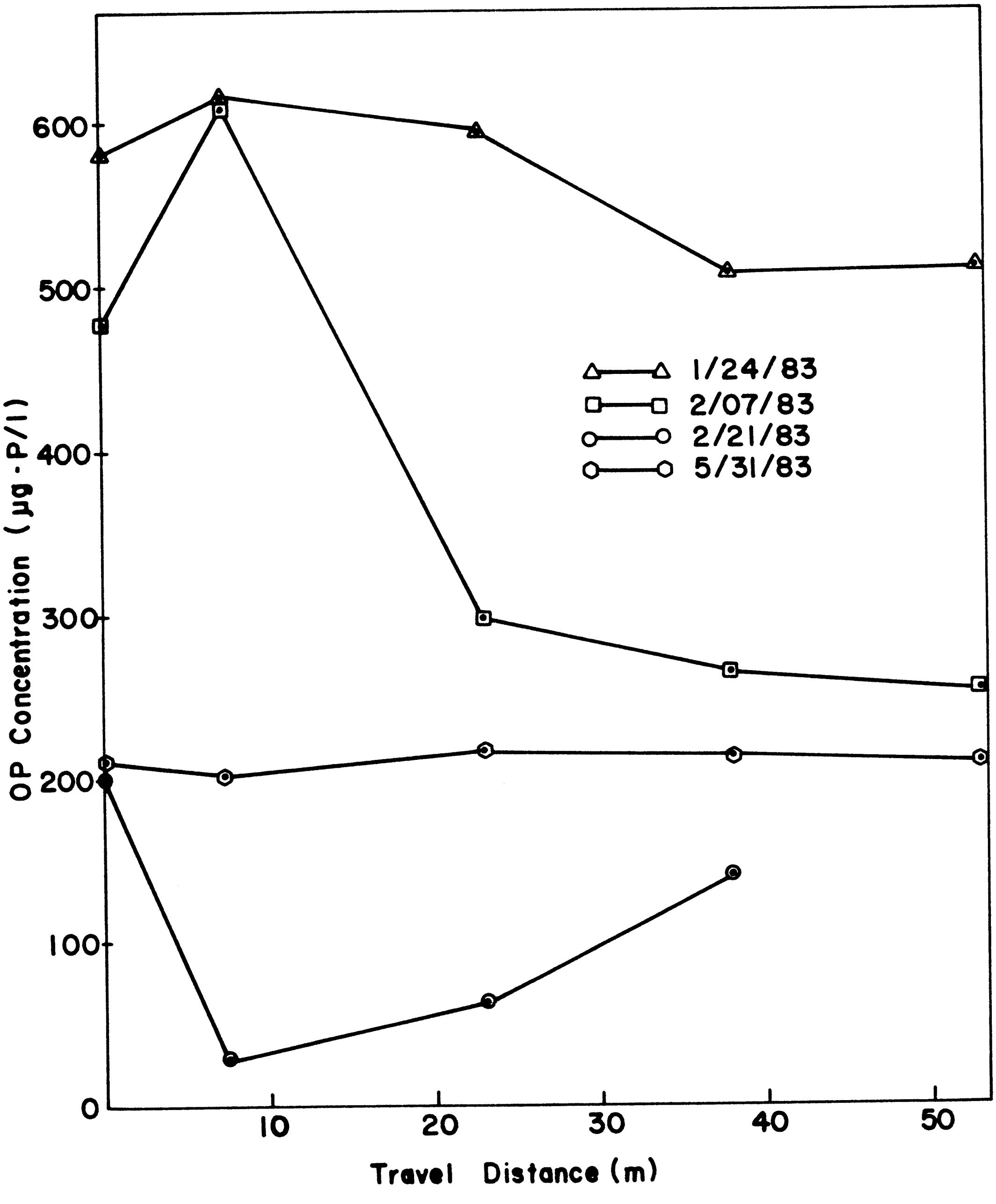

Figure 4-2: TRANSPORT OF DISSOLVED ORTHOPHOSPHORUS ALONG ROADSIDE SWALE RECEIVING SIMULATED HIGHWAY RUNOFF AT MAITLAND INTERCHANGE SITE.

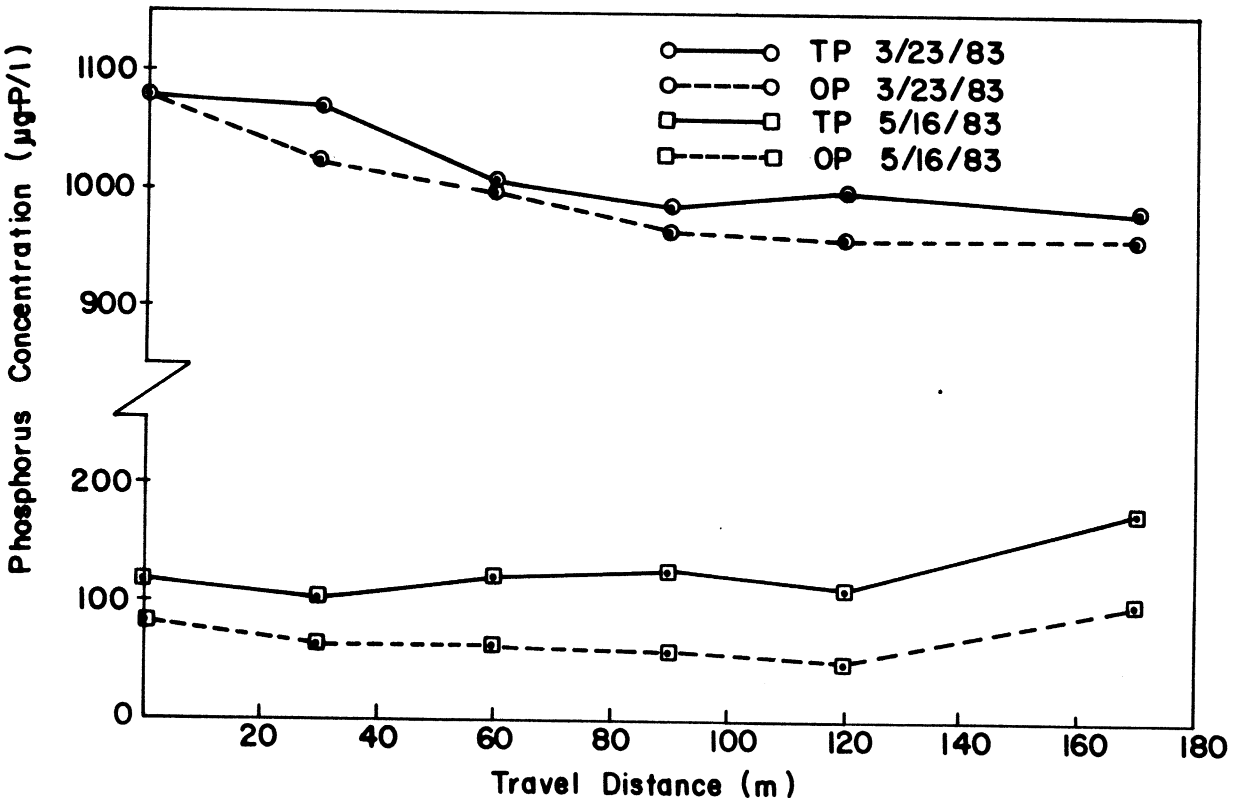

Figure 4-3: TRANSPORT OF DISSOLVED PHOSPHORUS ALONG ROADSIDE SWALE RECEIVING SIMULATED HIGHWAY RUNOFF AT EPCOT SITE.

7.5 m section. Total phosphorus dropped from 208 to 42 ug-P/l along this section, but then increased to 173 ug-P/l at the 38 m station. The flow of this experiment was very low (0.9 m/min) so that it never reached the end weir. The last Maitland study (5-31-83) showed no change in phosphorus concentration between the inlet and outlet concentrations with little or no removal of phosphorus by the end of the swale. The initial total phosphorus concentration was 237 ug-P/l and the final concentration was 222 ug-P/l with little variation in between.

The two EPCOT experiments (Table 4-5 and Figure 4-3) show different results. The first EPCOT experiment has a slow but steady decrease in phosphorus concentration along the length of the swale. Total phosphorus decreased from 1078 to 983 ug-P/l during this study. The second EPCOT study indicates little or no change in phosphorus concentration during the first 120 m section which is followed by an increase during the last 50 m section. The initial total phosphorus concentration is 120 ug-P/l with 113 ug-P/l at the 120 m station and 178 ug-P/l at the end weir. The data indicate that dissolved phosphorus concentration decreases from 83 to 50 ug-P/l during the first 120 m section but increases to 100 ug-P/l at the end weir. It was noted that this section was at a lower elevation and remained damp most of the time. The organic content of the soil in this section was also higher as observed by the darker color of the soil compared to the sandy soil of the first 120 m section.

The two EPCOT experiments are also different from Maitland studies because of the placement of a control weir at 90 m (mid-length of EPCOT swale). The weir blocked water, causing a water pool to occur and altering flows and hydraulic characteristics. However, Figure 4-3 does not indicate that the weir had a noticeable impact on phosphorus content.

The data are presented differently in Figures 4-4 and 4-5, which show the percent of inflow concentration versus swale length. These graphs are easier to use to determine when and where phosphorus was removed during flow through the swale. Only total dissolved phosphorus is shown, as it is practically identical to dissolved orthophosphorus. The first two Maitland studies (Figure 4-4) exhibit an initial increase of phosphorus followed by a decrease below the inflow concentration. Overall removal of dissolved phosphorus is about 10 percent for the first and 50 percent for the second. The third study, as indicated before, has a rapid decrease during the first 7.5 m section only to increase during the rest of the swale. Final removal is approximately 15 percent. The last Maitland study is relatively flat with the line dipping below and then above the initial concentration to finally end with about 5 to 10 percent overall removal.

The two EPCOT studies (Figure 4-5) exhibit fairly little change in phosphorus concentrations, ± 10 percent above and below the inflow concentration, except for the last 50 m of the second study (5/16/83). The phosphorus concentration increases about 50 percent during this 50 m section.

A summary of the average decline of the phosphorus concentration during flow through study swales is given in Table 4-6. The two EPCOT experiments are listed under three categories: swale length to 90 m weir, 90 m weir to final weir at 170 m, and entire 170 m swale. Only one of the swale experiments indicates an increase of phosphorus

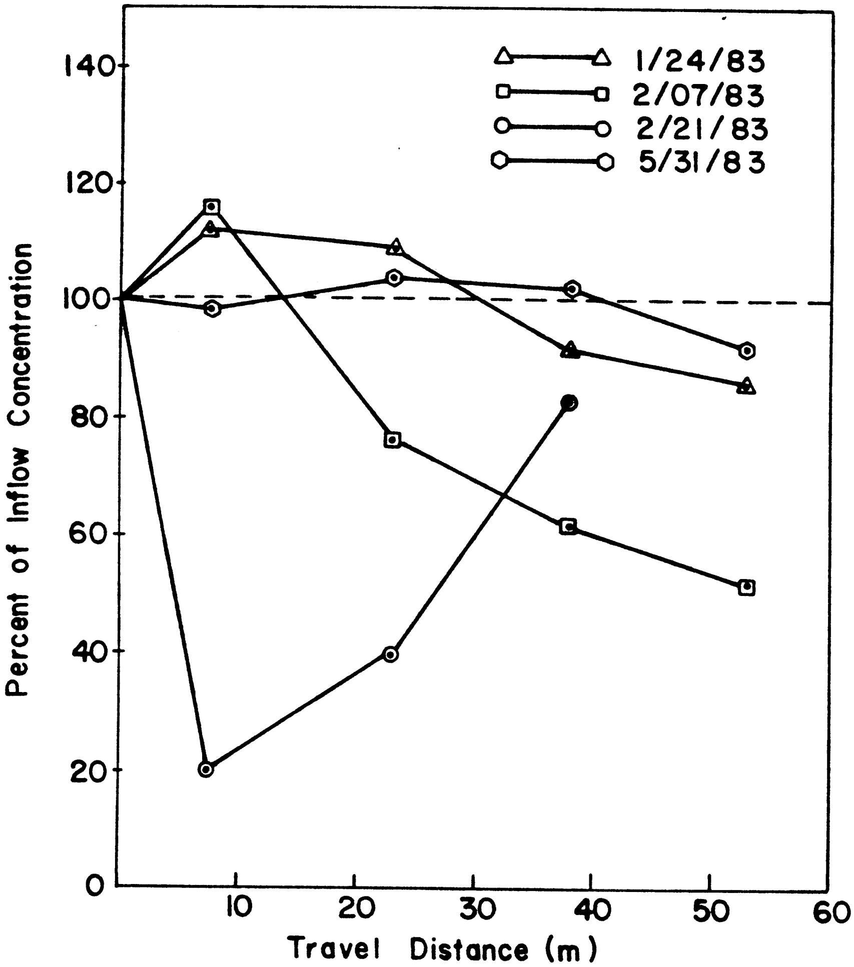

Figure 4-4: FATE OF TOTAL DISSOLVED PHOSPHORUS IN RUNOFF WATER FLOWING THROUGH A GRASSY SWALE AT MAITLAND INTERCHANGE SITE.

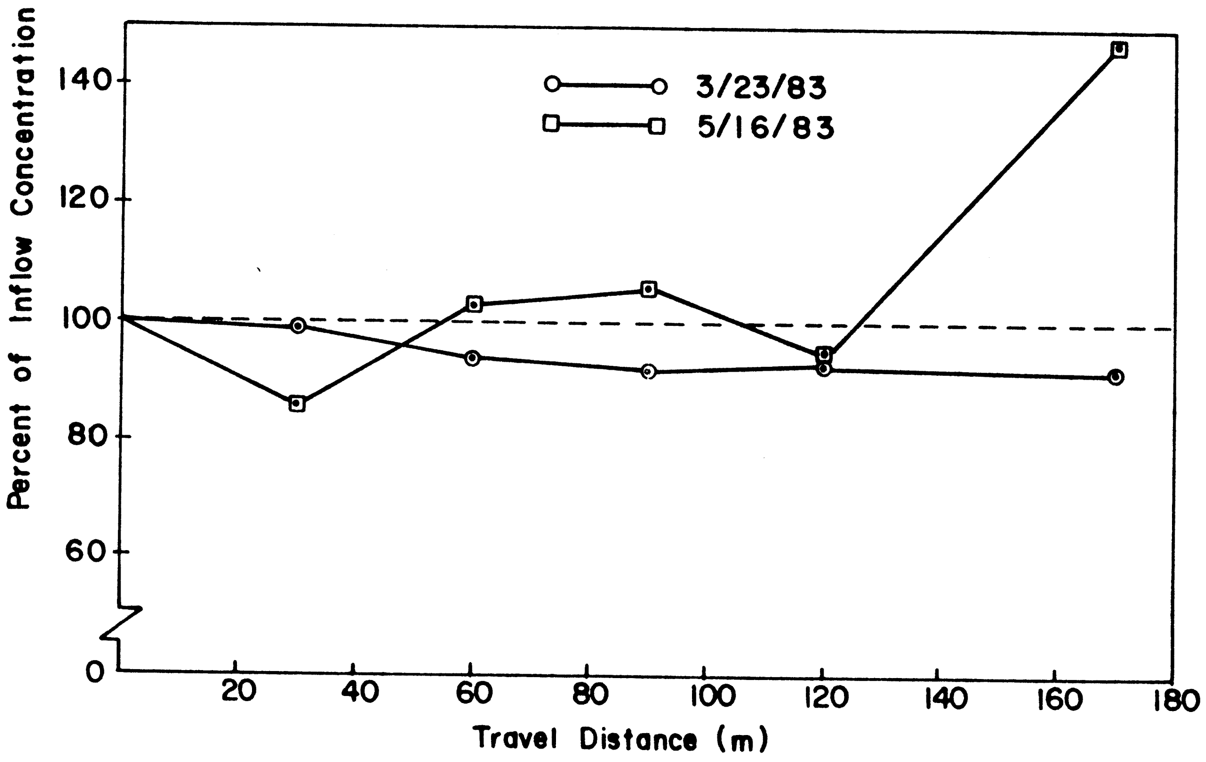

Figure 4-5 FATE OF TOTAL DISSOLVED PHOSPHORUS IN RUNOFF WATER FLOWING THROUGH A GRASSY SWALE AT EPCOT INTERCHANGE SITE.

concentration during flow through the swale. Percentage removal values for the other five studies range from 0 to 40 percent. One study (the last EPCOT) exhibits a phosphorus increase in concentration during the last 50 m section. Of further interest, the first EPCOT study had only 1 percent removal after flowing through the last 50 m section.

TABLE 4-6

Average Decline in Dissolved Phosphorus Concentration During Flow Through Swales

Experiment			Swale	% Change in Phosphorus	
No.	Location	Date	Length (m)	TP	OP
1	Maitland	1/24/83	0 - 53	14	12
2	Maitland	2/07/83	0 - 53	48	47
3	Maitland	2/21/83	0 - 38	17	30
4	EPCOT		0 - 90	8	10
			90 - 170	1	1
			0 - 170	9	11
5	EPCOT	5/16/83	0 - 90	+6*	28
			90 - 170	+40*	+67*
			0 - 170	+48*	+20*
6	Maitland	5/31/83	0 - 53	7	0

*Represents an increase in concentration - all other values indicate a decrease.

During the last swale study conducted at Maitland Interchange on May 31, 1983, additional samples were taken as the initial flow reached each sampling station. These samples were collected to determine the first flush effect of the initial flow volume. As Table 4-7 shows, phosphorus concentration increased gradually as the runoff water

TABLE 4-7

Initial Dissolved Phosphorus Concentration in Simulated Highway Runoff Water as it Reached Sampling Locations Along the Swale at Maitland Interchange on May 31, 1983.

	OP		TP	
Swale Position (m)	Conc. ug-P/l	% Initial Conc.	Conc. ug-P/l	% Initial Conc.
0	338	100	348	100
7.5	404	120	450	129
23	474	140	503	145
38	490	145	548	157
53	492	146	561	161

proceeded through the length of the swale. Dissolved orthophosphorus concentration increased from 338 to 492 ug-P/l and concentration of total phosphorus increased from 348 to 561 ug-P/l.

In addition to concentration changes in TP and OP of runoff water flowing over roadside swales, a mass balance was calculated for each swale that includes the flow of runoff water and losses by evaporation and infiltration into the soil. Maitland study results are listed in Table 4-8. Of course 100 percent of the phosphorus load was retained by the swale area if water did not reach the downstream end. Mass loading rates range from 7 to 96 mg/m^2/hr and mass retention rates range from 5 to 60 mg/m^2/hr. Percent mass retained by the swale ranged from 26 to 100. EPCOT study data indicate good removal during the first section (39%) and little removal during the second (9%). The second EPCOT swale experiment run on May 16, 1983, shows an increase in the mass of phosphorus in runoff water flowing over the last section (90 - 170 m) of the swale. This increase of phosphorus mass during the

TABLE 4-8

Retention of Dissolved Phosphorus Mass from Runoff Flowing Through Swale

Experiment								
No.	Location	Swale Length (m)	Form	Mass In (g)	Mass Out (g)	Inflow Loading Rate mg/m²/hr	Average Retention Rate mg/m²/hr	Percent Mass Retained by Swale
1	Maitland	0 - 53	TP	23.6	9.47	96.4	60.8	63
			OP	23.8	8.98	89.6	55.8	62
2	Maitland	0 - 53	TP	12.15	2.55	42.0	33.5	79
			OP	9.89	2.10	34.5	27.2	79
3	Maitland	0 - 49	TP	1.69	0	7.4	7.4	100
			OP	1.64	0	7.2	7.2	100
6	Maitland	0 - 53	TP	8.39	5.77	24.7	6.9	31
			OP	7.41	5.46	19.5	5.1	26

TABLE 4-8 (Continued)

Retention of Dissolved Phosphorus Mass from Runoff Flowing Through Swale

Experiment No.	Experiment Location	Experiment Swale Length (m)	Form	Mass In (g)	Mass Out (g)	Inflow Loading Rate mg/m²/hr	Average Retention Rate mg/m²/hr	Percent Mass Retained by Swale
4	EPCOT	0 - 90	TP	62.2	38.7	118.3	44.7	38
			OP	62.1	37.9	118.2	46.0	39
		90 - 170	TP	38.7	35.2	81.2	7.3	9
			OP	37.9	34.4	79.5	7.3	9
		0 - 170	TP	62.2	35.2	62.1	26.9	43
			OP	62.1	34.4	62.0	27.6	45
5	EPCOT	0 - 90	TP	5.57	3.91	11.3	3.4	30
			OP	3.84	1.85	7.8	4.0	52
		90 - 170	TP	3.91	4.09	6.7	+0.3*	+5.*
			OP	1.85	2.30	3.2	+0.8*	+24.*
		0 - 170	TP	5.57	4.09	5.2	1.4	27
			OP	3.84	2.30	3.6	1.4	40

*Mass increased through this section of the swale, so mass was released not retained.

$\frac{mg/m^2/hr}{100}$ = kg/ha/hr

last section might appear to be minimal; however, it is considered significant when compared to the removal values of the first section.

Phosphorus in Highway Runoff

A summary of the phosphorus content in highway runoff from the Maitland study area is presented in Table 4-9. These samples were not taken during the previously discussed Maitland and EPCOT swale studies, but represent actual highway runoff. Runoff water samples were divided into two groups. Some samples were taken directly from highway surface discharge while other samples were collected after runoff flow through the swale area. Sample Station #4 was taken directly from highway surface discharge, while station #6 is downstream of #4 after flowing over a grassy swale of approximately 60 meters. It is interesting to note that the phosphorus content in runoff water samples collected from

TABLE 4-9

Average Phosphorus Concentrations in Highway Runoff

Sampling Station Number	Site Characteristics	# of Samples	Phosphorus (ug-P/l)		
			DOP	TOP	TP
1	After Swale Area	5	247	375	631
2	Directly from Highway	14	58	106	201
3	Directly from Highway	15	145	232	505
4	Directly from Highway	16	49	71	137
5	After Swale Area	4	334	403	556
6	After Swale Area	16	144	155	212
1+5+6	After Swale Area	25	195	239	351
2+3+4	Directly from Highway	45	84	136	280

DOP = Dissolved Orthophosphorus
OP = Orthophosphorus
TP = Total Phosphorus

Station #6 (after swale) is higher than from Station #4 (direct runoff). Stations #1, #5, and #6 reflect stormwater samples flowing over a swale area, and Stations #2, #3, and #4 represent direct highway runoff. When all six stations are divided into the two groups, it is again interesting to note that the swale area runoff has a higher phosphorus content than the direct runoff from the highway.

DISCUSSION OF RESULTS

One of the overall goals of this swale study is to determine the extent of phosphorus removal by flow through swales. The efficiency of net decline in phosphorus concentration by flow through the entire length of the swale. Figure 4-5 reveals that the EPCOT experiments show phosphorus concentration decrease in the first 120 meters of the swale. Possible mechanisms for phosphorus removal include particular settling, sorption, precipitation, and uptake by plants and organisms.

Maitland Experiments

Each of the four Maitland experiments resulted in a net decrease in phosphorus concentration ranging from 1 to 48 percent (Table 4-6). These studies indicate that there is a trend between the decline in dissolved orthophosphorus concentration and the velocity of water through the swale. The relationship between dissolved orthophosphorus and velocity seems reasonable in that the slower velocity will allow longer contact time between soluble phosphorus and soil. If soluble swales in phosphorus removal can be shown in Figures 4-4 and 4-5. Figure 4-4 indicates that all of the Maitland experiments resulted in a phosphorus removal occurs by sorption, uptake by organisms, or precipitation, all mechanisms will be enhanced by slower velocities resulting in longer contact time. Also, the particulate phosphorus entering the swale will be expected to settle out and a certain amount of the soluble phosphorus might precipitate or coprecipitate along the length of the swale.

Maitland studies show an increase in phosphorus concentration at the first section close to the input pipe. The highway runoff samples collected during this study indicate that the organic fraction of suspended solids is 50 to 60 percent. Organic matter from highway runoff and grass clippings that settle is degraded by bacteria. Then, when the next rain fills the swale with water, phosphorus is recycled back into solution. This series of events might explain the initial release of soluble phosphorus recorded in the Maitland experiments, and explain the increase of initial dissolved phosphorus concentrations in the runoff water as it first reached sampling locations along the swale (Table 4-7). Any increase in particulate phosphorus must be caused by the resuspension of organic debris (dead grass and plant clippings) by the flow through the swale.

The soluble inorganic form of phosphate is very reactive and will have a negative charge associated with it (PO_4^{-3}, PO_3^{-}, $P_3O_{10}^{-5}$, etc.). These ions can be readily sorbed by soils and other surfaces that have positive reaction sites. Clay particles are able to sorb phosphorus (Hesse, 1973). However, the clay content of sandy Florida soil is very low. This may be partially responsible for the relatively low phosphorus removal by the swales.

Of course, the phosphorus mass retention rate is expected to correlate with the infiltration rate which is essential to calculate the mass balance. The relationships can be expressed as follows:

DOP = 815i - 15.1 r = .97

TDOP = 869i - 14.3 r = .95

where i = Infiltration rate, m^3/m^2-hr

DOP = Dissolved orthophosphorus mass retention rate, mg/m^2-hr

TDOP = Total dissolved phosphorus mass retention rate, mg/m^2-hr

These high correlation values appear to indicate that no other mechanism besides infiltration is responsible for mass removal. It is also important to mention that a poor correlation with a value of r = .46 is obtained when correlating infiltration rate to flow velocity. Apparently other parameters, such as soil moisture and time prior to previous storm event, are important factors affecting infiltration rate. Of course this makes sense; if the swale is saturated, little infiltration will occur and consequently little phosphorus mass retention will result.

The effect of swale length on phosphorus concentration is variable with the Maitland swale studies. The third experiment at Maitland, February 21, 1983 (see Figure 4-1), exhibits an increase in phosphorus concentration along the final sections of the swale, but this is not considered to be typical. The last experiment, May 31, 1983, reveals almost constant concentration along the entire swale length with little or no net concentration decline. The first two experiments, however, appear to exhibit a typical logarithmic concentration percentage decline (see Table 4-6). Of course the amount of phosphorus mass removed is increased with swale length because total infiltration will be increased. As in the third Maitland study, February 21, 1983, 100 percent of the phosphorus mass was retained by the entire swale area because of flow so slow that no runoff from the swale resulted.

In summary, there appears to be a trend between the flow velocity and the percent reduction in phosphorus concentration and the percent of phosphorus mass retained by the swale. This is reasonable in that the slower velocity will enhance phosphorus removal mechanisms. However, the actual mass retention rate is only partially correlated to flow velocity. In addition, infiltration rate is only partially explained by flow velocity. Apparently other factors influence infiltration rate and thus phosphorus mass retention rate.

EPCOT Experiments

The two EPCOT experiments resulted in a net decline of phosphorus concentration during the first 120 meters of swale area. During the last 50 meter section, the first experiment exhibited only a 1 percent decrease in phosphorus concentration while the phosphorus concentration actually increased during the second experiment. A weir control structure was located at the 90 m location, but this structure appears to have little influence on phosphorus removal mechanisms. The following discussion will treat the first 120 m section separately from the last 50 m section.

The swale ground covers of the two experiments were different. The first experiment, March 23, 1983, was conducted when the ground cover was sparse with little or no grass cover. The ground cover of the next experiment at EPCOT, May 16, 1983, exhibited a much denser and better developed grass growth. Based on these two experiments and by examination of the first 120 m section, the first experiment had a 10 percent decrease in orthophosphorus concentration while the second experiment had a 28 percent decrease (see Table 4-6). Thus it appears as if the grassy swale removes more phosphorus than bare soil. However, the total dissolved phosphorus concentration increased during the second EPCOT run. This might be partially explained by the resuspension of grass clippings and other solids during the swale flow.

The percentages of mass of phosphorus retained by the first 120 m section were fairly consistent for the two runs. The average retention rates of the two experiments are quite different but this is due to the initial concentrations of phosphorus during the run. The first experiment had a phosphorus concentration almost ten times that of the second and thus resulted in the higher retention rate. Infiltration rates for the two are very close so that the resulting percentage mass removal is comparable.

The release of phosphorus by the last section of the swale can be explained by noting that this section is at a lower elevation and continually wet. However, the first of the two EPCOT studies was performed during a dry spell so that this section was relatively dry then. This section is also characterized by darker organic soil than the sandy soil of the upper section. The chemical properties of a saturated soil are similar to those associated with water body bottom sediments (Hesse, 1973). This is due to the fact that the soil becomes oxygen deficient. The oxidized surface of the soil is reduced and this may result in the liberation of phosphates. Iron (III) hydroxide and iron (III) humate complexes accumulate as a flocculent layer on the surface of soils and strongly sorb ions such as phosphates. When these compounds are destroyed by the reduction of iron, phosphates and other ions are released into the water.

Highway Runoff Data

As presented earlier in Table 4-9, several samples were collected to characterize direct highway runoff as well as runoff flowing through a swale. It is very interesting to note that the phosphorus concentrations of swale samples are higher than direct highway samples. Samples #4 and #6 can be compared because #6 sampling site is downstream of #4 highway discharge. A comparison of highway runoff versus swale runoff is presented in Table 4-10. The t-test probability that the dissolved orthophosphate mean values are different for #4 and #6 is 95.3. When all six sampling locations are divided into two groups, the t-test probability for unequal means is 98.8. Slightly lower values are calculated for total orthophosphate and total phosphate species (Table 4-10). At any rate, there is a significant difference between the two.

TABLE 4-10

Comparison Between Water Samples Collected
From Highway Runoff and Swale Runoff

Location	% Increase in P Conc.			t-Test Probability		
	DOP	TOP	TP	DOP	TOP	TP
Highway Runoff #4 Swale Runoff #6	194	118	55	95.3	91.6	87.5
Highway Runoff (2+3+4) Swale Runoff (1+5+6)	132	76	25	98.8	96.5	84.4

This data might indicate that the swale does not decrease phosphorus concentration at all, but in fact, increases phosphorus concentration to downstream waters. This could occur as organic particulates are degraded and soluble phosphorus released. The locations of the swale samples were in low lying areas that are occasionally wet. Therefore, these sections might act as the last section of the last EPCOT swale did; that is, release phosphorus. On the other hand, the swale data might be a reflection of the first flush effect. Data from Table 4-7 indicate that initial phosphorus concentrations increase as a function of swale length. It is possible that the swale samples collected do not represent a composite sample, but rather a grab sample during the first flush.

Of course this analysis does not take a mass balance into account, nor is this possible with the data collected. One real question that arises is whether there is any net mass removal of phosphorus from the system by the swale. Is phosphorus removed downward into the soil, or is it held near the surface, utilized by plants and organisms, but eventually released to the first flush of large storm events? This might be quite possible especially in swale areas (or end of swale areas) that are continually wet or wet a great deal of the time.

CHAPTER V

FATE OF NITROGEN IN ROADSIDE SWALES

INTRODUCTION

A number of treatment practices, including roadside swales, have been used to abate contaminant discharge from highway runoff. However, the effectiveness of these swales in removing these contaminants, namely nitrogen, phosphorus, and heavy metals, is not known. This chapter will concentrate on the capacity of roadside swales 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 use of receiving waters. 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.

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 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, (Gupta et al., 1981). Using average time periods of two to ten 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).

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).

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 from natural atmospheric processes. Qualitative data on nutrients added to precipitation by the leaching of vegetation are lacking, but it is known that some nitrogen and phosphorus are added to the storm runoff through leaching of the ground vegetation.

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.

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 ug/l or so, N is likely to be the limiting nutrient (Lee et al., 1980).

Physiochemical and biological processes determine the fate of nutrients associated with stormwater discharged into the receiving stream. Nutrients may remain in solution of suspension, settle to the bottom, chemically interact and precipitate or adsorb to sediments, plants or other surfaces. Some of the nutrients may be released back to solution.

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 (Burthouex, 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 results from the Maitland Interchange stormwater monitoring program and the six swale studies were analyzed for nitrogen concentrations and nitrogen mass balance. These results are discussed in the following sections.

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 5-1, most of the nitrogen in runoff samples analyzed was organic and ranged between 72 and 92 percent of the total nitrogen. The organic fractions averaged approximately 80 percent of total nitrogen in water samples collected from direct highway runoff or adjacent swale areas. The weighted average concentration of inorganic fraction is 388 ug-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 5-1 indicate no significant difference between concentration of 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 5-2. It appears that the organic nitrogen fraction during the warmer months is higher in swale runoff 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 5-2. The total nitrogen in runoff water increased by 37 percent during warm summer months and decreased by 7 percent 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 5-3 through 5-6 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 5-2, during the January 24, 1983 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 percent between the inflow concentration and the weir effluent concentration. In the second Maitland study on February 7, 1983, the average surface velocity through the swale was decreased to 1.37 m/min, and, consequently, the

TABLE 5-1

Average Dissolved Nitrogen Concentration in Stormwater Samples Collected from Maitland Interchange and I-4 Drainage Basin

Nitrogen Form	Average Concentration (ug-N/l)					
	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	178	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
No. of Samples	5	15	16	17	4	16

*Runoff water samples from grassy swales.
**Runoff water samples from direct highway.

TABLE 5-2

Change in Concentration Between Stormwater Sampling Stations 4 and 6 Over Two Different Time Periods

Average Percent Change Between Form	A Avg Concentration Over 8-19, 8-23, 9-8 and 10-1 Storm Events		B Avg Concentration Over 12-11, 1-21, 1-31, 2-3, 2-12, 2-14, 27 Events		Average Percent Change Between	
	Sta.4	Sta.6	Sta.4	Sta.6	4 & 6" Col.A	4 & 6" Col.B
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
potal-N	750	1031	1631	1511	37	-7

TABLE 5-3

Transport of Nitrogen Species Through Swale at Maitland Interchange Site on 1-24-83

Distance Along Swale (m)	Average Concentration (ug/l)					
	NH_4-N	NO_2-N	NO_3-N	Inorganic-N	Organic-N	Total-N
0	759	2	340	1101	202	1303
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	-110	-7	-10	-1	-9	

TABLE 5-4

Transport of Nitrogen Species Through Swale at Maitland Interchange Site on 2-07-83

Distance Along Swale (m)	Average Concentration (ug/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	-320	-33	-32	1	-25	

TABLE 5-5

Transport of Nitrogen Species Through Swale at Maitland Interchange Site on 2-21-83

Distance Along Swale (m)	Average Concentration (ug/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 (at 38 m)	-80	-33	-19	-74	134	-30

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

TABLE 5-6

Transport of Nitrogen Species Through Swale at Maitland Interchange Site on 5-31-83

Distance Along Swale (m)	Average Concentration (ug/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

decline in concentration of total-N along the swale increased to 25 percent. The average surface velocity through the swale during the February 21, 1983 Maitland study was further decreased to 0.90 m/min and the decline of total-N increased once again to 30 percent at a distance of 38 m 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 percent. 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 5-1 through 5-4. 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 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, 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 5-3 through 5-6 the inorganic forms of nitrogen declined by 10 percent during the January 24, 1983 and February 7, 1983 experiments, 74 percent during the March 21, 1983 experiment, and 13 percent during the May 31, 1983 study. It is interesting to note that there

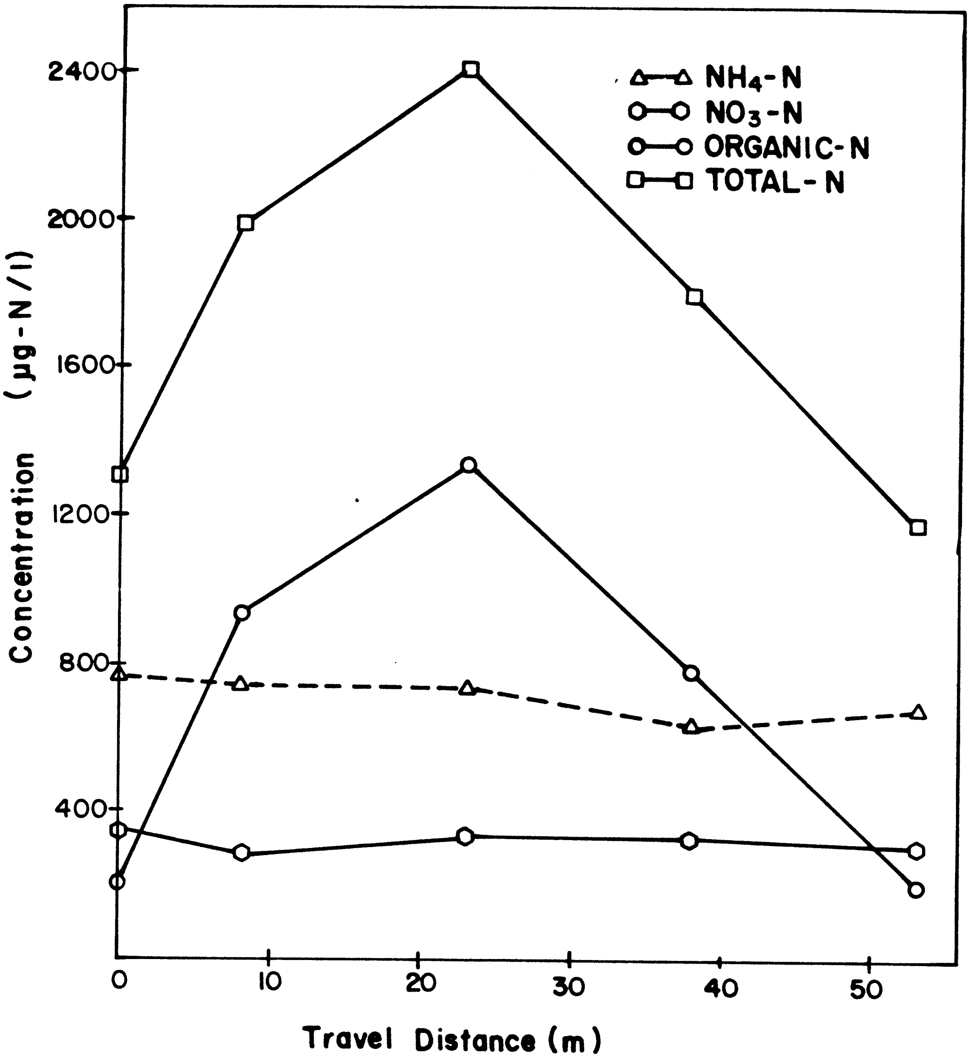

Figure 5-1: CHANGES OF NITROGEN CONCENTRATIONS IN SIMULATED HIGHWAY RUNOFF ALONG ROAD-SIDE SWALE AT MAITLAND INTERCHANGE ON 1-24-83.

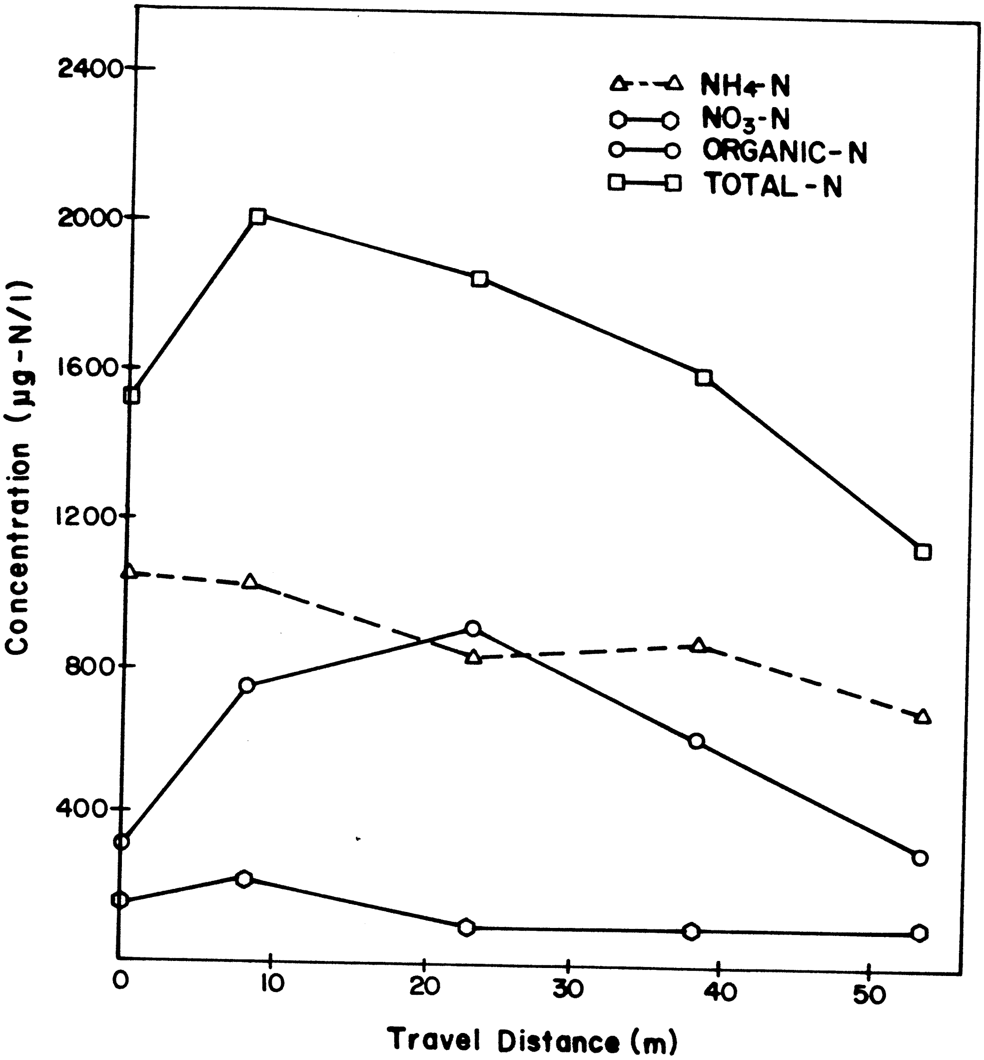

Figure 5-2 : CHANGES OF NITROGEN CONCENTRATIONS IN SIMULATED HIGHWAY RUNOFF ALONG ROADSIDE SWALE AT MAITLAND INTERCHANGE ON 2-7-83.

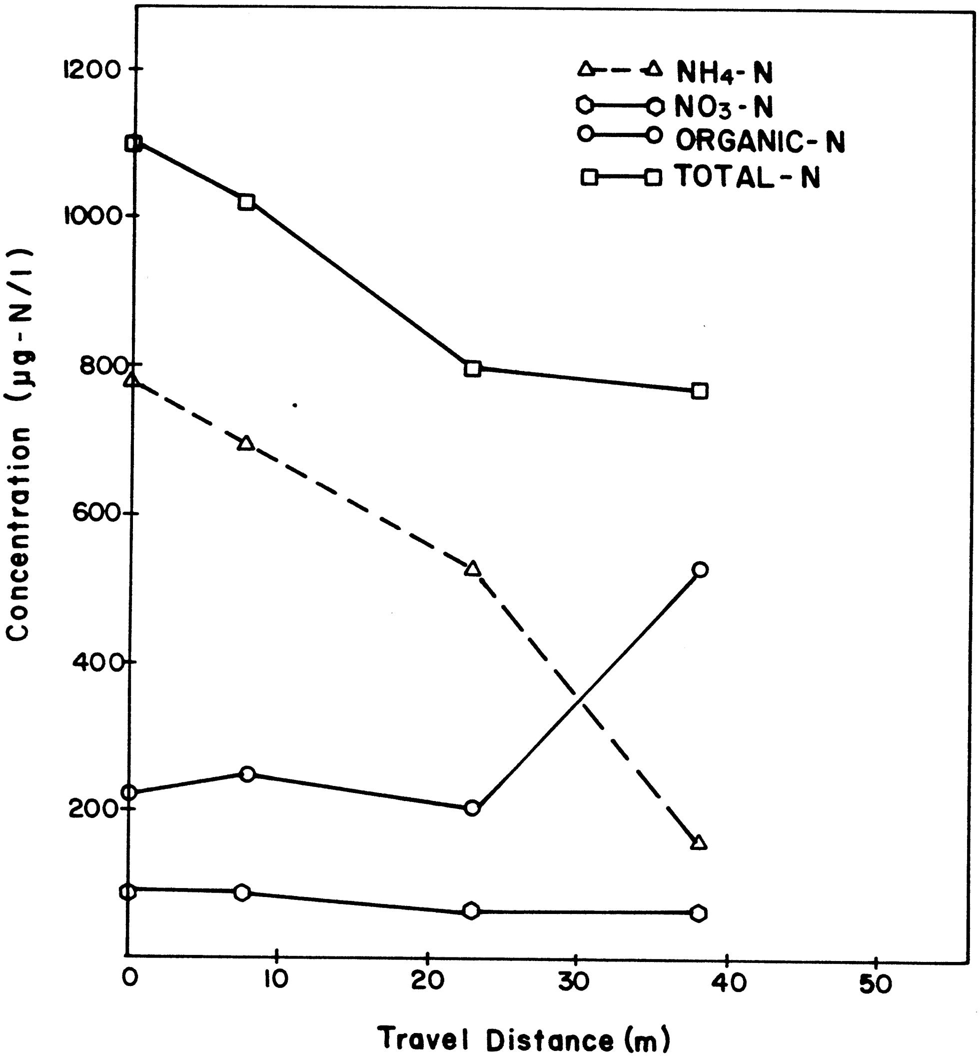

Figure 5-3: CHANGES OF NITROGEN CONCENTRATIONS IN SIMULATED HIGHWAY RUNOFF ALONG ROADSIDE SWALES AT MAITLAND INTERCHANGE ON 2-21-83.

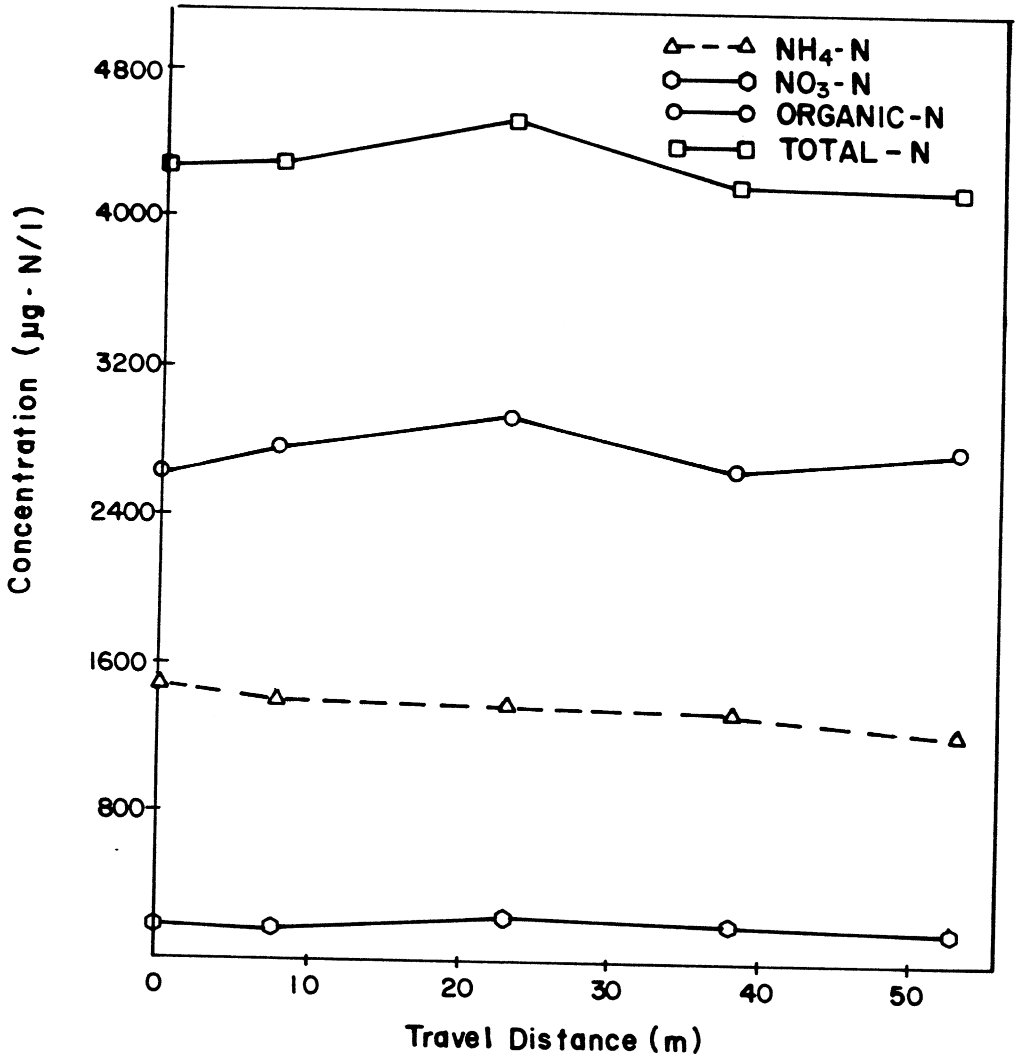

Figure 5-4: CHANGES OF NITROGEN CONCENTRATIONS IN SIMULATED HIGHWAY RUNOFF ALONG ROADSIDE SWALES AT MAITLAND INTERCHANGE ON 5-31-83.

was an increase of the total organic nitrogen species between intermediate sampling stations during the January 24, 1983, February 7, 1983, and May 31, 1983 studies, although there was a total inorganic net decline over the entire test length. These percent changes in nitrogen are further illustrated in Figures 5-5 through 5-8 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 observed at the site may affect the organic content in highway runoff flowing over swale areas.

At the EPCOT Interchange site, two studies were performed. The March 23, 1983 study was performed in a recently completed earthen swale. The May 16, 1983 study was performed at the same site after the 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 5-7 and 5-8, and in Figures 5-9 and 5-10. The Total-N concentration declined 2 percent during the March 23, 1983 study and increased 14 percent during the May 16, 1983 study. The percent changes of all nitrogen forms during the two EPCOT studies are further represented in Figures 5-11 and 5-12.

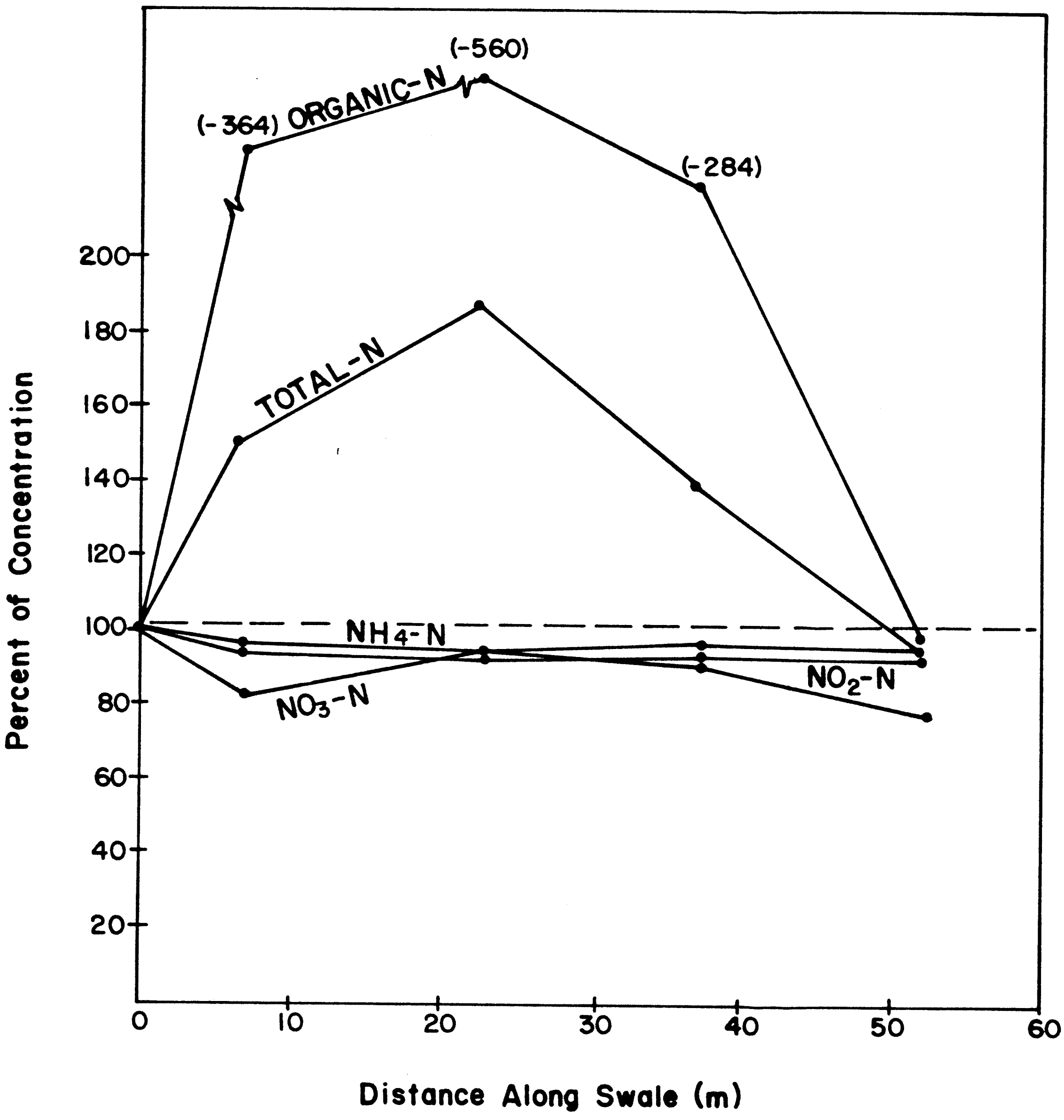

Figure 5-5: AVERAGE PERCENT CHANGES OF NITROGEN SPECIES ALONG THE LENGTH OF THE SWALE DURING THE MAITLAND SWALE STUDY ON 1-24-83.

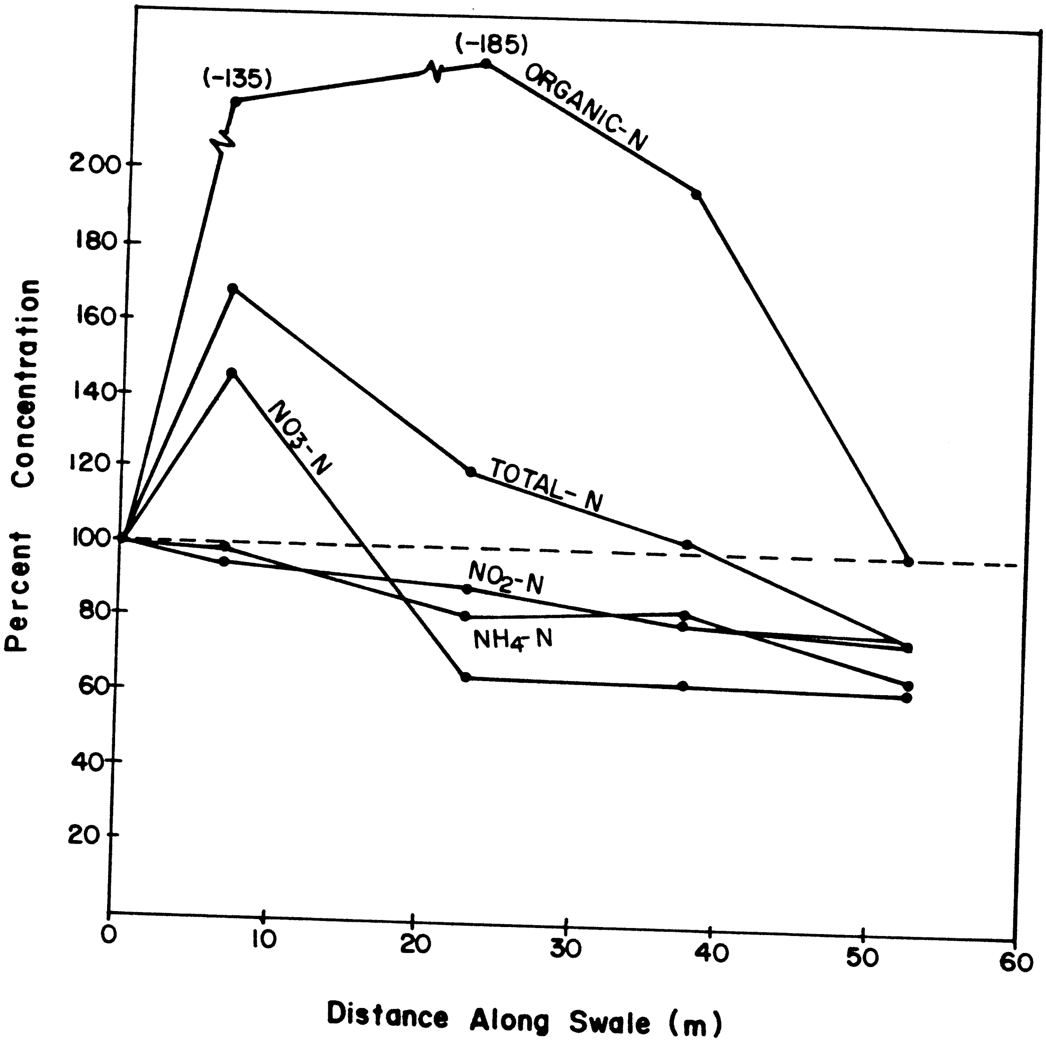

Figure 5-6: AVERAGE PERCENT CHANGES OF NITROGEN SPECIES ALONG THE LENGTH OF THE SWALE DURING THE MAITLAND SWALE STUDY ON 2-7-83.

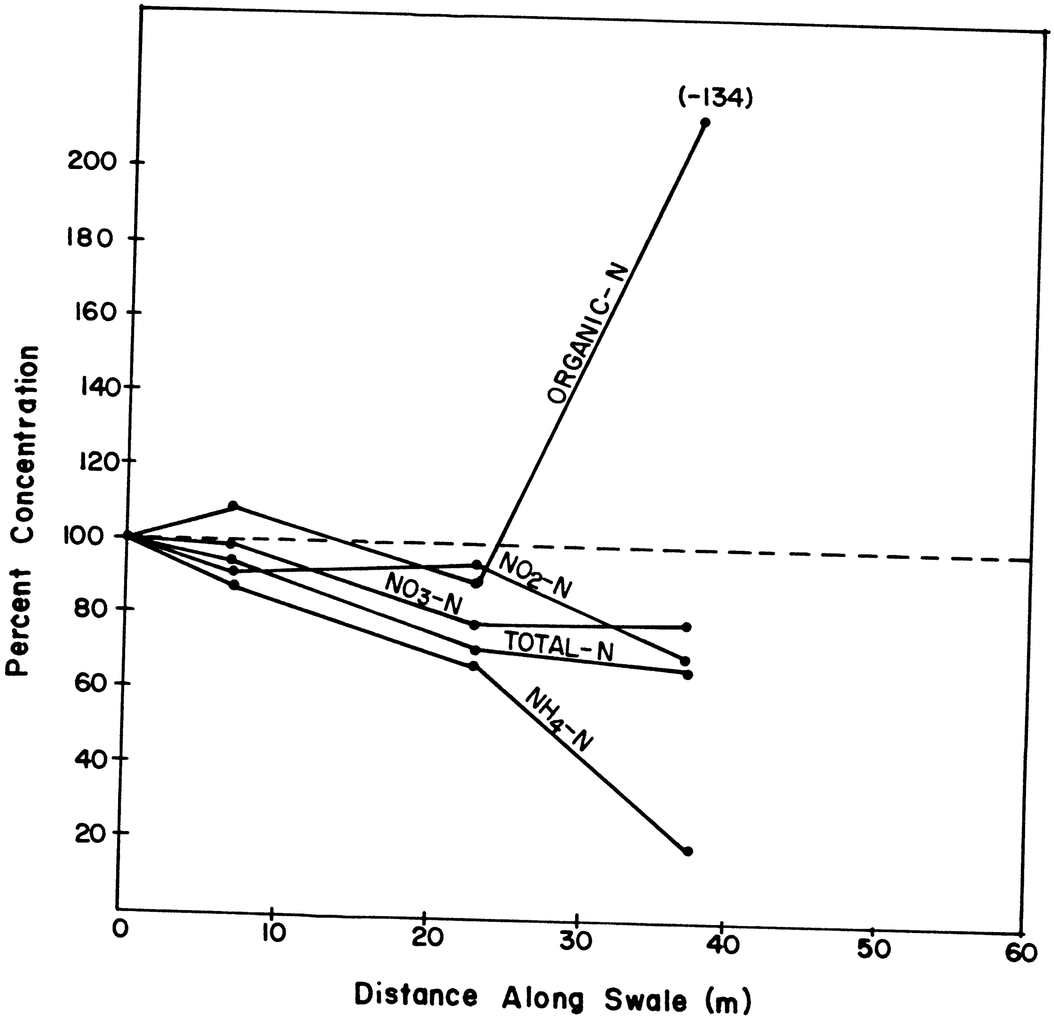

Figure 5-7: AVERAGE PERCENT CHANGES OF NITROGEN SPECIES ALONG THE LENGTH OF THE SWALE DURING THE MAITLAND SWALE STUDY ON 2-21-83.

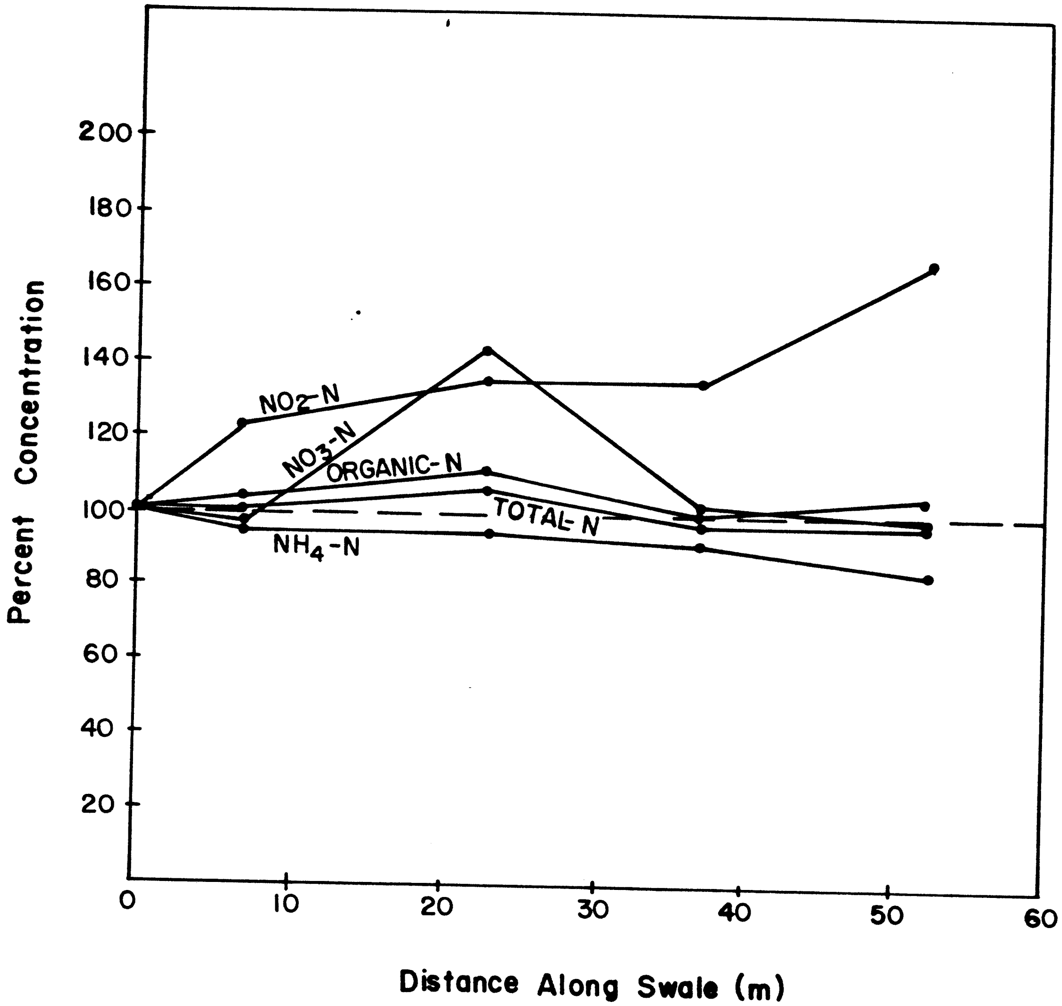

Figure 5-8: AVERAGE PERCENT CHANGES OF NITROGEN SPECIES ALONG THE LENGTH OF THE SWALE DURING THE MAITLAND SWALE STUDY ON 5-31-83.

TABLE 5-7

Transport of Nitrogen Species Through Swale at EPCOT Interchange Site on 3-23-83

Distance Along Swale (m)	Average Concentration (ug/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 5-8

Transport of Nitrogen Species Through Swale at EPCOT Interchange Site on 5-16-83

Distance Along Swale (m)	Average Concentration (ug/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

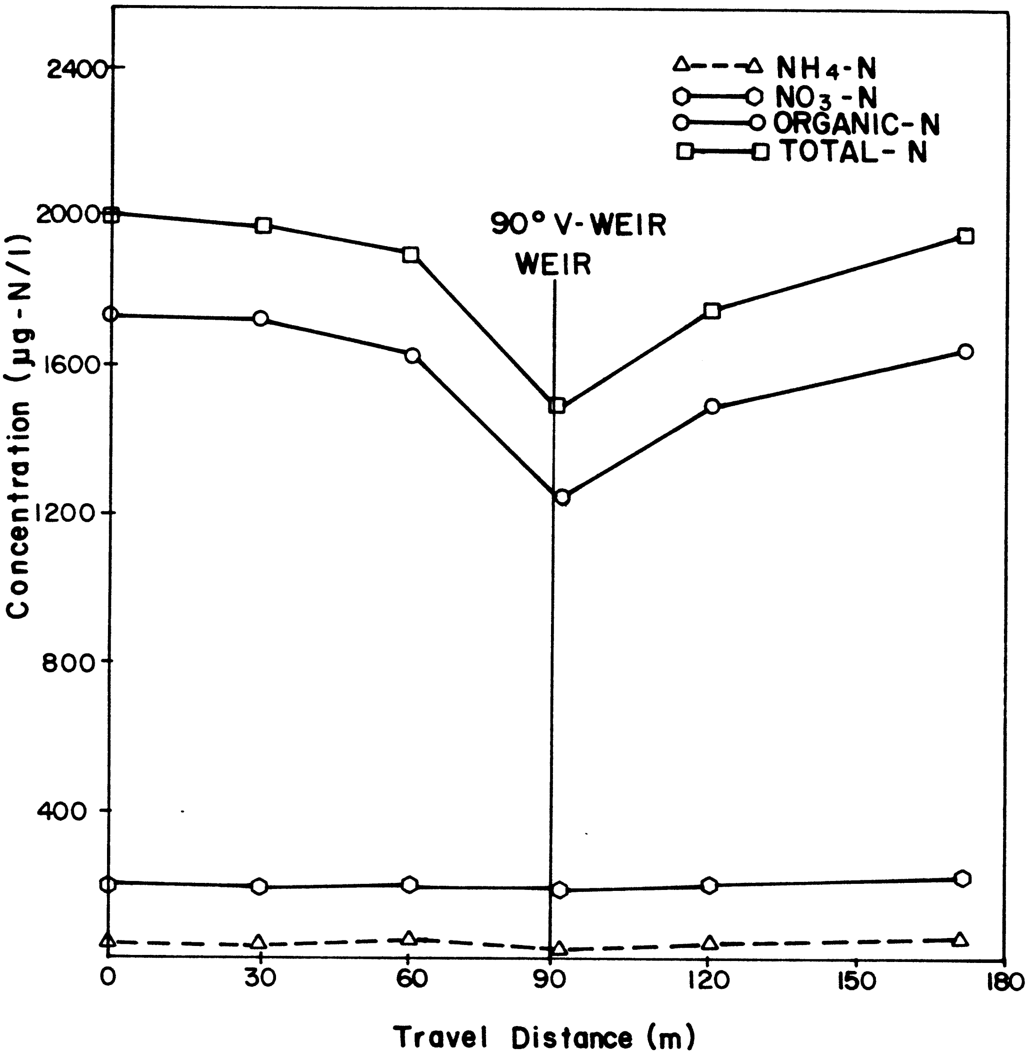

Figure 5-9: CHANGES OF NITROGEN CONCENTRATIONS IN SIMULATED HIGHWAY RUNOFF ALONG ROADSIDE SWALES AT EPCOT INTERCHANGE ON 3-23-83.

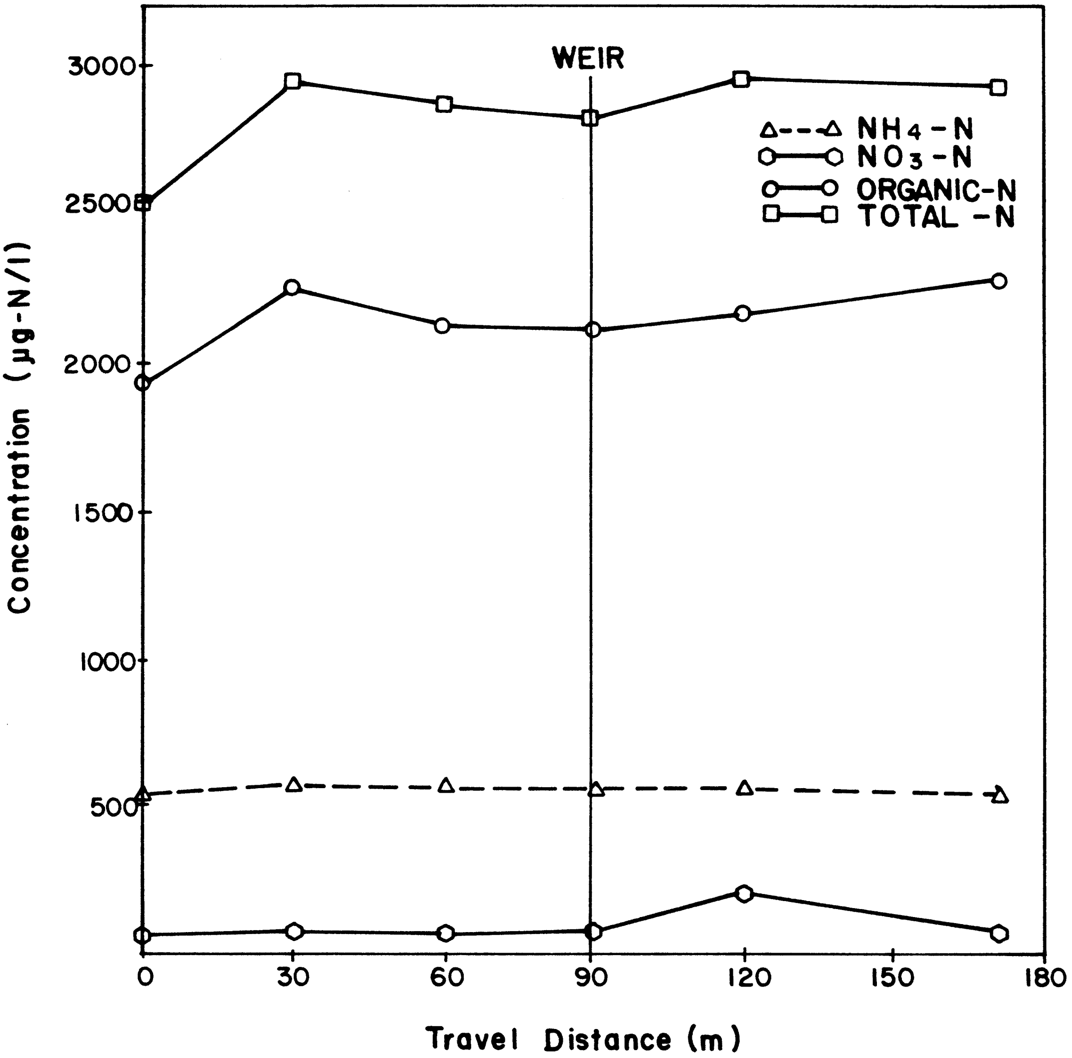

Figure 5-10: CHANGES OF NITROGEN CONCENTRATIONS IN SIMULATED HIGHWAY RUNOFF ALONG ROADSIDE SWALES AT EPCOT INTERCHANGE ON 5-16-83.

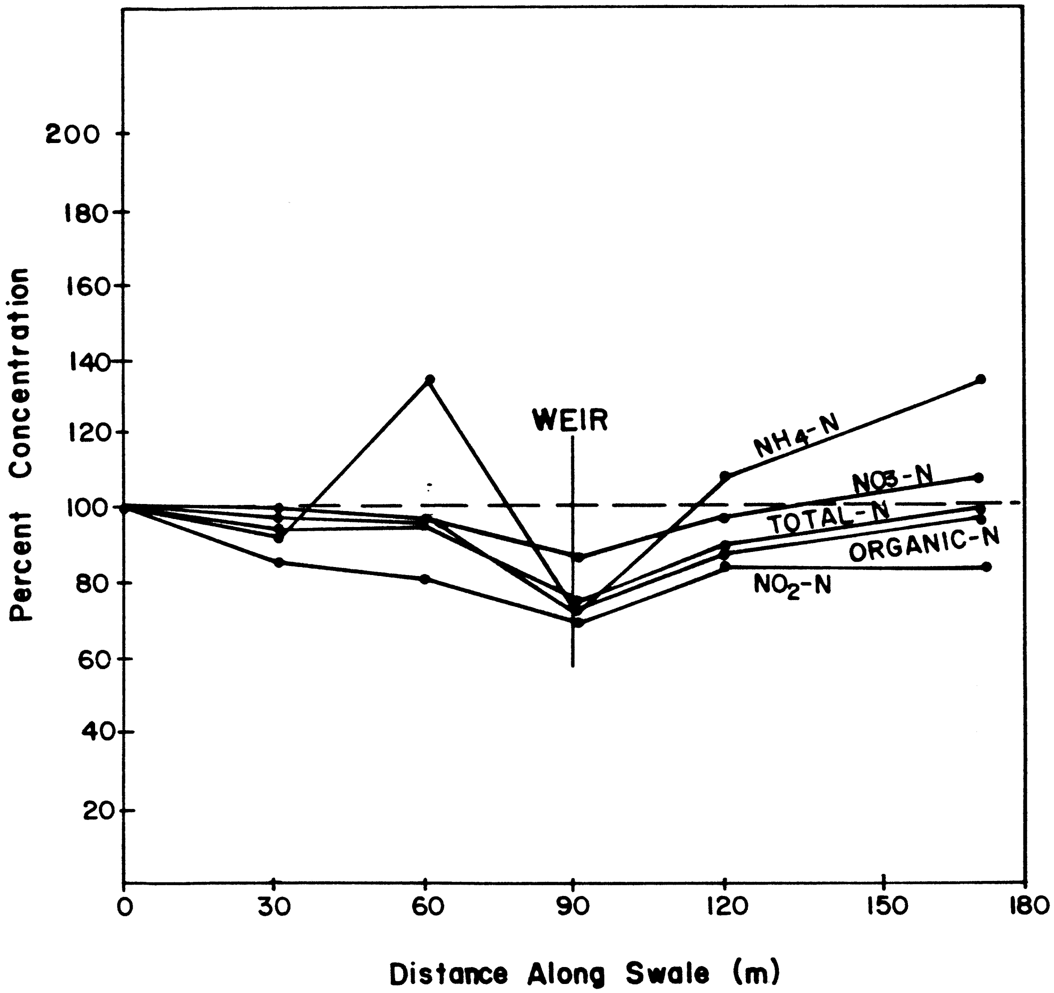

Figure 5-11 : AVERAGE PERCENT CHANGES OF NITROGEN SPECIES ALONG THE LENGTH OF THE EARTHEN SWALE DURING THE EPCOT SWALE STUDY ON 3-23-83.

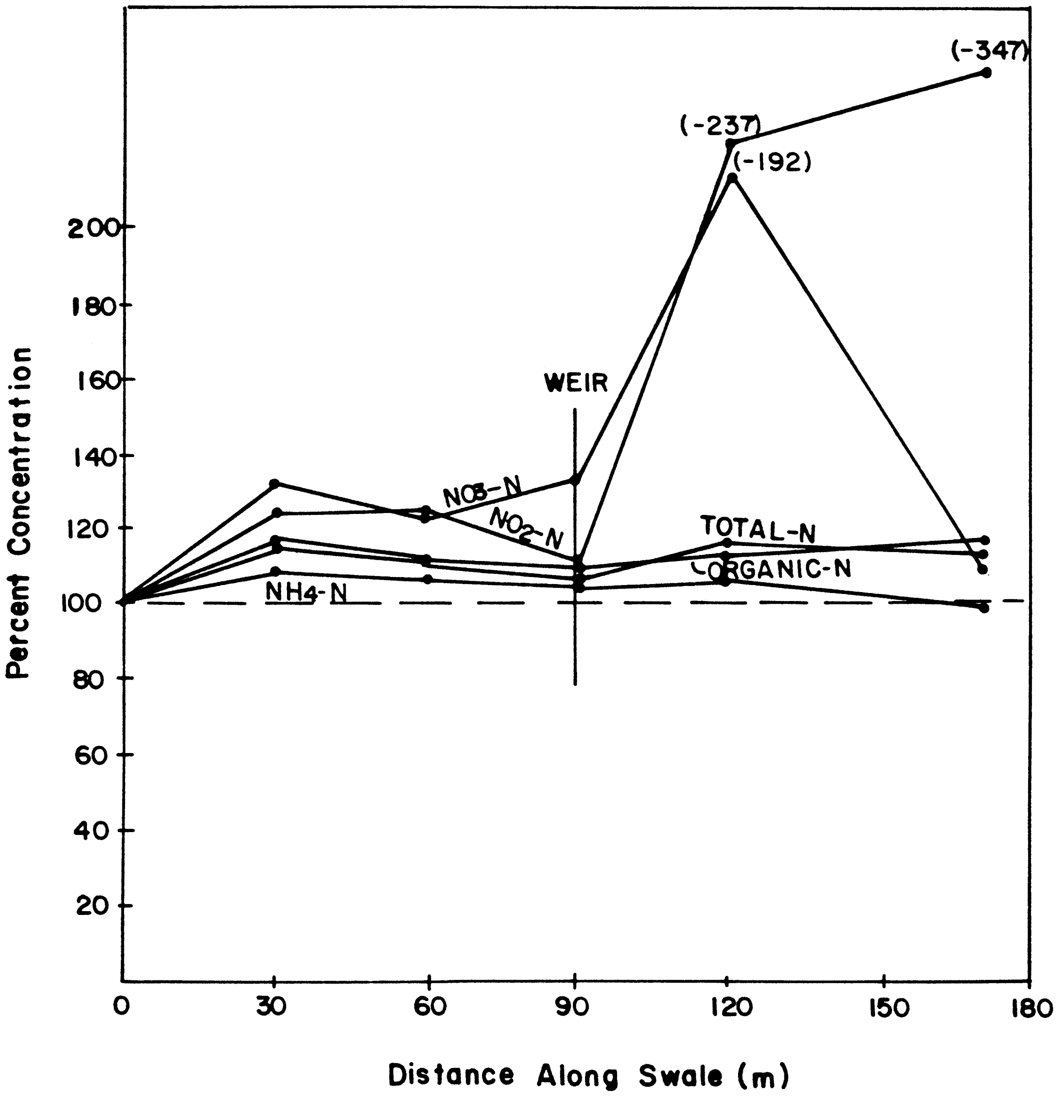

Figure 5-12: AVERAGE PERCENT CHANGES OF NITROGEN SPECIES ALONG THE LENGTH OF THE GRASSY SWALE DURING THE EPCOT SWALE STUDY ON 5-16-83.

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 5-7 shows that Organic-N declined 4 percent during the March 23, 1983 study. In contrast, Table 5-8 shows that Organic-N increased 18 percent during the second EPCOT study on May 16, 1983. The Inorganic-N concentration increased in both EPCOT studies, 6 percent during the March 23, 1983 study and 10 percent during the May 16, 1983 study. Table 5-9 provides a summary of the percent change of nitrogen forms along the length of the swale in all six swale studies.

TABLE 5-9

Comparison of Percent Changes of Nitrogen Forms Along the Length of the Swale During Six Swale Studies

Swale Study	Percent Change Along Length of Swale					
	NH_4-N	NO_2-N	NO_3-N	Inorganic-N	Organic-N	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

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 5-10. The percent nitrogen mass removal along the swale varied between 24 percent and 100 percent during the six swale studies. Nitrogen mass is retained in the swale area by infiltration, seepage, transpiration and soil-grass nitrogen interactions. The study at Maitland performed on February 21, 1983 showed 100 percent nitrogen removal through the swale area since the flow did not reach the end weir. Table 5-11 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 5-13. 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

TABLE 5-10

Total Mass of Nitrogen Through the Swales During the Six Swale Studies and the Corresponding Percent Change Between Outflow and Inflow

Swale Study	Mass of Total N (g) Inflow	Outflow	% Change
Maitland (1-24-83)	5	2	-61
Maitland (2-07-83)	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

TABLE 5-11

Detention of Nitrogen Mass, From Simulated Highway Runoff Through Swales

	Average Percent Change of Nitrogen Mass					
Experiment	NH_4-N	NO_3-N	NO_2-N	Organic-N	Total-N	Infiltration %
Maitland (1-24-83)	59	57	58	64	61	57
Maitland (2-07-83)	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.

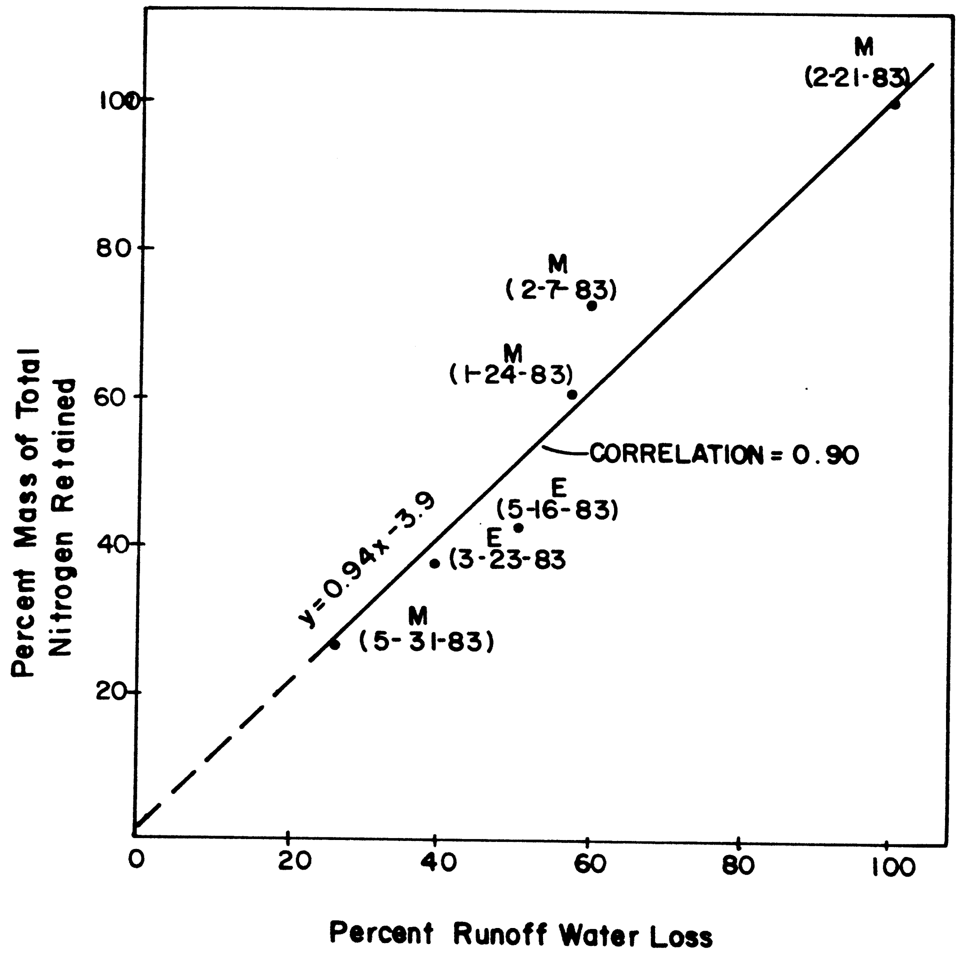

Figure 5-13: CORRELATION BETWEEN RUNOFF WATER LOSS AND NITROGEN RETETION DURING THE SWALE STUDIES AT MAITLAND AND EPCOT INTERCHANGES.

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 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 5-12. 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 5-13 and Figure 5-14 show a definite relationship between the average flow velocity permitted through the length of the swale and the percent 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 vegetative and climatic conditions. When vegetative and/or climatic conditions

TABLE 5-12

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 (2-07-83)	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 5-13

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

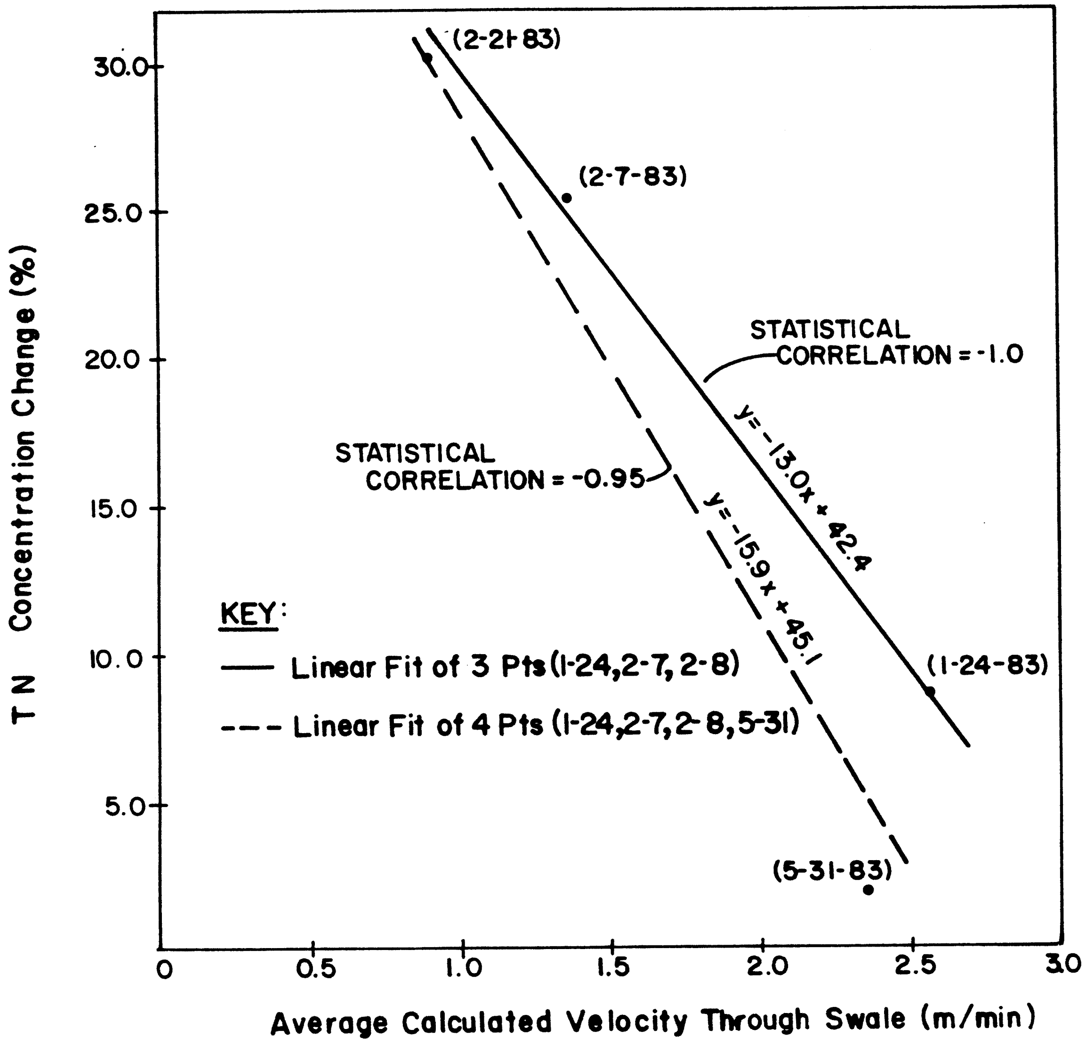

Figure 5-14: PERCENT CHANGES IN NITROGEN CONCENTRATIONS ALONG THE SWALE VS AVERAGE CALCULATED VELOCITY (m/min) FOR THE FOUR MAITLAND SWALE STUDIES.

change, the inverse relationship between velocity and nitrogen removal along the swale may be reduced.

The velocity in the Maitland swale study on May 31, 1983 was comparable to the study on January 24, 1983 (2.35 m/min and 2.58 m/min, respectively). The percent removals of total nitrogen, however, differ slightly. The study on May 31, 1983 produced a removal of 2 percent, while the study on January 24, 1983 produced about 9 percent nitrogen removal. The decreased removal in the May 31, 1983 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 May 16, 1983. The EPCOT swale study on March 23, 1983 produced a total nitrogen removal rate of 2 percent while the EPCOT swale study performed on May 15, 1983 produced an increase of total nitrogen along the swale of 14 percent. 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 5-14 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).

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 were 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 was not removed as efficiently as studies performed in January and February, and actually increased along the length of the swale during studies in May. Since the hydraulic characteristics at each site were comparable, other site conditions must be scrutinized.

TABLE 5-14

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

Average Swale Study Total-N	Velocity (m/min)	Site Conditions		Percent Change Along Swale		
		Cover	Season	NO_3-N	Organic-N	
Maitland (2-24-83)	2.58	Grass	Winter	-7	-3	-0
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

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 phenomenon of sorption is similar to the lake and wetland sediment sorption of nitrogen 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 5-11. 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 5-1. Table 5-2 shows the average percent changes for storm events in the summer (8-19-82, 8-23-82, 9-08-82, 10-01-82) and for storm events of winter (12-11-82, 1-21-83, 1-31-83, 2-03-83, 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 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 5-9 to 5-11 reveals that in the March 23, 1983 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 weir 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 May 16, 1983, as shown in Figures 5-10 and 5-12.

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 5-11 and in Figure 5-13, a linear relationship exists between average percent removal of Total-N mass versus 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 infiltration, assuming equal soil conditions and length in the swale during all the studies at each site. If the flow does not reach the end of the swale, mass removal efficiencies are considered 100 percent.

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 percent and 33 percent, respectively. However, the infiltration rate over the last 80 meter section of each EPCOT study was 9 percent and 25 percent, 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 March 23, 1983 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 March 23, 1983 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 5-15. 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.

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 built on dry soils and sloped as mildly as possible. The flow through velocities should be kept at minimum values to allow maximum infiltration and to retain most of the nitrogen mass on the swale site.

TABLE 5-15

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.

CHAPTER VI

SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS

Roadside swales are commonly used along highways to convey rainfall excess and to infiltrate runoff water flowing 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. One phase of the study was to evaluate the effectiveness of roadside swales in the removal of highway pollutants.

Runoff samples were collected from highways and adjacent roadside swales at Maitland Interchange for analysis of heavy metals (Pb, Zn, Fe, Cu, Ni, Cr, Cd) and nutrients (N, 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.

QUANTITATIVE ANALYSIS

Existing swale design methodologies are generally based on the rational formula for peak flow since there are scarce or no data on infiltration, roughness coefficients and fate of runoff pollutants through swales. Infiltration and soil storage appear to be two important parameters that can be used in evaluating swale pollutant removal efficiency (Anderson, 1982). It has been reported that a well maintained grassy swale percolates higher rates than areas which are unkept. It has also been reported that grassy cover generally enhances infiltration through swales. This may be caused in part by soil disturbances upon application of water to the earth's surface and easy compaction of soil in earthen swales. Anderson (1982) summarized available information and introduced new design methodologies for swales. He outlined a design approach based solely on hydrologic considerations for storage within a swale.

The design must incorporate both quality and quantity considerations. Since the primary mechanisms for management are infiltration and flow rates, quantitative estimates must be made. Infiltration rates are usually estimated from a knowledge of the soil permeability characteristics and the depth to the water table. Flow rates are estimated knowing the slope, cross-section flow areas, rainfall excess and the roughness characteristics of the swale.

The data presented in Chapter 2 describe the swale characteristics and experimental conditions. The inflow rates for the Maitland area varied between 0.026 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.

Several flow hydrographs were developed for the Maitland area and the EPCOT area. In the February 21, 1983 experiment at the Maitland

Site, water did not reach the end of the swale and was totally retained on the site. Water is generally lost by infiltration, seepage, evaporation and on-site storage. The hydrographs reflect clearly the water retention and the excess runoff from swale areas under various inflow rates. 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 0.066 m/m^2-hr (0 to 2.6 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 flows less than 0.036 m^3/m^2-hr (1.42 inch/hr). Excess runoff reached more than 90 percent of average flow at areas of the EPCOT site when the soil was saturated with moisture.

The hydraulic characteristics of the swale experiments showed that water depth, which is defined as the cross sectional area divided by the top width of flow, did not exceed 0.041 m (1.6 in). The calculated water velocity varied from 0.90 to 2.98 m/min during the swale experiments under steady state conditions of flow.

The double ring infiltrometer was used to estimate infiltration rates in the swale area at Maitland. The limiting infiltration rate using this method was approximately 0.125 m^3/m^2-hr (5 in/hr) with higher values of 0.305-0.375 m^3/m^2-hr (12-15 in/hr) reported during the early times of infiltration. For the hydrograph input-output studies these infiltration rates were never realized. Thus, the double ring infiltrometer may over-estimate the true infiltration rate. The calculated infiltration rate from the measured hydrographs at the Maitland site did not exceed 0.088 m^3/m^2-hr (3.46 in/hr).

Since rainfall excess, slope, average time of concentration, and length are directly measured, a relatively accurate estimate of the channel roughness measured by Manning's coefficient "n" should result. These results are shown in Table 6-1 (Wanielista et al., 1983). Also, there are many formulas for estimating peak discharge. With the well defined areas and relatively short time of concentration, the rational formula should provide an accurate estimate. Since the runoff coefficient, flow area and rainfall excess rate were available, it would be

TABLE 6-1

Manning's Roughness Coefficient and Calculated vs. Field Measured Peak Discharge Values

				Peak Discharge (m^3/min)	
Location	Date	Runoff Coef. C	Manning's "n"	Measured	Calculated
Maitland	1/24/83	0.43	.058	0.098	0.098
	2/07/83	0.41	.096	0.038	0.035
	5/31/83	0.75	.055	0.118	0.110
EPCOT	3/23/83	0.68	.044	0.131	0.156
	3/23/83	0.91	.059	0.118	0.143
	5/16/83	0.66	.035	0.145	0.127
	5/16/83	0.75	.046	0.131	0.097
			$\bar{n}$ = .056		

$$Q_p = C\ r\ A \qquad (6\text{-}1)$$

where: Q_p = peak discharge (triangular hydrograph)
C = runoff coefficient
r = rainfall excess to swale
A = watershed area

easy to compare flow measured with calculated peak discharge (Equation 6-1). The shape of the discharge hydrograph also gives an indication that the rational formula may be used.

The average value for Manning's roughness coefficient was 0.056 for both areas. During one heavy growth period, the roughness coefficient was as high as 0.096. If this value was discarded when calculating an average value for roughness, the coefficient would be 0.05. This value may help other designers of grassy swale systems.

When using the rational formula, peak discharge for the shorter swale (53 meters) was accurately predicted. However, for the longer swale (90 meters), the calculated value was not as accurate as compared to the measured value. The average watershed area may be the variable which was difficult to measure accurately.

QUALITATIVE ANALYSIS

Swales are designed to transmit, treat and store stormwater runoff. Contaminants in runoff, such as heavy metals and 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 soils. Dissolved contaminants will interact with surrounding water, soil and biota.

From the results obtained in swale experiments, it appears that the chemistry of heavy metals in natural waters is a fairly complex and site specific phenomenon. In the studies conducted at Maitland in which only inorganic species were assumed to be present, the solubility and removal efficiencies obtained for dissolved species appeared to be related to the dominant inorganic complex present. Those metal species which were present as a charged ion, such as zinc and iron, were removed to a significant degree. Those which were complexed with inorganic species and carried a diffuse charge or zero charge were not removed.

The importance of organic complexing in regulating solubility was demonstrated in the EPCOT experiments. Of the metal ions present, copper and iron are known to form significant metal-organic complexes and, as a result, no removal was found to occur. Other metals which formed no important organic complexes were regulated by their inorganic species. It also appears that the removal of dissolved metal species occurs more rapidly in bare earth swales than in grass-lined channels. Concentrations of iron, copper and chromium were actually increased by passage through a grassed swale.

Data collected over an eight-month period, from both highway and swale areas, indicate lower removal efficiencies than were obtained in the experimental spiked runoff. It is possible that certain metals may change forms between storm events and become solubilized. This is particularly likely in species which have a change in species from a charged free ion to a neutral ion in the pH range of 6-7.5.

Similarly, ionic nitrogen species (NH_4^+, NO_2^-, NO_3^-) and phosphorus species ($H_2PO_4^-$, HPO_4^{2-}, PO_4^{3-}) may be retained on the swale site by sorption, precipitation, co-precipitation and biological uptake processes. These processes can reduce the nutrient concentrations in highway runoff flowing over swales. Also, it appears that the removal of dissolved heavy metal, nitrogen and phosphorus species occurs more rapidly in bare earth swales than in grass-lined channels. Additionally, thinner grass seems to be more efficient in decreasing contaminants 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), 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 highway contaminants were observed at intermediate stations during swale experiments, particularly close to inflow point. This appears possible due to the initial flow effects on resuspension and resolubilization of loosely bound contaminants. The swale experiments showed better removal efficiencies at slow flow rates than at high velocities. The removal of nitrogen in swales on a concentration basis (measured in this study as ug/liter), is inversely related to the velocity of the runoff through the swale, i.e., directly related to 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 heavy metals, nitrogen and phosphorus 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 highway contamination loadings to adjacent receiving waters.

CONCLUSIONS

1. Minimum observed infiltration rates were 0.013 and 0.036 m^3/m^2-hr (0.5 and 1.4 in/hr) and maximum rates were 0.033 and 0.086 m^3/m^2-hr (1.3 and 3.4 in/hr) for swales studied at EPCOT and Maitland Interchanges, respectively. These rates are much lower than rates measured by double ring infiltrometer which shows three to four times higher values.

2. The measured runoff coefficients depend on the degree of soil saturation and the antecedent dry period. They varied between 0.41 and 0.91 during this investigation. Also, the calculated Manning's friction coefficient "n" for flow calculations through the swales generally varied between 0.035 and 0.059 with an average value of 0.053 for most of the cases.

3. Swales built on dry soils with good drainage and high infiltration rates showed better removal efficiencies for highway contaminants. Results from EPCOT suggest that removal of heavy metals decreases significantly in swale areas which are built on low soil and are constantly wet.

4. Concentrations of pollutants in highway runoff flowing over roadside swales may decline, increase or remain constant depending on

contaminant species, flow characteristics and swale environment. However, significant mass removals can be achieved if infiltration losses are considered. If no water reaches the downstream discharge, then the removal efficiency is actually 100 percent. Where infiltration is significant, removals from runoff will also be significant.

5. Swales appear to be more effective in removal of dissolved species of Zn, Cd, Ni, Pb, and Cr than Cu and Fe. Also removal efficiency is higher for metals than nitrogen and phosphorus. These efficiencies are governed by the predominant ionic species and complexes. Charged species are retained by sorption processes. Swales filter out particulate heavy metal and incorporate them in the soil. Heavy metals in highway runoff with large particulate fractions show higher removal efficiencies.

6. Removal of heavy metals may be caused by precipitation and sorption processes. Therefore charged ions and complexes may be removed more efficiently than stable complexes and non-charged particles.

RECOMMENDATIONS

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

a. Reduce slopes as much as possible.

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

c. Swales are less efficient for pollutant removal in areas where portions remain wet most of the time.

d. 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.

e. Plant a thin cover crop for erosion control and follow effective maintenance procedures; removal of grass clippings, loose debris and litter is desirable if practical.

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

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